PREDATION AND DOWNSTREAM MOVEMENT OF AGE-0 LAKE STURGEON IN THE SAGINAW RIVER BASIN

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ABSTRACT

Stocking is a primary management action for restoring Lake Sturgeon (Acipenser fulvescens) across the Laurentian Great Lakes, yet post-stocking predation and movement dynamics remain major uncertainties that can influence restoration success. I conducted complementary field studies to evaluate how hatchery rearing environments—non-natal streamside versus traditional groundwater-fed facilities—affect post-stocking movement and predation risk of age-0 Lake Sturgeon stocked into four tributaries of the Saginaw River Basin, Michigan, during 2022–2023. Fish were implanted with acoustic transmitters programmed with either alternating 60-day on/30-day off cycles (2022) or extended transmission delays with no off cycles (2023). Predation was evaluated using transmitters designed to change transmission code upon ingestion. Among tagged fish, 78% (52/67) were detected at least once after release with nine predation events based on transmitter codes. Maximum treatment-level predation reached 29% (4/14) in a given year, and overall predation did not differ significantly between rearing types (streamside = 21 %, traditional = 14 %; Fisher's exact test, $p = 0.72$). A simulation-based power analysis also showed that detecting a 20% difference in predation between groups with 80% power would require approximately 70 detected fish per treatment group. Downstream dispersal timing was analyzed using a time-to-event model with a Weibull distribution in a maximum likelihood framework across two movement states: exit from release tributaries (State 1) and exit from the Saginaw River (State 2), with a modified likelihood function to incorporate transmitter off intervals. Across years and states, movement patterns were similar between rearing types, indicating no strong effect of rearing environment on downstream dispersal. Together, these results suggest that non-natal streamside rearing did not confer measurable differences in predation risk or downstream movement dynamics after stocking.

This thesis is dedicated to my parents and grandparents.
Thank you for always believing in my dreams.

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GENERAL INTRODUCTION

1. JUVENILE FISHES

Due to their small size, juveniles may not be efficiently sampled using the same standard gear types that capture adults of the same species (Christiansen et al. 2020; Pritt et al. 2014). Further, habitat use may shift during developmental phases (Schlosser 1991; Scheerer and McDonald 2003; Szedlmayer and Able 1996). Predators may also drive juveniles fishes to areas of dense cover to avoid being preyed upon (Gillman and Fraser 1987; Lima and Dill 1990; Allouche 2002), resulting in fish inhabiting areas that may be more difficult to sample. As a result, data on juvenile fish are often sparse compared to adults for many fish species. Nonetheless, juvenile ecology influences population dynamics, recruitment, and mortality, and, thus, is important to study (Houde 1987; Sogard 1997; Walters and Juanes 1993). Furthermore, numerous conservation efforts across many taxa have included efforts focused on increasing juvenile recruitment, because producing, rearing, and releasing juveniles generally represents a feasible option for human intervention to support population growth and conservation goals (e.g. Lorenzen 2005; Bell et al. 2006; Catlin et al. 2015; Tetzlaff et al. 2019).

2. FISH STOCKING PROGRAMS

Many fish species can be successfully reared in hatcheries, but surviving until release does not mean stocked fish will recruit to the population or fishery (Trushenski et al 2010; Paquet et al. 2011; Trushenski et al. 2018). Maximizing the success of juvenile fishes from stocking programs requires applying the best available ecological knowledge related to growth, survival, and post-stocking movements (Flagg and Nash 1999; Mobrand et al. 2005; Peterson et al. 2007). Hatchery programs stock juvenile fishes because they are assumed to offer a cost-effective trade-off between rearing expenses, number produced, and expected post-stocking survival (e.g., Leber et al. 2005). Arguably the most common measure of success for hatchery-based conservation efforts is producing a self-sustaining number of sexually mature adults such that hatchery supplementation is no longer required for population persistence (Flagg and Nash 1999; Naish et al. 2009).

Several interacting factors determine whether stocked fish will recruit to the population and eventually contribute to future spawning, such as abiotic (e.g., temperature) and biotic (e.g., predators and competitors) factors (Houde 1987; Walters and Juanes 1997). These factors must align with life stage-specific needs, and ultimately determine if the environment fish are released into is capable of supporting them. Sometimes these factors can be evaluated prior to release (Brown et al. 2000; Hirzel and Le Lay 2008). For example, if thermal regimes no longer fall within the tolerance range for the species (Cherry et al. 1977; Payne et al. 2016), then releases will likely not succeed regardless of any other factors or actions taken. However, not all factors determining success can be evaluated prior to releases of animals into the wild.

Behavioral factors, including movement, also influence survival and ultimately determine whether will recruit to the population. For example, olfactory imprinting to natal waters during early life underpins homing for Pacific salmonids, implying that using natal (or chemically similar) water during rearing aids orientation and return for these species (Dittman & Quinn 1996). Evolutionary, this allows fish to return to suitable water where previous generations of fish were successful (Quinn 1993). Additionally, some species exhibit strong local adaptation (higher fitness in natal-like environments) and, therefore, stock–environment mismatches can depress survival and reproduction (Fraser 2011; Araki et al. 2007, 2008; Christie et al. 2012). Together, the abiotic/biotic suitability and behavioral factors broadly argue for environmental matching (water chemistry/odor, temperature) and exposure to cues that guide movements (locally adapted stocks) when designing juvenile fish stocking programs to increase the chances of juvenile survival and contribution to the adult population post-stocking. As such, recent years have seen increased efforts to rear fish under hatchery programs that consider these potential advantages for released fish, for example through the use of both traditional groundwater fed hatcheries or streamside facilities that instead draw water from surface water sources, often those waterbodies where the animals will ultimately be stocked into at the end of rearing.

3. STURGEON

Sturgeon species (Acipenseridae) populations have declined across the world (Lenhardt et al. 2006; Haxton and Cano 2016). The IUCN 2022 report classified all 25 sturgeon species between vulnerable and critically endangered. Remnant populations persist where access to spawning habitat remains, but most populations are well below historical levels. In the Laurentian Great Lakes, the status of Lake Sturgeon is rated “poor” but with improving 10-year trends (Environment and Climate Change Canada and U.S. Environmental Protection Agency 2022). For example, in Lake Huron, only a subset of tributaries historically used by Lake Sturgeon for spawning currently support self-sustaining populations (U.S. Fish and Wildlife Service 2023). Hatchery stocking programs are being widely used across the Great Lakes basin to support conservation efforts.

Past efforts have noted the difficulty of sampling juvenile Lake Sturgeon post-stocking (Auer 1996; Barth et al. 2009). Recovery programs have stocked juvenile Lake Sturgeon (*Acipenser fulvescens*) to supplement existing populations and restore extirpated ones following widespread declines from anthropogenic influences, including overharvesting, dams, dredging, and degraded water quality (Auer 1996; Holey et al. 2000; Peterson et al. 2007; Scott and Crossman 1973; Welsh et al. 2010). One way to evaluate stocking efforts is through physical captures to monitor abundance (e.g. Leber et al. 2005; Schloesser et al. 2021) and growth (e.g. Hoxmeier et al. 2006; Hervas et al. 2010; Elwer et al. 2023). For over twenty years, thousands of juvenile age-0 Lake Sturgeon have been stocked annually at sites across the state of Michigan (Michigan Department of Natural Resources, Fisheries Division, 2025), representing a significant federal, state, and Tribal investment in this species.

Lake Sturgeon hatcheries can be broadly categorized into two groups: 1) groundwater facilities and 2) streamside facilities. Groundwater facilities typically use particle filters and boilers to maintain favorable temperatures for growth and water quality (Ebeling et al. 2012). These hatcheries are often considered the "traditional" hatchery approach to rearing. In contrast, streamside facilities use filtered river water to reduce particulates while exposing fish to natural temperatures and odorants from systems in which they will be released (Holtgren et al. 2007). However, natal streamside facilities may not always be feasible due to practical constraints. As such, for non-natal stocking programs, questions remain as to whether traditional or non-natal streamside hatcheries are adequate to achieve management goals and which approach may work best.

4. STUDY OBJECTIVES

The overall aim for my thesis was to evaluate the differences in post-stocking movement and predation of age-0 Lake Sturgeon reared in two different hatchery environments – non-natal streamside versus traditional groundwater facilities. The study was conducted on four tributaries of the Saginaw River Basin, Michigan, with stocking occurring in 2022 and 2023. My first objective was to quantify first-year predation mortality of age-0 Lake Sturgeon to determine if different hatchery rearing methods conferred any survival advantages. My second objective was to estimate the downstream migration timing and the temporal spread of downstream migration events for age-0 Lake Sturgeon and whether there were differences between the hatchery environments. To achieve these objectives, I implanted age-0 Lake Sturgeon from two unique hatchery rearing strategies with Innovasea Systems Inc. V7-2x transmitters (standard and predation models), deployed an acoustic receiver array, and analyzed downstream movements using a modified time-to-event framework using a Weibull distribution that explicitly incorporated transmitter off-intervals. Predation proportions were compared between the two hatchery environments using Fisher's exact test (Kim 2017). A simulation-based power analysis was conducted to determine per-group sample sizes needed to detect plausible differences in predation between two groups for future predation experiments.

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CHAPTER 1:
Predation on age-0 Lake Sturgeon in the Saginaw River Basin, Lake Huron, U.S.A.

1. ABSTRACT

Predation can be a critical source of post-stocking mortality for hatchery-reared fish, yet remains poorly quantified for juvenile Lake Sturgeon (*Acipenser fulvescens*). I evaluated post-stocking predation of age-0 Lake Sturgeon reared at a traditional groundwater hatchery and a non-natal streamside facility stocked into four tributaries of the Saginaw River Basin, Michigan. Fish were surgically implanted with InnovaSea Systems Inc. V7D-2x "predation" transmitters designed to change transmission codes after ingestion. Passive acoustic receivers and targeted active-tracking surveys were used to detect predation events over a one-year interval. Among released fish with predation transmitters, 52/67 (78%) were detected, with nine confirmed predation events documented. The maximum tributary-level predation rate was 29%; the proportion of predated individuals combined across years, and release tributaries did not differ between rearing environments (streamside = 21%, traditional = 14%; Fisher's exact test, p = 0.72). Active tracking increased detection probability and revealed predation events that would have been missed through passive monitoring alone. A simulation-based power analysis indicated that detecting a 20% difference in predation between groups with 80% statistical power would require approximately 70 detected fish per treatment group. My results suggest that non-natal streamside rearing did not confer statistically significant antipredator advantages relative to traditional rearing at the scale evaluated. This study provides the first field-based quantitative estimate of age-0 Lake Sturgeon predation using acoustic predation transmitters and underscores the importance of using power analyses to help design future telemetry studies.

2. INTRODUCTION

Losses from predation immediately after fish stocking is a significant concern for fish management agencies because it decreases the likelihood that the stocking program will achieve its ultimate goals (Bailey and Houde 1989; Sogard 1997). Because of space limitations in hatcheries, fish frequently are stocked in early-life stages (e.g. larvae, spring fingerlings, fall fingerlings, yearlings) and therefore may be prone to predation because of their small size (Pepin 1991; Sogard 1997; Walters and Juanes 1993; Hixon and Webster 2002). Additionally, stocked fish have been found to have poorly developed foraging and anti-predator behaviors compared to their wild counterparts as a consequence of rearing conditions within hatcheries (Brown and Laland 2001; Brown and Day 2002; Salvanes and Braithwaite 2006). Understanding post-stocking predation can improve fisheries management by informing stocking procedures, such as the stocking level, timing of release, and location of release with the primary goal of stocking the fewest number of fish possible while still maintaining a high likelihood of survival (e.g. Lorenzen 2000; Buckmeier et al. 2004; Glover and Stephen 2023).

Controlling the number of fish released is often the most readily available means for managers to influence stocking programs. Stocking programs often release relatively large numbers of juveniles under the expectation that some fraction will survive (e.g. Cowx 1994; Aprahamian et al. 2004; Klinard et al. 2020; Fonken et al 2023). However, overstocking can lead to density dependent inter- and intraspecific competition that causes adverse effects on stocked juveniles, which might delay population recovery (Cowx 1994, Rose et al. 2001; Lorenzen and Enberg et al. 2002). Genetic introgression may also dilute desirable wild strain genetic diversity or traits if not carefully considered for the stocking program (Araki et al. 2007; Christie et al. 2012; Karlsson et al. 2016). Stocking fewer individuals could extend the expected timeline for population recovery, and given the investment of resources (financial, personnel, etc.) that go into stocking programs, it is desirable to achieve conservation goals within the shortest possible timeline. Therefore, among the many variables that must be considered when developing stocking strategies, reliable estimates of post-stocking survival, particularly losses due to predation, could be used to inform the total number of fish stocked.

In addition to the number of stocked juvenile fish, rearing practices can be used to increase fish performance and reduce predation risk after release. Rearing fish to larger sizes before release is expected to reduce vulnerability to predation, for example by exceeding the gape limits for some predators (Hyvarinen and Vehanen 2004; Grausgruber and Weber 2020). Exposure to intended release water conditions (temperature, pH, turbidity, etc.) and predator cues (i.e. conspecific alarm cues and predator odorants) in a hatchery setting may enhance behavioral traits that reduce the risk of predation (Flagg et al. 1999; Maynard et al. 2004, Feng et al. 2025). The potential advantages of these exposures prior to release contributed to the development of streamside hatcheries where fish are reared in water pulled from a surface waterbody (rivers or lakes).

Traditional methods to evaluate predation on fishes include visual observations (e.g. Tidwell et al. 2019), gastric lavage (e.g. Waters et al. 2004; Blankman et al. 2018), stomach dissections (e.g. Buckland et al. 2017; Taylor et al. 2024), molecular diet analysis (e.g. Oehm et al. 2017; Waraniak et al. 2018; Rubenson et al. 2020), or tag recoveries (e.g. Skov et al 2014; Babey et al. 2020). While analyzing stomach contents can be an effective method for evaluating predation (Hyslop 1980; Hartleb and Moring 1995; Waters et al. 2004), these methods can be costly and labor intensive (Mousing et al. 2023) and prey items can be difficult to identify (Garvey and Chipps 2012; Buckland et al. 2017). These methods may also require that the predator be euthanized for sampling, which may not be desirable for its impact on the population of potential predators.

An alternative approach for assessing post-stocking predation is through biotelemetry methods where stocked fish are tracked to infer their possible fates. Biotelemetry methods, including satellite, radio, and acoustic telemetry, are broadly used in marine and freshwater fisheries science to study movement behavior and assess population dynamics of fishes (Hussey et al. 2015; Matley et al. 2023). Recent advancements in acoustic telemetry that have both reduced the transmitter size and added sensors (Haflyard et al. 2017; Weinz et al. 2020; Shorgan et al. 2025) that can provide reliable indicators of predation have opened their use for studying the fates of fish post-stocking. While acoustic predation transmitters are not reliable outside of water, they offer a potential method to quantify predation events without the direct capture of predators and allow for detections of consumption over the full transmitter battery life.

However, biotelemetry, such acoustic telemetry studies, can require considerable logistical and financial investment, including receiver infrastructure, tagging equipment, and field effort for fish capture and equipment maintenance. These costs can constrain sample sizes (i.e., the number of fish tagged), which can potentially limit statistical inferences (Stiedl et al. 1997). Power analyses (e.g., Cohen 1992) offer a method for evaluating necessary sample sizes to detect statistical differences between treatment groups at desirable effect sizes, such as differences in the proportion of predated fish, which may help guide future studies.

One group of fishes that especially benefit from biotelemetry research into post-stocking is sturgeon species (Acipenseridae). Sturgeon species are the focus of many programs due to their conservation status (IUCN 2022), success at being reared in hatcheries (e.g. Gisbert et al. 2000; Van Eenennam et al 2001; Ireland et al. 2002; Crossman et al. 2011; Chebanov and Galich

2013), and general vulnerability of larval and early juvenile life-stages to mortality (e.g.Gadomski and Parsley 2005; Scribner and Baker 2017; Steffensen et al. 2010). Sturgeon species worldwide have experienced significant population declines due to a variety of reasons ranging from overharvesting, habitat loss, and barriers to migration (Auer 1996; Graham and Murphy 2007; Congiu 2023). As a result, hatchery stocking programs have become a key tool for conservation programs. Because sturgeon have long maturation times – often exceeding 10 years (e.g. Dadswell 1979; Keenlyne and Jenkins 1993; Bruch 1999; Erickson and Webb 2007) – and rearing fish to this age may not be feasible, many stocking programs rear fishes to a size that maximizes survival and minimizes costs associated with rearing (e.g. Margenau 1992; Leber et al. 2005; Losee & Phillips 2017). Thus, understanding sources of mortality such as predation post-stocking is of interest to sturgeon stocking programs and biotelemetry provides a means to track individual fish, providing information on movements and possible predation risk.

Lake Sturgeon (*Acipenser* fulvescens) is a species with a broad geographic range in North America that historically included the St. Lawrence, Hudson Bay, Great Lakes, and Mississippi River drainages. Throughout its range, lake sturgeon populations have declined from overfishing, habitat fragmentation, and habitat degradation (Pollock et al. 2014). Within the Great Lakes, it has been estimated that lake sturgeon populations have been reduced to 1% of historical abundance (Tody 1974; Hay-Chmeilewski 1997). Coordinated restoration efforts by state, federal, and Tribal agencies are occurring throughout the Great Lakes, with stocking being a major management action to achieve restoration targets. The majority of stocking programs release hatchery reared juveniles into tributaries or lakes during the first year of life. One area of focus is Lake Huron's Saginaw River watershed, which is in eastern Michigan, U.S.A. (Figure 1.1). This population was considered functionally extirpated in 2017 by the Lake Huron Lake

Sturgeon Working Group, a multi-agency task group headed by the U.S. Fish and Wildlife Service. In an effort to restore this Lake Sturgeon population, fish are released annually from two hatchery sources that differ in rearing strategies: a traditional groundwater facility (Genoa National Fish Hatchery, Genoa, WI, U.S.A) and a non-natal streamside facility (Black River Streamside Rearing Facility, Onaway, MI, U.S.A). This stocking program is now in a phase of evaluating the effectiveness of these two hatchery stocking sources. This includes interest in mortality related to predation post-stocking.

Despite visual observations of predation events on stocked juvenile Lake Sturgeon (e.g. Crossman et al. 2011), the magnitude and frequency of predation events remain poorly quantified. Previous field studies that targeted piscivorous fishes by investigating stomach contents did not observe predation on age-0 Lake Sturgeon (Caroffino et al. 2010; J. Jolley, personal communication), nor was predation observed in a laboratory experiment that placed stocking-sized Lake Sturgeon with potential predators (Crossman et al. 2018). These results are consistent with the goal of Lake Sturgeon hatchery programs to raise fish to a total body size that allow for the development of bony scutes that will reduce predation vulnerability (Peterson et al. 2007). Thus, while I expect predation of stocked juvenile Lake Sturgeon to be low, the use of biotelemetry tools to track individual fish provides the opportunity to evaluate this after fish are released into the wild.

The overarching goal of this chapter is to improve the use of stocking as a management tool to support lake sturgeon restoration in the Lake Huron basin. To address this goal, I pursued two specific objectives focused on post-stocking predation, an important early source of mortality. First, I used specialized predation-sensing transmitters (InnovaSea Systems Inc. 2019) to estimate the proportion of age-0 Lake Sturgeon predated within one year of release for two rearing strategies commonly used in Laurentian Great Lakes restoration programs: a traditional groundwater hatchery and a non-natal streamside facility. Second, I developed a simulation-based power analysis to evaluate the sample sizes needed to achieve 80% statistical power (Cohen 1992) across a range of differences in predation rates between the two treatments. Together, these objectives provide one of the first field-based estimates of post-stocking predation on age-0 Lake Sturgeon using biotelemetry and offer guidance for designing future telemetry studies that balance cost, detection probability, and statistical power.

3. METHODS

3.1 Hatchery Origin

Juvenile Lake Sturgeon were reared in two hatcheries that differed in rearing practices and water sources. The Black River Streamside Rearing Facility (Onaway, MI, U.S.A; hereafter BRSF) uses wild-caught Lake Sturgeon larvae collected from drift nets in the Black River, a tributary to Lake Huron, U.S.A. Fish at the BRSF were reared via a flow-through system with water drawn from a reservoir on the Black River. The non-natal streamside treatment group was intended to subject fish to natural environmental conditions such as temperature variability, and other system-specific odorants (Holtgren et al. 2007). In contrast, fish reared at the Genoa National Fish Hatchery (Genoa, WI, U.S.A; hereafter GNFH) originated from eggs collected from approximately 8-10 females captured from the Upper St. Clair River (Lake Huron, U.S.A.). Eggs from gravid females were fertilized with milt from 3-4 flowing males and transferred to the GNFH for rearing in a recirculating groundwater-fed aquaculture system (hereafter "traditional" rearing). Water temperature was controlled to reduce extreme variability through the addition of water from a cold-water well or boiler. Fish from both hatcheries began feeding on Artemiidae and later transition to Chironomidae larvae and Euphasiacae. While the rearing environment and stock selection differ, fish originated from the same genetic stocking unit, a population or group of populations that may be used as a donor source for stocking within the unit, for Michigan (Welsh et al. 2010) and both sources were stocked into non-natal release waters.

3.2 Acoustic Transmitter Implantation

In fall 2022 and 2023, fish from both hatcheries were implanted with acoustic transmitters and released into the Cass, Flint, Shiawassee and Tittabawassee rivers. Fish were implanted with a V7D-2x (69 kHz; diameter 7 mm, length 22 mm, weight in air 1.7 g) acoustic transmitter that changes transmission codes following digestion (Innovasea Systems Inc., 2019), hereafter referred to as a predation transmitter. Each year, five fish per hatchery were released into each tributary (n=20 fish/hatchery/year, 80 total).

In 2022, all transmitters were programmed with a signal transmission rate of 150s ± 30s and had an estimated battery life of 324 days for standard and predation transmitters, respectively. Due to limitations in battery life because of transmitter size and the goal of obtaining a 1-year observation period, transmitters were programmed with a repeating sequence of off intervals (60d on, 30d off, 60d on, 30d off, etc.) until the battery died. This programming was selected based on an ongoing study with similarly sized Lake Sturgeon in the Maumee River, OH, that was also assessing Lake Sturgeon movement dynamics post-stocking (McKenna 2023). In 2023, all transmitters were programmed with a constant signal transmission rate of 300s ± 60s with an estimated battery life of 311 days. This change occurred to remove the off interval so that the observation period was continuous throughout the battery life of the transmitter while maintaining a similar expected battery life as in 2022 (see Majinska 2025, Chapter 2, for additional consideration of these transmission schemes).

A key assumption of tagging studies is that tagged fish experience similar survival rates and exhibit similar behaviors as untagged individuals (Rogers and White 2007). In the case of juvenile sturgeon, multiple studies have shown that transmitter burdens ≤ 5% of fish mass have minimal impacts on age-0 fish (e.g. Snobl et al. 2015; McCabe et al. 2019). Additionally, no negative effects were visually observed prior to release in age-0 Lake Sturgeon implanted with V7-2x transmitters in a study in the Maumee River, Ohio, U.S.A., even though transmitter burdens approached 8.5% (McKenna 2023). For this study, fish were selected based on a minimum size of 165mm total length and 20g (~8.5% transmitter burden).

Intracoelomic acoustic transmitter implantation surgeries were conducted at each hatchery following the surgical and fish husbandry practices established under Michigan State University's Institution for Animal Care and Use Committee permit PROTO202000023 (2022) and PROTO20230067 (2023). Prior to transmitter implantation, food was withheld for 24-36 hours to reduce the risk of viscera expulsion during surgery following observations from two test surgeries. To minimize the potential of sutures tearing out from gastric distension, standard rations were withheld for 12 h post-surgery and then provided at 50% for one week before returning to standard rations until release.

Fish were anesthetized individually with Tricaine Methanesulfonate (MS222) using a dose of 125mg/L for 3-5 minutes, until they reached stage-4 anesthesia (unresponsive to touch and unable to maintain equilibrium; Summerfelt and Smith 1990; Coyle et al. 2004; Hegna et al. 2019) and were then maintained on a lower dose of 100mg/L during surgery. Incisions (~12-14mm) were made ventrally, between the midline and ventral scutes, approximately half-way between the pectoral and pelvic fins (Figure 1.2). Three interrupted 5/0 monofilament or vicryl sutures were used to close the incision. To minimize handling stress, length, weight, and transmitter ID were recorded prior to surgery while fish were anesthetized. Surgical equipment was sterilized with iodine or replaced with sterile equipment between each surgery. Following transmitter implantation, fish were monitored in a recovery tank until they regained equilibrium and routine swimming movements resumed (i.e., Stage-0 anesthesia; Summerfelt and Smith 1990; Coyle et al. 2004; Hegna et al. 2019). Fish were held after implantation for a minimum of one week prior to release for observation. Incisions were visually inspected several days post-surgery to ensure wounds were closed, no viscera were exposed, and there were no obvious signs of infection. When necessary, moribund fish were euthanized, and new fish were implanted if time permitted. In 2022, no fish were euthanized post-surgery at either facility. In 2023, 9 of 80 tagged fish (11%) were euthanized following surgery at the GNFH and no fish were replaced, and 11% (9/80) were euthanized at the BRSF but all nine fish were replaced.

The mean total length of fish with predation transmitters in 2022 was 207 mm ± 11 mm and 181 mm ± 6 mm for streamside and traditionally reared fish respectively. Mean tag burden was 5.2% (range: 4.0% - 7.0%) for streamside fish and 7.6% (range: 6.3% - 8.8%) for traditional fish in 2022. The mean total length of fish with predation transmitters in 2023 was 198 mm ± 7 mm and 185 mm ± 8 mm for streamside and traditionally reared fish respectively. The mean tag burden was 6.2% (range: 5.3% - 7.4%) for streamside fish and 7.8% (range: 6.3% - 9.0%) for traditional fish in 2023.

3.3 Release Sites

The Saginaw River Basin is in the mid-east of Michigan, U.S.A., and drains into Saginaw Bay, Lake Huron, Michigan (Figure 1.1). The Cass, Flint, Shiawassee, and Tittabawassee rivers comprise the four major tributaries to the Saginaw River. Fish were stocked at (1) Cass River: Gunzenhausen Park (43.32, -83.74), Frankenmuth, MI, (2) Tittabawassee River: Bob G.

Caldwell Municipal Launch (43.57, -84.19), Midland, MI, (3) Shiawassee River: Cole Park (43.18, -84.11), and (4) Flint River: Montrose Barber Park (43.17, -84.87) in Montrose, MI (2022), or Paddler's Landing (43.01, -83.73), Flint, MI (2023). Each river provides access downstream to Saginaw Bay with all rivers having limited upstream access due to dams or rock ramps that may limit fish passage. Streamside reared fish were stocked on September 25th and September 18th in 2022 and 2023, respectively. Traditionally reared fish were released on October 6th and September 28th in 2022 and 2023, respectively. Although the stocking dates were approximately one month later than the standard stocking dates for the Saginaw River Basin, the time between the releases mirrors current stocking program, and the delay between the stocking events and the standard stocking events was necessary to rear fish to an appropriate body size for tagging.

3.4 Acoustic Receiver Grid

In addition to the passive acoustic receivers (VR2W-69 kHz or VR2Tx-69kHz model receivers; InnovaSea Systems Inc., Halifax, NS Canada) deployed throughout the Great Lakes basin as part of the Great Lakes Acoustic Telemetry Observation System (GLATOS; Krueger et al. 2018), acoustic receivers (*n*=25) were deployed during the fall of 2022 in the Cass, Flint, Shiawassee, Tittabawassee, and Saginaw rivers (Figure 1.1). Tributary receivers were tended (i.e., data downloaded, and batteries replaced) in spring and fall of each year. Additional receivers (*n*=5) were placed at each river mouth in fall of 2023 (Figure 1.1).

3.5 Data Filtering

Data were separated by riverine or Saginaw Bay detections. Lake data were passed through the *false_detection*() function from the GLATOS package (Holbrook et al. 2024) in R (R Core Team 2024; Version 4.4.1) which calculates the probability of subsequent detections based on transmitter transmission rates and false detection likelihood established by Pincock (2010) to identify potential false detections. Any lake detections that did not pass the filter were removed from the data. This filter usually assumes that events based on single detections are false. However, I believed that one-detection events were plausible in the river system and, therefore, were not filtered. Poor "line of sight" brought on by variable bathymetry and meandering in the rivers can hinder the detection range (Kessel et al. 2014). This coupled with longer transmission rates raised concerns that multiple detections by a fish at a receiver in the river may not always be possible. I am further confident in this decision because visual examination of these single detection events did not result in implausible movement inferences, e.g., skipping between multiple locations. I assumed that lake-based detections were not subject to the same potential biases as in the river system, therefore only detections that passed the *false_detection*() filter were retained.

3.6 Active Tracking (2023)

Active tracking was used to supplement passive detections of predation transmitters within the tributaries during the second year (2023, Figure 1.3). All active tracking was conducted with a VR-100-200 or VR-100-300 mobile receiver unit and omnidirectional hydrophone (InnovaSea Systems Inc., , Halifax, NS Canada). The Shiawassee River was actively tracked on 6 January 2024, upstream of the rock ramp from the Ditch Rd. kayak launch (43.14, -84.13), Chesaning, MI, to the rock ramp at Cole Park (43.18, -84.11), Chesaning, MI (~8.6rkm). Large scale active tracking occurred via kayaking during the summer of 2024 when all transmitters were expected to be active. The Cass, Flint, and Tittabawassee rivers were tracked twice, and the Shiawassee was tracked three times. Additional effort took place in the Shiawassee because at least 50% of released fish were never detected during the study each year

(2022: 20 of 40 detected; 2023: 12 of 36 detected) whereas at least 79% of released fish were detected each year in the other tributaries. Each river was tracked from the release site to the second receiver downstream of the release site. The second receiver downstream of the release site was chosen as the stopping point for active tracking because it represented the largest gap in coverage between stationary receivers in the tributaries, in most cases it occurred near the river mouths.

To evaluate active tracking detection probability, two sentinel transmitters were deployed in each tributary, except the Shiawassee where three were deployed (Figure 1.3). Sentinel transmitters were attached to the middle of a 3-m rope secured with concrete blocks and deployed on the bottom of the river to mimic the benthic behavior of Lake Sturgeon (Peterson et al. 2007). Tracking crews were not informed of their locations and were told not to wait in areas of previous detections. Because the survey areas were large, each tributary was divided into three reaches, and not all reaches contained sentinel transmitters.

When a potential transmitter was detected, the crew stopped and listened for at least twice the maximum transmitter transmission rate or until a full ID sequence was received. This was repeated after a transmitter was found to assess if other transmitters were present. Transmitter ID, location, and time were recorded. The initial desired speed of tracking was 3.2–4.8 kph. However, after only one of three sentinel transmitters was identified in the first completed survey, the desired speed was reduced to 1.6–2.4 kph in an attempt to increase the detection probability after confirmation was provided the sentinel transmitters were active. Establishing a slower speed for kayaking downriver increased the number of sentinel tag detections from 1 of 3 (33%) sentinel transmitters during the initial full tracking event of the Shiawassee River to 3 of 3 (100%) sentinel transmitters during the next pass. Given the tracking speed was changed after the first event in the Shiawassee River, data from this initial event were excluded in the analysis of active tracking detection probability and the river was re-done at the new speed.

Active tracking was evaluated based on detection probability. Detection probability was calculated by summing the number of unique sentinel transmitters detected per full pass of a tributary divided by the sum of known sentinel transmitters within the tributary:

Eq. 1 $$Detection\ Probability = \frac{\sum_{i=1}^{n_{passes}} n_{detected,i}}{n_{\ passes}\ x\ n_{\ sentinel}}$$

where n_{passes} is the total number of completed passes for a tributary, $n_{detected}$ is the number of unique transmitters detected during pass i, and $n_{sentinel}$ is the number of sentinel transmitters in the tributary. This metric provided relative context for evaluating the effectiveness of active tracking and confidence in detecting a transmitter.

3.7 Predation Analysis

The proportion of predated individuals was based on the subset of fish that were implanted with specialized V7D "predation" transmitters (n=20/treatment/year). Predation was identified when a transmitter indicated that a predation event occurred. Transmitter sensor values remain at one until the capsule on the end of the transmitter is dissolved in stomach acid, in which case the sensor value increases as time passes until the sensor value maxes out at a value of 255 (35 days, InnovaSea Systems Inc., 2019). Fish released in the Cass River were excluded from the analysis in 2022 due to two receiver malfunctions in the Cass River. In 2023, active tracking detections were combined with passive array data to supplement detections of predation transmitters. The proportion of individuals predated was calculated as the number of unique transmitters with a predation signal divided by the total number of predation tags detected during the study interval. Only detected transmitters were included in the analysis as the fate of undetected transmitters could not be determined. The proportions of predated fish were calculated for each year based on the passive array with the inclusion of active tracking in 2023. The proportions of predated fish were compared between treatment groups, combing the data from 2022 and 2023. Data was compared between hatchery treatments as counts of categorical data, predated vs not predated, in a contingency table using Fisher's Exact Test due to low sample size (Kim 2017). I did not investigate patterns in the timing or location of predation events given the ability to identify the time that a sensor value changed is limited to 35 days (Innovasea Inc. 2019), active tracking occurred towards the end of the battery life, and few predation events occurred.

3.8 Power Analysis

Using the statistical test from the analysis of predation of my field data, I conducted a simulation-based power analysis to estimate the per-group sample sizes needed to detect differences in the proportion of predated fish between treatments to direct future studies (e.g. Cohen 1992). I did not incorporate the off-periods used in the field study to inform this analysis. Power, in this case, was defined as the proportion of significant p-values ($\alpha \leq 0.05$) produced for a statistical test at a given sample size and effect size. Effect size was defined as the difference of proportions between groups. For each effect size (Δ = 10%, 20%, 30%, and 40%; Group A fixed at $P_A=0.10\%$, and Group B at $P_B=P_A+\Delta$) and each per-group sample size *n* (10–300 in increments of 20), I generated 10,000 replicate datasets by drawing counts of predated fish for each group from a Binomial distribution and forming a 2×2 contingency table (predated vs. not predated × hatchery treatment). Group A was set at 10% to represent a low proportion of predated fish. Differences in the proportion of predation between groups were tested with Fisher's exact test (two-sided, $\alpha=0.05$), and power was estimated as the proportion of replicates that were statistically significant. For each effect size, I identified the smallest *n*, per group sample size, to achieve 80% power (Cohen 1992). Cohen (1992) noted that 0.80 is not a mathematical requirement but a widely adopted convention that balances the risks of Type I and Type II errors. Using β = 0.20 (power = 0.80) keeps the probability of a false negative within the same order of magnitude as a false positive (when α = 0.05), while avoiding the impractically large sample sizes required to achieve higher power levels. Although the choice of power reflects a value judgment about acceptable risk, 80% remains the most commonly used standard in ecological research (Stiedl et al. 1997; Walsh et al. 1999; Johnson et al. 2015). I used a Fisher's Exact Test to evaluate each simulation to match the analysis used to evaluate predation in section 3.7 of this chapter.

4. RESULTS

4.1 Predation

A total of 67 fish were released with predation acoustic transmitters across both years of the study and the four release tributaries (Table 1.). Fish released in the Cass River in 2022 (*n* = 10) and three (*n* = 3) tagging mortalities from traditionally reared fish in 2023 were removed from the study. Of the fish released, 52/67 individuals (78%) were detected at least once following release. Detections for individual fish occurred for up to 312 days post-stocking (median = 127, range = 0.4 - 312).

A total of nine individuals (17% of all detected individuals) were detected with transmitters with a predation signal. In 2022, the stationary receivers, representing the passive array (Table 1.1) detected a single transmitter with a predation signal. In 2023, I detected three individuals with predation signals using the passive array (Table 1.1); but when I included detections from the active tracking transects, the number of individuals with predation signals increased to eight (see Table 1.1 for details). The highest observed annual predation proportion for a single hatchery type was 29% (traditional hatchery, 2023; see Table 1.1). The proportion of predated individuals when the data were pooled across years and with the inclusion of active tracking data was 21% (5 of 24) for streamside, and 14% (4 of 28) for traditional fish (Table 1.1). There was not a significant difference in the proportion of individuals detected post-stocking with predation signals between the two hatchery types with data grouped across years and the release tributaries (Fisher's Exact Test: p=0.72).

4.2 Active Tracking Detection Probability

Active-tracking detection probability at the reduced speed (<2.5 kph) varied among tributaries: Cass = 1.0, Flint = 0.5, Shiawassee = 0.75, and Tittabawassee = 1.0. Active tracking in 2023 identified three predation transmitters that were never detected on the passive array and two additional transmitters that were last observed on the passive array without a predation signal but were subsequently detected with predation signals during active tracking.

4.3 Power Analysis

The approximate per group sample size required to achieve 80% power for a two-group comparison of predation proportions using a Fisher's Exact Test (two-sided, α=0.05) was 210 fish for a 10% effect size, 70 fish for 20%, 30 fish for 30%, and 25 fish for 40% (Figure 1.4).

5. DISCUSSION

5.1 Predation

This study found that predation rates of non-natal hatchery reared juvenile lake sturgeon released into the tributaries of the Saginaw River ranged from 0.0 to 0.29 with no significant difference between rearing environments detected. Several factors likely contributed to these predation patterns. First, juvenile sturgeon hatchery programs are designed to release fish both at increased size and after the development of dermal scutes that are expected to reduce vulnerability to predation (Peterson et al. 2007; Baker and Scribner 2017). Laboratory trials on sturgeon species have demonstrated that piscivores selected alternate prey types when given the choice (Gadomski and Parsley 2005; French et al. 2010). However, unlike other studies that did not detect predation of age-0 Lake Sturgeon (Cariffino et al. 2010; Jeff Jolley personal communication), I detected multiple predation events following stocking. Overall, this study represents the first use of predation transmitters to evaluate post-stocking predation of age-0 Lake Sturgeon and these results indicated that predation should not be assumed to be absent (e.g. McDougall et al. 2014) as a factor influencing the overall survival of juvenile Lake Sturgeon in the first year after stocking.

In addition to the observed predation, the comparison of hatchery types showed no significant differences in the proportion of fish detected with predation signals. This similarity suggests that the characteristics, whether physiological or behavioral, influencing predation risk were comparable between the non-natal streamside- and traditionally reared juvenile Lake Sturgeon at the time of release. Streamside facilities are designed to expose fish to more natural water temperatures, chemical cues, and potential predator odors that could enhance antipredator responses before release (Berejikian et al. 1999; Flagg et al. 1999; Maynard et al. 2004). However, the absence of a detectable difference between the two treatment groups for juvenile Lake Sturgeon may indicate that 1) streamside conditioning benefits are not transferable when stocking occurs in a non-natal stream, 2) any potential benefits from streamside rearing were not related specifically to predation during this stage of development, or 3) that both hatchery programs produced juveniles that had already reached a size threshold that conferred similar vulnerability levels to predation.

5.2 Power Analysis and Acoustic Telemetry

My power analysis simulations highlight the potential logistical challenges of detecting treatment effects under low sample sizes, even with large effect sizes. Under conditions of low predation, even moderate differences between the simulated treatment groups would require large sample sizes to achieve adequate statistical power. For example, detecting a 10% difference between groups in smallest effect size scenario (Group A = 10%; Group B = 20%) would require 220 fish to be tagged per treatment group. Using the cost of V7D-2x transmitters ($750 per transmitter) at the time of this study, to tag approximately 440 fish would cost approximately $326,000 USD in transmitters, and this assumes that every fish is detected post-stocking. After completing the power analysis simulations, I consulted multiple managers involved with Lake Sturgeon stocking in the Great Lakes and they indicated a general desire to be able to detect a 20% effect size in predation between treatment groups (via personal communication from Ed Baker, Jeff Jolley, Justin Chiotti, and Tim Cwalinski). To achieve this with 80% power would require approximately 70 detected fish per group. Applying the same transmitter costs and assumption of all fish being detected, this would require approximately $104,000.00 USD in transmitters. Aside from the transmitter costs involved, these sample sizes require tagging a large portion of the annual per hatchery stocking quota (500 fish per hatchery) in the Saginaw River Basin. This highlights the importance of study design considerations both in the number of necessary transmitters and the potential methods that may be used to increase the detection probability of transmitters for future studies. The financial investment associated with such studies may be worthwhile if the information gained improves the cost-effectiveness of the stocking program.

I emphasize that I am not interpreting the current results via post-hoc (observed) power, which is mathematically redundant given power and p-values are correlated (Hoenig & Heisey 2001). Instead, my simulation-based power analysis—conducted after data collection—serves to guide future studies by using the information gathered here to quantify the necessary trade-off between the predicted effect size and different sample sizes. Such power analyses can be used to balance research objectives, plausible effect sizes, desired power, and cost (Steidl et al. 1997). In the real world, there is a detection probability that is not accounted for in my power analysis. This further highlights the necessity to evaluate methods to detect predation events, including the addition of active tracking along with the passive array, to maximize detections of the transmitters for inclusion in analyses to reduce "lost" data via undetected fish which would therefore reduce the need for extra transmitters to account for expected missing transmitters.

5.3 Active Tracking Considerations

The integration of passive acoustic monitoring with active acoustic tracking, as done here in 2023, has been employed in other study systems as well (Chavarie et al. 2022; Mensinger 2024). Passive receivers enable continuous long-term monitoring of relatively small total area of the river system, whereas active tracking provided intermittent monitoring of the gaps between receivers. I opted to use a kayak (see section 2.3.6), similar to other approaches deployed by Chavarie et al. (2022). Additionally, through the use of sentinel transmitters, I refined the active tracking speed to improve detection probability. Reducing the tracking speed to < 3 kph increased detection probability to 75% (see section 2.3.6).

The inclusion of active tracking in 2023 not only provided increased detections of predation events by covering more geographic area but also reclassified two fish that would have been considered 'alive' based on the passive receivers as 'predated' when they were detected downriver of the last passive receiver they were detected at. Thus, while the use of both active and passive tracking techniques in the same study can provide additional information about the fates and movements of tagged animals, they should be integrated with caution, and the assumptions of both methods should be considered in estimations of detection efficiency.

5.4 Sturgeon Management and Implications

Comparable predation levels of the streamside- and traditional-hatchery sourced Lake Sturgeon in this study suggests that both rearing strategies can effectively contribute to restoration efforts when viewed from the perspective of predation risk post-stocking in a non-natal environment. Streamside facilities are often promoted for their potential to promote imprinting or enhance conditioning for release into natural systems (Holtgren et al. 2007; Kimmel et al. 2023), but the absence of differential predation here indicates that these assumed benefits did not translate into a statistically significant difference in predation under the conditions examined here.

Results from this chapter should be interpreted with caution and not be taken as a measure of the overall success of the Michigan Department of Natural Resources and U.S. Fish and Wildlife Service Lake Sturgeon stocking program in the Saginaw River Basin. For example, predation only represents one component that affects the survival of juvenile fishes (Houde 1987; Walters and Juanes 1993). Furthermore, there are many other factors to consider when evaluating the overall efficacy of a stocking program (i.e. survival, return rates, spawning success, etc.; e.g. Jonsson et al. 2002; Saloniemi et al. 2004; Fraser 2008; Thériault et al. 2011). It is unclear how estimates of predation alone translate into juvenile survival or the number of spawning adults. As more fish are stocked into the system and as stocked fish reproduce, there may also be density dependent intraspecific and interspecific competition that may limit recruitment (Lorenzen and

Enberg 2002; Ward et al. 2006). Lastly, survival alone does not mean adults will return to the intended area (e.g. Quinn 1993; Jonsson et al. 2002), if they will be reproductively successful (e.g. Thériault et al. 2011), and if natural recruitment alone will be sufficient to maintain management desired population abundance. Therefore, while the results of this study inform predation of juvenile sturgeon during the first year after release, it does specifically address the overall success of the stocking program in the Saginaw River Basin.

5.5 Future Directions & Conclusions

While I was able to quantify the proportion of fish predated over 1-year post-stocking, the study likely had limited statistical power to detect any effect size less than 40% in hindsight as evidenced by findings from the power analysis I conducted. This was further exacerbated by the fact that I pooled data across years and tributaries to increase the sample size and, therefore, I did not compare predation between years or tributaries. Given this finding, I suggest future studies consider power analyses such as the one used here prior to conducting a field study.

Collectively, this study provided new insights into post-stocking predation of age-0 Lake Sturgeon and demonstrated the value of biotelemetry as a tool for estimating predation in the context of restoration programs. To my knowledge, this study represents the third time predation on age-0 Lake Sturgeon has been attempted to be quantified in the field (e.g. Cariffino et al. 2010; Crossman et al. 2011) but is the first based on acoustic detections using predation transmitters. Predation was observed across hatchery treatments which indicated that predation was and cannot be dismissed as a factor (e.g., McDougall et al. 2014) in the overall survival of juvenile Lake Sturgeon. Based solely on predation, these results support the continued use of both traditional and streamside hatchery sources in non-natal environments for Lake Sturgeon recovery efforts, assuming cost is not a limiting factor, as neither approach exhibited detectable increased susceptibility to predation. Additionally, using field-derived analysis methods from this study in a power analysis, I report that similar studies may need a relatively large investment in predation transmitters depending on the questions and effect size that researchers or managers desire to detect. Broader applications of these methods across additional tributaries will further refine understanding of early survival for Lake Sturgeon across the Laurentian Great Lakes basin.

6. TABLES AND FIGURES

Table 1.1 Proportion of predated fish over 1-year post-stocking by year, treatment, and data type. Active tracking only occurred in 2023. Fish stocked in the Cass River in 2022 were removed from the analysis due to receiver download issues. There was no significant difference in the portion of predated individuals between treatment types based on Fisher's Exact Test (p = 0.72).

Year	Treatment	Data	Number Released	Total Detected	Detected Predated	Detected Not Predated	Proportion of Predation	Fisher P-value
2022	Streamside	Passive Array	15	12	1	11	8%	
	Traditional	Passive Array	15	19	0	19	0%	
2023	Streamside	Passive Array	20	15	3	12	20%	
	Traditional	Passive Array	17	12	0	12	0%	
2023	Streamside	Passive and Active Tracking	20	16	4	12	25%	
	Traditional	Passive and Active Tracking	17	14	4	10	29%	
All Years	Streamside	Grouped with Active Tracking (2023)	35	24	5	19	21%	
	Traditional	Grouped with Active Tracking (2023)	32	28	4	24	14%	0.72

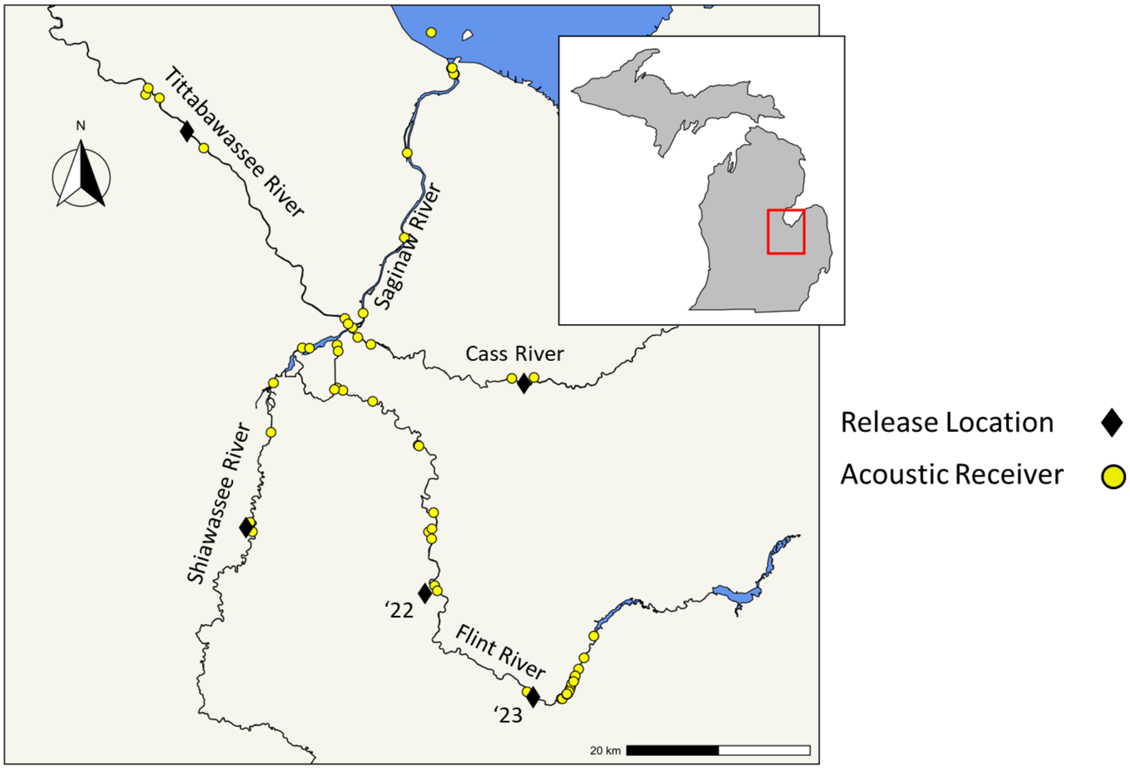

Figure 1.1 Map of Saginaw River Basin, including all four stocking tributaries. Black diamonds represent the stocking locations. Due to a chemical spill in 2022, fish were stocked downstream of the standard stocking location in the Flint River. Yellow dots represent acoustic receivers. Each tributary flows into the Saginaw River and ultimately to into Saginaw Bay, Lake Huron.

Figure 1.2 Image of incision location and length (~12-14mm) made for acoustic transmitter implantation of an InnovaSea Inc.V7-2x and V7D-2x acoustic transmitters in age-0 Lake Sturgeon. Transmitters were pushed anterior towards their head. Three interrupted sutures were spaced equidistant to close the incision.

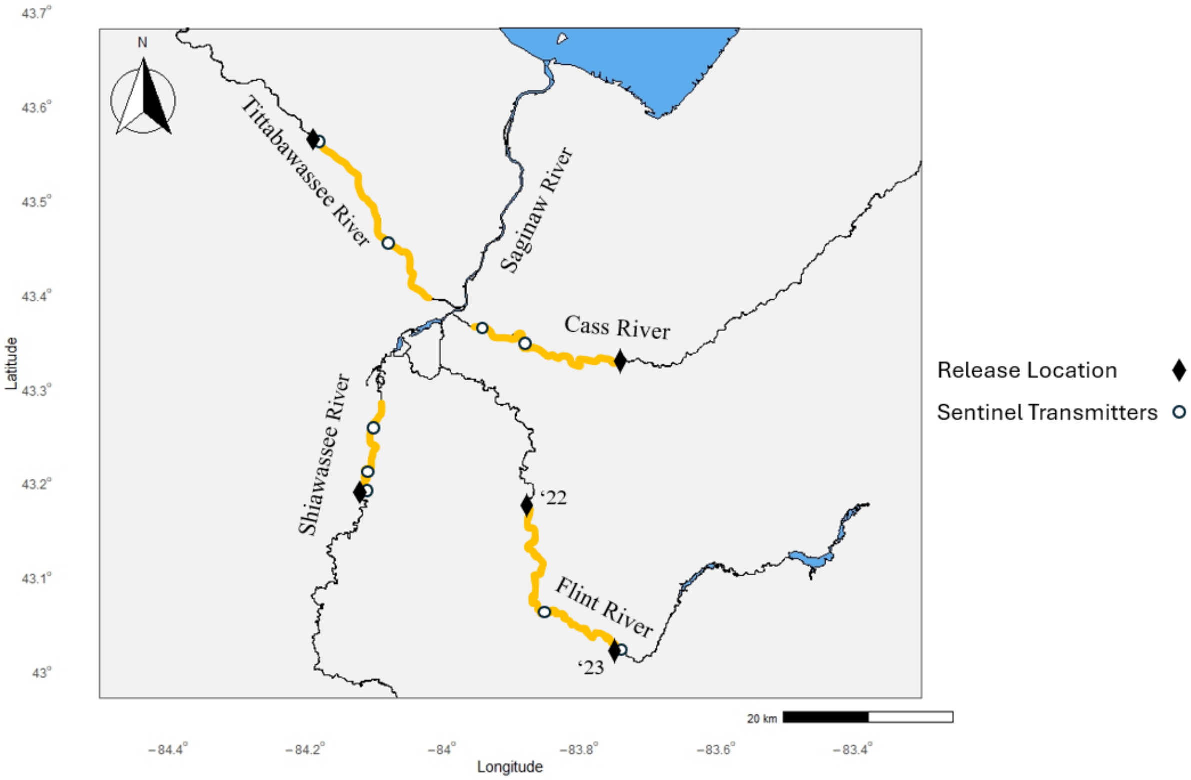

Figure 1.3 Map of active tracking routes (orange lines) with sentinel transmitter (white circles with black borders) locations relative to release locations (black diamonds). In 2022, fish were released downstream of the standard release site due to a potentially hazardous chemical spill upstream. In 2023, fish were released at the standard stocking site.

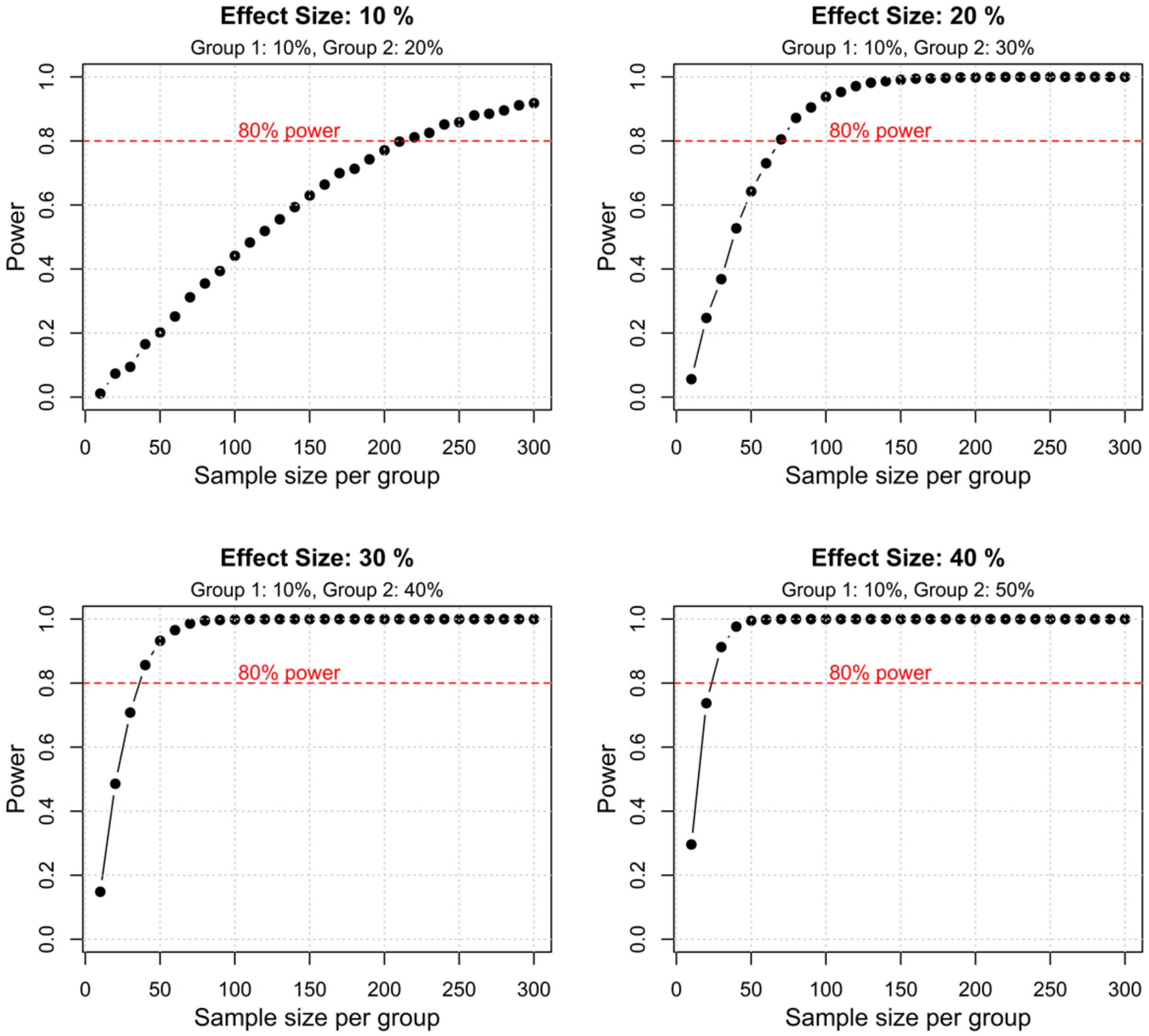

Figure 1.4 Estimated power to detect differences in the proportion of fish predated between groups for effect sizes of 10%, 20%, 30%, and 40%. Approximate sample sizes per group were 210, 70, 40, and 25 to achieve 80% power for each effect size, respectively. Sample size per group ranged from 10 to 300 fish. Power estimates were based on 10,000 simulations per sample size using a two-sided Fisher's exact test with $\alpha = 0.05$.

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CHAPTER 2:
Downstream dispersal patterns of age-0 Lake Sturgeon in the Saginaw River Basin, Lake Huron, U.S.A.

1. ABSTRACT

Stocking remains a primary management action for restoring Lake Sturgeon (*Acipenser fulvescens*), yet the influence of hatchery rearing environment on early post-stocking movement is poorly understood. I compared downstream dispersal of age-0 Lake Sturgeon reared at a non-natal streamside facility and a traditional groundwater hatchery and released into four tributaries of the Saginaw River, Michigan, during 2022–2023. Fish were implanted with V7D-2x (69 kHz) acoustic transmitters programmed with either alternating 60-day on/30-day off cycles (2022) or increased delays between transmissions (2023). Downstream movements were analyzed using a Weibull time-to-event framework the time to exit two movement states—release tributaries (State 1) and the Saginaw River (State 2). I modified a traditional time to event model to incorporate transmitter-off intervals and right censoring within a maximum likelihood framework. Time to tributary exits (State 1) were delayed (3.6 – 34.0 days), whereas exits from the Saginaw River (State 2) occurred soon after transition into this state (1.4 - 10.0 days). Across both years and states I tended to observe similar movement patterns by both hatchery types where fish exited the tributaries and the Saginaw River did so in pulse early on relative to when they were available to exit. Overall, results suggest rearing environment (non-natal streamside versus traditional) did not substantially influence first-year downstream dispersal timing.

2. INTRODUCTION

Throughout the Laurentian Great Lakes, managers have stocked juvenile Lake Sturgeon *Acipenser fulvescens* to supplement existing and reintroduce extirpated populations following widespread declines from anthropogenic influences such as overharvest, dams, dredging, and chemical spills (Auer 1996; Holey et al. 2000; Peterson et al. 2007; Scott and Crossman 1973; Welsh et al. 2010). At present, thousands of juvenile Lake Sturgeon are stocked annually at various sites throughout Michigan (Michigan Department of Natural Resources, Fisheries Division, 2025) by federal, state, and Tribal fisheries agencies to promote the recovery of Lake Sturgeon populations. These hatchery-based stocking programs have become a cornerstone of Lake Sturgeon restoration efforts aimed at rebuilding remnant populations and reintroducing the species to historically occupied habitats (Collier et al. 2022; Dittman et al. 2015).

Evaluating the post-stocking fates and movements of stocked juvenile Lake Sturgeon has been challenging because small-sized juvenile Lake Sturgeon can be difficult to relocate following release (Auer 1996; Barth et al. 2009). Understanding early dispersal patterns after stocking is therefore essential for assessing and informing restoration efforts for Lake Sturgeon. Whether or not stocking works and is cost-effective are critical questions for management. Stocking can be thought of in four phases. Phase one concerns post-release mortality and initial dispersal. Phase two concerns movement dynamics and feeding success. Phase three concerns homing and migration returns. Phase three concerns mate finding, fertilization success, and larval survival. In this chapter, I focus on phase one, specifically with immediate post-stocking downstream movement and the potential influence of the hatchery rearing environment.

Lake Sturgeon in the Great Lakes are reared in two primary hatchery environments: 1) traditional groundwater-fed facilities and 2) streamside facilities that use surface water drawn from a nearby waterbody (Holtgren et al. 2017; reviewed by Crossman et al. 2014). Groundwater facilities typically use particle filters and boilers to maintain optimal temperatures for growth and water quality (Ebeling et al. 2012). These hatcheries are often considered the “traditional” approach to hatchery rearing. Streamside facilities are designed to use filtered water to reduce particulates while exposing fish to natural temperatures and chemical cues (Holtgren et al. 2007). Streamside facilities are often argued for Lake Sturgeon rearing as exposure to local waters may facilitate post-stocking orientation and imprinting (DeHaan et al. 2006; Kimmel et al. 2023). However, operational constraints often necessitate using groundwater systems (traditional hatcheries) or non-natal streamside facilities, particularly in watersheds where streamside sites are unavailable or where populations become functionally extirpated. The Saginaw River Basin (Michigan, U.S.A.) provides a valuable case study for examining non-natal stocking efforts by both hatchery types. Lake Sturgeon were considered functionally extirpated from the Saginaw River Basin in 2017 (Lake Huron Lake Sturgeon Working Group) and stocking of juvenile Lake Sturgeon in four tributaries commenced the same year. Since 2018, roughly equal numbers of fish from both hatchery types have been used in annual stocking events of juvenile Lake Sturgeon in an effort to restore a self-sustaining population in Saginaw Bay (Hayes and Caroffino 2012).

Previous studies comparing hatchery rearing environments have yielded mixed evidence for movement differences among Lake Sturgeon stocked from different sources. In the lower Fox River (Wisconsin, USA), the spatial distributions of juvenile Lake Sturgeon (156–212 mm total length) of wild and non-natal streamside origin were similar (Tucker et al. 2022). Crossman et al. (2011) found that natal-streamside-reared fish exhibited movement patterns comparable to those of traditional-hatchery-reared fish by 17 weeks of age (~150 mm total length)—a size and age range consistent with the Lake Sturgeon released within the Saginaw River Basin. Similarly, age-0 juveniles (~200 mm total length) stocked in the Maumee River, another functionally extirpated watershed with an ongoing restoration program, showed comparable travel distances, river occupancy times, and spatial home range between fish reared in a traditional hatchery and those reared at a streamside facility on the Maumee River (McKenna 2023). Taken together, these studies suggest that potential behavioral effects of hatchery environment may be diminished with increased size at the time of release, leading to broadly similar post-stocking movement patterns between streamside- and traditionally reared juveniles. Yet, few studies have explicitly examined the timing and temporal distribution of downstream dispersal.

Acoustic telemetry provides a means to monitor and describe downstream movements at high temporal and spatial resolution (Welsh et al. 2008; Rechisky et al. 2012; Lothian et al. 2024). However, studying small-boded fish can entail trade-offs among transmitter sizes, battery life, and transmission frequency (Jepsen et al. 2004; Smircich & Kelly 2014; Thorstad et al. 2012). In this study, two transmitter configurations were deployed to achieve monitoring objectives: one set of transmitters operated with regular on-off transmission cycle and the other had longer gaps between transmissions. Both of these approaches were designed to provide approximately one-year of battery life for the transmitters to evaluate the movement dynamics during the first-year post-stocking. These transmitter-off intervals, resulted in heterogeneous detection intervals, which I approached using a time-to-event (TTE) framework. In the case of the Weibull distribution, a TTE model can be used to estimate two parameters: shape (k), which indicates whether events occur early or late in the observation period, and scale (λ), which describes the overall rate of dispersion where smaller values indicate fish exit in more of a pulse-like behavior. Thus, I believed that the TTE could be used to accommodate the different transmitter coding schemes and provided ecologically relevant information about the movements of tagged fish.

This goal of this study was to examine downstream dispersal patterns of age-0 Lake Sturgeon stocked into tributaries of the Saginaw River. To accomplish this goal, I used acoustic telemetry and a TTE framework aimed at characterizing the movement timing and dispersal of Lake Sturgeon reared in a non-natal streamside facility and a traditional hatchery environment, reflecting the current stocking strategy for the Saginaw Bay Basin. My specific objectives were to 1) describe the downstream movement dynamics of age-0 Lake Sturgeon across two ecologically relevant spatial states (release tributaries and the Saginaw River), and 2) evaluate whether rearing environment was associated with differences in these movements. By integrating acoustic telemetry data from two transmitter configurations and a TTE framework using a Weibull distribution, this work provides insights into the early dispersal ecology of stocked sturgeon, can contribute to evaluations of stocking methods for the Saginaw River Basin, and be used to inform the design of monitoring efforts for other restoration programs.

3. METHODS

3.1 Study Design

The study area, acoustic-receiver grid, stock sources and rearing methods, surgical procedures, release sites, and data-filtering steps were identical to those described in Chapter 1. In addition to the 20 fish implanted with V7D-2x predation transmitters per year used in Chapter 1, I implanted 60 fish per hatchery per year with InnovaSea “standard” V7D-2x acoustic transmitters (69 kHz; diameter 7 mm; length 19 mm; weight in air 1.5 g). Each tributary (n=4) was stocked with 15 fish with standard transmitters and five fish with predation transmitters so that each tributary was stocked with 20 fish per treatment per year.

3.2 Tagging Data

The mean total length of fish in 2022 was 208 mm ± 10 mm and 182 mm ± 7 mm for streamside and traditionally reared fish respectively. Mean tag burden was 4.7% (range: 3.6% - 7.0%) for streamside fish and 6.8% (range: 5.0% - 8.8%) for traditional fish in 2022. The mean total length of fish in 2023 was 199 mm ± 7 mm and 187 mm ± 9 mm for streamside and traditionally reared fish respectively. Mean tag burden was 5.7% (range: 3.9% - 7.4%) for streamside fish and 7.0% (range: 4.7% - 9.0%) for traditional fish in 2023. Tag burden is likely biased high as fish were withheld food for 36-hours prior to surgery to reduce the risk of viscera expulsion and weights were recorded at the time of surgery.

3.3 Downstream Dispersal Analysis

Transmitter programming matched that used in Chapter 1 for each year. The expected battery life of the standard transmitters was 380 days (2022) and 364 days (2023). Because predation transmitters had a shorter expected battery life within years (324 days in 2022 and 311 days in 2023), I excluded detections from fish with standard transmitters that occurred after the expected predation-transmitter battery life each year to standardize the detection windows across transmitter types. Due to receiver download issues in the winter of 2022 at the mouth of the Cass River, fish released in the Cass River were removed from the 2022 data analysis.

I evaluated downstream dispersal using a TTE model in terms of the relative timing and distribution of exit events from two states. My event was a fish being detected at a receiver marking the exit location (Figure 2.1). I assume that a detection at the receiver(s) meant that the fish was exiting the location and used the first detection at the receiver(s) at the river mouth as the exit event. I divided the study system into two distinct locations (State 1: a fish's release tributary; State 2: the Saginaw River) (Figure 2.1) I further assumed the transition between states was instantaneous whereby a detection marking the end of State 1 represented the entry time for State 2.

Fish were observable in the study system from the time they were released to the time the battery was expected to die, henceforth observation period. An important component of TTE models is establishing the time at which individuals are at risk of an event, hereafter t_0. I initially selected release date as t_0 for State 1 and the entry time as a t_0 for State 2. The time to the event was calculated as the difference in time between t_0 and the exit event detection. However, fish cannot be at risk of exiting either state immediately post-entry as they require a minimum transit time to get from the upper end of the tributaries to the receivers at the lower end of the tributaries and TTE models assume fish experience an instantaneous risk of the event. This was further complicated by releasing fish into multiple tributaries where there may be river-specific influences on movement such as minimum travel distance, water velocity (Hintz et al. 2024), and predator abundance (Gilliam and Fraser 2001). Therefore, I adjusted the time to the event by calculating the fastest known TTE by treatment, release location, state, and year and subtracted the difference in time in days between the release date and the exit detection using the appropriate time for each combination because I know fish are at risk of exiting at this time and accounts for potential tributary effects. This can be thought of as setting the event detection of the fastest known exit event for each combination of treatment, year, release location, and state as t_0.

I adjusted the transmitter-off intervals and observation period using the rationale above to ensure that the timing of the transmitter-off intervals and observation period matched the adjustments to t_0 for the analysis. Any fish where t_0 or the event detection could not be established in a state were removed from the analysis. For example, a fish was detected in State

2, but was never detected leaving State 1, therefore I could not establish an exit time for State 1. Additionally, I do not know when that fish entered State 2 to identify t_0 for this state. Fish that exited the state were given an event value of 1 and the time to the event was calculated. Fish that did not exit the state were given an event value of 0 and the time to the event was set to the maximum observation period because they were not observed, as a result they were right censored in the analysis.

I selected the Weibull distribution for the TTE model because it is a highly flexible continuous probability distribution for positive events (Rinne 2008) that is often used in event timing modeling (e.g. Castros-Santos and Perry 2012; Zabel et al. 2014; Benoít et al. 2015; Lowerre-Barbieri et al. 2025). Time-to-event models feature an underlying hazard function where hazard is defined as the instantaneous risk that a fish will exit a given state at time *t*, given it has not done so yet. The shape parameter *k* describes how the hazard shifts through time. When $k < 1$, the rate of movement events decreases through time, meaning most fish exit quickly and fewer exit events occur later in time (Figure 2.2). When $k = 1$, the rate of movement is constant through time, reducing the model to an Exponential distribution. Fish in this scenario have an equal probability of exiting the state of interest at any time. When $k > 1$, the rate of movement events increases through time, meaning on average fish were initially slow to exit and most fish exited at later times. The scale parameter λ describes distribution of time over which events occur (see Figure 2.2). A large λ stretches those events through time, leading to an overall low risk of an event through time, while a small λ means exit events occurred as more of a pulse, leading to a fast rise in the risk of event through time.

The Weibull distribution probability density function $f(t)$ can be expressed as

Eq. 1. $$f(t \mid k, \lambda) = \frac{k}{\lambda}\left(\frac{t}{\lambda}\right)^{k-1} exp\left[-\left(\frac{t}{\lambda}\right)^{k}\right],$$

where *t* is the time to the event and the other parameters are defined above. This probability density function (Eq. 1.) describes the distribution of event times. I can integrate Eq. 1. to generate the survival function, $S(t)$:

Eq. 2. $$S(t)=exp\left[-\left(\frac{t}{\lambda}\right)^{k}\right],$$

which describes the probability that the event (a fish moving out of the state) has not occurred by time *t*.

Time to event models often include censoring mechanisms, meaning some events are never observed. Fish may be censored for multiple reasons. For example, a fish may have been present at the event location but was not detected because it arrived during a transmitter-off interval (interval censoring) or a fish may have never arrived at the event location during the observation period (right censoring). These events are mutually exclusive because a fish cannot be detected moving past the event location during a transmitter-off interval and then move past the event location after the observation period ends. The probability of a non-detection event was described by:

Eq. 3. $$Pr(no\ detection) = \sum_j\left[S(a_j)-S(b_j)\right] + S(t_B)$$

where $\sum_j\left[S(a_j)-S(b_j)\right]$ describes the probability of a fish moving during a transmitter-off interval and the term $S(t_B)$ describes the probability of a fish moving after the observation period ended. In these equations, the term a_j is the beginning of transmitter-off interval j and the term b_j is the end of that transmitter-off interval. The term t_B describes the time after which fish are unobservable (the battery life of the transmitter has ended). Note that the transmitter-off interval portion of the likelihood was turned off for 2023 where transmitter-off intervals were not used.

Using the components in Eqs. 1-3, I can then derive a log-likelihood function for both detected and undetected fish i:

Eq. 4. $$logL = \sum_{i=1}^{n} \begin{cases} log\ (f(t_i)) & if\ fish\ i \text{ is } detected \\ log\big(S(t_B) + \sum_j \big[S(a_j)\text{-}S(b_j)\big]\big) & if\ fish\ i \text{ is } undetected \end{cases}$$

I fit Eq. 4. to data using *optim()* from the stats package in R (R Core Team 2024). I expanded the model described in Eqs. 1-4 to estimate treatment specific k and λ for each year (2022 and 2023) within each state (release tributaries and Saginaw River). Estimates were compared using asymptotic 95% confidence intervals (Schenker and Gentleman 2001).

3.4 Weibull Model Fits to Observed Data

I visually assessed model fits using a parametric bootstrap based on Kaplan-Meier (KM) curves. Kaplan-Meier curves are a widely used tool to visualize TTE data (Therneau and Grambsch 2000; Gail et al. 2010). My observed data was passed through the *survfit()* function from the survival package in R (Therneau 2024), which produced KM curves for each combination of state (State 1 or State 2), year (2022 or 2023) and treatment (traditional or streamside hatchery). A problem with KM curves within the context of the study was that it does not account for the transmitter-off intervals, which means all unobserved fish are treated as right censored, and thus, the KM estimates are biased high. To address this issue and evaluate model fit given the complicated censoring processes present in this study, I simulated 10,000 replicate datasets from each fitted model to generate event times for each fish (Efron 1981; Davison and Hinkley 1997). I imposed the same censoring processes present in the study (see above) and then generated a KM curve for each simulated replicate. I then summarized the KM curves through time by their median and 95% quantile band and overlayed the KM curve from the observed data. If the estimated models described the observed data well, I expected the KM curve from the observed data to both fall within the 95% quantile band and overlap the median KM curve from the simulated data (e.g. see description of visual model evaluation in Buja et al. 2009).

4. RESULTS

4.1 State 1: Release Tributaries

In 2022, t_0 was adjusted by 11.4, 24.3, and 3.6 days for the Flint, Shiawassee, and Tittabawassee rivers for non-natal streamside reared fish. In 2022, t_0 was adjusted by 18.6, 18.6, and 11.7 days for the Flint, Shiawassee, and Tittabawassee rivers for traditionally reared fish. In 2023, t_0 was adjusted by 34.0, 27.0, 32.7, and 27.0 days for the Cass, Flint, Shiawassee, and Tittabawassee rivers for non-natal streamside reared fish. In 2023, t_0 was adjusted by 23.8, 20.9, 12.8, and 24.0 days for the Cass, Flint, Shiawassee, and Tittabawassee rivers for traditionally reared fish.

The estimated shape and scale parameters in 2022 were 0.33 (95% CI: 0.23-0.48) and 723 (95% CI: 163-3208) for non-natal streamside reared fish and 0.34 (95% CI: 0.25-0.48) and 174 (95% CI: 54-567) for traditionally reared fish (Figure 2.4). The estimated shape and scale parameters in 2023 were 0.0.35 (95% CI: 0.27-0.45) and 284 (95% CI: 121-665) for non-natal streamside reared fish and 0.0.30 (95% CI: 0.22-0.40) and 531 (95% CI: 169-1666) for traditionally reared fish (Figure 2.4). There was no evidence to suggest statistical differences between the two treatment types based on overlap of the asymptotic 95% confidence intervals in either year (Figure 2.3 and 2.4). Model Fits vs data for the release tributaries are shown in Figure 2.6.

4.2 State 2: Saginaw River

In 2022, t_0 was adjusted by 5.1, 10.0, 5.8 days for the Flint, Shiawassee, and Tittabawassee rivers for non-natal streamside reared fish. In 2022, t_0 was adjusted by 5.2, 112.2, and 9.3 days for the Flint, Shiawassee, and Tittabawassee rivers for traditionally reared fish. In 2023, t_0 was adjusted by 2.3, 5.9, 3.3, and 1.4 days for the Cass, Flint, Shiawassee, and Tittabawassee rivers for non-natal streamside reared fish. In 2023, t_0 was adjusted by 1.2, 3.9, 3.9, and 1.7 days for the Cass, Flint, Shiawassee, and Tittabawassee rivers for traditionally reared fish.

The estimated shape and scale parameters in 2022 were 0.29 (95% CI: 0.19-0.46) and 19 (95% CI: 3-106) for non-natal streamside reared fish and 0.30 (95% CI: 0.20-0.46) and 107 (95% CI: 23-510) for traditionally reared fish. The estimated shape and scale parameters in 2023 were 0.26 (95% CI: 0.19-0.35) and 118 (95% CI: 31-442) for non-natal streamside reared fish and 0.31 (95% CI: 0.24-0.40) and 6 (95% CI: 2-19) for traditionally reared fish. There was no evidence to suggest that there are differences in the shape and scale parameters between hatchery stocks in 2022 based on overlap in the 95% CIs (Figure 2.5). However, there was evidence to suggest that differences in the scale parameter between hatchery stocks in 2023 (Figure 2.3 and 2.5). Non-natal streamside reared fish had a larger estimated scale parameter than traditionally reared fish, suggesting exit times for non-natal streamside reared fish were more broadly dispersed through time than traditionally reared fish. Model Fits vs data for the Saginaw River are shown in Figure 2.7.

5. DISCUSSION

This study used acoustic telemetry and a TTE analysis using a Weibull distribution to describe the downstream dispersal of stocked age-0 Lake Sturgeon in the Saginaw River Basin and considered whether rearing environment (non-natal streamside or traditional hatchery) influenced movement timing. Overall, the results indicated that rearing-origin effects on downstream dispersal were generally weak, with evidence that the distribution of the probability of exiting the Saginaw River was more widely distributed through time for non-natal streamside reared fish than traditionally reared fish in 2023. I consider the implication of these results below and discuss the application of applying TTE models to descriptions of downstream movements using acoustic telemetry.

5.1 Downstream Movement

My study demonstrates that the downstream movement dynamics, as described by the shape and scale parameters, were broadly similar between non-natal streamside reared and traditionally reared age-0 Lake Sturgeon in the release tributaries and Saginaw River with only evidence of a single difference between the scale parameters observed in the Saginaw River in 2023, suggesting that the probability of a fish exiting was more widely distributed through time for non-natal streamside reared fish than traditionally reared fish. This general lack of consistent rearing-origin effects aligns with previous studies that have reported convergence in movement behaviors once juveniles reached a sizes comparable to the fish released in this study (e.g., Crossman et al. 2011, Tucker et al. 2022, McKenna 2023). A study in the Black River (Michigan, U.S.A.) found that early differences in the downstream distributions of streamside and traditional hatchery fish were no longer present when fish were reared 17 weeks of age prior to stocking (Crossman et al. 2011), consistent with the fish released as part of this study. The dominant patterns of quick transitions between both the release tributaries and Saginaw River are consistent with Lake Sturgeon juvenile movements dynamics in the Saginaw River system being largely robust to hatchery origin, at the ages and sizes stocked here, with both sources contributing similarly to downstream dispersal.

The difference in the Weibull scale parameter for State 2 in 2023, where non-natal streamside-reared fish showed more temporally dispersed exit times than the traditionally reared

Lake Sturgeon, could be interpreted biologically or in light of the different tag programming used between years. Although the TTE analysis was designed to account for the on-off periods of transmitters in 2022, it remains possible that the results were influenced, in part, by the transmission schemes. Potential influences of transmitter programming and monitoring effort have been noted in other acoustic telemetry studies, including receiver coverage, tag transmission intervals, and receiver detection range which can bias inferences about movement (e.g., Kessel et al. 2014, Hussey et al. 2015, Lennox et al. 2017). Passive acoustic monitoring of marine mammals, which also relies on intermittent 'detection' events based on the activity of deployed sound recorders, has shown that gaps in observations periods (i.e., when the sound recorders were active) can alter inferences about movements by missing rare or infrequent events when sampling schedules are misaligned with underlying behavior (e.g., Stanistreet et al. 2017). Because the two transmitter programming schemes in this study were used in separate years rather than simultaneously, I cannot distinguish whether the 2023 contrast reflects a year effect (environmental or cohort variation) or a programming effect. Future studies could include simultaneous deployments of transmitters with both programming schemes to better quantify how programming choices influence TTE inferences.

In this study, downstream movements differed between the spatial states, as reflected in the parameter estimates for the release tributaries and the Saginaw River. These patterns support the interpretation that immediately after release into tributaries (State 1), fish may exhibit a wider array of responses to being in a novel environment. It is possible that a stress response is related to spread of downstream movements in State 1. For example, Birnie-Gauvin et al. (2019) found that Brown Trout (*Salmo trutta*) and Atlantic Salmon (*Salmo salar*) with higher cortisol levels (i.e., more stressed) left rivers sooner than fish with lower cortisol levels. It has also been observed that hatchery-reared juvenile Atlantic Salmon were more likely to remain near the location of release than wild counterparts (Symons et al. 1969), suggesting an acclimation process following release that delayed the onset of downstream movements. In comparison, by the time fish were detected exiting State 2, individuals from both hatcheries may have acclimated to the release environment and entered a behavioral state in resulted in directed movements downstream in a way that reduced variability in movements in State 2.

Not all tagged fish left each state during the study period. In contrast to McKenna (2023), who observed juvenile age-0 Lake Sturgeon moving out of the Maumee River into Lake Erie within days of release, a number of fish in this study remained in the river based on the TTE analysis. This pattern of retention in the river is similar to the findings of Tucker et al. (2022) that reported juveniles staying in the Fox River (WI, U.S.A.) after release. For example, within the tributaries (State 1), approximately 40% of released fish in both years were never detected leaving for State 2. It is possible that some of the released Lake Sturgeon opted for long-term river residence, but it is also possible that there was unidentified mortality following release which contributed to this pattern of residency that I could not account for. Further work should focus on the tributaries to evaluate if the fish remaining in the tributaries are alive and, if so, what sections of the river.

5.2 Management Implications

In the absence of consistent rearing-origin effects on downstream movement dynamics, hatchery choice may instead be guided by other management priorities. These priorities could include logistics, cost (Leber et al. 2005), disease concerns (Winton 2001), or genetic objectives (Busack and Currens 1995; Welsh et al. 2010). Promoting and maintaining genetic diversity to avoid inbreeding depression and founder effects have been a goal for multiple Lake Sturgeon stocking programs (Tringali et al. 1998; Welsh et al. 2010). Fish from the Black River Streamside Facility and Genoa National Fish Hatchery are sourced from the same genetic management unit, Genetic Stocking Unit 1 (Welsh et al. 2010) but originate from different donor populations (upper Black River and St. Clair River). Thus, given that the lack of movement differences observed here suggest no disadvantages associated with either source, continuing to use both hatchery programs may support multiple priorities for Lake Sturgeon stocking programs.

5.3 Future Directions

Future studies should evaluate the length of the transmitter off intervals and how this might influence inferences of movement dynamics. Transmitter-off intervals inherently add uncertainty because researchers cannot know if a fish was present but could not be detected or never arrived at the location. If this uncertainty is increased via the length of the off intervals, researchers risk losing more observations that would otherwise inform model estimation. The current study expanded upon a TTE model using a Weibull distribution to create a model that can account for this experimental design complexity. One potential study could be to simultaneously release fish with and without transmitter-off intervals and compare the inferences drawn from these data or simulations to evaluate the length of such transmitter-off intervals and how it might affect inferences.

The transmission frequency additionally is an important telemetry setting that should be evaluated, particularly in areas with limited line of site, such as rivers (Kessel et al. 2014; Carlson et al. 2023). The current practice for many telemetry studies is to use filters, for example the false_detection() function from the GLATOS package (Holbrook et al. 2024), to remove isolated detections based on work from Pincock (2012). This filtering approach assumes that one detection events are false detections, possibly due to tag collisions between multiple overlapping transmissions at the same frequency but longer signal transmission rates may reduce the probability of multiple detections on a receiver when a fish is actively migrating, therefore risking the potential removal of informative detections that then may influence analyses. Longer transmission times also risk the possibility of missed detections at a receiver due to a fish swimming out of range before a detection can be fully coded on a receiver.

5.4 Conclusions

Biologically, these results provide little evidence that downstream movement patterns of age-0 Lake Sturgeon differed between non-natal streamside-reared and traditionally reared fish in the Saginaw River Basin. Instead, movement dynamics varied more strongly between spatial states than between hatchery sources. Transitions from release tributaries (State 1) were characterized by generally broader temporal dispersion, whereas exits from the Saginaw River (State 2) among fish that moved into Saginaw Bay occurred over a comparatively shorter timeframe, consistent with the Saginaw River functioning as a corridor for more rapid downstream dispersal towards Saginaw Bay. Together, these findings support the continued use of both hatchery programs and highlight the value of flexible time-to-event modeling frameworks for interpreting telemetry data and informing restoration strategies for Lake Sturgeon.

This study also demonstrated the importance of aligning transmitter programming with planned analytical approaches and explicitly accounting for duty-cycled transmissions in TTE models. As acoustic telemetry technology advances and tag programming options become more variable, study designs that do not anticipate how on–off cycles, detection gaps, and censoring will be handled risk generating data that are poorly matched to standard statistical tools or, more critically for species of conservation concern, could lead to biased or misleading inferences. In this case, applying a standard KM framework to 2022 data would have ignored interval censoring induced by transmitter off-intervals and likely biased estimates of downstream movement timing, whereas the TTE approach, combined with parametric bootstrap assessment, provided a more appropriate and robust description of movement dynamics.

6. TABLES AND FIGURES

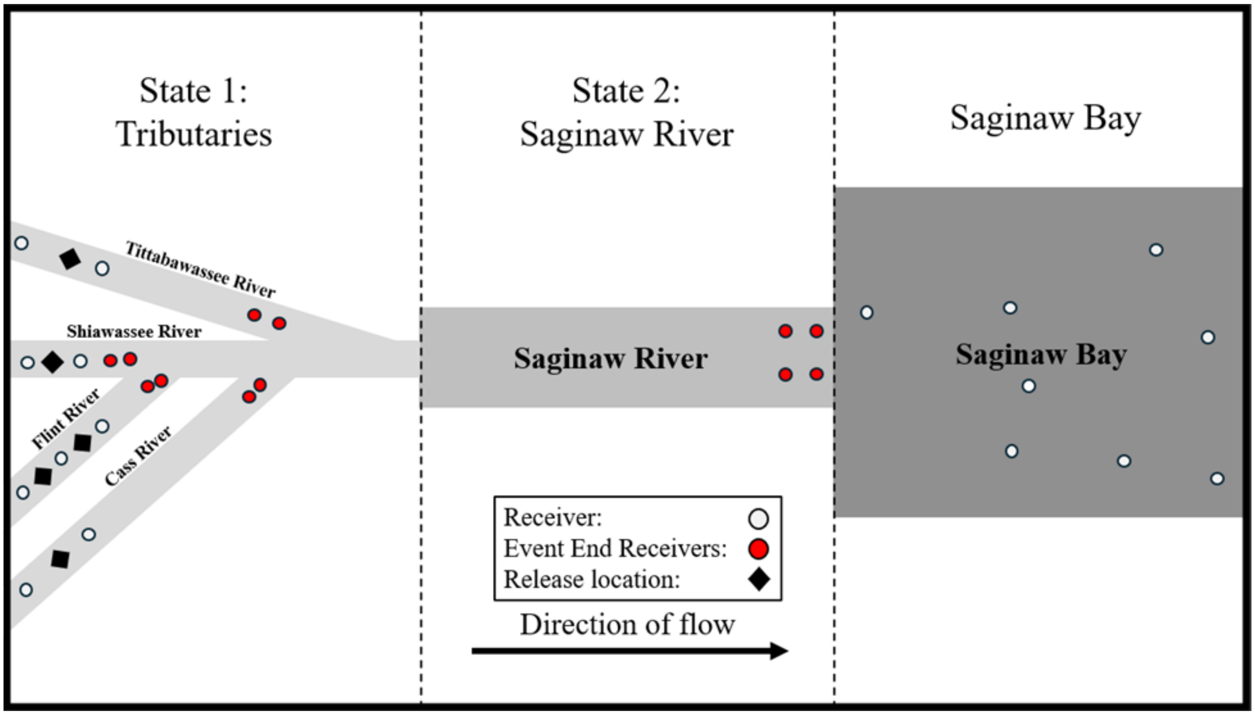

Figure 2.1 Schematic of unique states used in the analysis. Note that this is not to scale. State 1 is the tributary that a fish was released into and State 2 was the Saginaw River. Fish have unimpeded access to Saginaw Bay from the release sites. Black diamonds represent the release locations, white dots represent acoustic receivers, and red dots represent the acoustic receivers used to define the event. The event of interest was the time a fish took to exit the tributary following the first known exit event.

		Shape (*k*)	
		$k_1 \neq k_2$	$k_1 = k_2$
Scale (λ)	$\lambda_1 \neq \lambda_2$	***"Survival" Curve:*** Events are concentrated at different times, and the proportion of "survivors" are different over time ***Hazard Function:*** Both the concentration of time when risk of the event is the highest and the distributions of the risk through time are not similar.	***"Survival" Curve:*** Events are concentrated at similar times, but the proportion of "survivors" are different over time ***Hazard Function:*** The concentration of time when risk of the event is the highest occur at similar times, but the distributions of the risk through time are not similar.
	$\lambda_1 = \lambda_2$	***"Survival" Curve:*** Events are concentrated at different times, but the proportion of "survivors" are similar over time ***Hazard Function:*** The concentration of time when risk of the event is the highest occur at different times, but the distributions of the risk through time are similar.	***"Survival" Curve:*** Events are concentrated at similar times, and the proportion of "survivors" are similar over time. ***Hazard Function:*** Both the concentration of time when risk of the event is the highest and the distributions of the risk through time are similar.

Figure 2.2 Pannel of general interpretations of parameters, shape (*k*) and scale (λ), for time to event framework in terms of the "survival" curve and hazard function using a Weibull distribution.

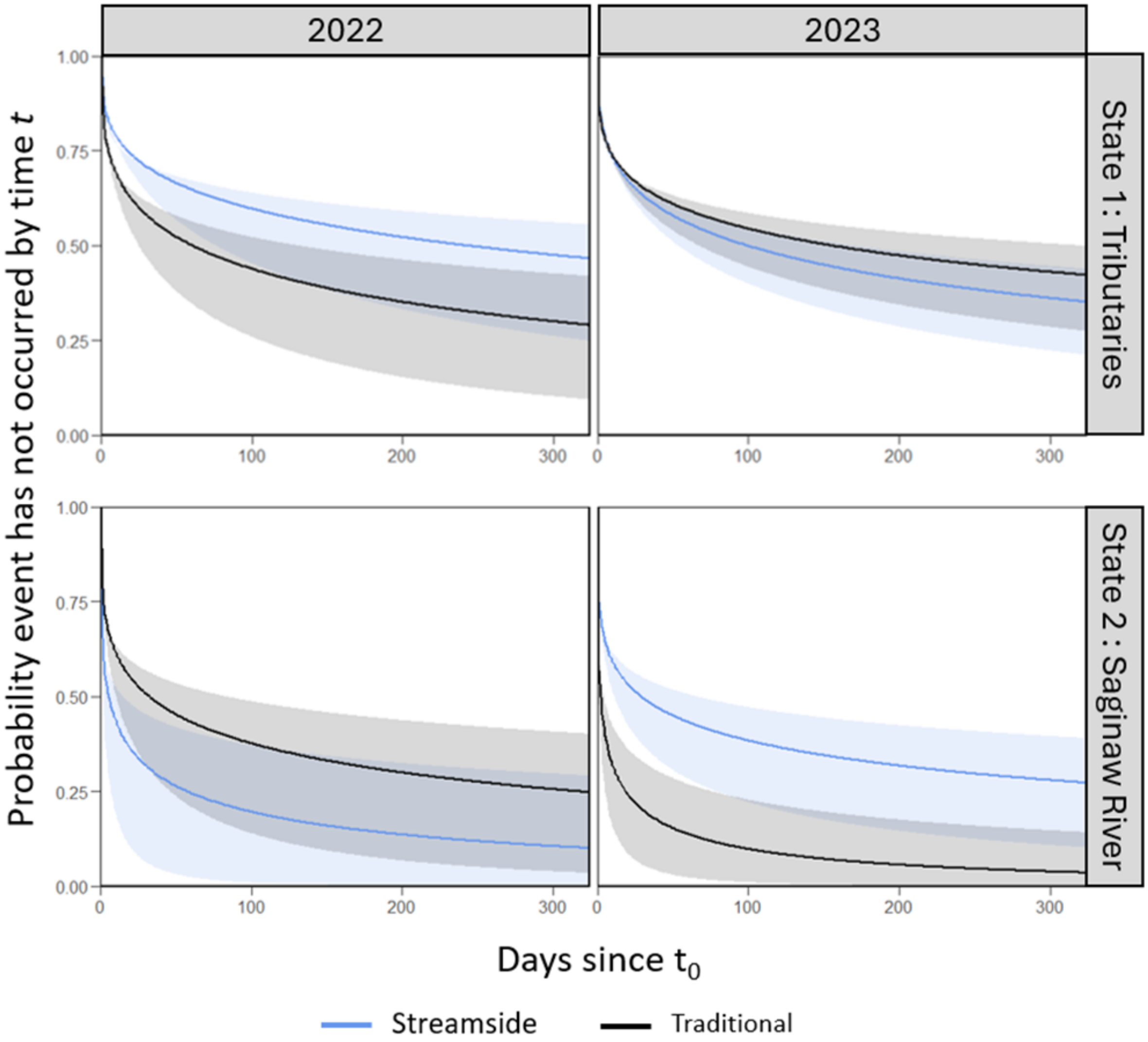

Figure 2.3 Survival curves by state, year and treatment based on estimated shape and scale parameters with asymptotic 95% confidence intervals.

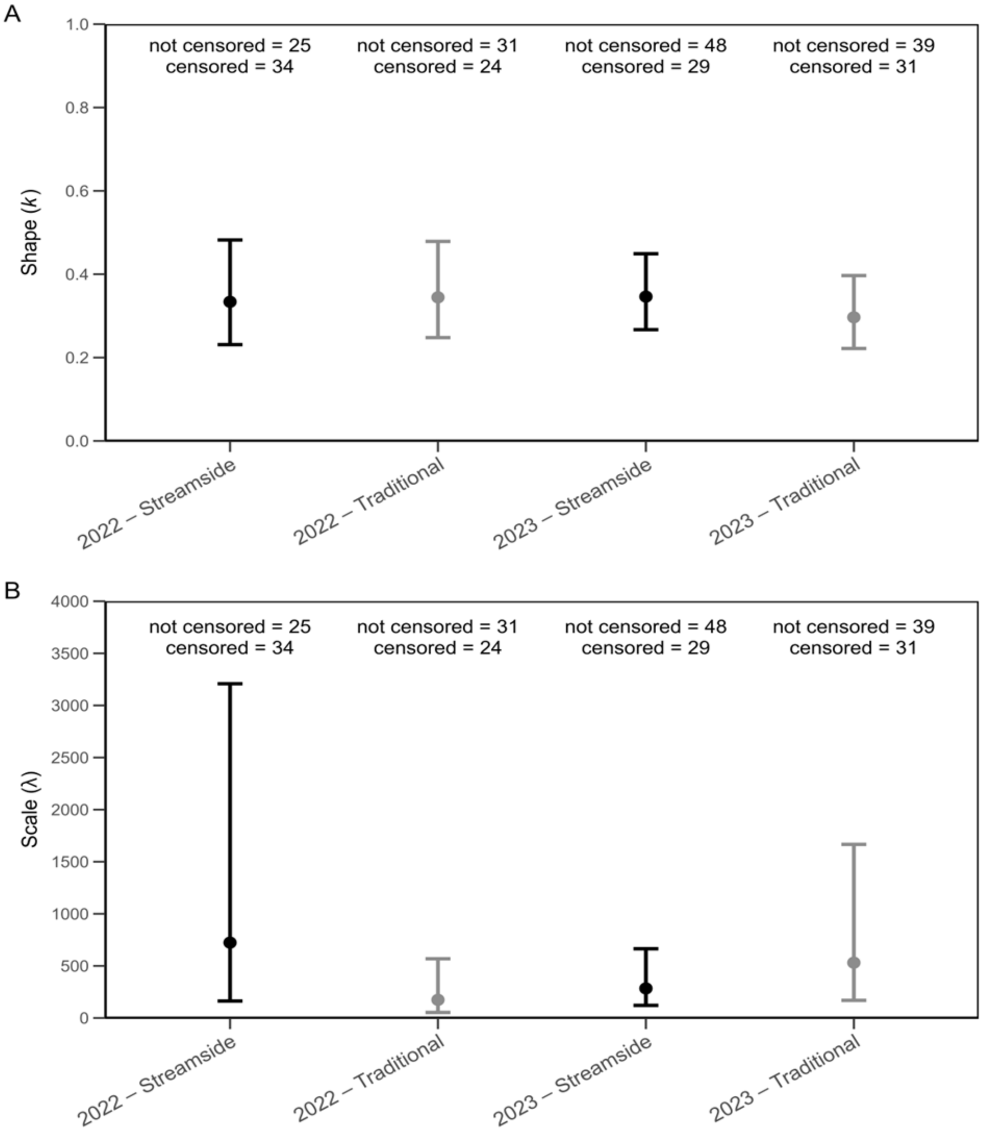

Figure 2.4 Plot of model parameter estimates for A) shape (*k)* and B) scale (λ) with asymptotic 95% confidence intervals for each combination of treatment (streamside vs traditional) and year (2022 & 2023) in State 1: Release Tributaries. The number of fish that are not censored is the number of fish that experienced the event. The number of fish that are censored is the number of fish that were not observed experiencing the event due to a transmitter-off interval or never arriving at the event location.

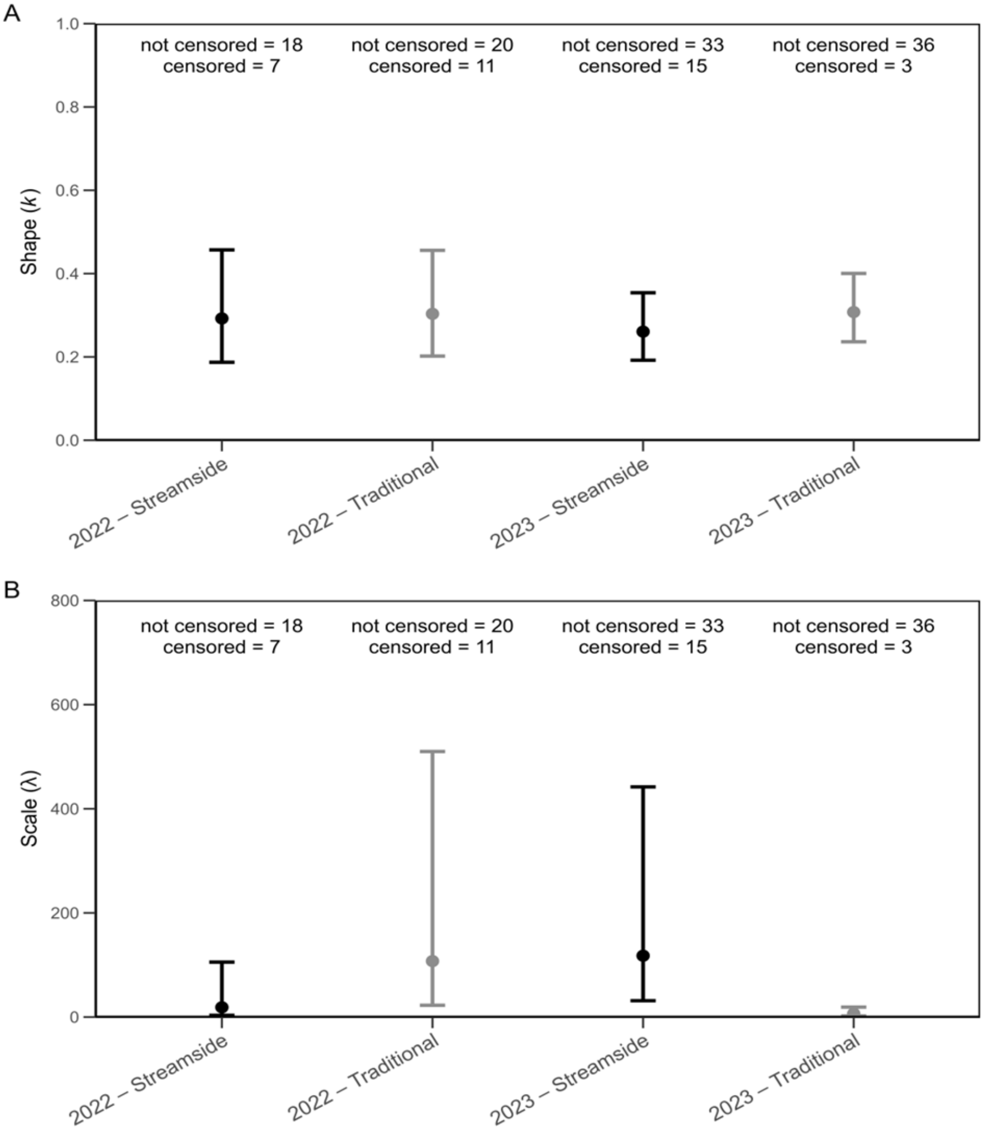

Figure 2.5 Plot of model parameter estimates for A) shape (*k)* and B) scale (λ) with asymptotic 95% confidence intervals for each combination of treatment (streamside vs traditional) and year (2022 & 2023) in State 2: Saginaw River. The number of fish that are not censored is the number of fish that experienced the event. The number of fish that are censored is the number of fish that were not observed experiencing the event due to a transmitter-off interval or never arriving at the event location. There is evidence of potential differences in scale parameter in 2023 based on a lack of overlap in the asymptotic 95% confidence intervals.

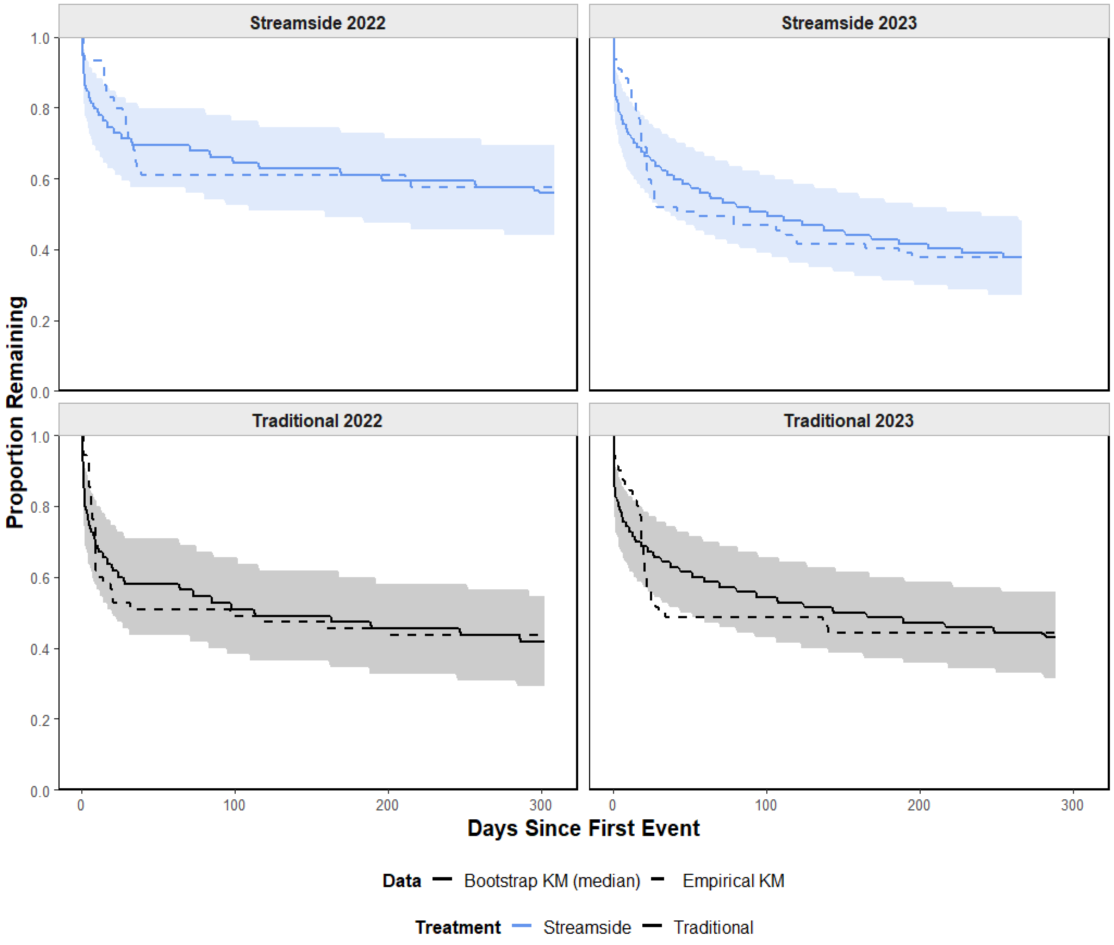

Figure 2.6 Parametric-bootstrap Kaplan-Meier (KM) curves versus empirical KM curves from the observed data for State 1: Release Tributaries. Panels show streamside (blue) and traditional (black) fish by release year. Solid lines represent the bootstrap median Kaplan–Meier (KM) curve from 10,000 simulated datasets generated with from the estimated Weibull parameters and censored using the same censoring mechanisms as the observed data. The shaded ribbon is the 95% quantile envelope from the summarized bootstrap data. The empirical KM curves fall outside the 95% quantile envelope from the bootstrap fits less than 5% of the observation period in all cases except for traditional fish in 2023; however, the general trends of the empirical KM curves from the observed data are similar in all cases, suggesting no underlying bias between years or treatments.

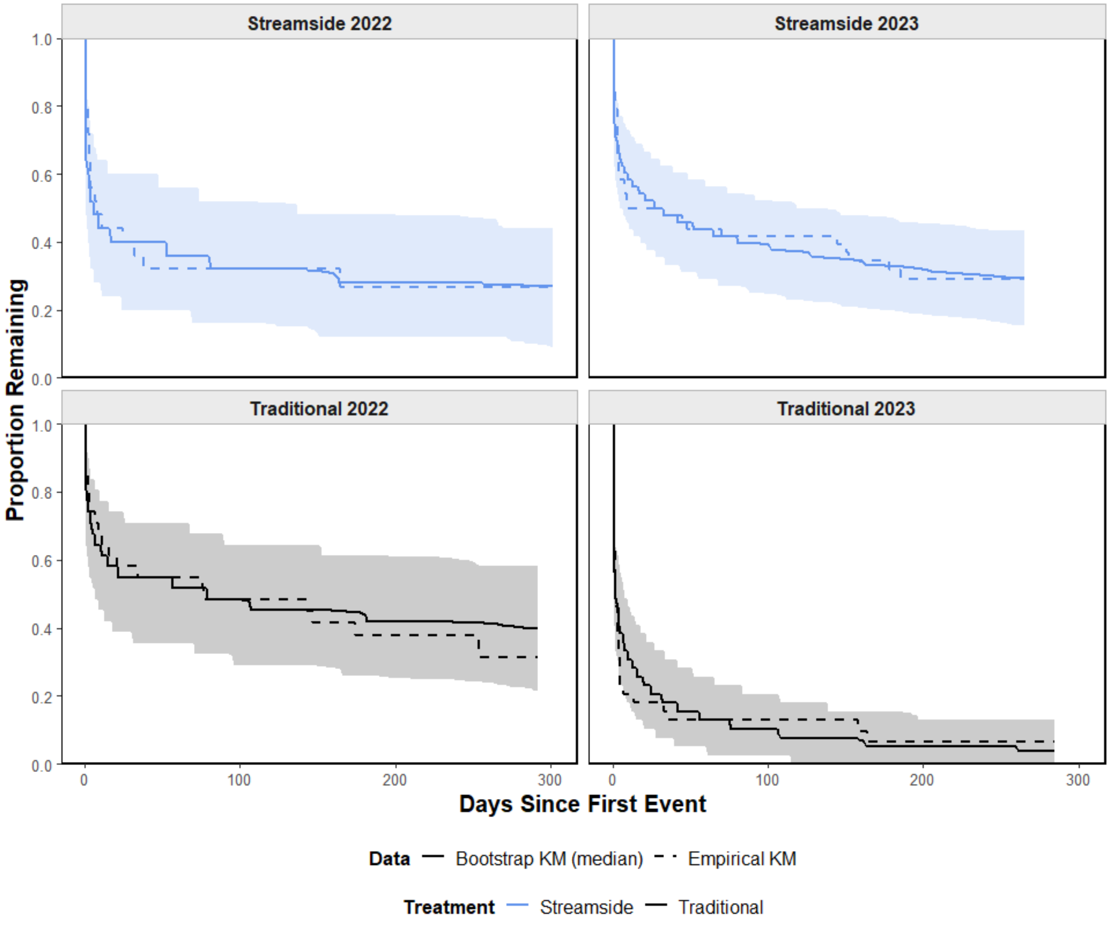

Figure 2.7 Parametric-bootstrap Kaplan-Meier (KM) curves versus empirical KM curves from the observed data for State 2: Saginaw River. Panels show streamside (blue) and traditional (black) data by release year. Solid lines represent the bootstrap median Kaplan–Meier (KM) curve from 10,000 simulated datasets generated with from the estimated Weibull parameters and censored using the same censoring mechanisms as the observed data. The shaded ribbon is the 95% quantile envelope from the summarized bootstrap data. Close alignment of the empirical curve within the bootstrap envelope indicates adequate fit.

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