Assessing Atlantic Salmon (*Salmo salar*) Recovery Across Five Atlantic Canadian National Parks Through Juvenile Abundance Modelling

by

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Abstract

For decades, Atlantic salmon in Eastern North America have precipitously declined. In response, Fundy National Park (FNP) and Cape Breton Highlands National Park (CBHNP) implemented smolt-to-adult supplementation strategies within marine and freshwater rearing environments, respectively, Kouchibouguac National Park (KNP) planted fertilized eggs, and Gros Morne (GMNP) and Terra Nova National Park (TNNP) used community-based restoration approaches. To assess juvenile salmon population abundance through decline and restoration implementation of these parks, a Bayesian hierarchical model was developed to standardize single-pass and multi-pass electrofishing methodology, and estimate an index of juvenile density, at site and annual scale, using an electrofishing dataset covering 536 sites, over 49 years, across 24 rivers.

Single-pass sites lack the accuracy of multi-pass methods but are an efficient sampling approach from a full catchment perspective. Increasing population trends were shown in some rivers (FNP, KNP), while others exhibited continued declines (CBHNP, KNP, GMNP) or modelling challenges (CBHNP, TNNP).

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List of Symbols, Nomenclature, or Abbreviations

USR: Upper Salmon River

PWR: Point Wolfe River

FNP: Fundy National Park

KNP: Kouchibouguac National Park

CBHNP: Cape Breton Highlands National Park

GMNP: Gros Morne National Park

TNNP: Terra Nova National Park

DFO: Fisheries and Oceans Canada

SARA: Species At Risk Act

COSEWIC: Committee on the Status of Endangered Wildlife in Canada

DU: Designatable Unit

SAS: Smolt-to-Adult Supplementation

iBoF: Inner Bay of Fundy

Introduction

1.1 Biology

Atlantic salmon (*Salmo salar*) is a species of fish in the salmonidae family, found throughout the Atlantic Ocean, in a large number of northern hemisphere rivers (i.e., North America, Greenland, Iceland, Europe, and Western Russia). As an anadromous species, adult salmon enter rivers to spawn, where females dig a redd (*i.e.,* a nest) and lay their eggs while a nearby male(s) broadcast fertilize the eggs, which are then buried by the female. The eggs hatch in the substrate, where the alevins absorb their energy-rich yolk sac while they grow and develop. Upon emerging in the late spring, they live in freshwater during their juvenile years (between one to eight years) (Figure 1). Young-of-the-year are commonly referred to as fry (0+ when sampled during the first summer post-hatching), while older juveniles are called parr (X+, X referring to the number of full years post-hatching, with the + referring to the portion of the current year) (Figure 1). Based on physiological status and environmental cues, juveniles start their migration to sea during spring (Metcalfe, 1998; Otero et al., 2014). Juveniles migrating to sea are called smolts and undergo several physiological, morphological, and behavioral transformations to adapt to the saltwater environment (McCormick, 2012) (Figure 1). At a regional scale, smolt generally migrate at the same time and follow similar migration routes (see Chaput et al., 2019 for Eastern Canada). In the ocean, adults feed in different areas of the North Atlantic based on their age and region of origin (Bradbury et al., 2021; Dadswell et al., 2010; Olmos et al., 2020). After generally spending one to three years at sea, most adults return to their natal stream to spawn, although straying is possible, ranging between 4.4 to >20% in some populations (Keefer and Caudill, 2014). As an

iteroparous species, post-spawned adults can recondition in freshwater before migrating to sea again (a phase of the adult life stage called kelt) and potentially returning to freshwater to spawn again in subsequent years.

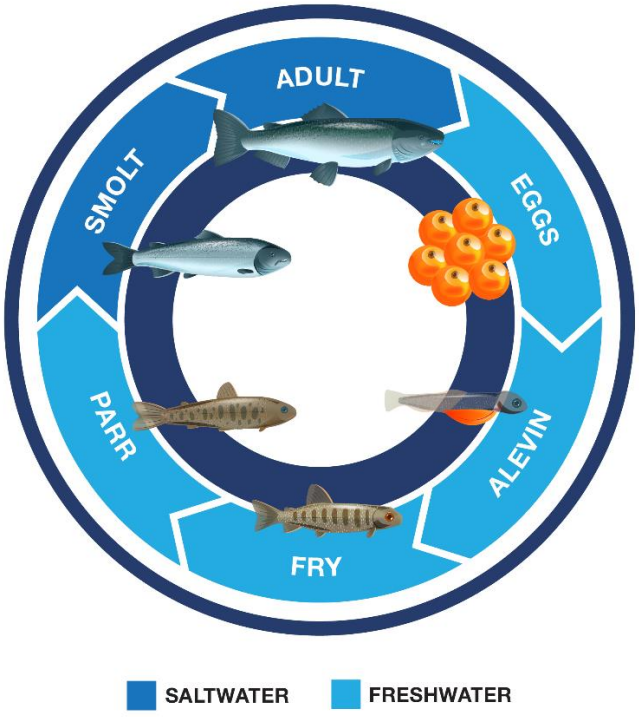

Figure 1. Simplified lifecycle of the Atlantic salmon.

In the freshwater environment, Atlantic salmon prefer cool, clear, well oxygenated water with low to moderate gradient, and gravel, cobble, and boulder substrate (COSEWIC 2006; NASCO 2010). Atlantic salmon tolerance and growth-limiting temperatures can vary widely but are optimum from 16-20°C, with ideal egg development around 4-7°C (Elliott and Elliott, 2010; Jonsson et al., 2001; Stehfest et al., 2017). Juvenile growth

begins when temperatures are above 7°C, however, temperatures approaching 22°C result in juveniles seeking thermal refuge (Breau et al., 2007) with lethal temperature limits surpassing >27°C (Corey et al., 2023; Elliott and Elliott, 2010).

In the marine environment, Atlantic salmon prefer temperatures from 1.6–8.4°C (Strøm et al., 2020) but frequently inhabit ranges of 0 to 15°C (Lacroix, 2013; Strøm et al., 2020; Thorstad et al., 2021). Lethal temperatures occur below -0.75°C (COSEWIC, 2010; Elliott and Elliott, 2010).

1.2 Cultural and Economic Importance

Atlantic salmon hold significant cultural importance to many First Nations around the world, particularly in regions where salmon are native and were historically abundant. For centuries, salmon have played a central role in the traditions and heritage of Indigenous Peoples, as well as local communities closely tied to rivers and coastal areas. In many Indigenous cultures, Atlantic salmon are sacred and spiritual, often symbolizing resilience, wisdom, and interconnectedness with nature. Salmon ceremonies, rituals, and storytelling have been passed down through generations, celebrating the salmon's annual return as a time of renewal, abundance, and community unity. These cultural practices foster a deep appreciation and emphasize the importance of sustainability and conservation (Per. Comm. Betty Ward, Metepenagiag (Red Bank) First Nation).

Atlantic salmon also have significant socio-economic value as both a commercial and recreational fishery. Once widely fished commercially with landings exceeding 750 tonnes annually in rivers like the Miramichi, New Brunswick, Canada (Dadswell et al., 2022), and 12670 tonnes globally (ICES, 2024), they are now primarily farmed, with Norway being the largest producer of farmed salmon, and the global salmon fish market

valued at nearly USD 15 billion in 2021 (Grand View Research, 2022). Meanwhile, recreational angling dates back hundreds of years where anglers from around the world would travel to fish in famous rivers along the eastern coast of Canada and the UK. This generates tourism, promotes local businesses, and fosters a sense of place and recognition for many of the surrounding communities. However, wild populations of Atlantic salmon have declined significantly due to habitat loss, overfishing, and many other factors, resulting in many jurisdictions implementing conservation measures, such as catch-and-release policies, restrictions, and closures (ICES, 2024).

1.3 Threats

Atlantic salmon have been declining over the past several decades in the North Atlantic, with some populations on the verge of extinction (COSEWIC, 2010; ICES, 2024). It is difficult to identify an exact cause to these declines, although thought to be due to an accumulation of limiting factors, possibly population specific (Cairns, 2001; Gibson et al., 2008). These factors are occurring in both the freshwater (Gregory et al., 2019; Otero et al., 2014; Russell et al., 2012) and marine environments (Beaugrand et al., 2002; Beaugrand and Reid, 2012; Friedland et al., 2014; Harvey et al., 2022; Mills et al., 2013), with varying magnitudes of adverse impacts. Currently, the greatest obstacle preventing recovery of Atlantic salmon appears to be mortality at sea, documented to be greater than 99% in some populations (Amiro et al., 2008; Jones et al., 2020). There are many potential causes, but the overall impacts of each are not well understood (Hutchings and Jones, 1998; Trzcinski et al., 2004).

1.3.1 Environmental factors

Several environmental limitations exist in both the freshwater and marine environments including water quality and chemistry issues, migration obstructions, predation, and competition pressures.

Water chemistry can be a critical element to certain life stages of salmon, particularly the egg and juvenile development stages. In addition to warming temperatures, acidic water can threaten freshwater populations (*e.g.*, Scotia Upland Designatable Unit in Eastern Canada) (COSEWIC, 2010). Acidic precipitation, in combination with minimal buffering capacity, affects pH, making the water too acidic for juvenile salmon to thrive, resulting in many extirpations (Brown et al., 1990; COSEWIC, 2010; Marmorek et al., 1998). A pH between 4.7 and 5.0 is considered moderately impacted, with values below 4.7 considered acidified and can impact survival (Caissie, 2000). Pollutants can also affect water chemistry increasing the toxicity of water and/or eutrophication, and affect fish directly by suppressing immune response, and harming gills and the epithelial layer (Kumaraswamy et al., 2020; Malik et al., 2020).

Physical obstructions like dams, causeways, mills, etc. can affect a population by directly impeding juvenile movement and adult upstream migration (Babin et al., 2021; Clark et al., 2020; COSEWIC, 2006; Dunfield, 1985). Obstructions can also influence flooding events, affecting juvenile rearing habitat and spawning grounds (Grabowski and Isely, 2007; Holbrook et al., 2009; Lapointe et al., 2004; Clarke et al., 2008), thereby locally eliminating an entire next generation of offspring.

Predation in rivers occurs from numerous aquatic, terrestrial, and avian species (Cairns, 2006). Avian predators, such as common mergansers (*Mergus merganser*) and belted kingfishers (*Megaceryle alcyon*), can have an impact on juvenile salmon populations,

with an estimated harvest of 21-45% of the juvenile population annually in some studies (see Cairns, 2001). Other predators known to harvest salmon in freshwaters include eagles, herons, bears, mink, and otters. In estuaries, predators like gadoids, eagles, ospreys, and seals feed at the mouths of rivers where smolts and adults enter the marine environment (Cairns, 2001; Hansen et al., 2003; White, 1939). Several studies suggest smolt predation is high as fish initially enter estuaries, with estimates as high as 50% mortality before entering the marine environment (Dieperink et al., 2002; Hvidsten and Lund, 1988; Larsson, 1985). At-sea predation is difficult to study but predation by several large pelagic fish, including porbeagle sharks (*Lamna nasus*) and Atlantic bluefin tuna (*Thunnus thynnus*) have been reported (Cairns, 2006; Lacroix, 2014).

Aquaculture is suspected to have various effects on wild populations of Atlantic salmon either due to farm escapees interacting with wild returning adults (Castellani et al., 2018; McGinnity et al., 2003) or the density of farmed fish in cages at proximity of estuaries being a source of parasites and diseases for wild salmon swimming nearby. This exchange can potentially result in a transmission of various diseases and cause competition pressure where farmed fish can out-compete wild stocks (Cairns, 2001). Farmed fish can spawn and reproduce in rivers containing wild salmon, producing viable offspring that can be less fit, experience lower survival, and produce less viable future generations, posing a threat to the genetic structure of wild Atlantic salmon (Amiro et al., 2008; Cairns, 2001; Lawlor, 2003; McGinnity et al., 2003).

While greatly improved rearing practices decrease many of these threats, this may be offset by the continuous increase of farmed Atlantic salmon production since 1980, reaching over 1.9 million tonnes in 2022, up from the previous five year mean of 1.75

million tonnes in the North Atlantic, while total worldwide production has been estimated at over 2.9 million tonnes (ICES, 2024). The vastly increasing global production of the salmon aquaculture industry is needed to compensate for the continued declining nominal catches of wild salmon (4000 times less than aquaculture production in 2022) in the North Atlantic (ICES, 2024).

1.3.2 Commercial Fisheries & Angling

1.3.2.1 Coastal/Estuarine/Riverine Fisheries

Commercial fishery landings have largely impacted Atlantic salmon populations such that fishery moratoriums have been implemented across many regions (COSEWIC, 2006). Nominal catches in coastal waters, estuaries, and rivers of the North Atlantic have severely declined since the mid 20th century, once peaking around 12000 tonnes in 1973, declining six-fold with the current five year mean less than 2000 tonnes, and many countries recently recording their lowest ever catches since the 1960s (ICES, 2024). The majority of catches in the last decade come from in-river fisheries, with a combination of coastal and estuarine fisheries depending proportionally by region/country (ICES, 2024). Many of the recent landing records reflect measures to reduce exploitations in several countries and/or the decline of wild Atlantic salmon stocks globally.

1.3.2.2 Greenland Mixed-Stock Fishery

One unique aspect of the Atlantic salmon at-sea commercial fishery is the mixed stock fishery in Greenland. Each year, thousands of salmon from rivers originating in North America and Europe migrate to Greenlandic waters to feed before returning to their natal rivers to spawn. In Greenland, the fishery is still open for locals although exporting has been banned since 1998. With this, it is difficult to determine how many individuals from

which populations are being harvested, meaning it can result in the exploitation of depleted salmon populations (Bradbury et al., 2016; Chase, 2003; Crozier et al., 2004).

1.3.2.3 Angling

Besides commercial riverine fisheries in some countries, angling for recreational purposes can affect and decrease populations. Poaching can occur from recreational angling, with an estimated unreported catch of 16.4 tonnes globally (ICES, 2024). Recreational fisheries remove a significant number of individuals, at nearly 23 tonnes in 2023 (ICES, 2024).

In Canada, recreational angling for salmon is popular with over 42000 salmon recorded as catch-and-release in 2023 (ICES, 2024), although this should be considered a known minimum as there is no mechanism to enforce anglers to report their catches, except in Quebec. Even if catch and release angling, 3-10% of individuals can be impacted due to the stress, exertion, and potential injury (COSEWIC, 2010), with greater stress and increased mortality rates found at increased temperatures (Van Leeuwen et al., 2020).

1.3.3 Depressed Population

When a population has declined to very low numbers, they can then suffer from a genetic bottleneck or depressed population phenomena (COSEWIC, 2010). Genetic bottlenecking occurs when genetic diversity and overall fitness is reduced within a population, resulting in less viable and conditioned progeny, more likely to experience adverse deleterious mutations and decreased survival (Allendorf and Luikart, 2007; Houde et al., 2011; Riquet et al., 2019).

Depressed population phenomenon describes how the recovery of small salmon populations is limited by the opportunistic predation pressure experienced in the open ocean from large populations of predators (DFO, 2010). Even if an individual predator only consumes small numbers of salmon, the predation impact as a whole can have a proportionally large effect on the salmon population. Additionally, low populations result in spatial contraction where some resource patches may be at capacity, while other patches remain completely unused. This limits the ability of a population to rapidly increase in abundance when depressed to low levels (Atlas et al., 2015; Rose et al., 2001).

1.4 Mitigation measures to population decline

1.4.1 Policy

Regardless of the causes of population decline, without human intervention and implementing recovery actions, many populations would have already disappeared. When a population becomes at risk, policy and legislative actions are crucial in species protection. In Canada, the Species at Risk Act (SARA) is the primary legislative tool used to protect species at risk, providing a framework for identifying and protecting endangered and threatened species and their habitats through the creation of recovery strategies, establishment of protected areas, and the regulation of activities that may harm at-risk species. Similarly in the United States, the Endangered Species Act is used to protect at-risk species, serving a similar purpose to the SARA to provide legal protections to conserve critical habitat and their population (US Fish and Wildlife Service, 1973). Under these acts, industrial, agricultural, and recreational activities which potentially cause disturbance to species or their habitat in any form can be regulated and penalties

assessed appropriately, where necessary, to deter adverse impacts and/or harm (Goble et al., 2006).

It's not uncommon to see fishery closures implemented indefinitely or temporarily to conserve a particular species if conditions are deemed unsuitable and could cause harmful stress (Breau, 2012).

1.4.2 Habitat Restoration

Suitable habitat requirements are essential for any salmon population to thrive. Habitat restoration initiatives are a critical area to begin when these requirements are not met. This includes improving water quality by identifying and reducing pollution inputs, and/or controlling potential excess sedimentation. Increased sedimentation caused by erosion can be improved by restoring natural stream channels and the protection or addition of riparian vegetation to provide shade, cover, and food sources (Owens et al., 2005; Wang et al., 2016). Additionally, the construction of fishways and removal of barriers like dams or culverts can facilitate upstream migration and access to important adult spawning grounds and rearing areas for juveniles (Hill et al., 2019; Jonsson and Jonsson, 2009). Improving the quality and availability of important spawning and rearing habitat can involve enhancing gravel beds by removing accumulated sediments and debris, introducing appropriate substrate, and maintaining a natural flow regime by regulating flow from reservoirs to ensure suitable water levels and temperatures for successful spawning (Jonsson and Jonsson, 2009; Meyer et al., 2008; Webster, 1962). These efforts should also take the broader ecosystem into account, including conserving and restoring the connectivity of river systems, as well as managing land use practices in surrounding areas to minimize impacts such as deforestation, agriculture, and

urbanization (Wang et al., 2016). By addressing habitat degradation and promoting healthy, resilient ecosystems, habitat restoration initiatives provide essential support for the survival, growth, and recovery of Atlantic salmon populations at various life stages.

1.4.3 Stocking

If more intensive intervention is required, various restoration strategies directed at population augmentation or supplementation through stocking can be explored (NASCO, 2017). The idea of stocking has existed for centuries worldwide, with fish culture thought familiar to China more than 2000 years ago, but much development with this strategy for the purpose of augmenting Atlantic salmon populations was documented through the 1800's in Europe and North America (Dunfield, 1985). In Canada, stocking juvenile salmon occurred as early as 1858 by Richard Nettle, with the first government-sponsored hatchery operating in New Castle, Ontario in 1866. This was founded by Samuel Wilmot, who also aided in the establishment of the Miramichi Fish Hatchery, constructed in 1873 and remains the oldest operating fish hatchery in Canada, stocking Atlantic salmon for over 150 years (COSEWIC, 2006; Dymond et al., 2019; Wilmot, 1873).

Stocking programs have been used worldwide for many purposes from maintaining or increasing genetic diversity, off-set losses from commercial or recreational fisheries, ecosystem benefits, and restoring or preventing populations from going extinct, among other reasons (Clarke et al., 2016; Cowx, 1994; NASCO, 2017; Samways and Cunjak, 2015). Although there are benefits to stocking, it should be approached with great caution, as the disadvantages and potential risks it presents (NASCO, 2017). Risk likelihood often depends on the rearing environment, practices, and precautions taken within the program design, the state of the wild stocks, as well as many other factors.

Identifying whether the causes which contributed to the populations' initial decline are still present, the current state of the population, and the potential threats and consequences that may impede recovery, all need to be considered when determining future stocking initiatives to determine the acceptable degree of risk. Where possible, risk can be minimized by selecting the appropriate life stage (*e.g.,* eggs, fry, parr, smolt, adults) for intervention or when creating a captive-breeding program.

1.4.3.1 Egg Planting

Incubation boxes for artificial stocking of eggs are often used in areas of rivers that are inaccessible to adults or with decreased spawning activity, areas of low egg survival, or improving local adaptation to natural conditions by minimizing captive rearing exposure (Conley et al., 2020; Kennedy and Strange, 1981). Freshly fertilized ("green") or eyed ova are placed into separate compartments within a plastic box which is submerged and buried within the substrate or suspended in the water column just above the substrate. Stocking with both egg stages have resulted in varied success within different environments (Kennedy and Strange, 1981; Jonsson and Jonsson, 2009). While incubation boxes can protect eggs from predation or being scoured away in high flow events, one of the biggest disadvantages to this approach, similarly with hatchery reared juveniles as well, is the human influence on genetic diversity. An insufficient number of crosses used to produce a large number of eggs/offspring can potentially increase the amount of inbreeding and reduce the total effective population size (Laikre et al., 2010; Laikre and Ryman, 1996), and narrow genetic diversity (Naish et al., 2013).

1.4.3.2 Fry/Parr/Smolt

If juvenile survival in a population is low, fry, parr, or smolt stocking are potentially viable stocking recovery strategies. Fry stocking is common and can take the form of translocation juveniles from a nearby river system (NASCO, 2017) or releasing captive born individuals originating from hatchery facilities (DFO, 2018). Parr and smolt life stages can also be produced in hatchery facilities, with eggs reared, hatched, and grown from captive spawners, then released into rivers at the desired life stage. This approach bypasses much of their juvenile development and growth in the wild where survival could be low, potentially due to water quality and temperatures issues, high predation, competition, or low resource availability (Gray and Cameron, 1980; Romakkaniemi, 2008). Juvenile stocking can be beneficial when egg deposition or juvenile survival is low, or to supplement the existing population (Bacon et al., 2015; Gray and Cameron, 1980). If managed correctly, release strategies of juveniles of various life stages from unfed and 6-week feeding fry, Age-0 and Age-1 parr, and Age-1 and Age-2 smolts can be incorporated to reduce any potential effects of domestication on the population (Jones et al., 2020). Research shows the less captive exposure an individual endures earlier in life, the greater their fitness throughout their lifecycle (Clarke et al., 2016). Therefore, if juvenile stocking is employed, minimizing time in captivity and releasing juveniles at the earliest life stage possible is recommended (Huntingford, 2004).

1.4.3.3 Adult

Although less common, adult stocking is also used for salmon river restoration by introducing adults to a system, either “rescue” from a nearby source population of the same local wild stock, through captive-breeding or smolt-to-adult supplementation (SAS). SAS has been used in a freshwater rearing environment for the Saint John and

Miramichi Rivers (DFO, 2016, 2018; Jones et al., 2014), and in a marine rearing environment with Fundy Salmon Recovery in the Bay of Fundy (Bryson et al., 2022; Clarke et al., 2016). This can be an effective strategy as SAS typically uses local fish, avoids captive-rearing at early life-stages, allows natural mate selection in the wild, avoids any captive exposure to the next generation of offspring being produced, and additionally has the value of adults out-migrating, surviving in the marine environment, and returning to spawn again with a higher fitness (Nason et al., 2022). However, there are many risks with this strategy that could produce unintentional effects if conducted ineffectively or irresponsibly. These include, but may not be limited to domestication effects, relaxed selection to natural environmental pressures, depressed fitness on wild stocks, loss of genetic variation, and epigenetic effects (Christie et al., 2012; NASCO, 2017).

1.5 Population Assessment

Understanding the effectiveness of stocking strategies, and how they translate into population abundance, requires adequate monitoring and population assessment. Although, monitoring and assessment cannot always be used interchangeably and need to be contextually defined. Regular monitoring is important in determining the relative status and/or stability of a population over time. Monitoring facilitates the development of trends to aid in detecting relative population fluctuations and can identify the beginning of species declines (Allendorf and Luikart, 2007; Olney et al., 1994). However, this may not be sufficient to provide an accurate assessment of population dynamics (*i.e.*, recruitment, survival, abundance, etc.), especially when population abundance has diminished past an effective population target. Monitoring data needs to

be collected consistently, within and across rivers, with similar methodology in order to be used in a robust assessment, allowing for the comparison of population metrics over time and space. Even then, if monitoring data only looks at a single specific life stage, population dynamics cannot be inferred based on these data.

1.5.1 Electrofishing

To assess population abundance of freshwater fish species, in particular salmonids, electrofishing is a commonly used monitoring method in wadeable rivers. The general principle is that a power source (*e.g.*, generator, battery) will generate a high-voltage current between a submerged anode and cathode. When a fish encounters a large enough potential gradient, it will induce galvanotaxis in the fish (*i.e.*, uncontrolled muscular convulsion) making it swim towards the anode, allowing an easy capture of individuals to be sampled. Electrofishing can be used in a range of sampling techniques to estimate local population abundance such as capture-mark-recapture, multi-pass successive removal, or single-pass Catch Per Unit Effort (CPUE) (Matson et al., 2018; Rosenberger and Dunham, 2005). Capture-mark-recapture methods are often found more accurate but require multiple visits to the same sites, which may be difficult depending on location access (Rosenberger and Dunham, 2005; Ruetz et al., 2015). Multi-pass and single-pass removal sampling typically only requires a single site visit to provide a metric of population abundance. Multi-pass techniques often involve using fence/barrier/block nets and are more labour intensive than single-pass sampling but can provide accurate estimates of population abundance (Carle and Strub, 1978; Seber and Whale, 1970; Zippin, 1958). Single-pass methods are often seen as more flexible than multi-pass methods as they are typically less resource intensive. However, they only provide an

index of abundance and, in order to provide a population abundance estimate they need to be calibrated with sampling techniques producing population estimates (Dauphin et al., 2009; Dauphin et al., 2019). It is recommended that calibration should be completed regularly as changes in gear, crew, or river hydraulics can affect the shape of the relationship (Glover et al., 2018; Malcolm et al., 2019; Millar et al., 2016; Dauphin et al., 2019). When estimating population abundance at the river- or basin-scale, a combination of multi-pass and single-pass sampling methods are often used as single-pass methods allow for a broader spatial coverage, while the multi-pass depletion methods allow more accurate population abundance estimates and can be used to calibrate the single-pass sampling methods (Hanks et al., 2018; Myvold and Kennedy, 2017).

1.6 Atlantic National Parks with Active Atlantic Salmon Restoration

1.6.1 Terra Nova National Park

Terra Nova National Park (TNNP) is located on the East coast of Newfoundland (within the Northeast Newfoundland DU (Figure 2)Figure 2) and encompasses part of the Northwest River (Figure 3). The salmon population in this river experienced declines in their adult salmon population through the 1990's and early 2000's, recording an all-time low of 151 individuals in 2001 on the Northwest River (Cote et al., 2021). The following year, a working group was formed within the community to address poaching, identified as the main driver of decline and on-going threat to recovery. An *Adaptive Fisheries Management* approach was implemented, allowing a small recreational retention fishery through a quota system of typically 75-100 tags, with the option of allocating more if adult escapement reaches conservation targets pre-determined for that year. This has

proven successful in the Northwest River as adult abundance increased to more than 2500 in 2019 (Figure 4, Cote et al., 2021).

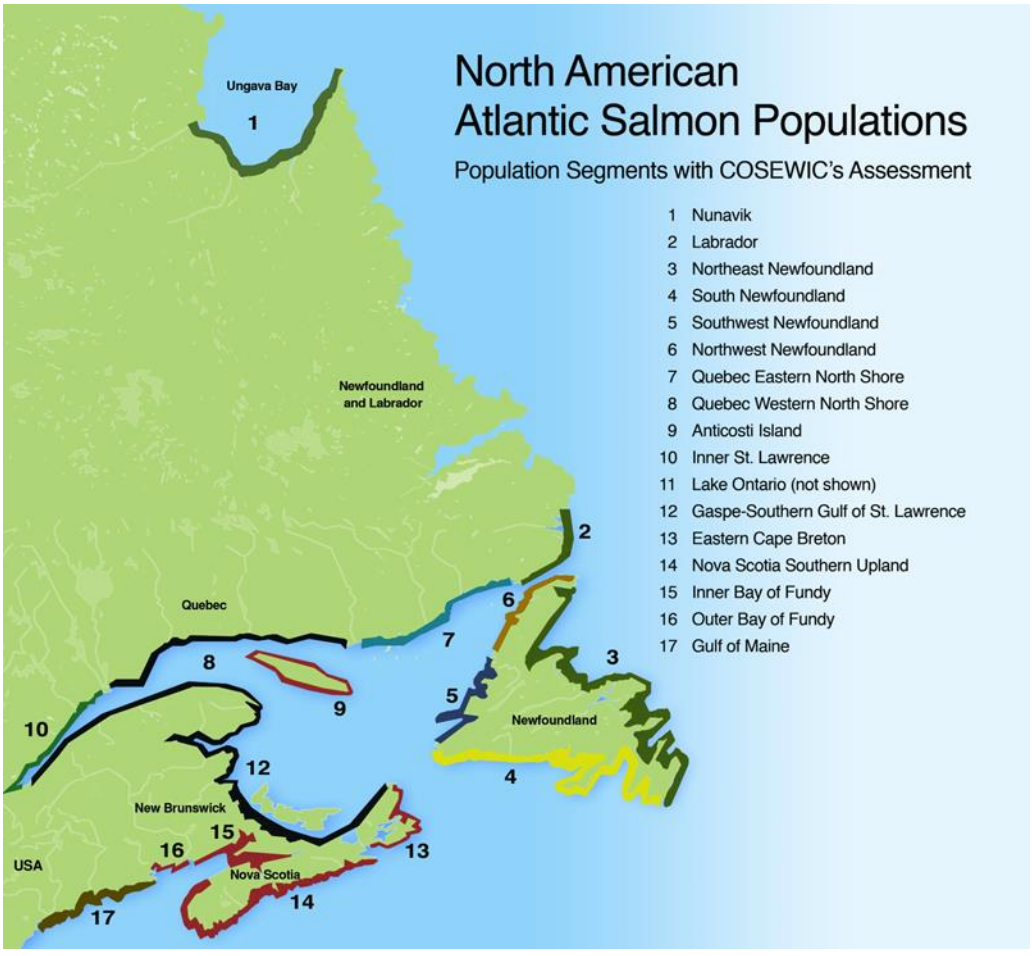

Figure 2. Map of Atlantic salmon Designatable unit populations for North America. (Atlantic Salmon Federation).

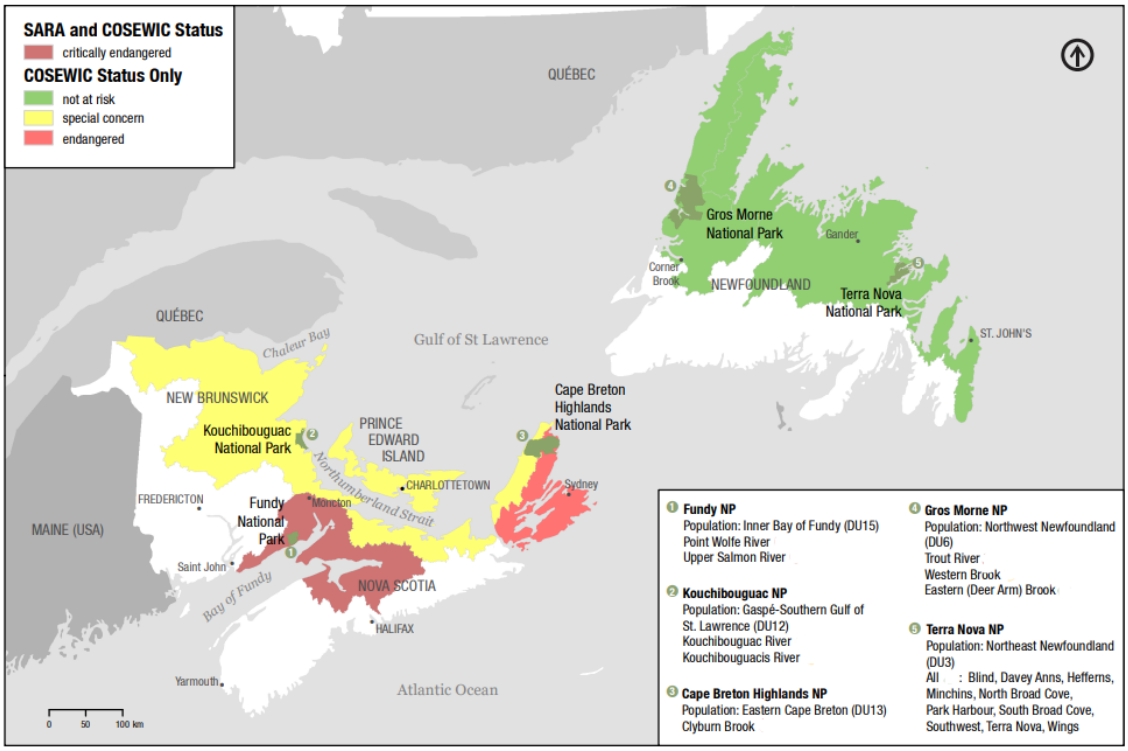

Figure 3. Map of each National Park being studied (dark green, numbered one to five) and their Designatable Unit (coloured based on COSEWIC/SARA status). The bottom right corner legend indicates rivers of interest in each Park (GeoNB, Federal Open Source database).

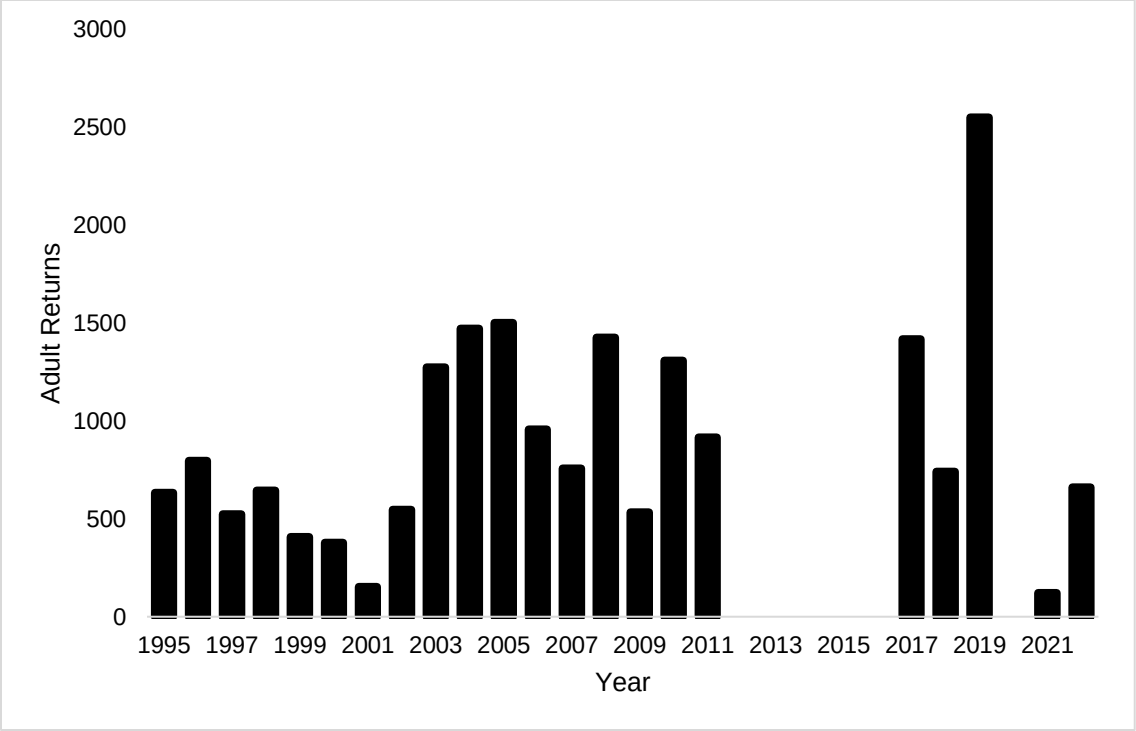

Figure 4. Adult Atlantic salmon returns counted using a counting fence at Northwest River in Terra Nova National Park from 1995-2022. In 2002, 2008, 2009, 2018, 2021, and 2022 the counting fence was removed early due to warm temperatures, storm events, water levels, and/or blow outs.

1.6.2 Gros Morne National Park

Gros Morne National Park (GMNP) is located on the West coast of Newfoundland, containing Western Brook, Deer Arm Brook, and Trout River (Figure 3). These rivers are a part of the Northwest Newfoundland DU (Figure 2), which is not at risk under the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) (COSEWIC, 2010), however, Trout River recorded a low of 13 returning adults at their counting fence in 2017 (GMNP unpub.). This status differs from other rivers within the National Park, Western Brook and Deer Arm Brook, which still see stable spawning escapements. Like the Northwest River of Terra Nova, the greatest threat identified to Trout River salmon

populations is poaching. For this reason, GMNP is following the Terra Nova approach and focusing on community engagement and educating others of their salmon program and state of Trout River. These efforts helped contribute 51 returning adults in 2022 (Figure 5).

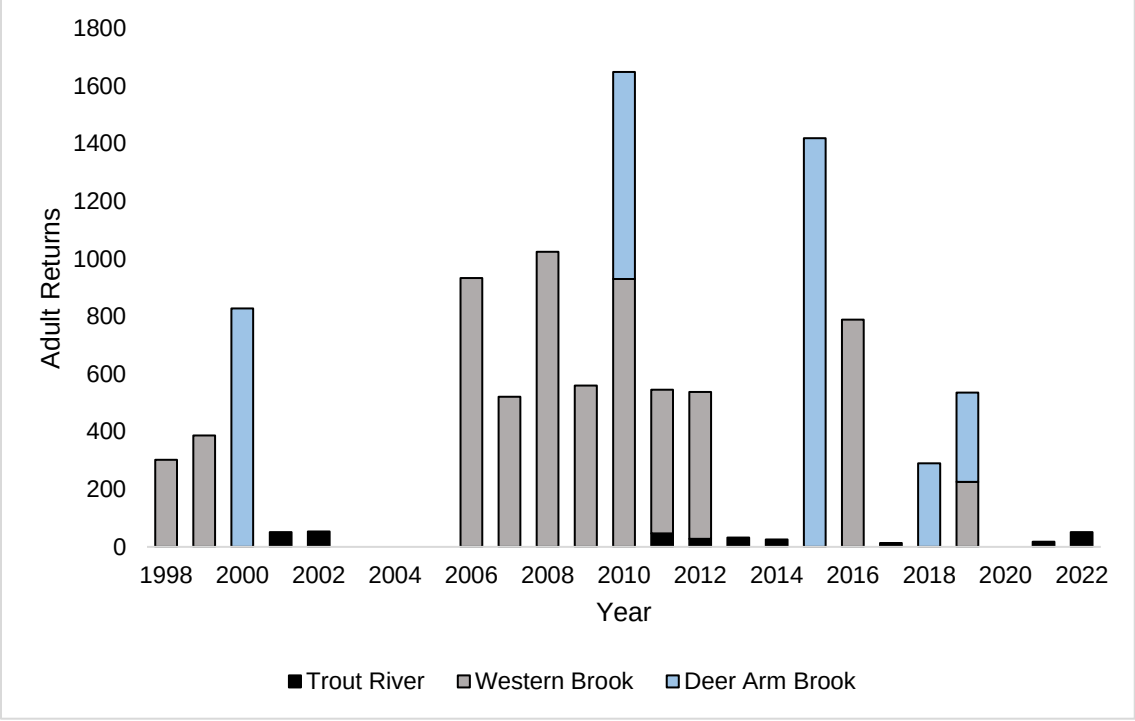

Figure 5. Adult Atlantic salmon returns counted using counting fences on Trout River, Western Brook, and Deer Arm Brook in Gros Morne National Park from 1998-2022.

1.6.3 Cape Breton Highlands National Park

On the Northern coast of Nova Scotia, Cape Breton Highlands National Park (CBHNP) consists of Clyburn Brook on the eastern side of the Park, belonging to the Eastern Cape Breton DU (Figure 2), and listed as endangered (COSEWIC, 2010). Adult abundance in Clyburn Brook, estimated via snorkel surveys, observed a low of five adults in 2019. In

response, CBHNP, in partnership with the Aquatron at Dalhousie University, has started a smolt-to-adult supplementation program, where out-migrating smolt are collected from Clyburn Brook and transported to Halifax, NS, grown to maturity and subsequently released back to Clyburn Brook, where they can spawn naturally. CBHNP has facilitated annual adult releases since 2021, with the following year counting 60 returning adults in their snorkel survey, the most since 1997 (Figure 6).

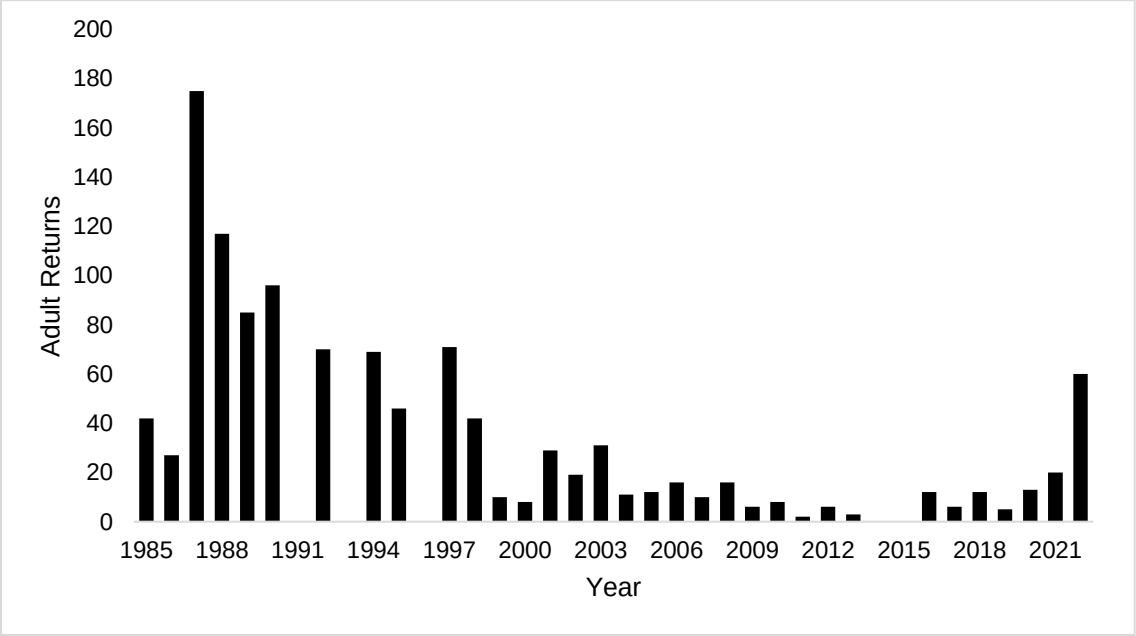

Figure 6. Adult Atlantic salmon returns based on snorkel counts in Clyburn Brook in Cape Breton Highlands National Park from 1985-2022. No surveys were completed in 1991, 1993, 1996, and 2015.

1.6.4 Kouchibouguac National Park

On the East coast of New Brunswick is Kouchibouguac National Park (KNP) (Figure 3). Located within is the Kouchibouguac and Kouchibouguacis rivers, a part of the Gaspé-Southern Gulf of St. Lawrence DU, designated as Special Concern (Figure 2, COSEWIC,

2010). In KNP rivers, the adult population status generally remains unknown as adults are intercepted using modified-smelt box traps in the fall, but typically for targeting adults for their egg incubation program, rather than assessment purposes. A portion of captured adult salmon are temporarily retained at the Miramichi Salmon Conservation Centre until they are ready to spawn. At which time, the eggs are collected, fertilized, and placed in Jordan-Scotty incubator boxes for protection against predation and environmental conditions. While the traps are operating, the nets cover a portion of the waterway, therefore, only a small percentage of the returning population is known.

1.6.5 Fundy National Park

Fundy National Park (FNP) is located on the Southern coast of New Brunswick alongside the Bay of Fundy (Figure 3). Here, the Upper Salmon (USR) and Point Wolfe Rivers (PWR) have a long history of decline, near extirpation, with a variety of recovery strategies trialed (Figure 7). The Inner Bay of Fundy (iBoF) DU (Figure 2) has precipitously declined for decades which led to the initiation of a Live Gene Bank program in 1998 at the Mactaquac Biodiversity Facility, operated by Fisheries and Oceans Canada, to conserve the remaining genetic diversity of this stock. The iBoF Atlantic salmon population is the only DU in Canada listed as endangered under SARA (DFO, 2010). Specifically, FNP is conserving Point Wolfe River High Ancestry genetics, which are specific to the Point Wolfe and Upper Salmon rivers, identified within the 10 critical habitat watersheds within the iBoF. Beginning in 2006, years of juvenile stocking occurred before the initiation of a pilot project to release adults to the Upper Salmon River, allowing them to spawn naturally, with the next generation of juveniles having no captive exposure (Clarke et al., 2016). This was the foundation of Fundy Salmon

Recovery with the strategy of collecting a portion of out-migrating smolts, raising them to maturity at the world's first wild salmon marine conservation farm located on the island of Grand Manan, and then releasing them back to their natal river to spawn naturally. Because of the extensive work in FNP and declining status of the iBoF population, there has been long-term consistent monitoring of different life stages, making FNP the most data rich park in terms of population modelling.

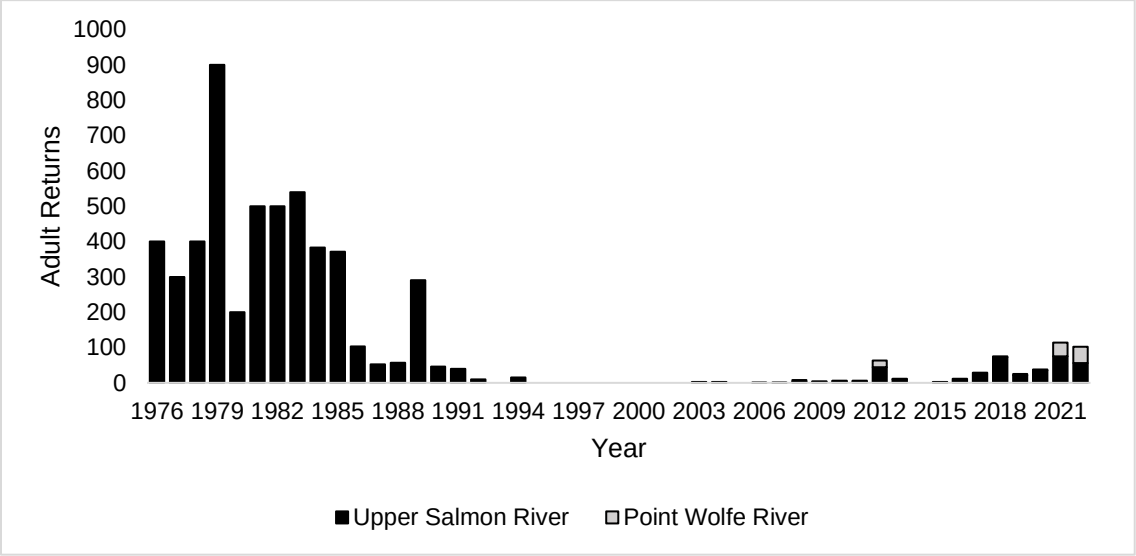

Figure 7. Adult Atlantic salmon returns based on shore estimates from 1976-1982, snorkel surveys from 1983-2015, and a combination of snorkel surveys and PIT telemetry from 2016-2022 to the Upper Salmon and Point Wolfe Rivers in Fundy National Park. Annual counts of 0 typically represent no surveys except 1998, 2002, 2007, and 2014 in the Upper Salmon River, and 2002, 2003, 2014, 2016, and 2017 in the Point Wolfe River, where surveys occurred with no salmon observed.

1.7 Study Purpose/Objectives

In 2019, Parks Canada initiated a multi-park Atlantic salmon recovery project across Fundy, Kouchibouguac, Cape Breton Highlands, Gros Morne, and Terra Nova National

Parks, to assess the decline of salmon on a regional landscape, outside of their respective rivers or Designatable Unit (DU). This initiative aimed at bringing all conservation and restoration programs together to centralize and standardize datasets and attempt to provide comparable population assessments across parks to provide new insights to salmon recovery on a larger scale. Because of this, multiple salmon initiatives are occurring within each park simultaneously, some with long-standing monitoring programs and recovery history (*i.e.*, Fundy National Park), while others are just entering the recovery process (*i.e.*, Gros Morne National Park).

Although adult populations are monitored across most parks, this abundance doesn't necessarily equate to a healthy or self-sustaining population. Understanding the status of various key life stages as well as the impacts of recovery actions taken is needed to assess progress and implications of each recovery program and/or any intervention measures. Because of differing recovery strategies, population abundances, monitoring objectives, and resource availability to each park, there lacks a standardized protocol across National Parks. This leads to different methodologies being used and monitoring data recorded in different ways. As a consequence, the full juvenile/electrofishing dataset is heterogeneous and its standardization requires a range of hypotheses and modelling techniques to be able to reliably compare population estimates across parks.

This study focuses on juvenile population abundance using a dataset spanning multiple rivers across all five National Parks, collected with varying methodologies of single-pass and multi-pass successive removal sites over a time series of several decades. Covariates available for each monitoring program are related to environmental and habitat characteristics, gear used, and within-site sampling procedure however, for most

programs one or several of these covariates are missing. The objectives of this study were: (i) provide standardized site density estimates regardless of the method used (single vs. multiple pass) and the river in which the data was collected, (ii) conduct robustness testing to assess how accurate (or robust) the single-pass densities are predicted compared to multi-pass densities, (iii) provide average annual density estimates within each river to assess population trends before and during recovery implementation, and (iv) provide managers with a practical and powerful assessment tool. The first objective of reconciling two sampling methodologies is a common problem and has been well described in the literature (Dauphin et al., 2009; Chaput et al., 2005; Wyatt, 2002). It typically involves a calibration process where sites are sampled first with the single-pass technique and then followed up by successive removal passes to relate the index of abundance from the first pass to the population abundance estimate of the successive removal passes. This method is commonly used by Fisheries and Oceans Canada in larger systems like the Miramichi and Restigouche rivers to assess Atlantic salmon populations by covering a large area rapidly. In this study there are no true calibration sites. Therefore, at the sites where a multi-pass successive removal sampling occurred, the probabilities of capture were modelled accounting for all covariates available, then at the sites where a single-pass methodology was used, the pass was modelled as the first pass of a successive removal sampling.

Methods

This study catalogues and analyses electrofishing data collected from 24 rivers across Eastern Canada. All Rivers are located in one of five National Parks in New Brunswick (FNP and KNP), Nova Scotia (CBHNP) and Newfoundland (GMNP and TNNP) (Table 1, Figure 3). All rivers belong to a DU, "*a unit of Canadian biodiversity that is discrete and evolutionarily significant"* (COSEWIC, 2010), where Atlantic salmon vary in status from Not at Risk (*e.g.*, Newfoundland populations) to Critically Endangered/SARA listed (*e.g.*, Inner Bay of Fundy population) (Table 1, Figure 3) (COSEWIC, 2010, DFO, 2010). In Eastern North America, there are 17 distinct populations/DUs of Atlantic salmon, 16 of which are in Eastern Canada (Figure 2), with each National Park belonging to a different DU (Table 1).

In Eastern Canada, the conservation status of Atlantic salmon populations have varied greatly over the past several decades, but with a common trend of steady declines or relatively low abundance (COSEWIC 2010). In response to these changes, each of the National Parks have had a range of responses, some initiated long-term, intensive recovery programs (*e.g.,* FNP) while others are in the initial development or implementation stages (*e.g.*, CBHNP, GMNP) of their program. These differences are reflected in their monitoring efforts (spatially and/or temporally) and protocols, with some parks having long-term electrofishing programs of nearly 50 years, while others only started their monitoring programs in the early 2000s' (Table 2).

Table 1. List of National Parks with their associated Atlantic salmon Designatable Units, and SARA or COSEWIC population status (COSEWIC, 2010).

National Park	Designatable Unit	Population Status
Fundy National Park	Inner Bay of Fundy	Critically Endangered
Kouchibouguac National Park	Gaspé-Southern Gulf of St. Lawrence	Special Concern
Cape Breton Highlands National Park	Eastern Cape Breton	Endangered
Gros Morne National Park	Northwest Newfoundland	Not at Risk
Terra Nova National Park	Northeast Newfoundland	Not at Risk

Table 2. List of National Parks and their respective study rivers, their drainage and juvenile rearing habitat when available. (KNP, CBHNP, GMNP: O'Connell, 1997; FNP: Kerekes and Schwinghamer, 1978; FNP: Alexander and Gailbraith, 1982), electrofishing data time series available, and total number of sites sampled by method type (open site: single-pass; closed site: successive removal/multi-pass).

National Park	River	Year Range	Open Sites	Closed Sites	Total Sites	Juvenile Rearing Habitat (100 m^2)	Drainage Area (km^2)
Fundy National Park	Upper Salmon River	1984-2022	29	104	133	798	173
	Point Wolfe River	1984-2022	4	133	137	1707	130
Kouchibouguac National Park	Kouchibouguac River	1974-2022	59	19	78	5880	420
	Kouchibouguacis River	1974-2022	48	8	56	5490	362
	Black River	2021-2022	0	8	8	-	420
Cape Breton Highlands National Park	Clyburn Brook	2012-2022	17	5	22	1165	70
Gros Morne National Park	Trout River	2002-2022	0	39	39	1120	254
	Western Brook	2020	0	1	1	2150	192
Terra Nova National Park	Arnold's	2002-2003	0	4	4	NA	NA
	Terra Nova	2002-2012	0	16	16	NA	NA
	Big Pond	2003	0	2	2	NA	NA
	Big	2003-2005	0	4	4	NA	NA
	Cobblers	2003	0	2	2	NA	NA
	Minchins	2003-2011	0	3	3	NA	NA
	Wings	2003-2012	0	8	8	NA	NA
	Blue Hill	2003	0	1	1	NA	NA
	Blind	2005-2012	0	6	6	NA	NA
	Southwest	2005-2011	0	4	4	NA	NA
	Saltons	2005	0	1	1	NA	NA
	Park Harbour	2005-2012	0	5	5	NA	NA
	North Broad Cove	2005-2011	0	3	3	NA	NA
	South Broad Cove	2010	0	1	1	NA	NA
	Hefferns	2011	0	1	1	NA	NA
	Davey Ann's	2011	0	1	1	NA	NA

2.1 Electrofishing Protocols

Backpack electrofishing was conducted within each of the five National Parks to monitor juvenile Atlantic salmon populations, beginning as early as 1974 (*e.g.*, KNP). As of 2022, this amounted to 536 sampling sites across all parks spread over a 49-year time-series (Table 2, Figure 8).

In all parks, fish collected during a sampling event were identified to the species level. All individuals were counted, and their length and weight measured. In some instances, caudal fin clips and/or scales were collected for genetic or aging purposes, respectively. Typically, Atlantic salmon juvenile ages are determined based on length frequency distribution (Sethi et al., 2017). Usually, the length differences between young of the year (0+) and older juvenile fish are obvious with very little overlap in their length distributions. However, distinguishing a one-year-old (1+) from a two (2+) or three year (3+) old juvenile based on their length alone can be challenging either because of differential growth rate and/or too low number of juvenile fish of these age classes collected at a given site, in a given river, for a given year. For these age classes, scale reading is needed to confirm the age of an individual. Unfortunately, this information was not often available and therefore in this study juveniles were classified as fry (0+) or parr (1+ and older).

Sampling protocols and data collected varied greatly among and/or within parks or across years making direct comparisons not possible. Common metrics of site data collected are shown in Table 3.

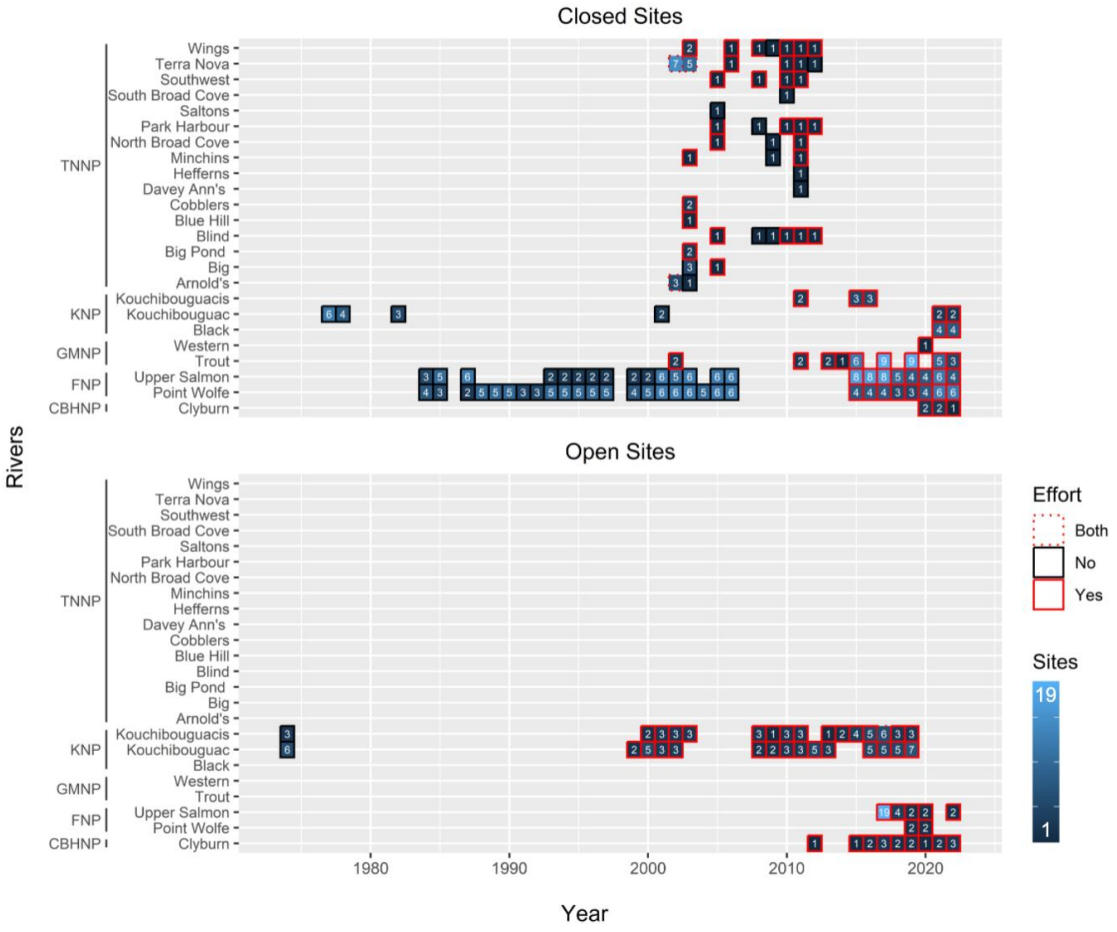

Figure 8. Number of electrofishing sites surveyed each year within each study river, grouped by park. Top panel shows closed sites sampled using the multi-pass method and bottom panel shows open sites sampled using the single-pass method. The number within each box and colour gradient corresponds to the number of sites sampled per associated year/river (lighter = more, darker = less). A border around each box represents if there was effort recorded for all sites throughout the year (solid red), a mixture of effort and non-effort sites (dotted red) or not (solid black).

Table 3. Site data collected by each National Park during electrofishing sampling.

	Fundy 1984-2006	Fundy 2015-Present	Kouchibouguac	Cape Breton Highlands	Gros Morne	Terra Nova
River	Yes	Yes	Yes	Yes	Yes	Yes
Date	Yes	Yes	Yes	Yes	Yes	Yes
GPS Location	Yes	Yes	Sometimes	Yes	Yes	Yes
Fish Counts	Yes	Yes	Yes	Yes	Yes	Yes
Sweep/Pass	Yes	Yes	Yes	Yes	Yes	Yes
Effort	No	Yes	Sometimes	Yes	Yes	Sometimes
Habitat %	No	Sometimes	Sometimes	Sometimes	No	Sometimes
Habitat Type - Riffle/Run	Yes	Yes	Sometimes	Sometimes	No	Sometimes
Site Length	No	Sometimes	Sometimes	Yes	No	No
Site Width	No	Sometimes	Sometimes	Yes	Sometimes	Sometimes
Site Area	Yes	Yes	Yes	Yes	Yes	Yes
Depths	No	Sometimes	Sometimes	Sometimes	Sometimes	Sometimes
Substrate %	No	Sometimes	Sometimes	No	No	No
Water Temperature	Sometimes	Sometimes	Sometimes	Yes	Sometimes	Sometimes
Crew	No	Sometimes	Sometimes	Yes	Sometimes	No
Collection Equipment	Yes	Yes	Sometimes	Sometimes	Yes	No
Efisher Type	No	Yes	Sometimes	Yes	Yes	No
Voltage	No	Sometimes	Sometimes	Yes	Yes	No
Duration	No	Sometimes	No	Sometimes	No	No

2.1.1 Fundy National Park

Fundy National Park has the most consistent sampling protocol of all parks. The FNP time series data is split into two periods, (i) 1984-2006 and (ii) 2015-2022. This is because there was a fundamental change in the sampling protocol to standardize effort which had a significant impact on density estimates.

Six Permanent Sample Plots (PSPs) were established for each of the USR and PWR. These plots were categorized as upper, middle, and lower sections to reflect their location within the river and split between riffle and run sites within each section. Riffle type habitats were identified by shallow, fast moving water with a broken flow over the substrate, while run habitats, often found at the end of pools, had fast flow but more uniform with no broken water surface, and often slightly deeper. The location of these PSPs were determined based on ease of access as well as knowledge of historic spawning sites (*i.e.*, sites were chosen to be downstream of known spawning grounds). GPS coordinates are available for all PSPs allowing for consistency in annual return of the sampling location. The only exception for the modification of a site's coordinates occurs when river morphology changes, and the site is no longer suitable. In that scenario, location may be shifted upstream or downstream by up to a hundred meters to maintain similar habitat characteristics between sampling years.

When sampling closed sites (*i.e.*, multiple pass), 25-meter long block or barrier nets are installed perpendicular to the river banks at the lower and upper boundaries of the site in order to 'close' the site for the duration of the sampling and prevent fish movement in or out of the site. Typically, nets are made of nylon with a 7 mm mesh size, allowing water to flow but no fish of interest to swim through them. In order, to maintain the fish

population as close as possible to its state prior the installation of the nets, the lower net is installed first to minimize the number of fish escaping the site during net installation, as disturbed fish tend to escape primarily following the current (FNP, pers. obs.). Rocks from the riverbed are used to anchor the net bottom, while wooden stakes are used to elevate the net top out of the water and seal the site. The size of the site is characterized by two length measurements (one on each side) and two width measurements (upstream, downstream), forming a sampling area of approximately 250 m^2.

2.1.1.1 1984-2006 Electrofishing Protocol

From 1984-2006, the sampling crew consisted of an electrofisher operator and two to three crew members with dip nets used to collect incapacitated fish and store them in a water-filled bucket until the end of the pass. Sampling started at the upstream end of the site and the crew performed a bank to bank transect. Once a transect was completed, the crew moved downstream and conducted another transect until they reached the downstream net. The general principle was to go over each square meter of the site once. Following the first pass through the site, all fish were brought to the processing area where fish are placed in aerated buckets or live wells waiting to be processed. Fish are kept outside of the sampling area until all the passes were completed. The second pass started no less than 10 minutes after the preceding one ended to let the remaining fish settle back in their habitat. Three to six passes were conducted at each site with the objective of observing a depletion (*i.e.*, less and less fish captured after each pass). Data collected at the sampling site consisted of the number of passes, fish captures per pass (including length and weight), site area, and site location.

2.1.1.2 2015-2022 Electrofishing Protocol

In 2015, the protocol evolved from having two-three dip netters, to having two crew members using a 1.5 m collection lip seine downstream of the electrofisher operator, while one dip netter remained on the four-person crew. The switch to a lip seine was seen as a more efficient way to capture smaller and more evasive fish. Additionally, sampling pass direction was reversed as to be conducted from the downstream to upstream end of the site to mimic how open sites are sampled (see below), as they were planned to continue in the coming seasons. Starting in 2015, power was standardized across sites and voltage was systematically adjusted to ensure an ~80-watt output, using a Smith-Root LR24 backpack electrofisher, set to 30Hz, 12% duty cycle, and pulsed DC. Due to the low conductivity of freshwater systems in FNP, voltage in excess of 800 volts is not uncommon.

In FNP, the single-pass sites (open) started being conducted in 2017. The protocol for single-pass sites is similar to the one used within multi-pass sites, except barrier nets are not installed and only one pass is conducted. The lower operational requirements in comparison to the closed site method allow for a larger number of sites to be sampled throughout the catchment. These sites have been incorporated in annual assessments to compliment the PSPs or to replace closed sites if time and human resources become limited.

From 2015 onward, in addition to the number of passes, fish captures per pass (including individuals' length and weight), site area and location, additional information on the site characteristics such as substrate and habitat type, depths, and water velocity were also collected, along with, effort, gear types and power settings. The variability in the type and

amount of data available for each site throughout the time-series is a significant challenge to consider when attempting to standardize density estimates in FNP and/or across parks throughout the time-series.

2.1.2 Gros Morne National Park

Electrofishing monitoring started in 2002 in Trout River of GMNP, where they used a closed site, multiple pass methodology. Barrier nets were installed, downstream then upstream, to average a 100-200m^2 area. A Smith-Root LR24 electrofisher was used with settings established using the "quick setup" option which adjusts voltage, frequency and duty cycle according to conductivity, temperature and depth of the anode in the water (GMNP, unpub.). Voltage often ranges from 350-400 volts. Similarly to the FNP protocol used between 1984 and 2006, sampling occurred from bank-to-bank moving from upstream to downstream, with a team of three (one electrofisher operator and two dip netters) (GMNP, unpub.). Sampling stopped or did not occur if temperatures were >18°C.

2.1.3 Cape Breton Highlands National Park

Electrofishing monitoring data was available beginning in 2012 in Clyburn Brook of CBHNP, using primarily single-pass methods, as CBHNPs monitoring program used Catch per Unit Effort methodology based on existing historic data within the park. Beginning in 2020, multiple pass methods were resumed to recalibrate estimates (Figure 8). Barrier nets were used to enclose closed sites, typically averaging 100-200m^2, with the electrofishing protocol being similar to the one currently used in FNP, starting at the downstream end of the site and moving upstream (CBHNP, unpub.). Collection equipment used is either dip nets or a smaller one-person collection lip seine, with a Smith-Root LR24 electrofisher, typically using 400-500 volts.

2.1.4 Kouchibouguac National Park

KNP had the rivers with the longest ongoing electrofishing time series starting in 1974, with a large gap from the mid-1980s to late 1990s (Figure 8). Most sites were sampled using the single-pass method, with only 25% of the total number of sites sampled using the multiple pass method. Contrary to other parks, data was collected by a number of organizations often in conjunction with KNP (*e.g.*, Fisheries and Oceans Canada, Friends of the Kouchibouguacis). Because of this, protocols and data collection can vary. When KNP had conducted the sampling, typically an electrofisher operator and 2-3 dip netters were used with a Smith-Root APEX electrofisher. Similar to GMNP, the "quick set up" feature was used to set electrofisher settings. Barrier nets were used for the closed sites, targeting a 200-300m^2 area, where the site was then fished from downstream moving upstream. Sampling is stopped if water temperature reached >21°C (KNP, Pers. Comm.).

2.1.5 Terra Nova National Park

In TNNP, monitoring began in 2002 and stopped in 2012 (Figure 8). Sites were sampled using the multiple pass methodology exclusively, in multiple rivers, often meaning only 1-2 sites per river were completed (Figure 8). A Smith-Root LR24 electrofisher was used with a team of 3-4 people, with often only 2-3 people to complete the electrofishing, fishing in the downstream direction, while 1-2 people conducted the onshore sampling simultaneously after the first pass. Sites were established by marking one end with flagging tape and measuring 50 m in length, where the site was enclosed using barrier nets. Sampling was stopped if the water temperature reached 19ºC (TNNP, unpub.).

2.2 Modelling Framework

The two most common approaches for statistical inference are frequentist and Bayesian approaches. Frequentist statistics assumes population parameters are fixed and the sample is random with estimations derived from the probability of observing the sample data (*i.e.*, solely influenced by the data itself), whereas Bayesian statistics assumes the parameters are random variables and incorporates prior knowledge (*i.e.*, background or previous knowledge about a parameter) in form of a likelihood function to determine a posterior distribution, which are updated based on observational data (Gelman et al., 2015; Smid et al., 2020; van de Schoot et al., 2021). With the development of technology leading to higher computational power and statistical ability, popularity in using a Bayesian framework has grown. These models can be computationally and time intensive for programming and model fitting, however, they are extremely powerful because of the flexibility and elements such as estimating uncertainty, variance incorporated within estimates, the use of prior information, strength borrowing, etc. that are included in Bayesian modelling (Clark, 2005; Ellison, 2004; van de Schoot et al., 2021).

When conducting electrofishing surveys using multi-pass successive removal methods, several assumptions do apply depending on the modelling method. These assumptions include: 1) the population remains stationary within the site, meaning no immigration or emigration occurs through closed site boundaries; 2) capture probability remains constant from pass to pass; and 3) capture probability remains fixed and equal from individual to individual (Carle and Strub, 1978; Seber and Whale, 1970; Zippin, 1958). The first assumption can often be met using barrier or block nets to prevent movement, which is important to avoid potential avoidance behavior if left unused (Peterson et al., 2005). However, the other assumptions can rarely be met, for example, numerous studies have

shown capture probability typically decreases through consecutive passes and varies based on size of some individuals, not remaining constant (Glover et al., 2018; Malcolm et al., 2019; Millar et al., 2016; Peterson et al., 2004). The assumption of depleting or regressing captures through consecutive passes in multi-pass successive removal methods (Zippin, 1958), cannot be met if capture probability increases with successive passes or new individuals continue to enter the site. While a well-defined depletion is ideal, smaller population numbers can make this difficult to achieve, often creating greater uncertainty or bias (Korman et al., 2009; Riley and Fausch, 1992; Ruetz et al., 2015), which can often be observed when working on depleted and endangered populations, requiring greater attention or more complex estimation techniques. Bayesian modelling is ideal for these applications as a hierarchical structure can be implemented to these models where strength can be borrowed from data-rich sites and applied to those lacking information, improving estimates where data is missing (Dauphin et al., 2009). Doing so can result in more accurate and precise estimates in the indices of abundance when modelling capture probability and abundance together (Glover et al., 2018; Millar et al., 2016).

The review of the electrofishing data available for each park identified a range of inconsistencies between and/or within parks, which included differences in sampling protocols and types of data recorded (Table 3). Additionally, the sampling effort (*i.e.*, the number of sites sampled in a given year) varied from park to park and year to year and was generally low, with all rivers having less than ten electrofishing sites per year and the majority of them with less than five sites per year (with except of USR in 2017; Figure 8). It is important to keep in mind these small sample sizes when reviewing the output of any model using this dataset. With the number of uncertainties in the dataset, Bayesian

inference has many advantages of incorporating and reflecting this, particularly in ecological datasets (Ellison, 2004).

2.2.1 Model Description

To account for the discrepancies in protocols and data availability, a hierarchical Bayesian model was developed to estimate Atlantic salmon juvenile densities at the sampling site and river scale (Figure 9). Due to data availability across time and parks (Table 3, Figure 8), the only covariates available to include in the model, which affect capture probability, were effort (time/m^2 spent electrofishing), day of year, life stage, and park (an overall metric reflecting non exhaustive potential differences in methodology, crew, habitat at the park scale) (Table 4, Table 5). The model was developed following the approach described in Craigmile et al., (2009) where hypotheses are tested iteratively, only retaining the models consistent with the data (Figure 9).

To address the first objective of providing standardized density estimates at the site scale, a component in the model was created to address the best-case scenario: multi-pass closed sites with effort data available (section 2.2.1.1.1a). Since effort data was not available across all rivers in some years, particularly in the older part of the time-series (Table 3), a similar component was created including all covariates except the effort (section 2.2.1.1.1b).

Additionally, several parks conducted single-pass open sites sampling (Table 2, Figure 8), requiring another model (section 2.2.1.1.2). Typically, to use single-pass catch data as a temporally consistent index of abundance, it has to be calibrated against a measure of total abundance (Dauphin et al., 2019). In absence of this, single-pass sites were treated similar as multi-pass closed sites, but having only the first pass conducted. Considering

that the methodology used for the open sites is similar to the first pass of a multi-pass closed site, this was considered the best approximation. This is done by sharing all parameters and variance described by and estimated within the closed sites and applying them to open sites after the first pass. Again, because some sites lack available effort data, the effort covariate had to be dropped to retain all available sites. A summary of all indices used in the following sections can be found in Table 5.

Table 4. Parameters and their prior distributions used.

Parameter	Distribution	Description
$\mu^{Z\,p}$	Normal(0,0.01)	Mean capture probability of a given pass
$\mu^{d}_{t,k,l}$	Normal(0,0.01)	Average annual density
σ^{p}	Uniform(0,10)	Capture probability variance
σ^{d}	Uniform(0,10)	Inter-site density variance
α^{E}	Normal(0,0.01)	Effort covariate
α^{DoY}	Normal(0,0.01)	Day of year covariate
α^{LS}	Normal(0,0.01)	Life stage covariate
α^{Park}	Normal(0,0.01)	Park covariate

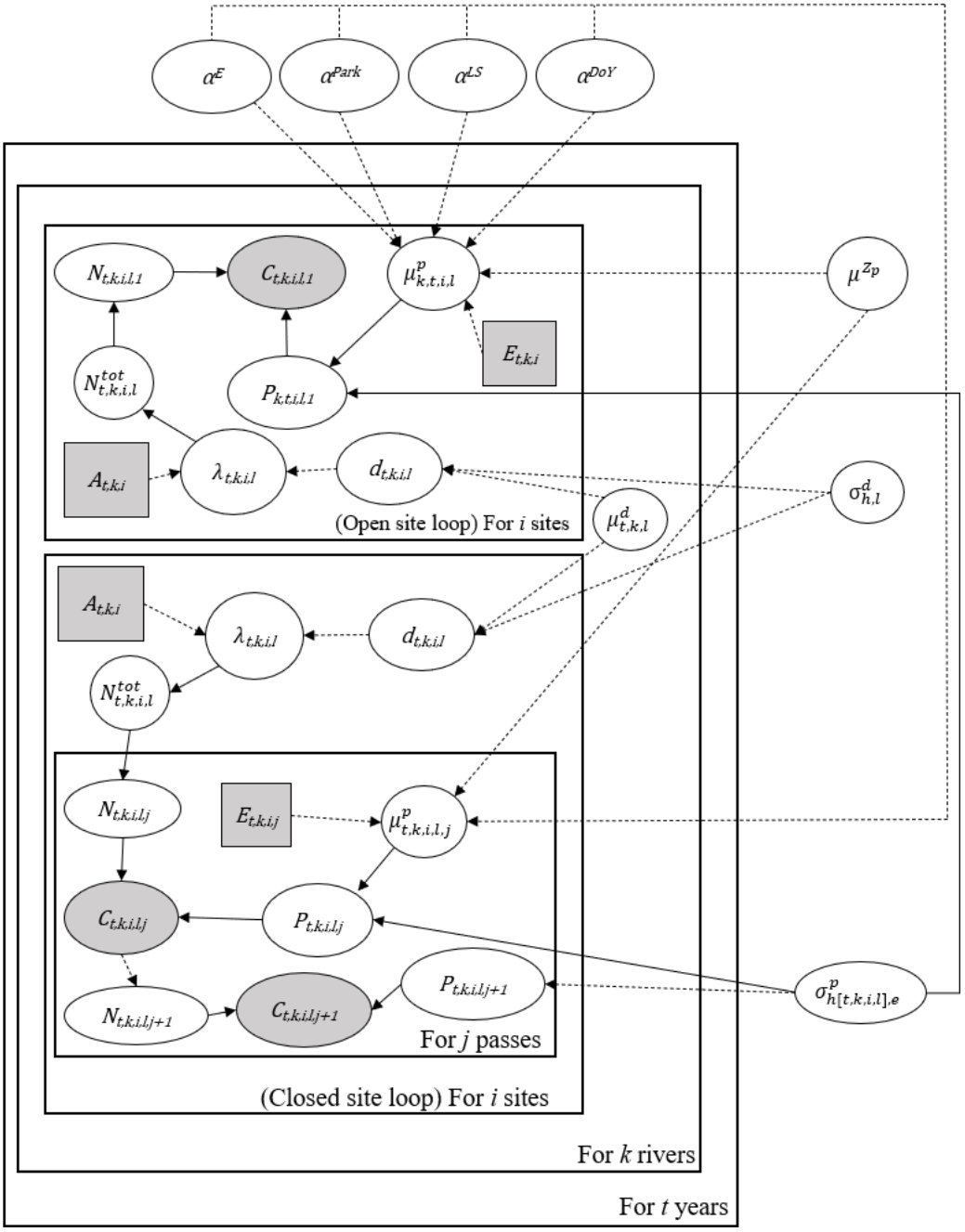

Figure 9. Directed Acyclic Graph (DAG) of the hierarchical bayesian model used to estimate juvenile salmon abundance using electrofishing data. Circular nodes represent random variables, whereas squares represent fixed quantities. Any nodes shaded in gray are observed quantities. Arrows represent dependencies between the different nodes with solid arrows showing probabilistic relationships and dashed arrows as deterministic relationships. The surrounding frames represent repetition which correspond to a loop within the model of multiple passes, sites, rivers and years being run several times.

Table 5. Summary of indices used in Bayesian model.

Index	Description	Values/Levels
h	Park	5
i	Site	536
j	Successive removal pass	Up to 6
k	River	24
l	Life stage	2 (fry and parr)
t	Year	49

2.2.1.1 Local site densities

For any given year *t* and river *k*, the total abundance of an Atlantic salmon at life stage *l* in site *i* is drawn from a Poisson distribution, as this distribution aligns well with the characteristics of count data (*e.g.*, fish counted), commonly used in other studies (Dauphin et al., 2009; Dauphin et al., 2019; Hanks et al., 2018; Malcolm et al., 2019; Parent and Rivot, 2012; Wyatt, 2002).

Equation 1. $N^{tot}_{t,k,i,l} \left| \lambda^{N}_{t,k,i,l} \right. \sim \text{Poisson}(\lambda^{N}_{t,k,i,l})$

where $\lambda_{t,k,i,l}$ is the product of the density, $d_{t,k,i,l}$ (in fish of life stage *l* per m^2), multiplied by the area, $A_{t,k,i}$, of the sites (in m^2). Densities are described in further detail in section 2.2.1.2.

Equation 2. $\lambda_{t,k,i,l} = d_{t,k,i,l} \times A_{t,k,i}$

2.2.1.1.1 Multiple-pass sites

In the following section, and for conciseness, the indices *t,k,i,l* are replaced by *I* (*e.g.*, $N_{t,k,i,l}^{tot}$ becomes N_I^{tot}).

The number of fish caught after the first pass $C_{I,1}$ is assumed to follow a Binomial process with a probability of capture $p_{I,1}$. A Binomial distribution is used as it reflects the binary nature of capture events (captured or not) and is commonly used in other abundance models (Dauphin et al., 2009; Dauphin et al., 2019; Hanks et al., 2018; Parent and Rivot, 2012; Wyatt, 2002).

Equation 3. $$C_{I,1}|p_{I,1}, N_I^{tot} \sim Binomial(p_{I,1}, N_I^{tot})$$

Fish caught during the consecutive 2:*n* pass (where *n* is the total number of passes) also follow a binomial process:

Equation 4. $$C_{I,2:n}|p_{I,2:n}, N_{I.1:(n-1)} \sim Binomial(p_{I,1}, N_{I.1:(n-1)})$$

Where $N_{I,1:(n-1)}$is the number of fish left in the site after the previous pass such that:

$N_{I,1} = N_I^{tot} - C_{I,1}$ and $N_{I,2:(n-1)} = N_{I,1:(n-2)} - C_{I,(n-2)}$

The probabilities of capture are modeled differently depending on the availability of effort data.

2.2.1.1.1a with Effort

When effort data is available, the probabilities of capture $p_{I,n}$ (logit scale) are drawn from a normal distribution.

Equation 5. $Logit(p_{I,n})|\mu^{p}_{I,n}, \sigma^{p}_{h[I],e[I]=1}$ ~ Normal $(\mu^{p}_{I,n}, \sigma^{p}_{h[I],e[I]=1})$

Equation 6. $\mu^{p}_{t,i,l,j} = \mu^{Z_p} + \alpha^{LS_e} + \alpha^{Park_e} + \alpha^{JDay_e} + \alpha^{E}$ x $E_{t,k,i,j}/A_{t,k,i}$

Variance in the probability of capture, σ^{p}, is park and sampling effort availability specific, $h[I], e[I]$. These variables are used as to incorporate the possible variance differences across parks caused from their differences in protocols and methodology of fishing, equipment differences, crew and organizations conducting the survey, etc. Ideally, this data would be available across all parks and time series to include as covariates, however in absence of such, adding a park covariate is the best approach. Because there were notable changes in field methodology in FNP, a variance coefficient was estimated for each time series, whereby they were treated as separate parks (FNP A and FNP B). As effort is known to affect capture probability (Dauphin et al., 2019), it is important to include when available, and because different models were created when effort data does not exist, different capture probability parameters need to be used.

Size of fish can affect capture probabilities with larger juveniles captured at high probabilities. Where size data is not always available, day of year was used as other studies have used day of year to serve as a proxy for juvenile size due to the relationship of juveniles being larger later in the year (Dauphin et al., 2019; Glover et al., 2018; Millar et al., 2016).

2.2.1.1.1b without Effort

When effort data is not available Eq 5 and 6 become:

Equation 7. $Logit(p_{I,n})|\mu^{p}_{I,n}, \sigma^{p}_{h[I],e[I]=0} \sim$ Normal $(\mu^{p}_{I,n}, \sigma^{p}_{h[I],e[I]=0})$

Equation 8. $\mu^{p}_{t,i,l,j} = \mu^{Z_p} + \alpha^{LS} + \alpha^{Park} + \alpha^{DoY}$

2.2.1.1.2 Single-pass Sites

Some parks/rivers have a varying number of single-pass sampling sites within their datasets (Table 2, Figure 8). Because single-pass sites are conducted in the same manner as the first removal of the multi-pass depletion sites, they are treated as such. To estimate a site density in these sites, the capture probability variance ($\sigma^{p}_{h[I]}$) and inter-site density variance ($\sigma^{d}_{h,l}$) from multi-pass sites are applied to the number of fish captured on that single-pass ($C_{t,i,l,1}$). This is effort availability dependent as well, meaning effort must be available for the multi-pass sites from the corresponding river and year, to use the associated variance parameters for single-pass sites with effort data.

For single-pass sites when effort is available in both single-pass and multi-pass sites of the same river and year, Equation 5 and Equation 6 are used, with the exception of *n* and *j* remaining as 1, indicating that only a single/first pass.

For single-pass sites when effort is not available within multi-pass sites of the same river/year, Equation 7 and Equation 8 are used, and again with *n* and *j* equaling 1.

When single-pass sites are not conducted, this single-pass model is skipped and only the multiple pass sites are used.

2.2.1.2 Annual Densities

Once site scale densities were estimated, they could then be used to estimate an index of annual average density. Site densities (log scale) were drawn from a normal distribution

of the mean annual density in log scale, $\mu^{d}_{t,k,l}$, which is river, year, and life stage specific, due to possible differences between parks and/or rivers, and has a variance of $\sigma^{d}_{h,l}$.

Equation 9. $\text{Log}(d_l) \mid \text{Log}(\mu^{d}_{t,k,l}), \sigma^{d}_{h,l} \sim \text{Normal}(\text{Log}(\mu^{d}_{t,k,l}), \sigma^{d}_{h,l})$

2.3 Robustness Testing

To assess the robustness of the single-pass model, only the FNP dataset was used due to a larger dataset covering the whole time period, and having a mixture of single- and multi-pass sites whereas other parks lacked the amount of multi-pass sites needed to accurately estimate their single-pass sites (Table 2). To assess robustness, only closed sites from the FNP dataset were used as they are the only sites which can be used to estimate density based on one or several electrofishing passes. Using the same dataset, densities were estimated under several scenarios; the baseline (Scenario 0) treated all sites as closed sites and produced the 'reference' density estimate for any given site. For the three other scenarios, a fraction of the closed sites was treated as single-pass sites (*i.e.*, only the first pass data used). Density estimates under Scenario 1-3 were then compared to density estimates obtained under Scenario 0.

i) Scenario 0: All closed sites included,
ii) Scenario 1: 10% of the 1984-2006 sites randomly turned to single-pass,
iii) Scenario 2: 10% of the 2015-2022 sites randomly turned to single-pass,
iv) Scenario 3: 10% of sites across the whole time series randomly turned to single-pass (5% from each time series).

Scenarios 2 and 3 were tested to account for FNP's methodology change between the 1984-2006 and 2015-2022 periods (and absent/presence of effort data), and assess if the methodological changes introduced systematic biases.

Results

3.1 Capture Probability

Capture probability for FNP was modelled separately for the 1984-2006 data, which did not include any effort data, whereas the 2015-2022 data was modelled with an effort coefficient included. Differences in capture probability within the first pass were evident with the 1984-2006 data showing greater fluctuations in the interquartile ranges and overall lower average capture probability compared to the 2015-present capture probabilities (Figure 10, Figure 11). Similar observations occurred with KNP and TNNP, being the only other parks with a mixture of effort and non-effort sites, showing capture probability being more inconsistent and uncertain when modelled without an effort covariate (Figure 10, Figure 11). When effort data was available FNP and TNNP had, on average, higher probabilities of captures than the other parks with CBHNP having the lowest probability of capture overall.

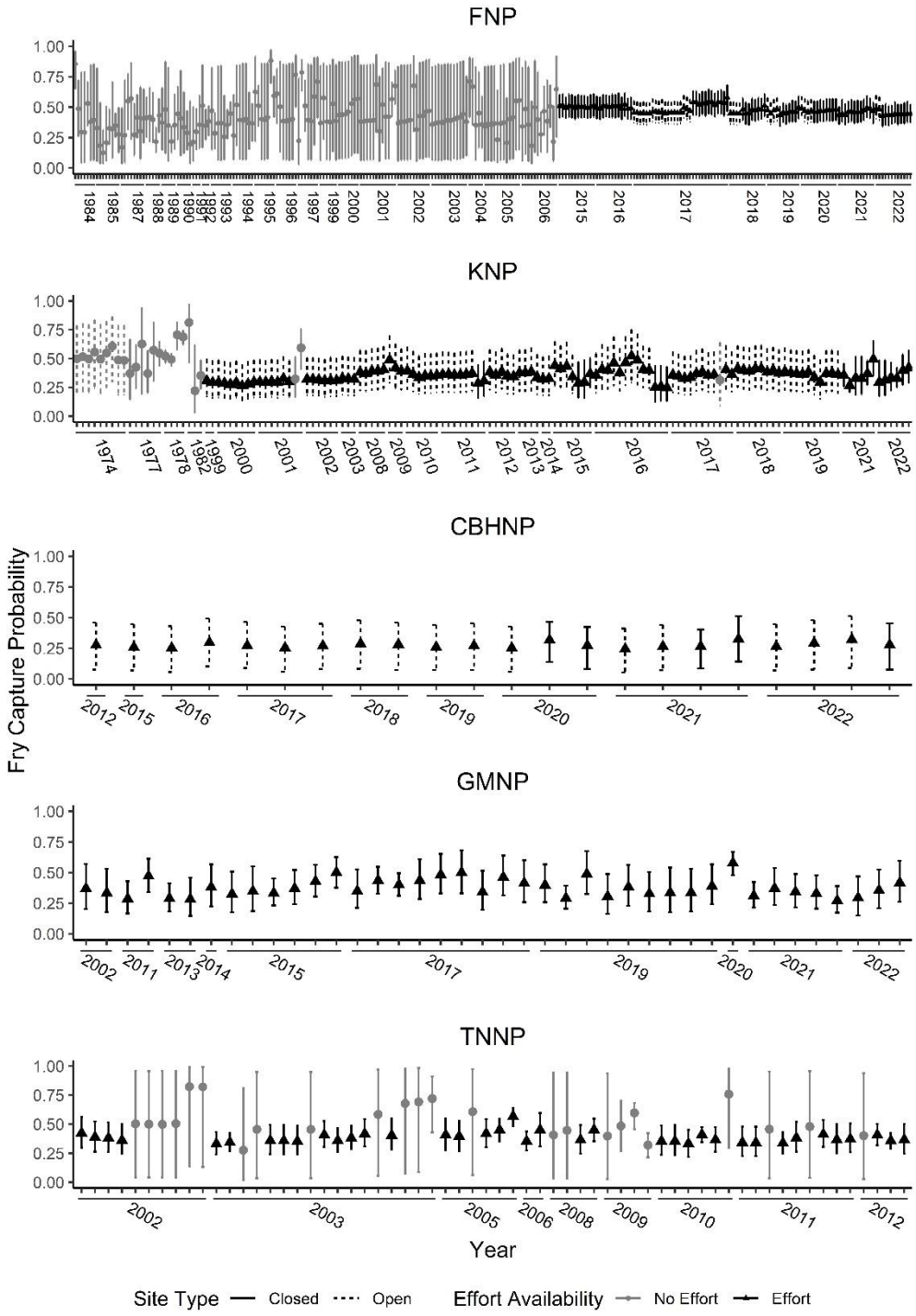

Figure 10. Summary of the posterior distributions of fry capture probability on the first pass of each site for each National Park. Points represent the medians, with gray circles representing sites with no effort recorded and black triangles sites with effort. Whiskers represent the 2.5th and 97.5th percentiles, with dashed gray lines representing open (single-pass) sites and solid black lines closed (multi-pass) sites.

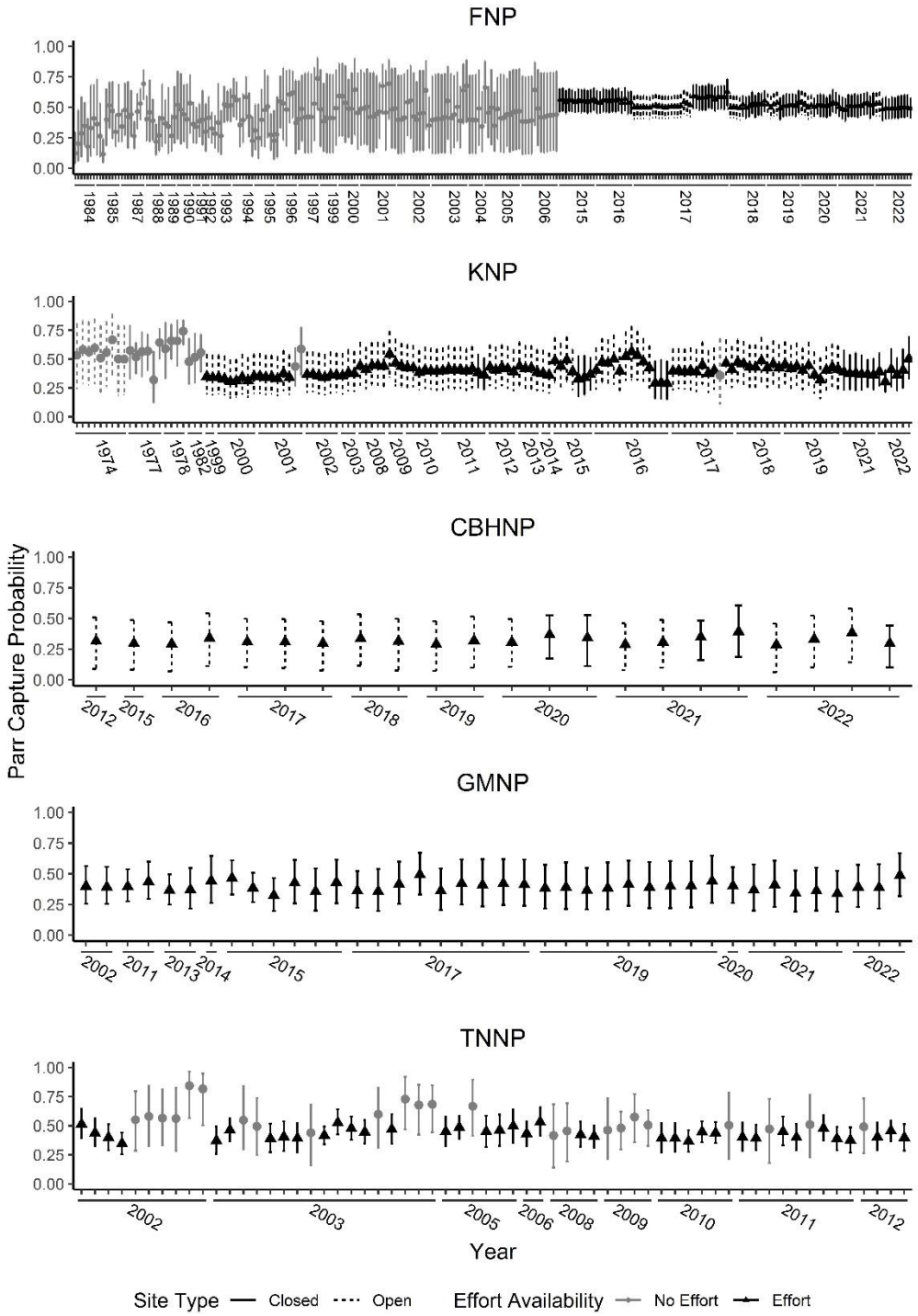

Figure 11. Summary of the posterior distributions of parr capture probability on the first pass of each site for each National Park. Points represent the medians, with gray circles representing sites with no effort recorded and black triangles sites with effort. Whiskers represent the 2.5th and 97.5th percentiles, with dashed gray lines representing open (single-pass) sites and solid black lines closed (multi-pass) sites.

These differences were also reflected in the capture probability variance parameter σ^p, as greater uncertainty was found across all sites modelled with no effort covariate (Figure 12). Specifically, the two FNP time periods having the lowest and highest variances for the FNP 2015-2022 (FNP B) and FNP 1984-2006 (FNP A) data, respectively.

For the sites where effort data was available, there was an effect of effort (α^E) on the probability of capture, meaning that as more time per square meter was spent electrofishing, a greater number of fish were captured (Figure 13). Regardless of the availability of effort data, day of year (α^{DoY}) had a positive effect on the probability of capture, meaning that the probabilities of capture were higher later in the sampling season (Figure 13). The life stage effect (α^{LS}) showed that fry had smaller probabilities of capture than parr, however this difference was more pronounced when effort data was available (Figure 13).

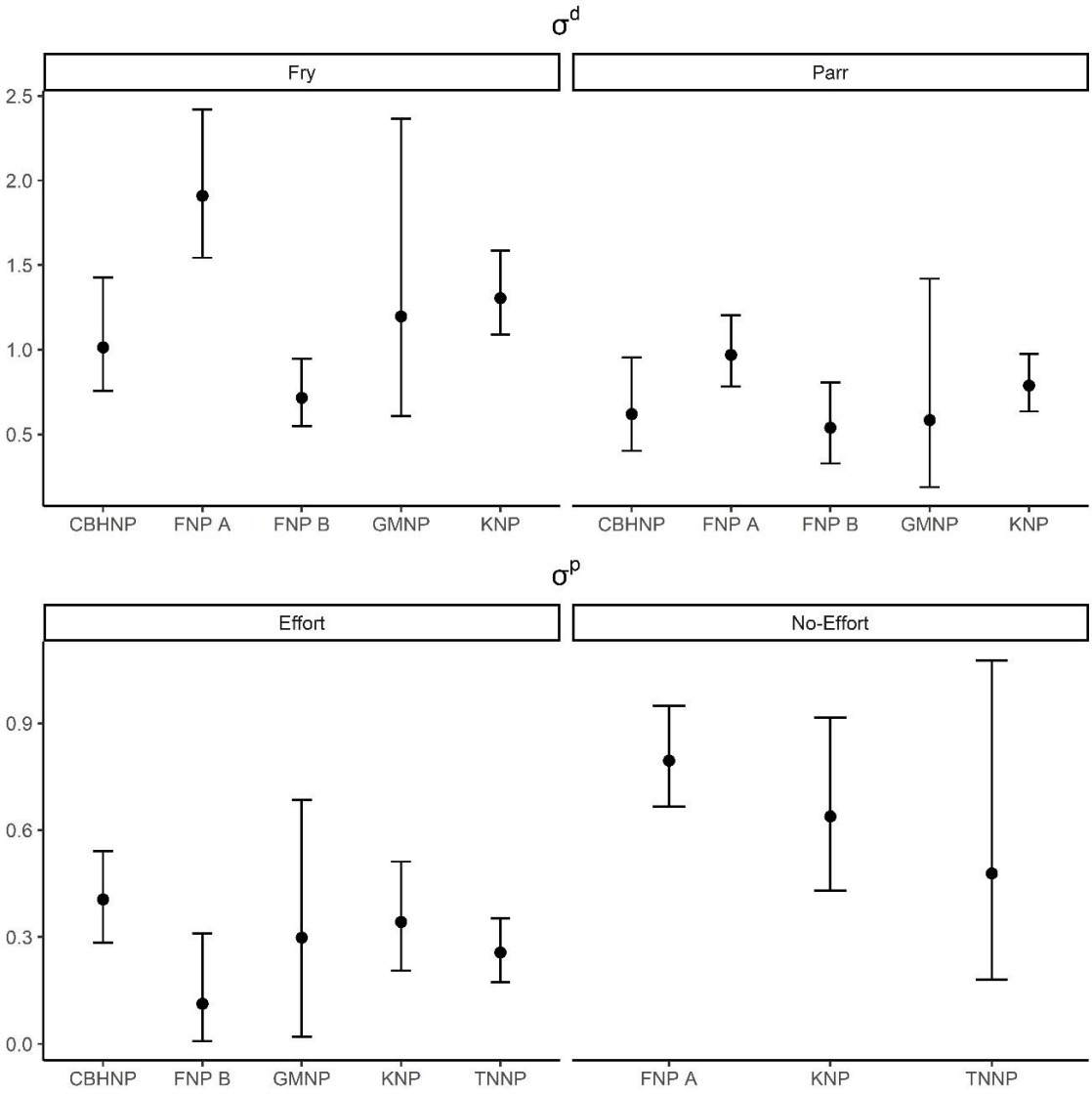

Figure 12. Posterior distributions of the variance parameters associated with density, σ^d, for the two life stages and capture probability, σ^p, for effort and no effort sites across five National Parks. FNP was separated in two time periods, being 1984-2006 shown as FNP A, and 2015-2022 shown as FNP B. There is no density variance for TNNP as no annual densities were calculated. Points represent the medians, and whiskers represent the 2.5th and 97.5th percentiles.

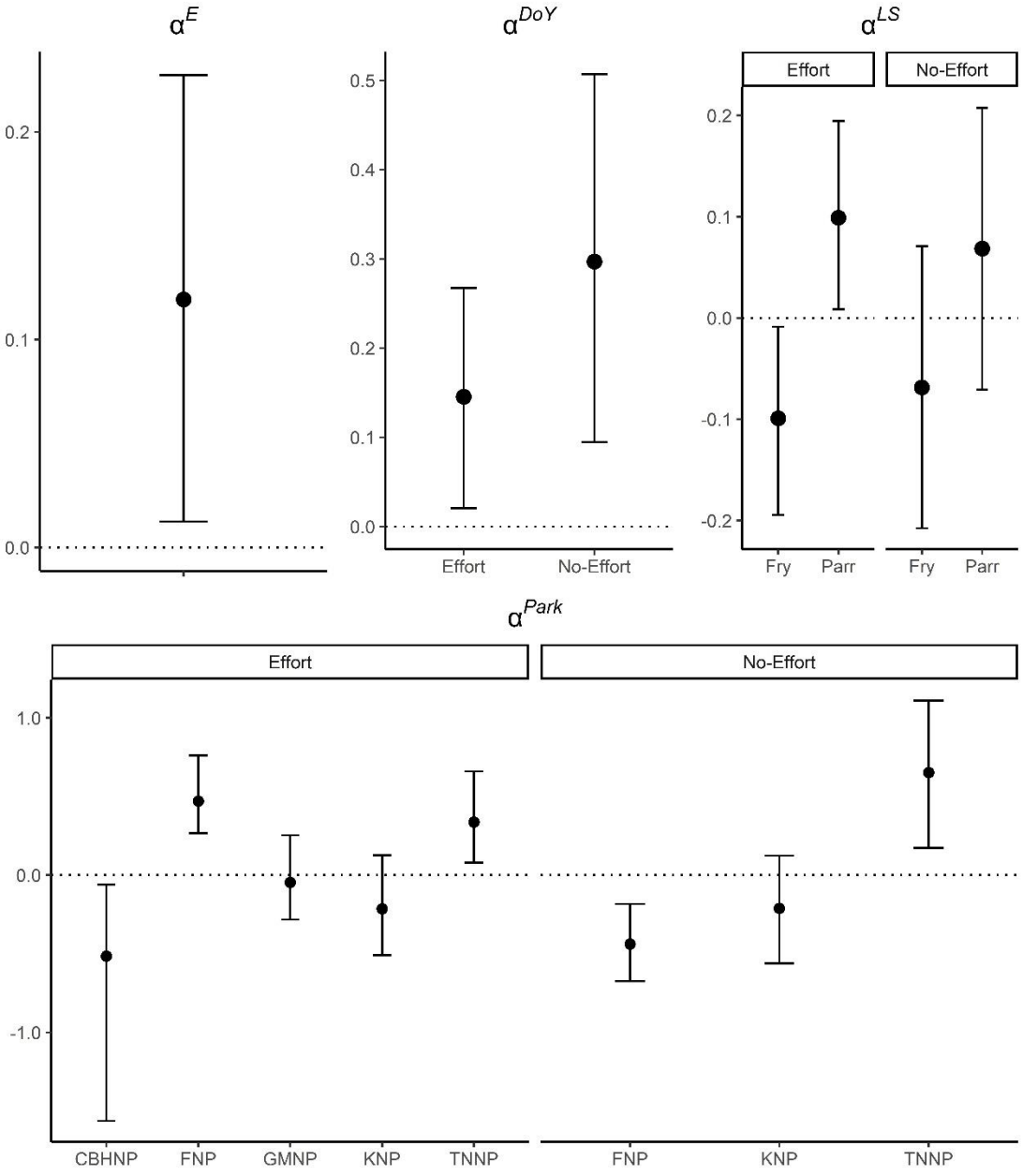

Figure 13. Posterior distributions (logit scale) of the covariate effects on capture probability associated with effort, day of year, life stage, and Park, being α^{E}, α^{DoY}, α^{LS}, and α^{Park}, respectively. Points represent the medians, and whiskers represent the 2.5th and 97.5th percentiles.

3.2 Site Density

For inter-site density variance, σ^d, the 1984-2006 FNP data (FNP A), had greater variance than any other park for both fry and parr life stages (Figure 12). GMNP had the greatest interquartile range and uncertainty amongst the parks for either life stage, while the 2015-2022 FNP data had the lowest median variance and more consistent, narrowed percentiles (FNP B) (Figure 12). Because TNNP was unable to calculate annual densities, they have no inter-site density variance distributions.

Density at site scale was estimated for each site for fry (Figure 14) and parr (Figure 15). Maximum densities, in fish per m^2 for fry and parr, respectively, in each park were 2.04 and 0.60 for TNNP, 1.88 and 0.80 for KNP, 1.08 and 0.62 for FNP, 1.01 and 0.17 for GMNP, and 0.32 and 0.17 for CBHNP. However, some of these values, like FNP (both life stages) and KNP (fry) have not been recorded since the 1980's and 1970's, respectively. While others like GMNP (fry) and CBHNP (parr) were recently recorded in the 2020's. All site density estimate values are in Appendix I.

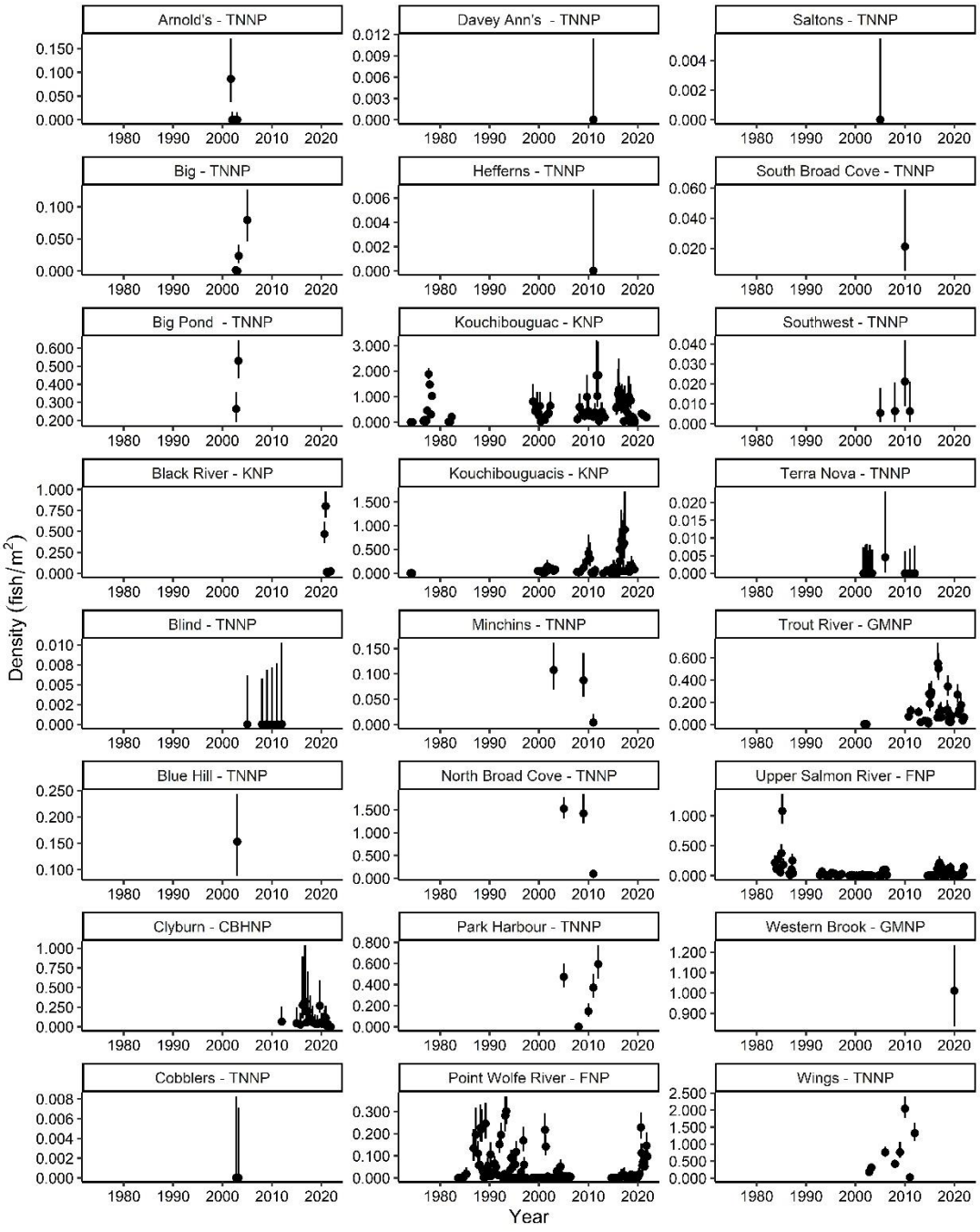

Figure 14. Summary of posterior distributions for salmon fry (0+) site density from 1974-2022 for 536 sites across 24 rivers within 5 National Parks. Circular point representing the median, and bars showing the 2.5th and 97.5th percentiles.

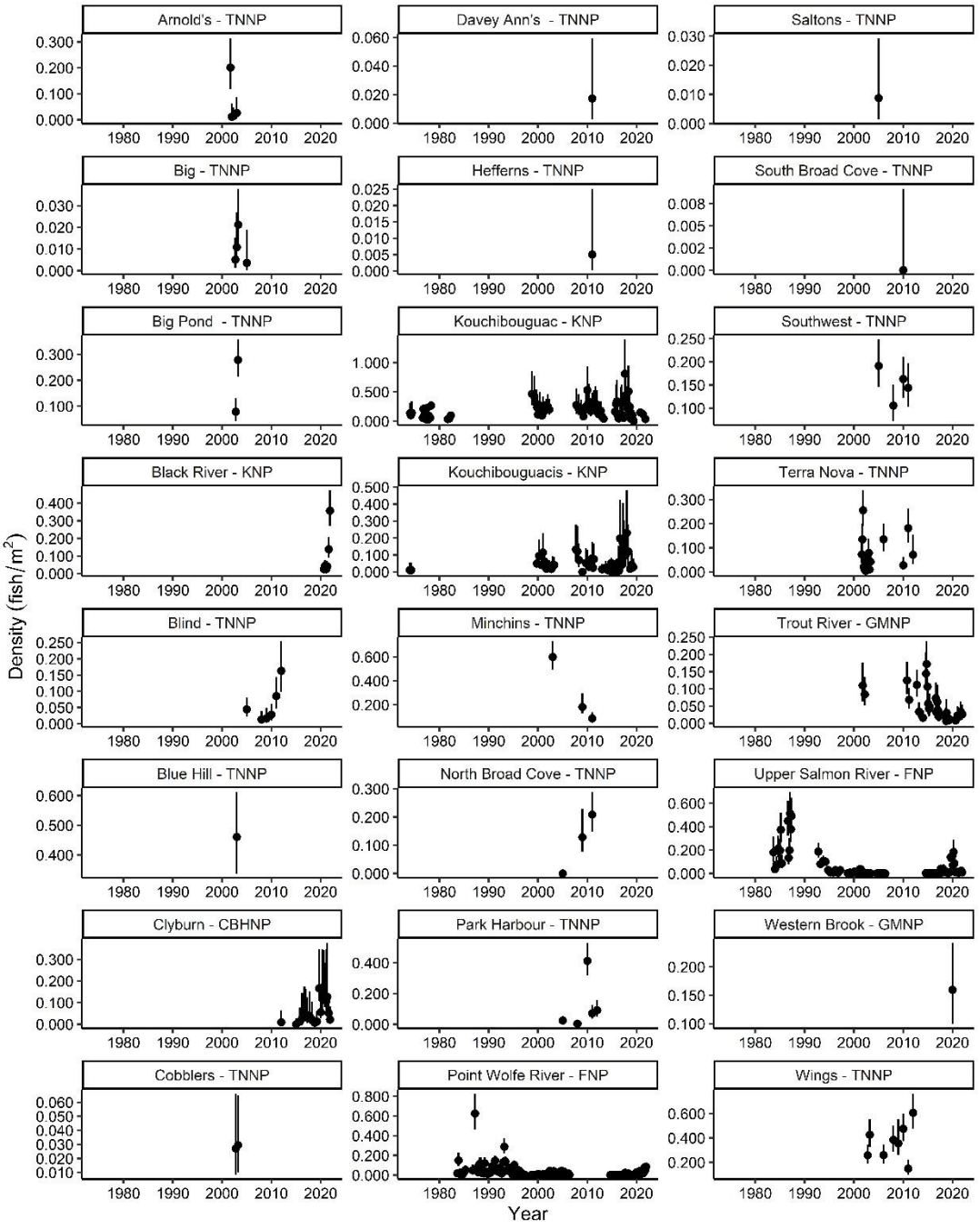

Figure 15. Summary of posterior distributions for salmon parr (1+, 2+, etc.) site density from 1974-2022 for 536 sites across 24 rivers within 5 National Parks. Circular point representing the median, and bars showing the 2.5th and 97.5th percentiles.

3.3 Robustness Testing

Robustness testing was completed to assess how robust the model was in predicting densities from single-pass data. The closed site dataset from FNP was used, where densities were estimated under several scenarios; Scenario 0, keeping all sites (1984-2022) closed as the baseline, Scenario 1 treating 10% of sites from the earlier time series (1984-2006) as single-pass, which does not have effort data available, Scenario 2, following the same method but with 10% of sites from the later time series (2015-2022),which includes effort data, and Scenario 3 treating 10% of sites as single-pass throughout the entire series (5% from each time series). After running each model (or scenario) for 800,000 iterations, burning the first 100,000 and thinning every 1 in 140 values, for 5000 total sample values, no evident differences were observed in the posterior distributions of the annual densities when comparing Scenarios 1-3, which converted 10% of their sites to single-pass, to the Scenario 0 baseline multi-pass densities for either the USR or PWR for either life stage (Figure 16).

Looking at specific individual sites converted to single-pass during Scenarios 1-3, relative to the Scenario 0 multi-pass sites (baseline), the points representing the medians, remained relatively similar, as well as the interquartile intervals, representing uncertainty. The uncertainty was much lower in sites with lower densities, and exhibited increased uncertainty as densities increased, particularly evident in densities estimated through Scenario 1 (Figure 17). Under Scenario 1 (10% of sites sampled from 1984-2006, treated as single-pass sites) density estimates displayed weaker correlations compared to their multi-pass counterparts than under Scenario 2 (10% of sites sampled 2015-2022, treated as single-pass), 0.51 vs. 1.0, and 0.45 vs. 0.81 for fry and parr, respectively. Under

Scenario 3, which converted multi-pass sites to single-pass from both the 1984-2006 (non-effort sites) and 2015-2022 (effort sites) time series, both fry and parr had median estimates that varied greatest from the multi-pass site densities (Figure 17, Figure 18). However, Scenario 3 had a higher correlation amongst its single-pass estimates when compared to multi-pass than Scenario 1.

Overall, robustness testing showed that Scenario 1, converting single-pass sites from no-effort sites only, produced estimates with the greatest uncertainty and variability, as well as the lowest R^2 value. For each life stage, Scenario 2, converting sites from effort sites only, produced the most accurate and precise estimates when compared to multi-pass sites and the highest R^2 values (Figure 18). This indicates that treating the single-pass (open) sites like the first pass of a multi-pass (closed) site was not as reliable for the first data series, when the effort covariate was not used, as it was for the second time series, when the effort covariate was used.

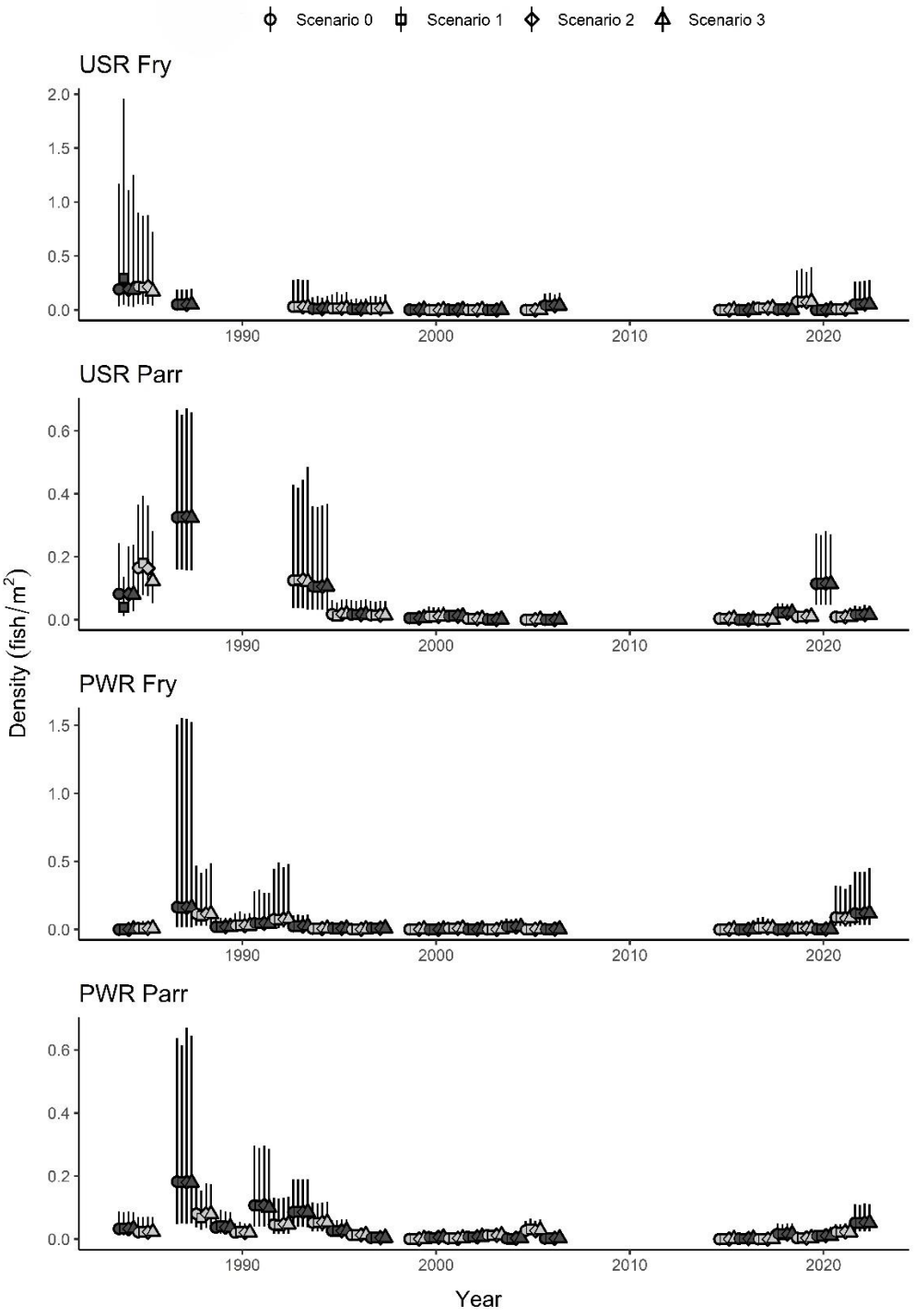

Figure 16. Comparison of posterior distributions from robustness testing of fry and parr annual densities in the Upper Salmon and Point Wolfe rivers from 1984-2022. Points represent the medians, with shapes corresponding to the respective scenario. Whiskers represent the 2.5th and 97.5th percentiles. Colours alternate to differentiate each year easily.

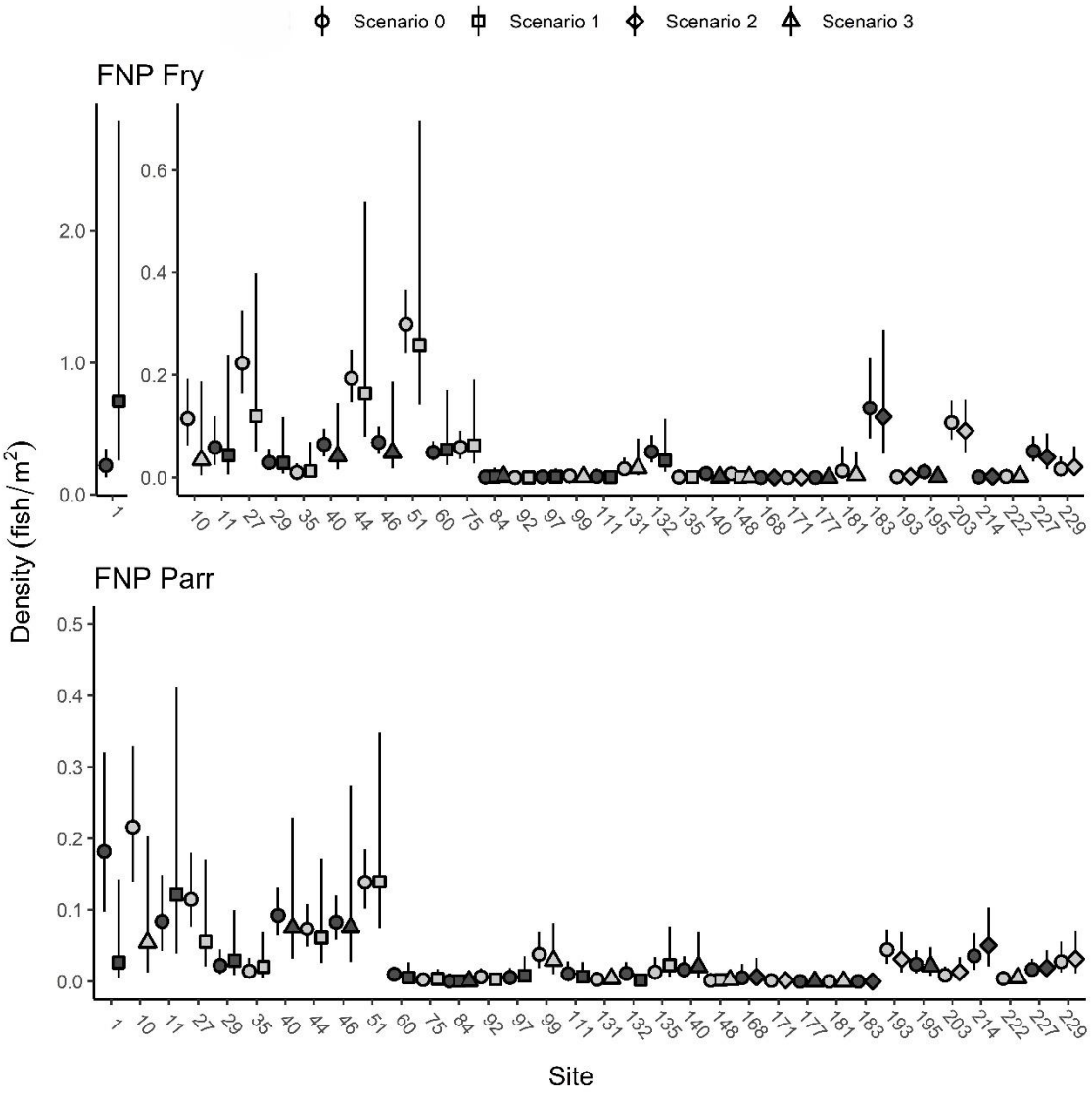

Figure 17. Comparison of posterior distributions from robustness testing of fry and parr site densities of all sites turned single-pass in Scenario 1, Scenario 2, or Scenario 3 (square, diamond, triangle, respectively), compared to Scenario 0 (circles). Points represent the medians, with whiskers representing the 2.5th and 97.5th percentiles. Colours alternate by site for easier differentiation.

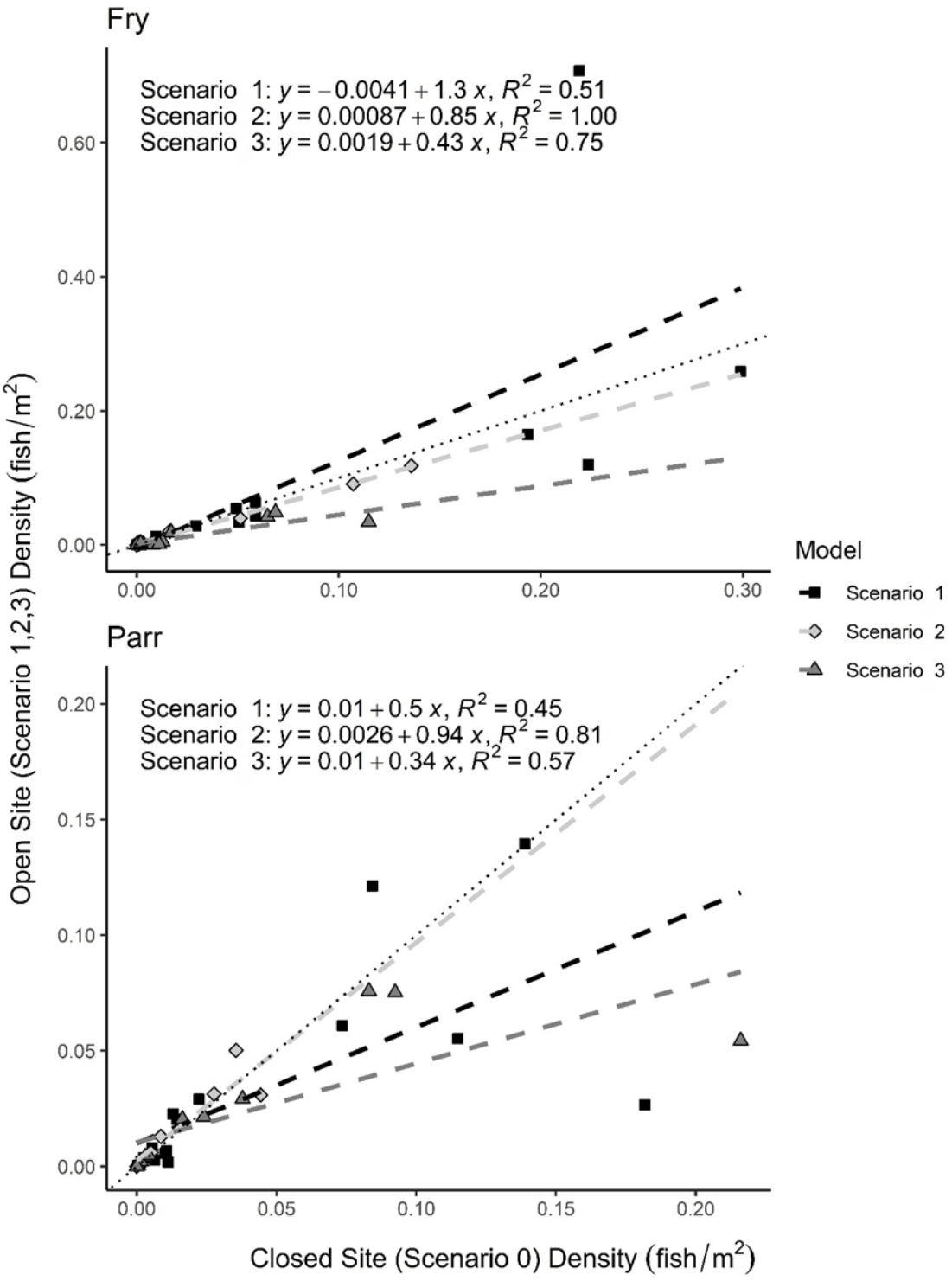

Figure 18. QQ-plot of posterior distributions from robustness testing of fry and parr site densities of all sites turned single-pass in Scenario 1, Scenario 2, or Scenario 3 compared to their multi-pass density (Scenario 0). Points represent the medians, with shape and colour of the points representing model type. Colour also corresponds the dashed linear trendline for each model, while the dotted black line marks the 1:1.

3.4 Annual Density

Attempts were made to achieve annual density estimates in TNNP, however due to the vast number of different rivers being sampled each year and often only one site surveyed per year, accurate annual estimations were not possible. Annual salmon fry density varied across river and parks with medians ranging from 0-0.027 in CBHNP, 0-0.064 in FNP, 0-0.13 in GMNP, and 0-0.261 in KNP (Figure 19). Similarly for Parr, density estimations ranged from 0-0.054 in CBHNP, 0-0.146 in FNP, 0.003-0.13 in GMNP, and 0-0.192 in KNP (Figure 20). Uncertainty in estimates tended to increase as density increased or when the amount of sites sampled annually decreased, the most being observed in Western Brook of GMNP, representing only a single-site completed within the river, with estimations possible using the variance data associated with Trout River. All annual density estimates are in Appendix II.

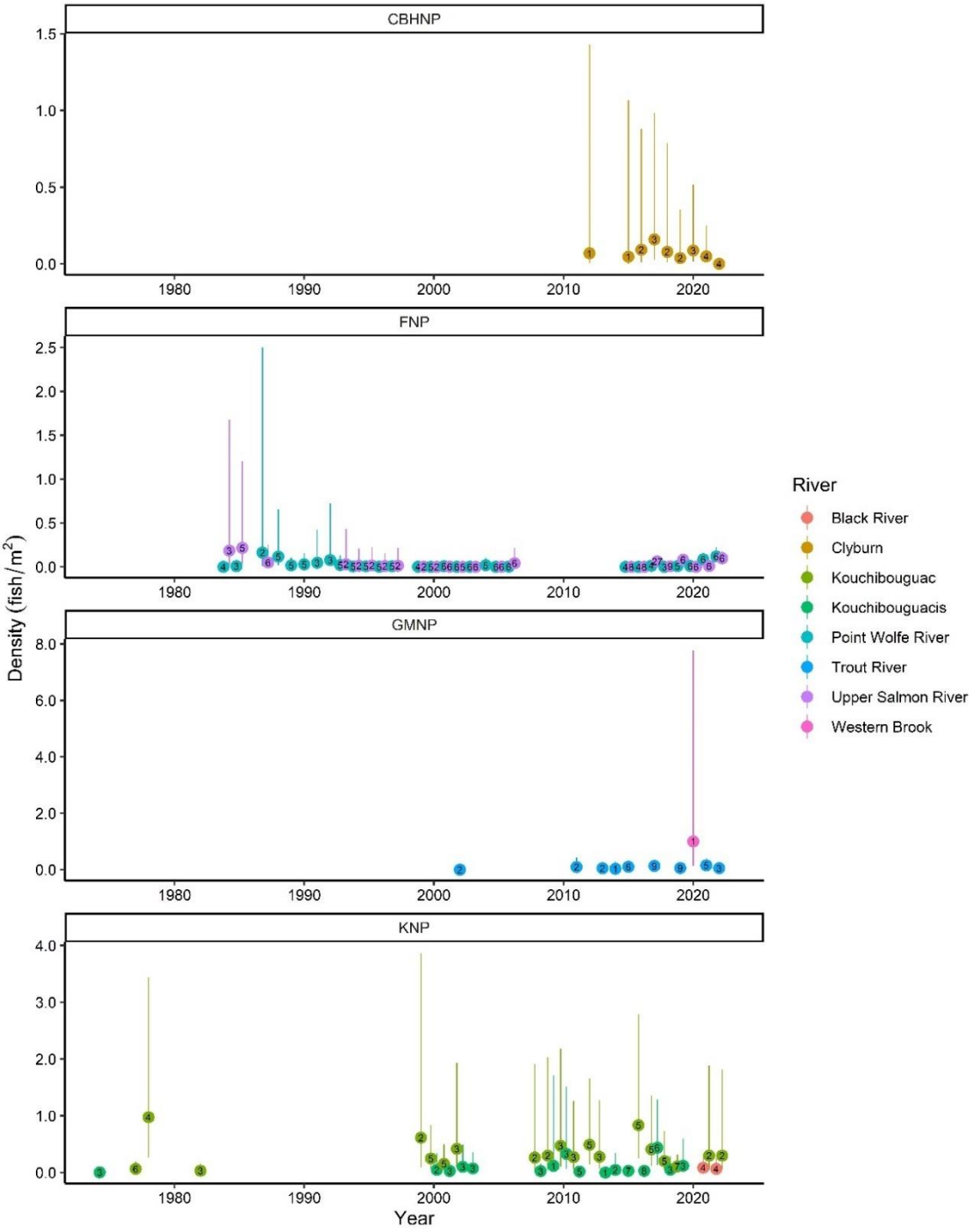

Figure 19. Summary of posterior distributions for salmon fry (0+) annual density from 1974-2022 for 8 rivers within 4 National Parks. Colours specify each river with the circular point representing the median, and bars showing the 2.5th and 97.5th percentiles. Numbers within each point correspond to the number of sites surveyed that year.

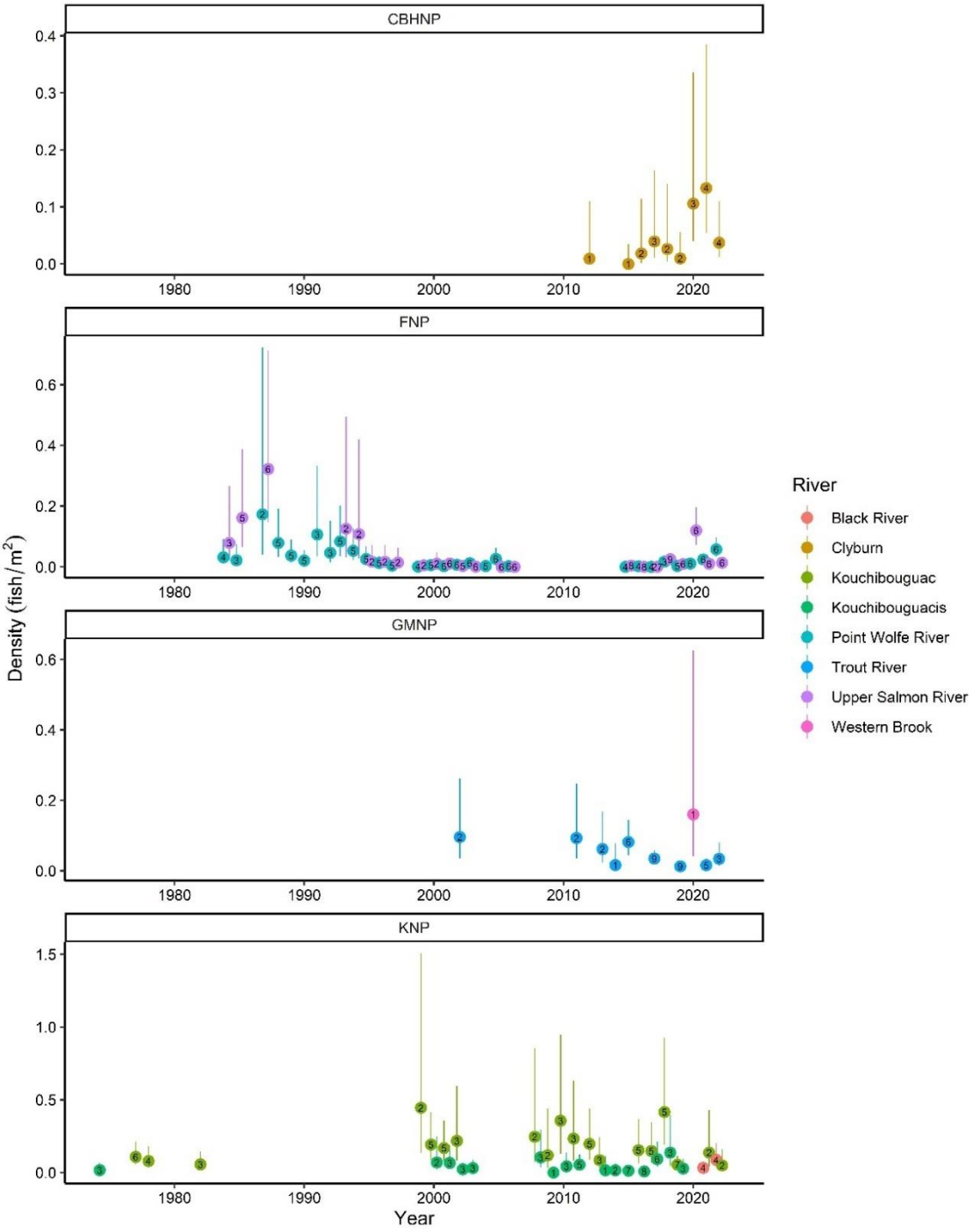

Figure 20. Summary of posterior distributions for salmon parr annual density from 1974-2022 for 8 rivers within 4 National Parks. Colours specify each river with the circular point representing the median, and bars showing the 2.5th and 97.5th percentiles. Numbers within each point correspond to the number of sites surveyed that year.

When the variance and uncertainty of estimates were removed, keeping only the average annual median values and applying a polynomial trend line to the 2nd order, unique trends were observed showing how FNP populations of the Upper Salmon and Point Wolfe rivers declined through the 1980's and 1990's, then have been increasing over the past 5-10 years as recovery measures were implemented (Figure 21, Figure 22). In KNP, the Kouchibouguacis River is showing stable to increasing trends, whereas the Kouchibouguac River has exhibited a declining trend in their fry over the past 10 years, while parr appear to have peaked in abundance nearly 20 years ago. Trout River in GMNP shows interesting trends where the fry population appears stable at lower densities but the parr population is showing steady declines. Finally, Clyburn Brook in CBHNP shows the most dramatic population crash within recent years in their fry populations (Figure 21, Figure 22).

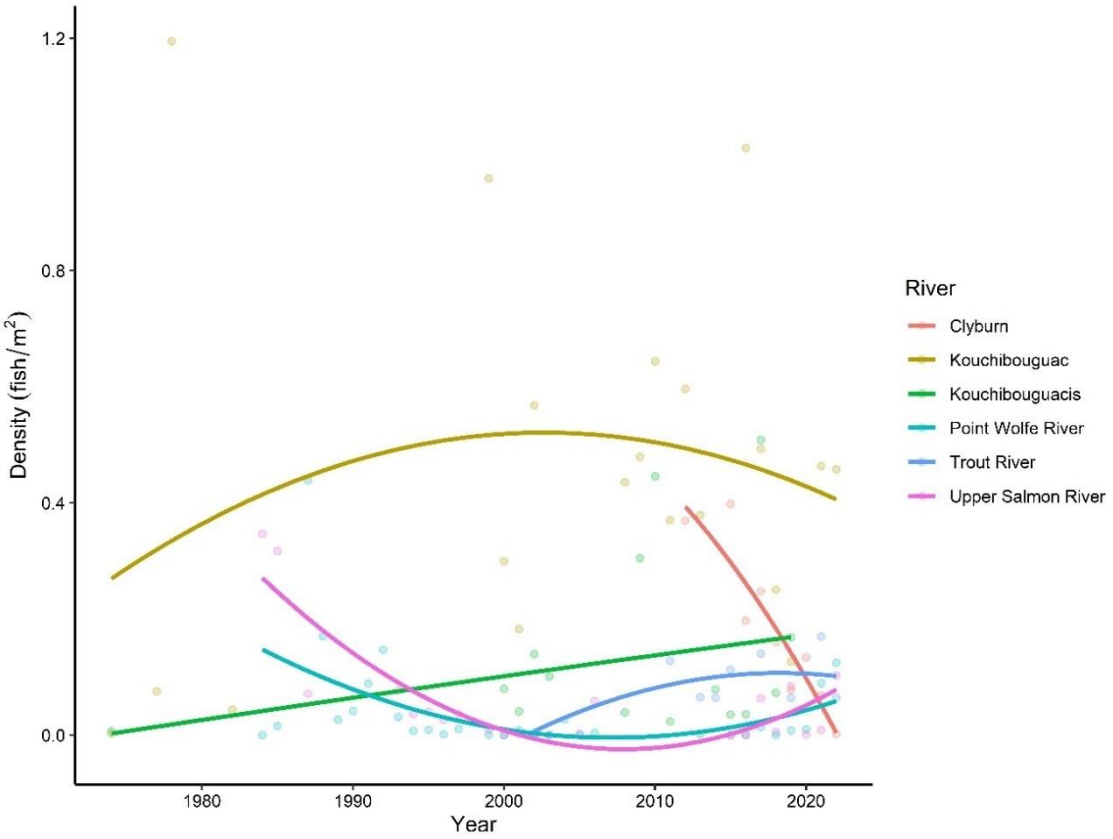

Figure 21. Summary of posterior distributions for salmon fry (0+) annual density from 1974-2022 for 6 rivers within 4 National Parks. Colours specify each river with the circular point representing the median, and polynomial trendlines showing best fit to the 2nd order.

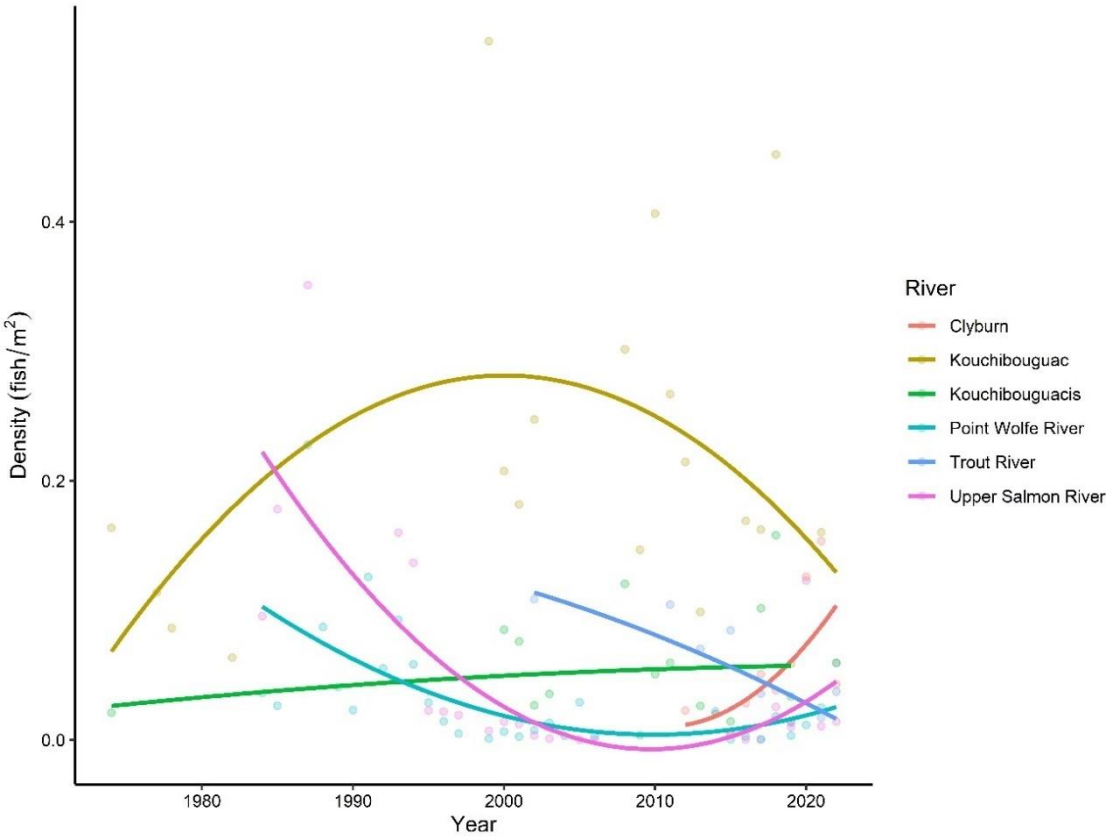

Figure 22. Summary of posterior distributions for salmon parr annual density from 1974-2022 for 6 rivers within 4 National Parks. Colours specify each river with the circular point representing the median, and polynomial trendlines showing best fit to the 2nd order.

3.5 Restoration Effectiveness

To assess how restoration/recovery efforts in each of the parks has impacted salmon abundances, adult abundance and fry density were plotted with linear trend lines representing the pre-restoration and current/ongoing restoration phases. In KNP from 1974–2017 in the Kouchibouguac River, fry density showed fluctuations but appeared relatively stable, even possibly increasing, with a sharp decline beginning in 2016 (Figure 23). Starting in 2018, egg incubation efforts were implemented, leading to an apparent increase in fry abundance, however, the concurrent adult population trends remain unclear, and it is unknown whether the increase in fry density aligns with a growing spawning population during this period or the result of egg planting efforts. In FNP, from 1976 to 2014 in the Upper Salmon River, both adult and fry populations experienced sharp declines. A notable shift occurred in 2015 with the introduction of marine-reared adults. Since then, both adult and fry populations have shown consistent growth (Figure 24). Additionally, in the Point Wolfe River, releases of marine-reared adults began in 2020 and have also resulted in increased adult and fry populations (Figure 25). In CBHNP, Clyburn Brook adult populations declined from 1985 to 2020, while electrofishing efforts showed fry density remained relatively stable but low between 2012 and 2020. Following the release of freshwater-reared adults in 2021, there was a significant return of adults in 2022 (Figure 26). However, fry density remained low. With this being a single data point, further monitoring is required to assess long-term trends and the 2022 adult spawning class. In GMNP, both adult and fry populations of Trout River have remained steadily low. However, community engagement efforts initiated post-COVID have led to an increase in adult populations in 2022, although fry densities

have not changed significantly (Figure 27). As with CBHNP, these trends are based on limited data, with only a single data point following their restoration efforts.

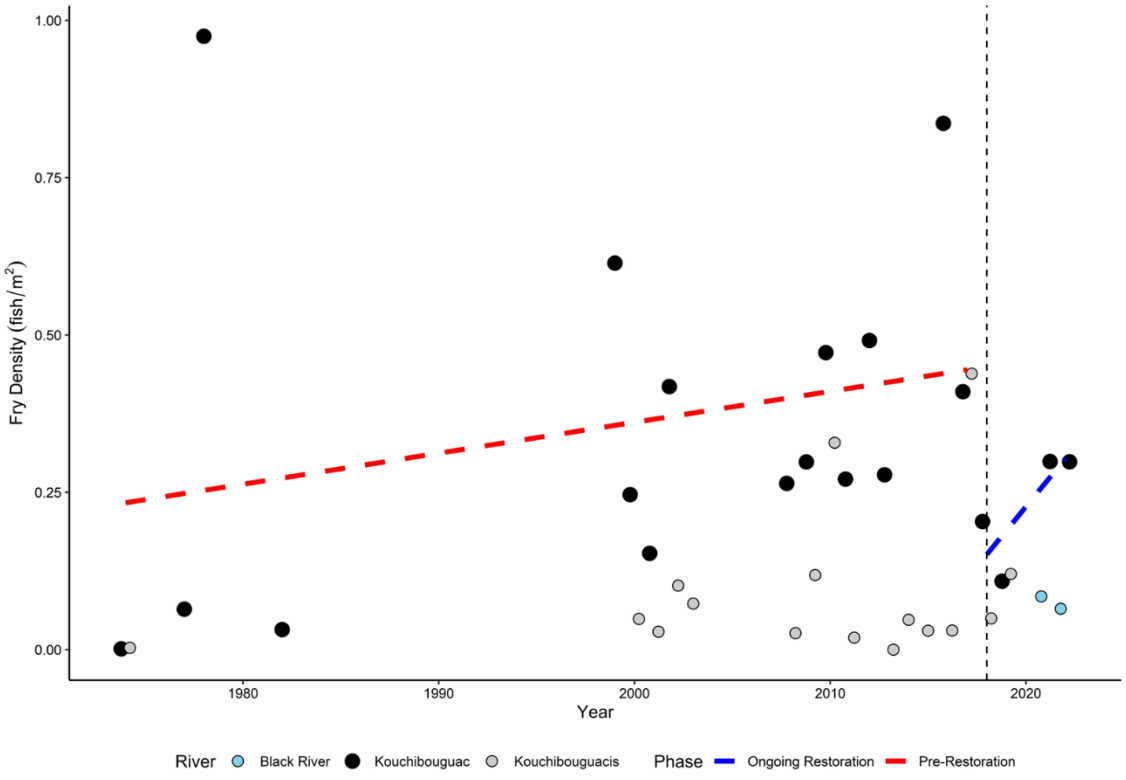

Figure 23. Population trends of the posterior distributions for salmon fry average annual density from 1974-2022 within the Kouchibouguac River, Kouchibouguac National Park, before and during restoration efforts. Circular points represent the median densities, with colour representing river. Dashed trend lines represent the linear trend of densities, coloured in red for pre-restoration and blue for ongoing restoration. The vertical black dashed line represents the start of restoration efforts.

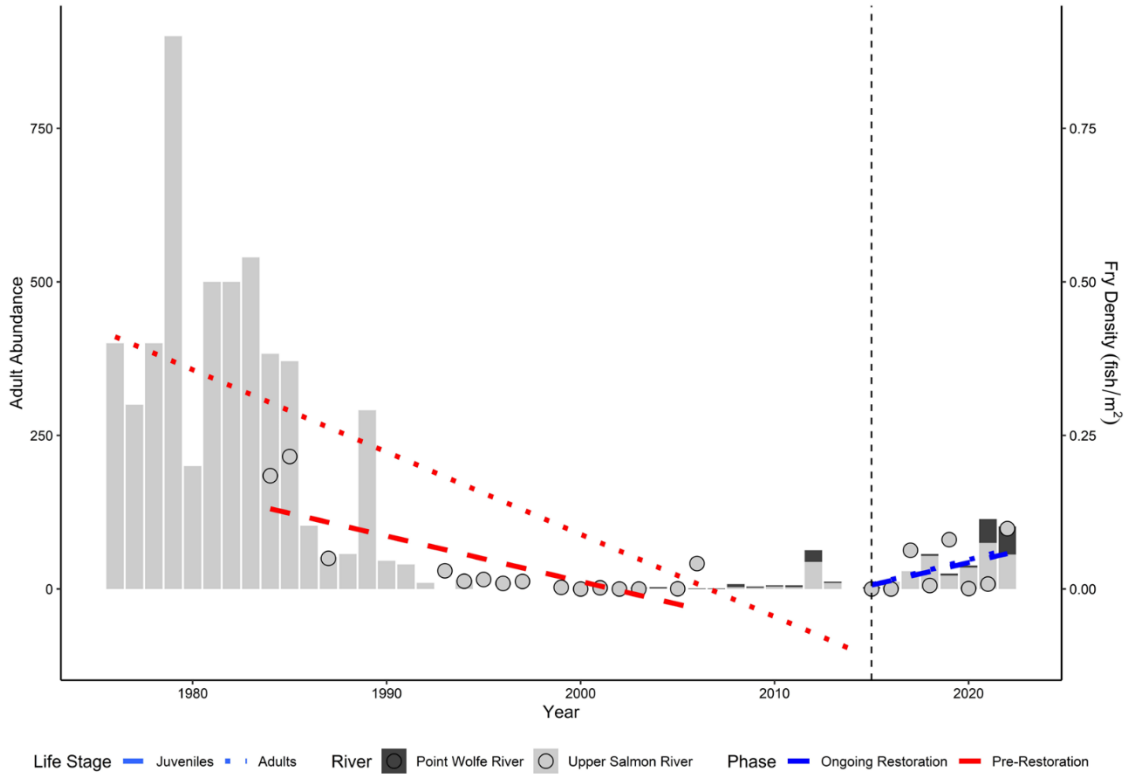

Figure 24. Population trends of returning adult salmon and posterior distributions for salmon fry average annual density from 1976-2022 within the Upper Salmon River, Fundy National Park, before and during restoration efforts. Bars represent adult salmon abundance, coloured by river. Circular points represent the median fry densities. Dotted and dashed trend lines represent the linear trend of adult abundance and fry densities, respectively, coloured in red for pre-restoration and blue for ongoing restoration. The vertical black dashed line represents the start of restoration efforts.

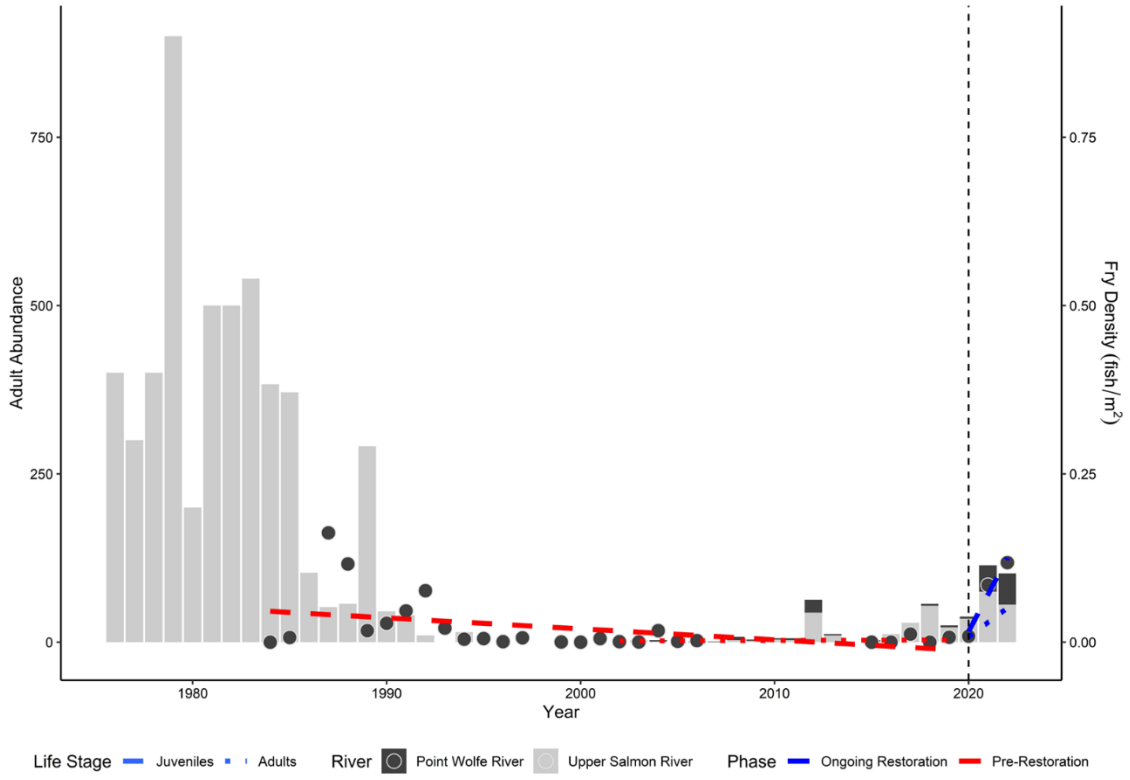

Figure 25. Population trends of returning adult salmon and posterior distributions for salmon fry average annual density from 1984-2022 within the Point Wolfe River, Fundy National Park, before and during restoration efforts. Bars represent adult salmon abundance, coloured by river. Circular points represent the median fry densities, with colour representing river. Dotted and dashed trend lines represent the linear trend of adult abundance and fry densities, respectively, coloured in red for pre-restoration and blue for ongoing restoration. The vertical black dashed line represents the start of restoration efforts.

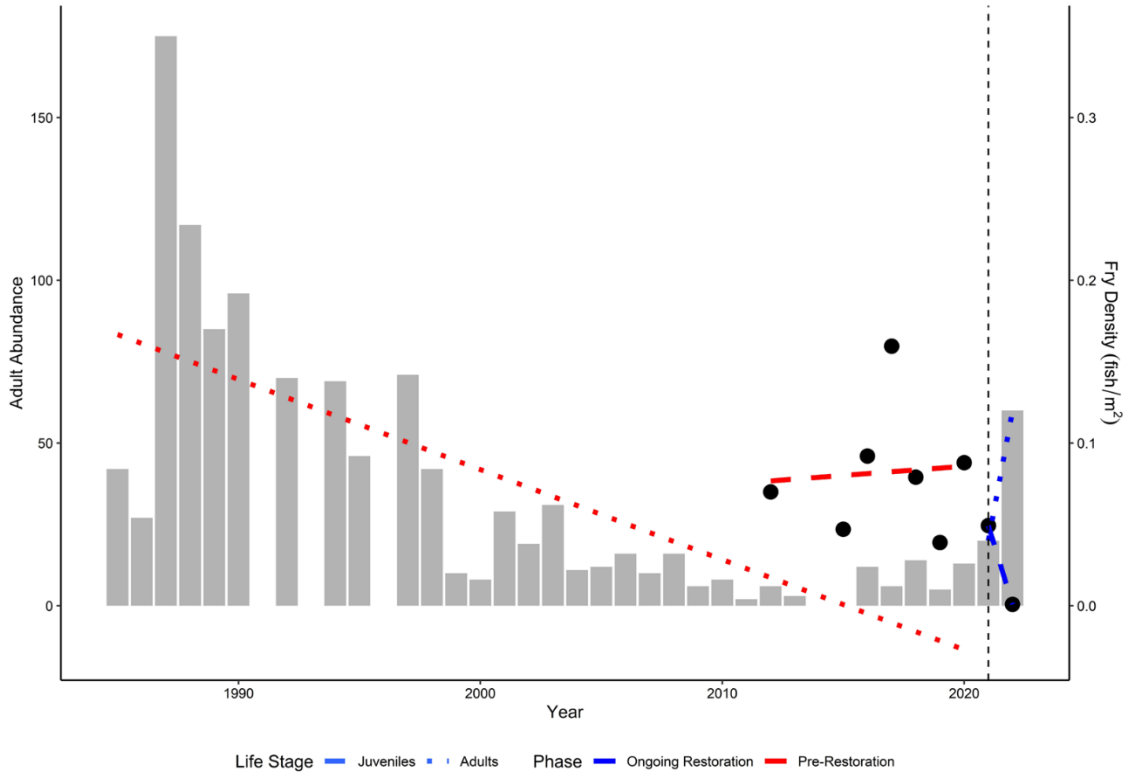

Figure 26. Population trends of returning adult salmon and posterior distributions for salmon fry average annual density from 1985-2022 within Clyburn Brook, Cape Breton Highlands National Park, before and during restoration efforts. Bars represent adult salmon abundance. Circular points represent the median fry densities. Dotted and dashed trend lines represent the linear trend of adult abundance and fry densities, respectively, coloured in red for pre-restoration and blue for ongoing restoration. The vertical black dashed line represents the start of restoration efforts.

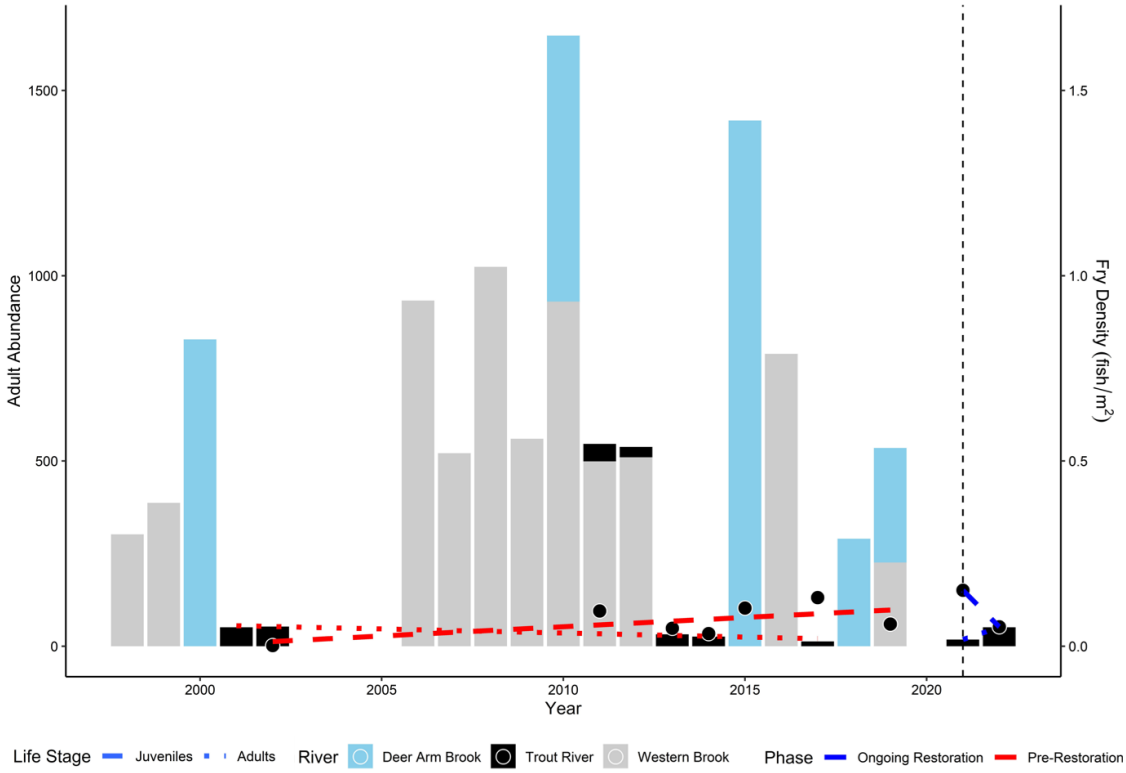

Figure 27. Population trends of returning adult salmon and posterior distributions for salmon fry average annual density from 1998-2022 within Trout River, Gros Morne National Park, before and during restoration efforts. Bars represent adult salmon abundance, coloured by river. Circular points represent the median fry densities. Dotted and dashed trend lines represent the linear trend of adult abundance and fry densities, respectively, coloured in red for pre-restoration and blue for ongoing restoration. The vertical black dashed line represents the start of restoration efforts.

Discussion

This study took advantage of an extensive 49 year dataset of 536 sites, across 24 rivers, in Fundy, Kouchibouguac, Cape Breton Highlands, Gros Morne, and Terra Nova National Parks, to develop a Bayesian hierarchical model to standardize electrofishing monitoring efforts, to assess indices of juvenile Atlantic salmon abundance (*i.e.,* density). Model results showed capture probability was more consistent and produced less uncertainty when the effort covariate was included and could more effectively estimate density from single-pass data when accounting for such covariates. Although single-pass sites may lack the accuracy of the multi-pass methods, they can be an efficient sampling approach from a full catchment scale perspective (Foley et al., 2015; Chaput et al., 2005; Honkanen et al., 2018). However, there are risks and uncertainties involved if not accounting for any calibration methods or modelling without the use of multi-pass sites and available parameters (Dauphin et al., 2019; Glover et al., 2018; Hedger et al., 2013; Malcolm et al., 2019; Millar et al., 2016). This can be challenging to account for and standardize across multiple years for one system, let alone comparing across multiple rivers from different regions. These challenges were considered in this model where possible and examined on their impact on annual density estimations across each park. For objective i), the model was able to standardize and estimate density at site scale all sites, regardless of protocol or other methods that took place within each park. In objective ii), site estimates for single-pass sites were estimated more accurately and with less uncertainty when the effort covariate was involved, than when it was not. To address objective iii), annual densities were calculated for all rivers except those in TNNP, which lacked the number of sites per year per river to accurately represent abundance for the entire system. These results allow

for biologically meaningful comparisons, and insights into population declines and impacts of restoration efforts across rivers, regions, and/or programs, showing the utility a tool like this can be for fishery managers, as per objective (iv).

4.1 Capture Probability

The capture probability of fish during electrofishing operations can be influenced by numerous variables, including but not limited to, organization, equipment, pass, year, altitude, distance to sea, stream width, effort, day of year, and the life stage of the fish being targeted (Dauphin et al., 2019; Glover et al., 2018; Hanks et al., 2018; Malcolm et al., 2019; Millar et al., 2016). As data was often collected inconsistently, with much of the metrics or meta-data missing, a number of these covariates act as proxies to provide further insight on their potential effect on capture probability.

Effort, defined as the amount of time and resources dedicated to electrofishing, plays a crucial role in determining the overall success of monitoring effort. Typically, an increased level of effort results in a greater likelihood of capturing fish (Dauphin et al., 2019). This positive relationship between effort and capture probability results from the increased coverage and intensity of electrofishing activities, which enhances the chances of encountering fish within the surveyed area. This factor was critical, as demonstrated in FNP where variability was greatly reduced amongst sites that had effort data available compared to the pre-2006 data without any available. Likewise, KNP and TNNP sites with effort data produced results with less uncertainty than those without effort data.

Day of year contributes to variations in fish behavior and habitat use (Dauphin et al., 2019; Millar et al., 2016). As the day of year progresses or gets later in the year, fish populations undergo changes in their size, distribution, and activity patterns. Most evident, as the day of year increases, fish are bigger, making them a larger target and easier to detect and capture during electrofishing surveys (Hedger et al., 2013; Korman et al., 2009; Peterson et al., 2004). This explains the positive effect day of year had on capture probability estimates. As the day increases within a particular year, increased capture probability could be observed through the progression of sites completed within FNP. This was more evident in some years in FNP (*e.g.* 2017) as full catchment-scale electrofishing 'blitz's' have taken place for genetic samples, therefore surveys can start as early as July and finish in September. However, this may only be one factor contributing to the increase in capture probability as the day of year gets later. Typically later in the season is represented by warmer water temperatures, resulting in increased fish activity, or increased fish activity can be observed if electrofishing sites happen to be near cold water refuge locations (Bohlin et al., 1989; Malcolm et al., 2019). Water levels and flows are typically reduced later in the season as well, making fish easier to spot, with reduced capacity to evade or find refuge, making them easier to capture. As the season progresses, typically the crew becomes more seasoned and skilled as well. A new crew or one that has not electrofished for a while might be more 'rusty' early in the season or on the initial sites, but as they complete more sites, continue to gain experience, they increase their skill level, and therefore are potentially more efficient later in the day of year (Malcolm et al., 2019; Millar et al., 2016).

Another critical factor influencing capture probability is the life stage of the fish captured (Glover et al., 2018; Malcolm et al., 2019; Millar et al., 2016). Juvenile salmon encompass distinct life stages, including fry and parr, each characterized by unique physiological and behavioral characteristics. Generally, parr, which are older and larger than fry, tend to be more efficiently captured when electrofishing (Korman et al., 2009; Peterson et al., 2004). The larger size of parr increases their detectability and vulnerability to electrical stimuli, making them more responsive to electrofishing compared to the smaller, younger fry. Consequently, they are easier to capture which results in a higher capture probability, therefore less uncertainty and more concise abundance estimates.

As Millar et al., (2016) notes, habitat characteristics, equipment, personnel and hydro-climatological variability are important factors that should be accounted for if accurate estimates of abundance are to be obtained. In absence of many of these variables, our park effect represents the most appropriate, non-exhaustive approach between many of these park rivers and their associated methods and protocols. CBHNP had the greatest negative effect on capture probability, followed by the older time series in FNP and KNP. Some of these differences could stem from local features or differences within the separate river systems (Dauphin et al., 2019). Width is another common factor that can effect capture probability as fish within wider systems tend to evade capture easier (Dauphin et al., 2019; Hanks et al., 2018; Millar et al., 2016), but most significant effects often are due to protocols, organizations, and the personnel or skill-level within the crew (Malcolm et al., 2019; Millar et al., 2016). Without further variables available, it's

difficult to determine what aspect needs improvement in these parks protocols, solely based on the park effect.

4.2 Robustness Testing

Robustness testing of models in their ability to predict open site electrofishing densities based on closed sites provides valuable insights into the reliability and accuracy of the predictive models under different conditions (Dauphin et al., 2019). As the FNP data in the USR and PWR were the only rivers used for this exercise, results demonstrate how robust the model predicts density from single-pass sites for FNP rivers and the assumption this is likely the case for other rivers.

Regarding both site and annual juvenile density estimates, robustness testing indicated that the models produce very close estimates, particularly in lower density sites and years. This suggests that the predictive models maintain consistency and reliability in estimating fish populations across sites, especially when densities are relatively low. However, in higher density sites, some variation in estimates was observed. For annual estimates, the overall median similarity remained relatively consistent, implying the models are generally robust in predicting annual estimates, with minor discrepancies, reflecting increased uncertainty. For sites, estimates become more variable from one other, often with multi-pass medians being greater than single-pass, but single-pass having increased uncertainty, regardless of scenario. Chaput et al. (2005) noted this as well, as in their study the Catch Per Unit Effort index at a closed site could be inflated relative to the index obtained in an open site and the density at the open site would be underestimated, due to barrier nets being present in closed sites. Fish pushed upstream or

displaced downstream during sampling could potentially return within the range, while fish within open sites can evade the site undetected.

Examining the specific line equations for each scenario provides further clarity regarding their predictive capabilities and limitations. In Scenario 1 for fry estimates, an R^2 of 0.51, suggests a moderate level of predictive power. This could be due to the absence of the effort covariate in this model, as it does not account for the potential influence of effort (*i.e.* time and area covered) on fish abundance estimates which makes it more unpredictable at higher densities. However, it is noteworthy that higher densities were more common in the 1980's and 1990's, found within Scenario 1, reflecting data before or the start of the iBoF population collapse in FNP rivers. This bias could potentially influence the predictive accuracy, particularly in comparison with lower densities during extirpation and recovery phases. In Scenario 2, an R^2 of 1.0 indicates a strong predictive capability, again noting that effort covariates are only used in this model, and lower densities are more common. Scenario 3, with an R^2 of 0.75, falls between scenarios 1 and 2 in terms of predictive power. While it provides reasonable estimates, its performance may be affected by the mixed exclusion of the effort covariate, potentially leading to increased variability.

Overall, robustness testing of predictive models in electrofishing demonstrates the importance of considering various factors when estimating fish populations using single-pass methodology. While the models exhibit overall reliability, their performance may vary under different conditions and time series, highlighting the need for cautious interpretation and refinement to enhance predictive accuracy and robustness. Scenario 2, which included the effort covariate, is recommended when predicting single-pass

densities. But for this to occur, all sites must have effort recorded for each pass of their multi-pass sites, and within the single-pass sites. Without it, estimates are much more variable and not as reliable.

4.3 Density

Densities were estimated annually, both at the site and catchment scale for each river of interest within each park, except for TNNP. In TNNP, due to the lack of multiple river sites surveyed annually, estimating annual densities using only one or two sites was not possible or responsible. The sample size was too small or did not create enough strength and certainty in the data to estimate an accurate variance of density, and therefore no system should have an annual density represented by a single-site or two throughout an entire river. Using the hierarchical structure, a strategy of "pooling" all sites together, regardless of river within TNNP was attempted, however with such varying densities across rivers even within a single year, estimations were not possible, nor informative. Although it did not prove helpful for TNNP, the hierarchical structure helped to improve the accuracy and consistency of parameters to achieve density estimates by borrowing strength between sites/parks.

In CBHNP, uncertainty was greater in years with a minimal number of sites sampled. A large portion of these were open sites, leaving too few closed sites available to estimate variance, especially those with clean depletion data through consecutive passes. Poor depletion through consecutive passes can result in unusually low capture probability and affect estimates in some sites (Hedger et al., 2013; Wyatt, 2002). For FNP, greater uncertainty was observed in years with higher densities, particularly in years of higher

density when effort was not available in the 1984-2006 time series. Based on the large variations observed in capture probabilities, and negative park effect, which lead to a greater capture probability variance, this could be an effort issue specifically, although possibly compounded by a combination of effort and high density, because of the protocol and/or collection equipment. It is noteworthy that the older time series/protocol in FNP, which involved dip nets and fishing in the downstream direction, had a negative park effect, a significantly higher capture probability variance, and a much larger but more uncertain density variance. Meanwhile the newer time series/protocol in FNP, which involved a single dip netter and large collection lip seine, had a positive park effect, a lower and more consistent capture probability variance, and a much more narrowed density variance, implying a much more efficient protocol, with less uncertain or variable density estimates.

Increased variability and uncertainty were associated with fry more often than parr. This could be due to fry having overall higher densities in some locations or their negative capture probability effect when compared to parr. Because of this, future modelling efforts could look at making the density variance specifically, but also the mean capture probability as life stage specific, rather than just different based on the presence/absence of effort data (Glover et al., 2018; Malcolm et al., 2019).

4.4 Program Recommendations

Developing, testing, and refining a Bayesian hierarchical model for electrofishing data is a complex and meticulous process to ensure accuracy and robustness. This work ensures that the model can handle the variability and imperfections of ecological data, ultimately

leading to a reliable statistical framework for estimating juvenile salmon populations. However, one goal of this exercise was to provide resource managers with a practical and powerful assessment tool. By making strategic adjustments to their existing sampling protocols or programs, which in turn improves standardization amongst a number of rivers and areas, managers can effectively collect the likewise data required to feed into the model, which can then produce a robust and defensible estimate of juvenile salmon abundance. This estimate is critical for making informed and effective management decisions, helping to ensure the sustainability of salmon populations and the overall health of aquatic ecosystems.

Moving forward, it is strongly recommended for programs like CBHNP and KNP to prioritize the completion of more closed sites within their electrofishing programs. KNP did not encounter issues when running model diagnostics looking at the convergence of their estimates, meaning although they had significantly more single-pass sites than multi-pass sites, the depletion patterns through their multi-pass sites were sufficient enough to estimate appropriate variance parameters, and approximate the posterior distribution of the desired density parameters. This can be done be ensuring a negative linear trend is achieved through consecutive passes, if this is not achieved, then additional passes should be completed. Otherwise, managers may have to rely on more complex statistical models, inspecting model diagnostics, and ensuring appropriate convergence of estimates (Gelman et al., 2015), and make adjustments within the model to make sense of the data which is often undesirable. This is where CBHNP faced greater challenges, particularly in terms of producing enough reliable variability data within their closed sites that can then be applied to their single-pass data. Because of this, they were originally

modelled to share data variability strength with FNP, until they underwent adjustments in the last couple years to incorporate more closed sites into their electrofishing program, which allowed the model to run solely from their own data. Closed sites provide valuable baseline data and facilitate the assessment of population dynamics and trends over time (Dauphin et al., 2019). By incorporating more closed sites into their sampling protocols, programs can improve data reliability and strengthen statistical analyses, thereby enhancing the accuracy of fish population estimates and assessments (Wyatt, 2002). The variability in capture probability in the model is created from closed site electrofishing, which is primarily reflected by how consistent the depletions are through successive passes, and is directly related to convergence during the statistical analysis and modelling. Resolving the depletion issue may involve refining sampling protocols, adjusting sampling frequencies, and/or implementing additional passes to display a stronger depletion pattern. By ensuring proper data collection, programs can mitigate biases and uncertainties, ultimately improving the robustness and reliability of electrofishing data for informed management decisions. This can enhance the effectiveness of their electrofishing programs and contribute to the conservation and management of aquatic ecosystems and fish populations within and beyond their respective regions, making them more comparable. By collecting standardized, comprehensive data across multiple river systems, researchers and managers can identify common patterns, assess regional differences, and establish reference points for evaluating the ecological health and integrity of river ecosystems. This comparability is essential for identifying potential threats, prioritizing conservation actions, and evaluating the effectiveness of management interventions across different river systems.

To provide biologically meaningful estimates, especially when a population is at risk, programs need to push beyond merely collecting the bare minimum information to comply with monitoring population trends. This does not mean that existing programs need to change drastically, or that individual park monitoring metrics need to change, but consistency in basic electrofishing methodology and in data being collected (especially effort) must be ensured to allow for reasonable and robust population assessment that is comparable among years, and between parks. Given that many populations are already at risk, programs should invest extra effort and resources to conduct thorough assessments that provide valuable insights into the status and areas of concern within their fish populations. By conducting more detailed assessments, electrofishing programs can better interpret the health and trajectory of fish populations, allowing for better informed management decisions regarding habitat restoration efforts and population management strategies to ensure the long-term sustainability of fish populations and their ecosystems.

The decline of juvenile Atlantic salmon in FNP was detected because of consistent monitoring efforts, however, monitoring can often only detect trends. To understand population dynamics to implement conservation and/or restoration initiatives, population assessment metrics need to be put in place. Comprehensive assessment measures should be standard protocol to detect refined trajectories within the population collapse, as well as fine-scale results of any implemented actions. This underscores the importance of ongoing monitoring and assessment to track population trends, evaluate the effectiveness of management interventions, and inform adaptive management strategies.

With CBHNP (specifically fry), GMNP (specifically parr), and both life stages within the Kouchibouguac River, early signs of population declines have been detected, indicating

potential ecological stressors or management challenges within these ecosystems. Unfortunately, often only a year or two of data was available following the initiation of most parks restoration efforts so we cannot comment on their effectiveness at this point, but it is paramount that the systems undergo continued assessment to accurately gauge the efficacy of the ongoing and subsequent adaptive recovery measures and detect subtle changes in population dynamics over time. Recovery actions do appear to be increasing juveniles in the Kouchibouguac River, after a sharp decline, suggesting they started their efforts at an appropriate time. However, it's still unclear if the increase in juveniles is a result of their stocking efforts or a changing adult spawning population. Efforts in other parks did help facilitate the return of adult salmon, therefore, an increasing juvenile population could be likely in the years following. By assessing each life stage with a comprehensive approach, this can continue to aid managers in identifying drivers of population decline and tailor interventions accordingly. This may involve habitat restoration initiatives, fisheries management regulations, pollution control measures, and community engagement programs aimed at promoting sustainable resource use and conservation practices. Ongoing monitoring and assessment play a crucial role in evaluating the success of recovery measures and adjusting management strategies in response to changing ecological conditions. Long-term monitoring efforts provide valuable data on population trends, habitat conditions, and ecosystem health, enabling managers to track progress, identify emerging threats, and adapt management approaches as needed.

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Appendix I – Posterior distributions of site densities

Park	River	Year	Site	Life Stage	Site Type	Mean	SD	2.5%	50%	97.5%
CBHNP	Clyburn	2012	130	Fry	Open	0.086	0.102	0.020	0.065	0.263
		2012	130	Parr	Open	0.015	0.027	0.000	0.009	0.065
		2015	131	Fry	Open	0.067	0.105	0.006	0.046	0.247
		2015	131	Parr	Open	0.003	0.011	0.000	0.000	0.029
		2016	132	Fry	Open	0.044	0.075	0.004	0.028	0.176
		2016	132	Parr	Open	0.019	0.025	0.001	0.012	0.077
		2016	133	Fry	Open	0.339	0.318	0.104	0.277	0.895
		2016	133	Parr	Open	0.040	0.046	0.003	0.028	0.146
		2017	134	Fry	Open	0.391	0.321	0.150	0.322	1.041
		2017	134	Parr	Open	0.058	0.057	0.013	0.046	0.174
		2017	135	Fry	Open	0.079	0.100	0.012	0.055	0.273
		2017	135	Parr	Open	0.055	0.050	0.013	0.044	0.163
		2017	136	Fry	Open	0.247	0.240	0.085	0.199	0.708
		2017	136	Parr	Open	0.040	0.042	0.006	0.031	0.125
		2018	137	Fry	Open	0.134	0.145	0.032	0.104	0.404
		2018	137	Parr	Open	0.049	0.066	0.008	0.037	0.152
		2018	138	Fry	Open	0.080	0.110	0.011	0.057	0.273
		2018	138	Parr	Open	0.028	0.035	0.002	0.019	0.105
		2019	139	Fry	Open	0.051	0.054	0.015	0.040	0.160
		2019	139	Parr	Open	0.011	0.013	0.001	0.008	0.037
		2019	140	Fry	Open	0.047	0.056	0.008	0.035	0.151
		2019	140	Parr	Open	0.017	0.021	0.002	0.012	0.059
		2020	141	Fry	Open	0.050	0.063	0.007	0.036	0.173
		2020	141	Parr	Open	0.140	0.093	0.058	0.120	0.347
		2020	305	Fry	Closed	0.299	0.157	0.180	0.269	0.596
		2020	305	Parr	Closed	0.182	0.086	0.107	0.166	0.346
		2020	306	Fry	Closed	0.071	0.052	0.030	0.061	0.173
		2020	306	Parr	Closed	0.066	0.051	0.026	0.056	0.169
		2021	151	Fry	Open	0.023	0.035	0.001	0.014	0.096
		2021	151	Parr	Open	0.128	0.106	0.043	0.107	0.347
		2021	152	Fry	Open	0.046	0.048	0.007	0.034	0.148
		2021	152	Parr	Open	0.149	0.106	0.058	0.127	0.377
		2021	359	Fry	Closed	0.107	0.055	0.058	0.097	0.219
		2021	359	Parr	Closed	0.188	0.082	0.121	0.174	0.345
		2021	360	Fry	Closed	0.128	0.071	0.067	0.115	0.268
		2021	360	Parr	Closed	0.139	0.074	0.078	0.126	0.285
		2022	153	Fry	Open	0.002	0.006	0.000	0.001	0.014
		2022	153	Parr	Open	0.027	0.027	0.003	0.022	0.082
		2022	154	Fry	Open	0.002	0.004	0.000	0.001	0.012
		2022	154	Parr	Open	0.040	0.032	0.011	0.033	0.106

		2022	155	Fry	Open	0.002	0.004	0.000	0.001	0.012
		2022	155	Parr	Open	0.061	0.043	0.022	0.052	0.154
		2022	361	Fry	Closed	0.003	0.004	0.000	0.002	0.012
		2022	361	Parr	Closed	0.055	0.023	0.029	0.052	0.102
FNP	Point Wolfe River	1984	80	Fry	Closed	0.000	0.001	0.000	0.000	0.002
		1984	80	Parr	Closed	0.020	0.009	0.007	0.018	0.040
		1984	81	Fry	Closed	0.000	0.001	0.000	0.000	0.002
		1984	81	Parr	Closed	0.152	0.036	0.092	0.150	0.230
		1984	83	Fry	Closed	0.000	0.001	0.000	0.000	0.002
		1984	83	Parr	Closed	0.015	0.010	0.003	0.013	0.040
		1984	84	Fry	Closed	0.000	0.001	0.000	0.000	0.002
		1984	84	Parr	Closed	0.030	0.013	0.011	0.028	0.060
		1985	85	Fry	Closed	0.003	0.004	0.000	0.002	0.014
		1985	85	Parr	Closed	0.008	0.006	0.001	0.007	0.024
		1985	86	Fry	Closed	0.015	0.009	0.003	0.013	0.037
		1985	86	Parr	Closed	0.030	0.012	0.012	0.029	0.059
		1985	90	Fry	Closed	0.020	0.012	0.004	0.018	0.049
		1985	90	Parr	Closed	0.056	0.019	0.025	0.053	0.100
		1987	95	Fry	Closed	0.138	0.038	0.076	0.134	0.221
		1987	95	Parr	Closed	0.052	0.022	0.019	0.049	0.104
		1987	100	Fry	Closed	0.200	0.054	0.110	0.195	0.317
		1987	100	Parr	Closed	0.632	0.094	0.460	0.627	0.828
		1988	101	Fry	Closed	0.114	0.023	0.075	0.111	0.167
		1988	101	Parr	Closed	0.072	0.017	0.045	0.070	0.112
		1988	102	Fry	Closed	0.060	0.017	0.033	0.058	0.098
		1988	102	Parr	Closed	0.033	0.012	0.015	0.031	0.060
		1988	103	Fry	Closed	0.062	0.016	0.035	0.060	0.099
		1988	103	Parr	Closed	0.103	0.023	0.067	0.100	0.158
		1988	104	Fry	Closed	0.233	0.043	0.165	0.227	0.334
		1988	104	Parr	Closed	0.117	0.026	0.076	0.113	0.179
		1988	105	Fry	Closed	0.225	0.038	0.163	0.222	0.311
		1988	105	Parr	Closed	0.120	0.027	0.077	0.116	0.180
		1989	106	Fry	Closed	0.031	0.011	0.014	0.030	0.057
		1989	106	Parr	Closed	0.023	0.009	0.010	0.021	0.043
		1989	107	Fry	Closed	0.004	0.003	0.000	0.003	0.011
		1989	107	Parr	Closed	0.045	0.011	0.027	0.043	0.069
		1989	108	Fry	Closed	0.003	0.004	0.000	0.002	0.013
		1989	108	Parr	Closed	0.019	0.009	0.006	0.018	0.042
		1989	109	Fry	Closed	0.249	0.041	0.179	0.246	0.341
		1989	109	Parr	Closed	0.123	0.027	0.078	0.121	0.183
		1989	110	Fry	Closed	0.042	0.013	0.021	0.041	0.072
		1989	110	Parr	Closed	0.036	0.012	0.017	0.034	0.063
		1990	111	Fry	Closed	0.056	0.016	0.031	0.054	0.091
		1990	111	Parr	Closed	0.017	0.008	0.006	0.016	0.036
		1990	112	Fry	Closed	0.010	0.007	0.002	0.009	0.027

1990	112	Parr	Closed	0.015	0.007	0.005	0.014	0.033
1990	113	Fry	Closed	0.032	0.012	0.014	0.030	0.059
1990	113	Parr	Closed	0.036	0.012	0.017	0.034	0.062
1990	114	Fry	Closed	0.109	0.024	0.068	0.106	0.162
1990	114	Parr	Closed	0.021	0.009	0.008	0.020	0.041
1990	115	Fry	Closed	0.011	0.006	0.003	0.010	0.026
1990	115	Parr	Closed	0.027	0.009	0.013	0.026	0.049
1991	116	Fry	Closed	0.030	0.011	0.012	0.028	0.055
1991	116	Parr	Closed	0.091	0.020	0.057	0.090	0.135
1991	117	Fry	Closed	0.066	0.014	0.041	0.065	0.097
1991	117	Parr	Closed	0.093	0.017	0.063	0.092	0.130
1991	118	Fry	Closed	0.051	0.014	0.028	0.050	0.080
1991	118	Parr	Closed	0.148	0.023	0.105	0.146	0.198
1992	119	Fry	Closed	0.015	0.008	0.004	0.013	0.034
1992	119	Parr	Closed	0.043	0.013	0.021	0.041	0.071
1992	120	Fry	Closed	0.154	0.023	0.114	0.152	0.203
1992	120	Parr	Closed	0.034	0.009	0.018	0.033	0.055
1992	121	Fry	Closed	0.196	0.026	0.148	0.195	0.251
1992	121	Parr	Closed	0.075	0.015	0.048	0.075	0.109
1993	124	Fry	Closed	0.003	0.003	0.000	0.002	0.011
1993	124	Parr	Closed	0.019	0.008	0.007	0.017	0.038
1993	125	Fry	Closed	0.002	0.002	0.000	0.002	0.009
1993	125	Parr	Closed	0.046	0.012	0.026	0.045	0.074
1993	126	Fry	Closed	0.018	0.008	0.006	0.017	0.037
1993	126	Parr	Closed	0.134	0.022	0.095	0.133	0.181
1993	127	Fry	Closed	0.284	0.040	0.211	0.282	0.370
1993	127	Parr	Closed	0.291	0.040	0.220	0.289	0.376
1993	128	Fry	Closed	0.303	0.032	0.244	0.302	0.369
1993	128	Parr	Closed	0.140	0.021	0.101	0.139	0.186
1994	131	Fry	Closed	0.002	0.002	0.000	0.001	0.008
1994	131	Parr	Closed	0.044	0.012	0.024	0.043	0.072
1994	132	Fry	Closed	0.002	0.002	0.000	0.001	0.008
1994	132	Parr	Closed	0.056	0.014	0.032	0.055	0.087
1994	133	Fry	Closed	0.001	0.002	0.000	0.001	0.006
1994	133	Parr	Closed	0.051	0.011	0.032	0.050	0.075
1994	134	Fry	Closed	0.038	0.012	0.019	0.037	0.064
1994	134	Parr	Closed	0.074	0.016	0.046	0.073	0.109
1994	135	Fry	Closed	0.092	0.017	0.062	0.091	0.129
1994	135	Parr	Closed	0.047	0.012	0.027	0.046	0.073
1995	138	Fry	Closed	0.002	0.002	0.000	0.001	0.007
1995	138	Parr	Closed	0.023	0.008	0.011	0.022	0.041
1995	139	Fry	Closed	0.002	0.002	0.000	0.001	0.008
1995	139	Parr	Closed	0.008	0.005	0.002	0.007	0.019
1995	140	Fry	Closed	0.002	0.002	0.000	0.001	0.009
1995	140	Parr	Closed	0.028	0.010	0.012	0.026	0.050

1995	141	Fry	Closed	0.063	0.017	0.035	0.061	0.100
1995	141	Parr	Closed	0.105	0.022	0.067	0.104	0.152
1995	142	Fry	Closed	0.119	0.022	0.080	0.118	0.167
1995	142	Parr	Closed	0.030	0.011	0.013	0.029	0.056
1996	145	Fry	Closed	0.001	0.002	0.000	0.000	0.005
1996	145	Parr	Closed	0.009	0.005	0.002	0.008	0.023
1996	146	Fry	Closed	0.001	0.002	0.000	0.000	0.006
1996	146	Parr	Closed	0.008	0.005	0.001	0.006	0.020
1996	147	Fry	Closed	0.001	0.001	0.000	0.000	0.004
1996	147	Parr	Closed	0.028	0.008	0.014	0.027	0.047
1996	148	Fry	Closed	0.001	0.001	0.000	0.000	0.005
1996	148	Parr	Closed	0.047	0.012	0.027	0.047	0.073
1996	149	Fry	Closed	0.027	0.009	0.013	0.027	0.048
1996	149	Parr	Closed	0.006	0.004	0.001	0.005	0.016
1997	150	Fry	Closed	0.002	0.002	0.000	0.001	0.008
1997	150	Parr	Closed	0.010	0.006	0.003	0.009	0.024
1997	151	Fry	Closed	0.172	0.029	0.121	0.170	0.232
1997	151	Parr	Closed	0.017	0.008	0.005	0.016	0.037
1997	152	Fry	Closed	0.061	0.015	0.036	0.060	0.093
1997	152	Parr	Closed	0.003	0.002	0.000	0.002	0.009
1997	153	Fry	Closed	0.002	0.003	0.000	0.001	0.009
1997	153	Parr	Closed	0.003	0.003	0.000	0.002	0.010
1997	154	Fry	Closed	0.002	0.002	0.000	0.001	0.009
1997	154	Parr	Closed	0.003	0.002	0.000	0.002	0.009
1999	159	Fry	Closed	0.001	0.001	0.000	0.000	0.004
1999	159	Parr	Closed	0.001	0.001	0.000	0.000	0.004
1999	160	Fry	Closed	0.001	0.001	0.000	0.000	0.004
1999	160	Parr	Closed	0.001	0.001	0.000	0.000	0.004
1999	161	Fry	Closed	0.002	0.003	0.000	0.001	0.009
1999	161	Parr	Closed	0.001	0.001	0.000	0.000	0.004
1999	162	Fry	Closed	0.001	0.001	0.000	0.000	0.004
1999	162	Parr	Closed	0.002	0.002	0.000	0.001	0.007
2000	164	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2000	164	Parr	Closed	0.006	0.004	0.001	0.005	0.015
2000	165	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2000	165	Parr	Closed	0.006	0.005	0.001	0.005	0.019
2000	167	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2000	167	Parr	Closed	0.010	0.005	0.002	0.009	0.023
2000	168	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2000	168	Parr	Closed	0.005	0.004	0.001	0.004	0.014
2000	169	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2000	169	Parr	Closed	0.007	0.005	0.002	0.006	0.019
2001	172	Fry	Closed	0.002	0.002	0.000	0.001	0.009
2001	172	Parr	Closed	0.003	0.003	0.000	0.003	0.010
2001	173	Fry	Closed	0.002	0.002	0.000	0.001	0.008

2001	173	Parr	Closed	0.002	0.002	0.000	0.001	0.007
2001	178	Fry	Closed	0.002	0.002	0.000	0.001	0.009
2001	178	Parr	Closed	0.002	0.002	0.000	0.001	0.007
2001	179	Fry	Closed	0.002	0.002	0.000	0.001	0.008
2001	179	Parr	Closed	0.012	0.006	0.003	0.011	0.027
2001	180	Fry	Closed	0.220	0.035	0.158	0.218	0.293
2001	180	Parr	Closed	0.002	0.002	0.000	0.001	0.008
2001	181	Fry	Closed	0.144	0.024	0.101	0.143	0.194
2001	181	Parr	Closed	0.002	0.002	0.000	0.001	0.007
2002	182	Fry	Closed	0.001	0.001	0.000	0.000	0.005
2002	182	Parr	Closed	0.004	0.003	0.000	0.003	0.011
2002	187	Fry	Closed	0.008	0.005	0.001	0.007	0.019
2002	187	Parr	Closed	0.003	0.003	0.000	0.003	0.010
2002	188	Fry	Closed	0.004	0.004	0.000	0.002	0.014
2002	188	Parr	Closed	0.012	0.006	0.003	0.010	0.028
2002	189	Fry	Closed	0.001	0.002	0.000	0.000	0.005
2002	189	Parr	Closed	0.004	0.003	0.000	0.003	0.011
2002	191	Fry	Closed	0.001	0.002	0.000	0.000	0.006
2002	191	Parr	Closed	0.011	0.006	0.003	0.009	0.025
2002	192	Fry	Closed	0.001	0.002	0.000	0.000	0.007
2002	192	Parr	Closed	0.047	0.015	0.022	0.045	0.081
2003	199	Fry	Closed	0.001	0.001	0.000	0.000	0.004
2003	199	Parr	Closed	0.005	0.004	0.001	0.004	0.015
2003	200	Fry	Closed	0.001	0.001	0.000	0.000	0.004
2003	200	Parr	Closed	0.008	0.004	0.002	0.007	0.019
2003	201	Fry	Closed	0.001	0.001	0.000	0.000	0.004
2003	201	Parr	Closed	0.012	0.006	0.004	0.011	0.027
2003	202	Fry	Closed	0.001	0.001	0.000	0.000	0.004
2003	202	Parr	Closed	0.021	0.008	0.008	0.020	0.040
2003	203	Fry	Closed	0.001	0.002	0.000	0.000	0.007
2003	203	Parr	Closed	0.042	0.018	0.015	0.039	0.084
2003	204	Fry	Closed	0.008	0.005	0.001	0.007	0.021
2003	204	Parr	Closed	0.014	0.006	0.005	0.013	0.029
2004	205	Fry	Closed	0.033	0.011	0.015	0.031	0.058
2004	205	Parr	Closed	0.002	0.002	0.000	0.002	0.008
2004	206	Fry	Closed	0.032	0.012	0.013	0.030	0.059
2004	206	Parr	Closed	0.002	0.002	0.000	0.002	0.009
2004	207	Fry	Closed	0.003	0.004	0.000	0.002	0.013
2004	207	Parr	Closed	0.002	0.002	0.000	0.002	0.009
2004	208	Fry	Closed	0.019	0.009	0.006	0.017	0.039
2004	208	Parr	Closed	0.004	0.003	0.000	0.003	0.011
2004	209	Fry	Closed	0.053	0.014	0.029	0.051	0.084
2004	209	Parr	Closed	0.013	0.007	0.004	0.012	0.029
2005	210	Fry	Closed	0.001	0.002	0.000	0.001	0.008
2005	210	Parr	Closed	0.040	0.016	0.017	0.038	0.078

2005	211	Fry	Closed	0.001	0.002	0.000	0.000	0.005
2005	211	Parr	Closed	0.046	0.012	0.025	0.045	0.073
2005	212	Fry	Closed	0.001	0.002	0.000	0.001	0.007
2005	212	Parr	Closed	0.013	0.008	0.004	0.012	0.033
2005	213	Fry	Closed	0.001	0.002	0.000	0.001	0.008
2005	213	Parr	Closed	0.033	0.013	0.013	0.031	0.064
2005	214	Fry	Closed	0.009	0.007	0.001	0.007	0.027
2005	214	Parr	Closed	0.039	0.014	0.017	0.037	0.072
2005	217	Fry	Closed	0.009	0.006	0.002	0.008	0.023
2005	217	Parr	Closed	0.017	0.007	0.006	0.016	0.034
2006	222	Fry	Closed	0.001	0.002	0.000	0.001	0.008
2006	222	Parr	Closed	0.002	0.002	0.000	0.001	0.007
2006	223	Fry	Closed	0.005	0.004	0.000	0.003	0.017
2006	223	Parr	Closed	0.002	0.002	0.000	0.002	0.008
2006	224	Fry	Closed	0.001	0.002	0.000	0.001	0.006
2006	224	Parr	Closed	0.002	0.002	0.000	0.001	0.007
2006	225	Fry	Closed	0.009	0.005	0.002	0.008	0.022
2006	225	Parr	Closed	0.002	0.002	0.000	0.001	0.006
2006	226	Fry	Closed	0.005	0.005	0.000	0.004	0.018
2006	226	Parr	Closed	0.016	0.008	0.004	0.014	0.037
2006	227	Fry	Closed	0.006	0.004	0.001	0.005	0.017
2006	227	Parr	Closed	0.003	0.002	0.000	0.002	0.009
2015	234	Fry	Closed	0.000	0.001	0.000	0.000	0.003
2015	234	Parr	Closed	0.000	0.001	0.000	0.000	0.003
2015	235	Fry	Closed	0.000	0.001	0.000	0.000	0.003
2015	235	Parr	Closed	0.000	0.001	0.000	0.000	0.003
2015	240	Fry	Closed	0.000	0.001	0.000	0.000	0.003
2015	240	Parr	Closed	0.000	0.001	0.000	0.000	0.003
2015	241	Fry	Closed	0.000	0.001	0.000	0.000	0.003
2015	241	Parr	Closed	0.000	0.001	0.000	0.000	0.003
2016	246	Fry	Closed	0.000	0.001	0.000	0.000	0.003
2016	246	Parr	Closed	0.003	0.003	0.000	0.002	0.011
2016	247	Fry	Closed	0.000	0.001	0.000	0.000	0.003
2016	247	Parr	Closed	0.003	0.003	0.000	0.002	0.011
2016	248	Fry	Closed	0.000	0.001	0.000	0.000	0.003
2016	248	Parr	Closed	0.002	0.003	0.000	0.002	0.010
2016	249	Fry	Closed	0.000	0.001	0.000	0.000	0.003
2016	249	Parr	Closed	0.003	0.004	0.000	0.002	0.014
2017	258	Fry	Closed	0.014	0.010	0.002	0.012	0.040
2017	258	Parr	Closed	0.000	0.001	0.000	0.000	0.004
2017	259	Fry	Closed	0.014	0.010	0.002	0.011	0.040
2017	259	Parr	Closed	0.000	0.001	0.000	0.000	0.004
2017	268	Fry	Closed	0.015	0.011	0.002	0.012	0.044
2017	268	Parr	Closed	0.000	0.001	0.000	0.000	0.004
2017	269	Fry	Closed	0.016	0.012	0.002	0.013	0.049

2017	269	Parr	Closed	0.000	0.001	0.000	0.000	0.004
2018	273	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2018	273	Parr	Closed	0.026	0.008	0.012	0.025	0.045
2018	274	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2018	274	Parr	Closed	0.013	0.006	0.004	0.012	0.027
2018	275	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2018	275	Parr	Closed	0.018	0.007	0.007	0.017	0.033
2019	124	Fry	Open	0.006	0.005	0.001	0.005	0.019
2019	124	Parr	Open	0.003	0.003	0.001	0.003	0.010
2019	125	Fry	Open	0.006	0.004	0.001	0.005	0.018
2019	125	Parr	Open	0.003	0.003	0.000	0.003	0.010
2019	282	Fry	Closed	0.017	0.008	0.006	0.015	0.035
2019	282	Parr	Closed	0.005	0.003	0.001	0.004	0.013
2019	283	Fry	Closed	0.015	0.006	0.006	0.014	0.030
2019	283	Parr	Closed	0.003	0.002	0.000	0.002	0.008
2019	284	Fry	Closed	0.005	0.003	0.001	0.004	0.012
2019	284	Parr	Closed	0.004	0.003	0.001	0.004	0.012
2020	126	Fry	Open	0.018	0.011	0.005	0.016	0.045
2020	126	Parr	Open	0.011	0.006	0.003	0.010	0.026
2020	128	Fry	Open	0.020	0.011	0.006	0.018	0.047
2020	128	Parr	Open	0.011	0.006	0.003	0.010	0.025
2020	287	Fry	Closed	0.008	0.005	0.002	0.007	0.019
2020	287	Parr	Closed	0.008	0.004	0.002	0.007	0.018
2020	288	Fry	Closed	0.012	0.006	0.004	0.011	0.028
2020	288	Parr	Closed	0.008	0.004	0.002	0.007	0.018
2020	291	Fry	Closed	0.006	0.004	0.001	0.005	0.016
2020	291	Parr	Closed	0.029	0.011	0.012	0.028	0.056
2020	292	Fry	Closed	0.006	0.004	0.001	0.005	0.015
2020	292	Parr	Closed	0.015	0.006	0.006	0.014	0.030
2021	293	Fry	Closed	0.231	0.030	0.176	0.229	0.295
2021	293	Parr	Closed	0.042	0.011	0.023	0.041	0.068
2021	294	Fry	Closed	0.114	0.020	0.079	0.113	0.156
2021	294	Parr	Closed	0.030	0.009	0.016	0.029	0.050
2021	301	Fry	Closed	0.058	0.016	0.032	0.056	0.094
2021	301	Parr	Closed	0.019	0.008	0.008	0.018	0.037
2021	302	Fry	Closed	0.076	0.022	0.040	0.073	0.125
2021	302	Parr	Closed	0.018	0.008	0.006	0.016	0.037
2021	303	Fry	Closed	0.072	0.016	0.045	0.071	0.107
2021	303	Parr	Closed	0.033	0.010	0.017	0.032	0.055
2021	304	Fry	Closed	0.053	0.012	0.032	0.052	0.078
2021	304	Parr	Closed	0.020	0.006	0.009	0.019	0.034
2022	364	Fry	Closed	0.107	0.024	0.065	0.105	0.159
2022	364	Parr	Closed	0.082	0.020	0.048	0.080	0.126
2022	365	Fry	Closed	0.148	0.028	0.100	0.146	0.207
2022	365	Parr	Closed	0.054	0.014	0.030	0.053	0.087

	2022	366	Fry	Closed	0.100	0.021	0.063	0.098	0.146
	2022	366	Parr	Closed	0.087	0.019	0.054	0.085	0.127
	2022	367	Fry	Closed	0.108	0.026	0.064	0.106	0.165
	2022	367	Parr	Closed	0.029	0.012	0.011	0.028	0.056
	2022	370	Fry	Closed	0.113	0.023	0.073	0.111	0.162
	2022	370	Parr	Closed	0.048	0.013	0.026	0.047	0.077
	2022	371	Fry	Closed	0.157	0.027	0.109	0.156	0.213
	2022	371	Parr	Closed	0.080	0.018	0.050	0.079	0.118
Upper Salmon River	1984	78	Fry	Closed	0.223	0.056	0.128	0.218	0.345
	1984	78	Parr	Closed	0.187	0.056	0.097	0.181	0.315
	1984	79	Fry	Closed	0.114	0.031	0.062	0.111	0.185
	1984	79	Parr	Closed	0.040	0.017	0.014	0.037	0.079
	1984	82	Fry	Closed	0.280	0.056	0.180	0.277	0.400
	1984	82	Parr	Closed	0.080	0.029	0.033	0.076	0.148
	1985	87	Fry	Closed	0.117	0.033	0.063	0.114	0.191
	1985	87	Parr	Closed	0.217	0.047	0.138	0.213	0.321
	1985	88	Fry	Closed	0.060	0.024	0.024	0.057	0.117
	1985	88	Parr	Closed	0.085	0.026	0.042	0.083	0.144
	1985	89	Fry	Closed	0.380	0.068	0.261	0.375	0.530
	1985	89	Parr	Closed	0.201	0.046	0.122	0.198	0.299
	1985	91	Fry	Closed	1.090	0.127	0.866	1.081	1.365
	1985	91	Parr	Closed	0.378	0.064	0.265	0.374	0.517
	1985	92	Fry	Closed	0.187	0.047	0.109	0.182	0.290
	1985	92	Parr	Closed	0.091	0.030	0.043	0.087	0.158
	1987	93	Fry	Closed	0.042	0.020	0.012	0.039	0.090
	1987	93	Parr	Closed	0.456	0.076	0.324	0.450	0.622
	1987	94	Fry	Closed	0.013	0.010	0.001	0.011	0.040
	1987	94	Parr	Closed	0.138	0.037	0.077	0.135	0.220
	1987	96	Fry	Closed	0.042	0.020	0.012	0.039	0.091
	1987	96	Parr	Closed	0.202	0.046	0.125	0.199	0.302
	1987	97	Fry	Closed	0.109	0.033	0.054	0.105	0.183
	1987	97	Parr	Closed	0.521	0.080	0.380	0.515	0.697
	1987	98	Fry	Closed	0.258	0.053	0.165	0.254	0.372
	1987	98	Parr	Closed	0.381	0.063	0.268	0.378	0.516
	1987	99	Fry	Closed	0.043	0.021	0.013	0.040	0.094
	1987	99	Parr	Closed	0.495	0.072	0.367	0.491	0.646
	1993	122	Fry	Closed	0.013	0.007	0.003	0.012	0.030
	1993	122	Parr	Closed	0.190	0.031	0.137	0.187	0.261
	1993	123	Fry	Closed	0.070	0.014	0.046	0.069	0.101
	1993	123	Parr	Closed	0.083	0.015	0.057	0.082	0.117
	1994	129	Fry	Closed	0.017	0.007	0.005	0.016	0.034
	1994	129	Parr	Closed	0.110	0.020	0.076	0.109	0.152
	1994	130	Fry	Closed	0.011	0.006	0.002	0.009	0.025
	1994	130	Parr	Closed	0.102	0.019	0.069	0.101	0.142
	1995	136	Fry	Closed	0.005	0.003	0.001	0.004	0.013

1995	136	Parr	Closed	0.030	0.008	0.017	0.030	0.047
1995	137	Fry	Closed	0.050	0.010	0.033	0.050	0.071
1995	137	Parr	Closed	0.011	0.004	0.004	0.010	0.021
1996	143	Fry	Closed	0.033	0.010	0.018	0.032	0.055
1996	143	Parr	Closed	0.010	0.005	0.003	0.009	0.022
1996	144	Fry	Closed	0.003	0.003	0.000	0.003	0.010
1996	144	Parr	Closed	0.030	0.008	0.016	0.029	0.049
1997	155	Fry	Closed	0.006	0.004	0.001	0.005	0.017
1997	155	Parr	Closed	0.008	0.005	0.002	0.007	0.020
1997	156	Fry	Closed	0.028	0.010	0.012	0.027	0.051
1997	156	Parr	Closed	0.030	0.010	0.013	0.029	0.054
1999	157	Fry	Closed	0.001	0.002	0.000	0.001	0.006
1999	157	Parr	Closed	0.004	0.003	0.001	0.003	0.012
1999	158	Fry	Closed	0.012	0.006	0.003	0.011	0.027
1999	158	Parr	Closed	0.009	0.005	0.002	0.007	0.021
2000	163	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2000	163	Parr	Closed	0.007	0.004	0.002	0.006	0.015
2000	166	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2000	166	Parr	Closed	0.020	0.007	0.008	0.019	0.036
2001	170	Fry	Closed	0.001	0.002	0.000	0.001	0.006
2001	170	Parr	Closed	0.006	0.004	0.001	0.005	0.015
2001	171	Fry	Closed	0.001	0.002	0.000	0.001	0.006
2001	171	Parr	Closed	0.011	0.006	0.003	0.010	0.025
2001	174	Fry	Closed	0.002	0.003	0.000	0.001	0.010
2001	174	Parr	Closed	0.006	0.005	0.001	0.005	0.018
2001	175	Fry	Closed	0.007	0.004	0.001	0.006	0.016
2001	175	Parr	Closed	0.006	0.003	0.002	0.006	0.015
2001	176	Fry	Closed	0.004	0.004	0.000	0.003	0.016
2001	176	Parr	Closed	0.040	0.013	0.019	0.039	0.071
2001	177	Fry	Closed	0.017	0.008	0.005	0.016	0.036
2001	177	Parr	Closed	0.037	0.012	0.019	0.036	0.063
2002	183	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2002	183	Parr	Closed	0.002	0.002	0.000	0.002	0.007
2002	184	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2002	184	Parr	Closed	0.003	0.003	0.000	0.002	0.011
2002	185	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2002	185	Parr	Closed	0.011	0.006	0.003	0.009	0.025
2002	186	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2002	186	Parr	Closed	0.006	0.004	0.001	0.005	0.015
2002	190	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2002	190	Parr	Closed	0.002	0.002	0.000	0.002	0.008
2003	193	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2003	193	Parr	Closed	0.001	0.001	0.000	0.001	0.004
2003	194	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2003	194	Parr	Closed	0.001	0.001	0.000	0.001	0.004

2003	195	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2003	195	Parr	Closed	0.001	0.002	0.000	0.001	0.005
2003	196	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2003	196	Parr	Closed	0.001	0.001	0.000	0.001	0.004
2003	197	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2003	197	Parr	Closed	0.001	0.001	0.000	0.001	0.004
2003	198	Fry	Closed	0.000	0.000	0.000	0.000	0.001
2003	198	Parr	Closed	0.003	0.003	0.000	0.002	0.010
2005	215	Fry	Closed	0.001	0.002	0.000	0.000	0.006
2005	215	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2005	216	Fry	Closed	0.001	0.002	0.000	0.000	0.005
2005	216	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2005	218	Fry	Closed	0.001	0.002	0.000	0.000	0.005
2005	218	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2005	219	Fry	Closed	0.001	0.002	0.000	0.000	0.006
2005	219	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2005	220	Fry	Closed	0.003	0.003	0.000	0.002	0.010
2005	220	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2005	221	Fry	Closed	0.003	0.003	0.000	0.002	0.011
2005	221	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2006	228	Fry	Closed	0.096	0.022	0.058	0.094	0.143
2006	228	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2006	229	Fry	Closed	0.072	0.017	0.043	0.071	0.110
2006	229	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2006	230	Fry	Closed	0.025	0.011	0.009	0.023	0.049
2006	230	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2006	231	Fry	Closed	0.101	0.019	0.067	0.099	0.143
2006	231	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2006	232	Fry	Closed	0.026	0.010	0.011	0.025	0.049
2006	232	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2006	233	Fry	Closed	0.014	0.008	0.004	0.013	0.034
2006	233	Parr	Closed	0.000	0.000	0.000	0.000	0.001
2015	236	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2015	236	Parr	Closed	0.004	0.003	0.001	0.003	0.013
2015	237	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2015	237	Parr	Closed	0.004	0.003	0.001	0.003	0.013
2015	238	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2015	238	Parr	Closed	0.004	0.003	0.001	0.004	0.013
2015	239	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2015	239	Parr	Closed	0.004	0.003	0.001	0.003	0.012
2015	242	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2015	242	Parr	Closed	0.005	0.003	0.001	0.004	0.013
2015	243	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2015	243	Parr	Closed	0.004	0.003	0.001	0.003	0.011
2015	244	Fry	Closed	0.000	0.001	0.000	0.000	0.001

2015	244	Parr	Closed	0.007	0.005	0.001	0.005	0.019
2015	245	Fry	Closed	0.000	0.001	0.000	0.000	0.001
2015	245	Parr	Closed	0.006	0.004	0.001	0.005	0.017
2016	250	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2016	250	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2016	251	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2016	251	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2016	252	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2016	252	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2016	253	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2016	253	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2016	254	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2016	254	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2016	255	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2016	255	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2016	256	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2016	256	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2016	257	Fry	Closed	0.000	0.001	0.000	0.000	0.002
2016	257	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2017	99	Fry	Open	0.180	0.041	0.110	0.175	0.271
2017	99	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	100	Fry	Open	0.056	0.020	0.024	0.053	0.102
2017	100	Parr	Open	0.000	0.001	0.000	0.000	0.002
2017	101	Fry	Open	0.050	0.020	0.020	0.047	0.099
2017	101	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	102	Fry	Open	0.061	0.023	0.025	0.058	0.116
2017	102	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	103	Fry	Open	0.041	0.019	0.013	0.038	0.088
2017	103	Parr	Open	0.000	0.001	0.000	0.000	0.002
2017	104	Fry	Open	0.078	0.026	0.037	0.075	0.138
2017	104	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	105	Fry	Open	0.219	0.051	0.133	0.214	0.330
2017	105	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	106	Fry	Open	0.060	0.023	0.025	0.057	0.113
2017	106	Parr	Open	0.000	0.001	0.000	0.000	0.002
2017	107	Fry	Open	0.067	0.028	0.025	0.062	0.133
2017	107	Parr	Open	0.000	0.001	0.000	0.000	0.002
2017	108	Fry	Open	0.182	0.046	0.106	0.178	0.285
2017	108	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	109	Fry	Open	0.094	0.031	0.045	0.091	0.165
2017	109	Parr	Open	0.000	0.001	0.000	0.000	0.002
2017	110	Fry	Open	0.062	0.024	0.026	0.058	0.118
2017	110	Parr	Open	0.000	0.001	0.000	0.000	0.002
2017	111	Fry	Open	0.061	0.022	0.028	0.059	0.111
2017	111	Parr	Open	0.000	0.001	0.000	0.000	0.002

2017	112	Fry	Open	0.044	0.019	0.016	0.041	0.087
2017	112	Parr	Open	0.000	0.001	0.000	0.000	0.002
2017	113	Fry	Open	0.050	0.020	0.019	0.047	0.097
2017	113	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	114	Fry	Open	0.057	0.022	0.024	0.053	0.109
2017	114	Parr	Open	0.001	0.001	0.000	0.000	0.002
2017	115	Fry	Open	0.152	0.039	0.086	0.147	0.242
2017	115	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	116	Fry	Open	0.154	0.037	0.090	0.150	0.237
2017	116	Parr	Open	0.000	0.001	0.000	0.000	0.002
2017	117	Fry	Open	0.080	0.024	0.041	0.077	0.134
2017	117	Parr	Open	0.000	0.000	0.000	0.000	0.002
2017	260	Fry	Closed	0.119	0.029	0.070	0.116	0.184
2017	260	Parr	Closed	0.000	0.000	0.000	0.000	0.002
2017	261	Fry	Closed	0.117	0.034	0.060	0.113	0.193
2017	261	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2017	262	Fry	Closed	0.023	0.012	0.006	0.021	0.053
2017	262	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2017	263	Fry	Closed	0.029	0.016	0.007	0.026	0.067
2017	263	Parr	Closed	0.000	0.000	0.000	0.000	0.002
2017	264	Fry	Closed	0.042	0.025	0.010	0.036	0.106
2017	264	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2017	265	Fry	Closed	0.032	0.018	0.008	0.028	0.076
2017	265	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2017	266	Fry	Closed	0.043	0.018	0.015	0.040	0.085
2017	266	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2017	267	Fry	Closed	0.038	0.017	0.013	0.035	0.079
2017	267	Parr	Closed	0.000	0.001	0.000	0.000	0.002
2018	118	Fry	Open	0.008	0.006	0.002	0.007	0.023
2018	118	Parr	Open	0.041	0.016	0.018	0.039	0.080
2018	119	Fry	Open	0.007	0.007	0.001	0.005	0.025
2018	119	Parr	Open	0.032	0.019	0.009	0.028	0.080
2018	120	Fry	Open	0.006	0.004	0.001	0.004	0.017
2018	120	Parr	Open	0.028	0.012	0.011	0.027	0.057
2018	121	Fry	Open	0.006	0.004	0.001	0.005	0.017
2018	121	Parr	Open	0.017	0.009	0.005	0.016	0.038
2018	270	Fry	Closed	0.004	0.003	0.001	0.003	0.011
2018	270	Parr	Closed	0.042	0.011	0.023	0.041	0.067
2018	271	Fry	Closed	0.004	0.002	0.001	0.003	0.010
2018	271	Parr	Closed	0.023	0.007	0.011	0.022	0.039
2018	272	Fry	Closed	0.010	0.005	0.003	0.009	0.021
2018	272	Parr	Closed	0.026	0.008	0.013	0.025	0.043
2018	276	Fry	Closed	0.009	0.004	0.003	0.009	0.020
2018	276	Parr	Closed	0.013	0.005	0.005	0.012	0.025
2018	277	Fry	Closed	0.013	0.007	0.004	0.012	0.030

2018	277	Parr	Closed	0.032	0.011	0.015	0.031	0.056
2019	122	Fry	Open	0.141	0.036	0.083	0.137	0.221
2019	122	Parr	Open	0.010	0.005	0.003	0.009	0.023
2019	123	Fry	Open	0.059	0.021	0.026	0.056	0.108
2019	123	Parr	Open	0.010	0.005	0.003	0.009	0.023
2019	278	Fry	Closed	0.057	0.014	0.034	0.056	0.088
2019	278	Parr	Closed	0.010	0.005	0.004	0.010	0.022
2019	279	Fry	Closed	0.046	0.013	0.024	0.045	0.076
2019	279	Parr	Closed	0.016	0.007	0.006	0.015	0.032
2019	280	Fry	Closed	0.105	0.018	0.072	0.104	0.143
2019	280	Parr	Closed	0.010	0.004	0.004	0.009	0.020
2019	281	Fry	Closed	0.137	0.025	0.093	0.135	0.189
2019	281	Parr	Closed	0.008	0.004	0.002	0.007	0.018
2020	127	Fry	Open	0.002	0.003	0.000	0.001	0.009
2020	127	Parr	Open	0.189	0.046	0.112	0.185	0.292
2020	129	Fry	Open	0.001	0.002	0.000	0.001	0.005
2020	129	Parr	Open	0.089	0.027	0.045	0.086	0.150
2020	285	Fry	Closed	0.001	0.001	0.000	0.001	0.004
2020	285	Parr	Closed	0.142	0.024	0.099	0.140	0.192
2020	286	Fry	Closed	0.001	0.001	0.000	0.001	0.005
2020	286	Parr	Closed	0.132	0.026	0.086	0.130	0.188
2020	289	Fry	Closed	0.001	0.001	0.000	0.001	0.004
2020	289	Parr	Closed	0.149	0.024	0.105	0.148	0.201
2020	290	Fry	Closed	0.001	0.001	0.000	0.001	0.005
2020	290	Parr	Closed	0.072	0.018	0.041	0.071	0.113
2021	295	Fry	Closed	0.007	0.004	0.002	0.006	0.017
2021	295	Parr	Closed	0.012	0.005	0.004	0.011	0.024
2021	296	Fry	Closed	0.013	0.007	0.004	0.012	0.031
2021	296	Parr	Closed	0.011	0.005	0.003	0.010	0.024
2021	297	Fry	Closed	0.013	0.006	0.004	0.012	0.027
2021	297	Parr	Closed	0.015	0.006	0.006	0.014	0.029
2021	298	Fry	Closed	0.009	0.005	0.002	0.008	0.021
2021	298	Parr	Closed	0.010	0.005	0.003	0.009	0.021
2021	299	Fry	Closed	0.005	0.003	0.001	0.005	0.014
2021	299	Parr	Closed	0.007	0.004	0.002	0.006	0.016
2021	300	Fry	Closed	0.010	0.005	0.003	0.010	0.022
2021	300	Parr	Closed	0.013	0.005	0.005	0.012	0.025
2022	156	Fry	Open	0.292	0.054	0.198	0.288	0.408
2022	156	Parr	Open	0.010	0.005	0.003	0.009	0.023
2022	157	Fry	Open	0.211	0.046	0.132	0.207	0.313
2022	157	Parr	Open	0.013	0.007	0.004	0.011	0.029
2022	362	Fry	Closed	0.065	0.018	0.035	0.063	0.104
2022	362	Parr	Closed	0.013	0.006	0.004	0.012	0.028
2022	363	Fry	Closed	0.029	0.011	0.011	0.027	0.054
2022	363	Parr	Closed	0.025	0.010	0.011	0.023	0.047

		2022	368	Fry	Closed	0.148	0.028	0.098	0.146	0.209
		2022	368	Parr	Closed	0.011	0.005	0.003	0.010	0.024
		2022	369	Fry	Closed	0.062	0.018	0.033	0.060	0.101
		2022	369	Parr	Closed	0.023	0.009	0.009	0.022	0.045
GMNP	Trout River	2002	307	Fry	Closed	0.005	0.006	0.000	0.003	0.020
		2002	307	Parr	Closed	0.113	0.029	0.063	0.110	0.177
		2002	308	Fry	Closed	0.002	0.003	0.000	0.001	0.011
		2002	308	Parr	Closed	0.086	0.021	0.051	0.084	0.134
		2011	309	Fry	Closed	0.074	0.019	0.043	0.072	0.115
		2011	309	Parr	Closed	0.127	0.023	0.085	0.125	0.179
		2011	310	Fry	Closed	0.125	0.022	0.087	0.124	0.173
		2011	310	Parr	Closed	0.070	0.015	0.044	0.069	0.103
		2013	311	Fry	Closed	0.113	0.021	0.074	0.111	0.158
		2013	311	Parr	Closed	0.113	0.021	0.077	0.112	0.157
		2013	312	Fry	Closed	0.022	0.009	0.009	0.021	0.043
		2013	312	Parr	Closed	0.035	0.011	0.017	0.034	0.061
		2014	313	Fry	Closed	0.036	0.014	0.015	0.034	0.068
		2014	313	Parr	Closed	0.018	0.008	0.006	0.016	0.038
		2015	314	Fry	Closed	0.031	0.013	0.011	0.029	0.061
		2015	314	Parr	Closed	0.146	0.028	0.097	0.144	0.205
		2015	315	Fry	Closed	0.012	0.007	0.002	0.010	0.030
		2015	315	Parr	Closed	0.175	0.030	0.122	0.173	0.238
		2015	316	Fry	Closed	0.280	0.044	0.203	0.277	0.374
		2015	316	Parr	Closed	0.108	0.023	0.068	0.107	0.159
		2015	317	Fry	Closed	0.190	0.041	0.117	0.187	0.281
		2015	317	Parr	Closed	0.059	0.020	0.027	0.057	0.105
		2015	318	Fry	Closed	0.267	0.048	0.184	0.262	0.372
		2015	318	Parr	Closed	0.045	0.016	0.020	0.043	0.081
		2015	319	Fry	Closed	0.295	0.046	0.212	0.291	0.393
		2015	319	Parr	Closed	0.050	0.017	0.023	0.048	0.087
		2017	320	Fry	Closed	0.065	0.020	0.032	0.063	0.109
		2017	320	Parr	Closed	0.075	0.020	0.042	0.073	0.119
		2017	321	Fry	Closed	0.558	0.080	0.422	0.552	0.736
		2017	321	Parr	Closed	0.034	0.013	0.015	0.033	0.064
		2017	322	Fry	Closed	0.509	0.062	0.402	0.504	0.643
		2017	322	Parr	Closed	0.035	0.011	0.017	0.033	0.061
		2017	323	Fry	Closed	0.112	0.030	0.062	0.109	0.179
		2017	323	Parr	Closed	0.065	0.020	0.033	0.063	0.111
		2017	324	Fry	Closed	0.108	0.026	0.064	0.106	0.166
		2017	324	Parr	Closed	0.041	0.014	0.019	0.040	0.073
		2017	325	Fry	Closed	0.113	0.034	0.057	0.110	0.191
		2017	325	Parr	Closed	0.039	0.016	0.015	0.037	0.075
		2017	326	Fry	Closed	0.067	0.024	0.030	0.064	0.123
		2017	326	Parr	Closed	0.021	0.010	0.006	0.019	0.046
		2017	327	Fry	Closed	0.129	0.033	0.073	0.126	0.203

		2017	327	Parr	Closed	0.027	0.011	0.010	0.026	0.054
		2017	328	Fry	Closed	0.071	0.023	0.034	0.069	0.123
		2017	328	Parr	Closed	0.023	0.010	0.008	0.022	0.047
		2019	329	Fry	Closed	0.138	0.038	0.075	0.133	0.222
		2019	329	Parr	Closed	0.020	0.010	0.006	0.018	0.045
		2019	330	Fry	Closed	0.345	0.048	0.260	0.343	0.445
		2019	330	Parr	Closed	0.008	0.005	0.002	0.007	0.019
		2019	331	Fry	Closed	0.115	0.033	0.060	0.111	0.187
		2019	331	Parr	Closed	0.033	0.015	0.012	0.031	0.071
		2019	332	Fry	Closed	0.030	0.015	0.009	0.028	0.065
		2019	332	Parr	Closed	0.011	0.006	0.003	0.010	0.027
		2019	333	Fry	Closed	0.059	0.021	0.026	0.057	0.107
		2019	333	Parr	Closed	0.017	0.008	0.006	0.015	0.037
		2019	334	Fry	Closed	0.033	0.012	0.014	0.031	0.061
		2019	334	Parr	Closed	0.012	0.006	0.004	0.011	0.025
		2019	335	Fry	Closed	0.023	0.014	0.005	0.020	0.057
		2019	335	Parr	Closed	0.010	0.006	0.002	0.009	0.025
		2019	336	Fry	Closed	0.029	0.014	0.009	0.027	0.063
		2019	336	Parr	Closed	0.014	0.007	0.004	0.013	0.032
		2019	337	Fry	Closed	0.080	0.024	0.041	0.077	0.134
		2019	337	Parr	Closed	0.014	0.007	0.004	0.012	0.030
		2021	372	Fry	Closed	0.272	0.042	0.197	0.269	0.361
		2021	372	Parr	Closed	0.009	0.005	0.002	0.008	0.022
		2021	373	Fry	Closed	0.100	0.026	0.057	0.098	0.159
		2021	373	Parr	Closed	0.015	0.007	0.005	0.014	0.033
		2021	374	Fry	Closed	0.134	0.030	0.083	0.132	0.199
		2021	374	Parr	Closed	0.025	0.010	0.009	0.023	0.049
		2021	375	Fry	Closed	0.134	0.032	0.079	0.131	0.205
		2021	375	Parr	Closed	0.023	0.010	0.008	0.021	0.048
		2021	376	Fry	Closed	0.179	0.033	0.122	0.176	0.251
		2021	376	Parr	Closed	0.021	0.009	0.008	0.020	0.041
		2022	377	Fry	Closed	0.038	0.014	0.015	0.036	0.071
		2022	377	Parr	Closed	0.035	0.013	0.015	0.033	0.063
		2022	378	Fry	Closed	0.065	0.019	0.033	0.063	0.108
		2022	378	Parr	Closed	0.028	0.011	0.011	0.026	0.054
		2022	379	Fry	Closed	0.068	0.021	0.033	0.065	0.117
		2022	379	Parr	Closed	0.048	0.016	0.022	0.046	0.085
	Western Brook	2020	338	Fry	Closed	1.018	0.102	0.835	1.011	1.234
		2020	338	Parr	Closed	0.162	0.037	0.099	0.160	0.242
KNP	Black River	2021	347	Fry	Closed	0.475	0.065	0.362	0.470	0.618
		2021	347	Parr	Closed	0.027	0.012	0.010	0.026	0.056
		2021	348	Fry	Closed	0.806	0.078	0.667	0.801	0.974
		2021	348	Parr	Closed	0.046	0.013	0.024	0.045	0.076
		2021	349	Fry	Closed	0.019	0.011	0.005	0.017	0.047
		2021	349	Parr	Closed	0.027	0.012	0.010	0.025	0.055

	2021	350	Fry	Closed	0.009	0.006	0.002	0.008	0.024
	2021	350	Parr	Closed	0.042	0.013	0.022	0.040	0.070
	2022	353	Fry	Closed	0.023	0.011	0.007	0.021	0.048
	2022	353	Parr	Closed	0.141	0.029	0.092	0.138	0.207
	2022	354	Fry	Closed	0.029	0.013	0.009	0.027	0.061
	2022	354	Parr	Closed	0.361	0.052	0.270	0.358	0.473
	2022	357	Fry	Closed	0.191	0.039	0.125	0.187	0.277
	2022	357	Parr	Closed	0.030	0.013	0.011	0.028	0.060
	2022	358	Fry	Closed	0.175	0.036	0.113	0.172	0.254
	2022	358	Parr	Closed	0.044	0.016	0.020	0.042	0.079
Kouchibouguac	1974	1	Fry	Open	0.003	0.005	0.000	0.001	0.018
	1974	1	Parr	Open	0.151	0.074	0.056	0.136	0.338
	1974	2	Fry	Open	0.003	0.006	0.000	0.001	0.017
	1974	2	Parr	Open	0.224	0.095	0.101	0.206	0.452
	1974	3	Fry	Open	0.003	0.006	0.000	0.001	0.018
	1974	3	Parr	Open	0.219	0.097	0.093	0.199	0.457
	1974	4	Fry	Open	0.003	0.005	0.000	0.001	0.015
	1974	4	Parr	Open	0.153	0.066	0.067	0.140	0.317
	1974	5	Fry	Open	0.003	0.006	0.000	0.001	0.018
	1974	5	Parr	Open	0.118	0.065	0.039	0.104	0.278
	1974	6	Fry	Open	0.007	0.010	0.000	0.004	0.034
	1974	6	Parr	Open	0.158	0.071	0.067	0.143	0.339
	1977	63	Fry	Closed	0.046	0.016	0.021	0.044	0.085
	1977	63	Parr	Closed	0.062	0.019	0.033	0.059	0.105
	1977	64	Fry	Closed	0.081	0.018	0.051	0.079	0.119
	1977	64	Parr	Closed	0.214	0.033	0.160	0.210	0.288
	1977	65	Fry	Closed	0.009	0.005	0.002	0.008	0.022
	1977	65	Parr	Closed	0.108	0.026	0.070	0.104	0.172
	1977	66	Fry	Closed	0.088	0.019	0.055	0.086	0.130
	1977	66	Parr	Closed	0.130	0.024	0.091	0.128	0.185
	1977	67	Fry	Closed	0.068	0.020	0.035	0.066	0.114
	1977	67	Parr	Closed	0.043	0.017	0.019	0.040	0.082
	1977	68	Fry	Closed	0.460	0.049	0.374	0.457	0.562
	1977	68	Parr	Closed	0.224	0.033	0.166	0.222	0.294
	1978	69	Fry	Closed	1.892	0.106	1.697	1.886	2.114
	1978	69	Parr	Closed	0.028	0.010	0.012	0.027	0.052
	1978	70	Fry	Closed	1.482	0.080	1.336	1.478	1.648
	1978	70	Parr	Closed	0.091	0.017	0.060	0.090	0.128
	1978	71	Fry	Closed	0.306	0.038	0.237	0.304	0.386
	1978	71	Parr	Closed	0.061	0.016	0.034	0.060	0.097
	1978	72	Fry	Closed	1.030	0.067	0.903	1.028	1.168
	1978	72	Parr	Closed	0.271	0.034	0.210	0.269	0.345
	1982	73	Fry	Closed	0.014	0.006	0.005	0.013	0.026
	1982	73	Parr	Closed	0.039	0.010	0.023	0.039	0.061
	1982	74	Fry	Closed	0.014	0.007	0.004	0.013	0.030

1982	74	Parr	Closed	0.049	0.014	0.026	0.047	0.081
1982	75	Fry	Closed	0.208	0.030	0.154	0.207	0.272
1982	75	Parr	Closed	0.102	0.021	0.066	0.100	0.146
1999	7	Fry	Open	0.861	0.269	0.482	0.812	1.511
1999	7	Parr	Open	0.491	0.149	0.276	0.467	0.854
1999	8	Fry	Open	0.468	0.156	0.249	0.441	0.841
1999	8	Parr	Open	0.450	0.137	0.252	0.426	0.779
2000	9	Fry	Open	0.673	0.217	0.366	0.635	1.194
2000	9	Parr	Open	0.320	0.097	0.177	0.304	0.549
2000	10	Fry	Open	0.417	0.135	0.227	0.394	0.745
2000	10	Parr	Open	0.245	0.077	0.131	0.232	0.427
2000	11	Fry	Open	0.270	0.089	0.145	0.255	0.483
2000	11	Parr	Open	0.123	0.042	0.064	0.116	0.225
2000	12	Fry	Open	0.671	0.212	0.365	0.636	1.174
2000	12	Parr	Open	0.119	0.045	0.056	0.111	0.227
2000	13	Fry	Open	0.027	0.019	0.005	0.023	0.074
2000	13	Parr	Open	0.284	0.090	0.153	0.270	0.497
2001	14	Fry	Open	0.155	0.054	0.079	0.145	0.288
2001	14	Parr	Open	0.246	0.072	0.139	0.235	0.418
2001	15	Fry	Open	0.099	0.039	0.046	0.092	0.195
2001	15	Parr	Open	0.173	0.056	0.093	0.164	0.305
2001	16	Fry	Open	0.251	0.081	0.134	0.237	0.447
2001	16	Parr	Open	0.194	0.060	0.107	0.185	0.338
2001	76	Fry	Closed	0.158	0.039	0.097	0.153	0.247
2001	76	Parr	Closed	0.200	0.045	0.131	0.194	0.306
2001	77	Fry	Closed	0.161	0.029	0.112	0.158	0.225
2001	77	Parr	Closed	0.102	0.028	0.061	0.097	0.169
2002	17	Fry	Open	0.316	0.108	0.164	0.298	0.578
2002	17	Parr	Open	0.215	0.070	0.112	0.205	0.382
2002	18	Fry	Open	0.372	0.123	0.198	0.352	0.669
2002	18	Parr	Open	0.262	0.082	0.143	0.249	0.460
2002	19	Fry	Open	0.676	0.203	0.383	0.644	1.171
2002	19	Parr	Open	0.211	0.068	0.114	0.200	0.375
2008	20	Fry	Open	0.135	0.078	0.033	0.119	0.329
2008	20	Parr	Open	0.294	0.114	0.123	0.277	0.563
2008	21	Fry	Open	0.631	0.206	0.327	0.597	1.131
2008	21	Parr	Open	0.238	0.092	0.104	0.222	0.457
2009	22	Fry	Open	0.396	0.143	0.185	0.372	0.733
2009	22	Parr	Open	0.177	0.076	0.068	0.165	0.366
2009	23	Fry	Open	0.255	0.101	0.110	0.238	0.488
2009	23	Parr	Open	0.094	0.046	0.030	0.085	0.208
2010	24	Fry	Open	1.051	0.337	0.560	0.994	1.853
2010	24	Parr	Open	0.274	0.113	0.114	0.254	0.540
2010	25	Fry	Open	0.462	0.162	0.227	0.436	0.856
2010	25	Parr	Open	0.555	0.164	0.305	0.532	0.941

2010	26	Fry	Open	0.266	0.109	0.113	0.247	0.524
2010	26	Parr	Open	0.353	0.119	0.179	0.335	0.638
2011	27	Fry	Open	0.254	0.097	0.115	0.237	0.490
2011	27	Parr	Open	0.185	0.071	0.082	0.173	0.355
2011	28	Fry	Open	0.228	0.088	0.101	0.212	0.438
2011	28	Parr	Open	0.273	0.092	0.137	0.258	0.493
2011	29	Fry	Open	0.407	0.137	0.210	0.384	0.742
2011	29	Parr	Open	0.298	0.096	0.157	0.281	0.524
2012	30	Fry	Open	1.922	0.544	1.123	1.833	3.200
2012	30	Parr	Open	0.323	0.117	0.152	0.304	0.604
2012	31	Fry	Open	1.079	0.333	0.601	1.025	1.875
2012	31	Parr	Open	0.247	0.091	0.114	0.232	0.468
2012	32	Fry	Open	1.919	0.531	1.124	1.840	3.167
2012	32	Parr	Open	0.278	0.105	0.125	0.262	0.526
2012	33	Fry	Open	0.059	0.047	0.007	0.047	0.183
2012	33	Parr	Open	0.155	0.073	0.054	0.142	0.334
2012	34	Fry	Open	0.206	0.087	0.085	0.189	0.417
2012	34	Parr	Open	0.130	0.056	0.051	0.121	0.266
2013	35	Fry	Open	0.438	0.140	0.239	0.413	0.781
2013	35	Parr	Open	0.190	0.061	0.099	0.182	0.338
2013	36	Fry	Open	0.279	0.098	0.140	0.263	0.521
2013	36	Parr	Open	0.084	0.036	0.034	0.078	0.170
2013	37	Fry	Open	0.207	0.084	0.090	0.191	0.410
2013	37	Parr	Open	0.051	0.028	0.014	0.045	0.119
2016	38	Fry	Open	0.593	0.221	0.272	0.556	1.126
2016	38	Parr	Open	0.177	0.082	0.063	0.162	0.379
2016	39	Fry	Open	0.665	0.258	0.301	0.623	1.297
2016	39	Parr	Open	0.319	0.130	0.133	0.297	0.623
2016	40	Fry	Open	1.164	0.393	0.600	1.097	2.102
2016	40	Parr	Open	0.364	0.144	0.153	0.341	0.712
2016	41	Fry	Open	1.355	0.480	0.666	1.278	2.507
2016	41	Parr	Open	0.123	0.079	0.026	0.105	0.322
2016	42	Fry	Open	0.853	0.284	0.446	0.801	1.546
2016	42	Parr	Open	0.060	0.039	0.012	0.052	0.158
2017	43	Fry	Open	0.699	0.237	0.356	0.660	1.277
2017	43	Parr	Open	0.129	0.061	0.043	0.118	0.281
2017	44	Fry	Open	0.847	0.271	0.457	0.802	1.489
2017	44	Parr	Open	0.320	0.108	0.159	0.304	0.579
2017	45	Fry	Open	0.706	0.232	0.372	0.666	1.269
2017	45	Parr	Open	0.352	0.113	0.184	0.333	0.623
2017	46	Fry	Open	0.053	0.042	0.006	0.042	0.166
2017	46	Parr	Open	0.115	0.058	0.036	0.105	0.261
2017	47	Fry	Open	0.814	0.257	0.437	0.772	1.433
2017	47	Parr	Open	0.068	0.039	0.017	0.059	0.168
2018	48	Fry	Open	0.476	0.161	0.240	0.451	0.854

	2018	48	Parr	Open	0.844	0.229	0.500	0.809	1.392
	2018	49	Fry	Open	0.084	0.052	0.017	0.073	0.213
	2018	49	Parr	Open	0.471	0.151	0.246	0.448	0.835
	2018	50	Fry	Open	0.365	0.131	0.174	0.343	0.677
	2018	50	Parr	Open	0.327	0.111	0.161	0.310	0.594
	2018	51	Fry	Open	1.058	0.306	0.600	1.008	1.800
	2018	51	Parr	Open	0.232	0.090	0.103	0.217	0.448
	2018	52	Fry	Open	0.045	0.039	0.004	0.033	0.151
	2018	52	Parr	Open	0.539	0.175	0.276	0.511	0.953
	2019	53	Fry	Open	0.897	0.272	0.524	0.856	1.502
	2019	53	Parr	Open	0.258	0.080	0.140	0.244	0.453
	2019	54	Fry	Open	0.144	0.059	0.061	0.133	0.285
	2019	54	Parr	Open	0.054	0.026	0.018	0.049	0.118
	2019	55	Fry	Open	0.166	0.065	0.074	0.155	0.326
	2019	55	Parr	Open	0.147	0.051	0.070	0.139	0.266
	2019	56	Fry	Open	0.206	0.074	0.100	0.193	0.388
	2019	56	Parr	Open	0.050	0.024	0.017	0.045	0.110
	2019	57	Fry	Open	0.162	0.069	0.066	0.150	0.329
	2019	57	Parr	Open	0.109	0.045	0.045	0.102	0.222
	2019	58	Fry	Open	0.121	0.045	0.058	0.113	0.233
	2019	58	Parr	Open	0.032	0.015	0.011	0.030	0.070
	2019	59	Fry	Open	0.004	0.003	0.001	0.003	0.012
	2019	59	Parr	Open	0.008	0.004	0.002	0.007	0.019
	2021	351	Fry	Closed	0.337	0.049	0.251	0.334	0.442
	2021	351	Parr	Closed	0.153	0.028	0.104	0.151	0.214
	2021	352	Fry	Closed	0.270	0.048	0.189	0.266	0.375
	2021	352	Parr	Closed	0.125	0.028	0.077	0.123	0.188
	2022	355	Fry	Closed	0.198	0.039	0.132	0.195	0.285
	2022	355	Parr	Closed	0.042	0.014	0.019	0.040	0.075
	2022	356	Fry	Closed	0.446	0.054	0.350	0.442	0.564
	2022	356	Parr	Closed	0.062	0.017	0.034	0.061	0.099
Kouchibouguacis	1974	60	Fry	Open	0.010	0.011	0.000	0.006	0.041
	1974	60	Parr	Open	0.040	0.024	0.009	0.035	0.101
	1974	61	Fry	Open	0.006	0.009	0.000	0.002	0.031
	1974	61	Parr	Open	0.016	0.015	0.001	0.011	0.056
	1974	62	Fry	Open	0.005	0.009	0.000	0.002	0.030
	1974	62	Parr	Open	0.016	0.015	0.001	0.011	0.056
	2000	63	Fry	Open	0.057	0.026	0.022	0.052	0.121
	2000	63	Parr	Open	0.054	0.023	0.022	0.050	0.110
	2000	64	Fry	Open	0.049	0.026	0.014	0.044	0.116
	2000	64	Parr	Open	0.101	0.040	0.045	0.095	0.193
	2001	65	Fry	Open	0.021	0.013	0.005	0.018	0.053
	2001	65	Parr	Open	0.044	0.019	0.018	0.041	0.090
	2001	66	Fry	Open	0.065	0.032	0.021	0.059	0.143
	2001	66	Parr	Open	0.122	0.046	0.056	0.114	0.230

2001	67	Fry	Open	0.026	0.015	0.007	0.023	0.063
2001	67	Parr	Open	0.067	0.025	0.031	0.062	0.129
2002	68	Fry	Open	0.154	0.054	0.079	0.144	0.281
2002	68	Parr	Open	0.020	0.010	0.006	0.018	0.045
2002	69	Fry	Open	0.078	0.034	0.032	0.072	0.161
2002	69	Parr	Open	0.026	0.014	0.008	0.023	0.059
2002	70	Fry	Open	0.103	0.041	0.045	0.095	0.203
2002	70	Parr	Open	0.031	0.015	0.010	0.028	0.069
2003	71	Fry	Open	0.098	0.038	0.044	0.092	0.190
2003	71	Parr	Open	0.019	0.011	0.005	0.017	0.048
2003	72	Fry	Open	0.059	0.027	0.021	0.054	0.127
2003	72	Parr	Open	0.045	0.019	0.017	0.041	0.092
2003	73	Fry	Open	0.085	0.033	0.039	0.079	0.166
2003	73	Parr	Open	0.044	0.018	0.018	0.041	0.088
2008	74	Fry	Open	0.036	0.026	0.006	0.030	0.103
2008	74	Parr	Open	0.142	0.056	0.061	0.133	0.278
2008	75	Fry	Open	0.029	0.026	0.003	0.022	0.098
2008	75	Parr	Open	0.132	0.058	0.049	0.121	0.273
2008	76	Fry	Open	0.035	0.025	0.006	0.029	0.102
2008	76	Parr	Open	0.077	0.037	0.027	0.070	0.167
2009	77	Fry	Open	0.131	0.071	0.038	0.117	0.304
2009	77	Parr	Open	0.003	0.008	0.000	0.000	0.022
2010	78	Fry	Open	0.280	0.111	0.121	0.260	0.551
2010	78	Parr	Open	0.059	0.033	0.015	0.052	0.141
2010	79	Fry	Open	0.450	0.152	0.230	0.424	0.811
2010	79	Parr	Open	0.058	0.030	0.017	0.052	0.131
2010	80	Fry	Open	0.331	0.127	0.149	0.309	0.644
2010	80	Parr	Open	0.036	0.024	0.007	0.031	0.095
2011	81	Fry	Open	0.041	0.026	0.009	0.036	0.106
2011	81	Parr	Open	0.087	0.037	0.034	0.081	0.175
2011	82	Fry	Open	0.012	0.013	0.001	0.008	0.049
2011	82	Parr	Open	0.029	0.019	0.006	0.024	0.077
2011	83	Fry	Open	0.073	0.038	0.021	0.065	0.167
2011	83	Parr	Open	0.082	0.036	0.030	0.076	0.169
2011	345	Fry	Closed	0.016	0.007	0.006	0.015	0.033
2011	345	Parr	Closed	0.064	0.015	0.040	0.063	0.097
2011	346	Fry	Closed	0.012	0.006	0.003	0.010	0.026
2011	346	Parr	Closed	0.056	0.014	0.033	0.055	0.088
2013	147	Fry	Open	0.001	0.002	0.000	0.000	0.007
2013	147	Parr	Open	0.018	0.012	0.003	0.016	0.048
2014	148	Fry	Open	0.068	0.029	0.027	0.063	0.139
2014	148	Parr	Open	0.028	0.015	0.008	0.025	0.063
2014	149	Fry	Open	0.039	0.020	0.012	0.035	0.089
2014	149	Parr	Open	0.012	0.008	0.002	0.010	0.033
2015	84	Fry	Open	0.039	0.033	0.004	0.029	0.126

2015	84	Parr	Open	0.013	0.011	0.002	0.010	0.043
2015	85	Fry	Open	0.129	0.063	0.043	0.118	0.284
2015	85	Parr	Open	0.012	0.010	0.002	0.010	0.040
2015	86	Fry	Open	0.036	0.030	0.004	0.027	0.118
2015	86	Parr	Open	0.020	0.016	0.003	0.016	0.062
2015	150	Fry	Open	0.122	0.044	0.058	0.115	0.229
2015	150	Parr	Open	0.028	0.014	0.009	0.025	0.061
2015	339	Fry	Closed	0.008	0.007	0.001	0.006	0.026
2015	339	Parr	Closed	0.008	0.006	0.002	0.007	0.023
2015	340	Fry	Closed	0.005	0.004	0.001	0.004	0.014
2015	340	Parr	Closed	0.006	0.003	0.001	0.005	0.014
2015	341	Fry	Closed	0.113	0.022	0.075	0.111	0.162
2015	341	Parr	Closed	0.054	0.013	0.032	0.053	0.084
2016	87	Fry	Open	0.056	0.050	0.005	0.041	0.195
2016	87	Parr	Open	0.017	0.016	0.002	0.012	0.058
2016	88	Fry	Open	0.068	0.065	0.006	0.049	0.247
2016	88	Parr	Open	0.010	0.011	0.001	0.007	0.040
2016	89	Fry	Open	0.063	0.060	0.005	0.046	0.222
2016	89	Parr	Open	0.010	0.011	0.001	0.007	0.039
2016	142	Fry	Open	0.535	0.177	0.272	0.508	0.952
2016	142	Parr	Open	0.065	0.037	0.016	0.057	0.155
2016	143	Fry	Open	0.276	0.109	0.118	0.258	0.540
2016	143	Parr	Open	0.009	0.008	0.001	0.006	0.030
2016	342	Fry	Closed	0.009	0.007	0.001	0.006	0.029
2016	342	Parr	Closed	0.006	0.005	0.001	0.005	0.019
2016	343	Fry	Closed	0.004	0.003	0.001	0.003	0.012
2016	343	Parr	Closed	0.004	0.003	0.001	0.003	0.011
2016	344	Fry	Closed	0.005	0.004	0.001	0.004	0.015
2016	344	Parr	Closed	0.004	0.003	0.001	0.004	0.012
2017	90	Fry	Open	0.737	0.254	0.368	0.696	1.338
2017	90	Parr	Open	0.213	0.089	0.086	0.197	0.426
2017	91	Fry	Open	0.621	0.205	0.318	0.589	1.114
2017	91	Parr	Open	0.056	0.033	0.013	0.049	0.138
2017	92	Fry	Open	0.614	0.209	0.312	0.581	1.113
2017	92	Parr	Open	0.087	0.045	0.027	0.078	0.197
2017	144	Fry	Open	0.068	0.065	0.007	0.049	0.239
2017	144	Parr	Open	0.054	0.040	0.010	0.043	0.154
2017	145	Fry	Open	0.672	0.249	0.308	0.632	1.271
2017	145	Parr	Open	0.185	0.091	0.061	0.168	0.406
2017	146	Fry	Open	0.970	0.314	0.508	0.918	1.717
2017	146	Parr	Open	0.143	0.067	0.050	0.130	0.304
2018	93	Fry	Open	0.064	0.046	0.010	0.053	0.187
2018	93	Parr	Open	0.106	0.055	0.032	0.096	0.238
2018	94	Fry	Open	0.095	0.060	0.019	0.082	0.241
2018	94	Parr	Open	0.247	0.099	0.105	0.230	0.481

		2018	95	Fry	Open	0.038	0.032	0.004	0.030	0.120
		2018	95	Parr	Open	0.131	0.060	0.047	0.120	0.282
		2019	96	Fry	Open	0.185	0.073	0.083	0.172	0.361
		2019	96	Parr	Open	0.022	0.015	0.004	0.019	0.060
		2019	97	Fry	Open	0.134	0.059	0.050	0.123	0.279
		2019	97	Parr	Open	0.047	0.025	0.013	0.042	0.111
		2019	98	Fry	Open	0.089	0.040	0.033	0.082	0.186
		2019	98	Parr	Open	0.032	0.018	0.008	0.028	0.078
TNNP	Arnold's	2002	1	Fry	Closed	0.091	0.035	0.037	0.086	0.172
		2002	1	Parr	Closed	0.205	0.050	0.118	0.201	0.315
		2002	8	Fry	Closed	0.002	0.005	0.000	0.000	0.017
		2002	8	Parr	Closed	0.016	0.016	0.001	0.011	0.061
		2002	10	Fry	Closed	0.001	0.003	0.000	0.000	0.009
		2002	10	Parr	Closed	0.017	0.012	0.002	0.014	0.047
		2003	23	Fry	Closed	0.002	0.005	0.000	0.000	0.016
		2003	23	Parr	Closed	0.031	0.022	0.004	0.026	0.086
	Big	2003	13	Fry	Closed	0.002	0.002	0.000	0.002	0.008
		2003	13	Parr	Closed	0.006	0.004	0.001	0.005	0.015
		2003	14	Fry	Closed	0.000	0.001	0.000	0.000	0.003
		2003	14	Parr	Closed	0.012	0.006	0.004	0.011	0.027
		2003	27	Fry	Closed	0.024	0.007	0.012	0.024	0.041
		2003	27	Parr	Closed	0.022	0.007	0.011	0.021	0.038
		2005	31	Fry	Closed	0.081	0.021	0.046	0.079	0.127
		2005	31	Parr	Closed	0.005	0.005	0.000	0.004	0.019
	Big Pond	2003	11	Fry	Closed	0.266	0.042	0.190	0.263	0.356
		2003	11	Parr	Closed	0.081	0.023	0.043	0.079	0.131
		2003	12	Fry	Closed	0.533	0.053	0.434	0.530	0.643
		2003	12	Parr	Closed	0.281	0.036	0.215	0.279	0.357
	Blind	2005	28	Fry	Closed	0.001	0.002	0.000	0.000	0.006
		2005	28	Parr	Closed	0.046	0.015	0.021	0.045	0.081
		2008	37	Fry	Closed	0.001	0.002	0.000	0.000	0.006
		2008	37	Parr	Closed	0.015	0.009	0.003	0.013	0.038
		2009	40	Fry	Closed	0.001	0.002	0.000	0.000	0.007
		2009	40	Parr	Closed	0.019	0.012	0.004	0.017	0.050
		2010	44	Fry	Closed	0.001	0.002	0.000	0.000	0.007
		2010	44	Parr	Closed	0.030	0.013	0.010	0.028	0.062
		2011	50	Fry	Closed	0.001	0.002	0.000	0.000	0.008
		2011	50	Parr	Closed	0.088	0.024	0.048	0.085	0.143
		2012	62	Fry	Closed	0.001	0.003	0.000	0.000	0.010
		2012	62	Parr	Closed	0.166	0.041	0.097	0.163	0.254
	Blue Hill	2003	22	Fry	Closed	0.157	0.040	0.089	0.153	0.244
		2003	22	Parr	Closed	0.465	0.070	0.338	0.461	0.612
	Cobblers	2003	16	Fry	Closed	0.001	0.003	0.000	0.000	0.008
		2003	16	Parr	Closed	0.030	0.015	0.008	0.027	0.066
		2003	17	Fry	Closed	0.001	0.002	0.000	0.000	0.007

	2003	17	Parr	Closed	0.032	0.014	0.010	0.030	0.065
Davey Ann's	2011	55	Fry	Closed	0.001	0.004	0.000	0.000	0.011
	2011	55	Parr	Closed	0.021	0.015	0.003	0.017	0.060
Hefferns	2011	52	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2011	52	Parr	Closed	0.007	0.007	0.000	0.005	0.025
Minchins	2003	19	Fry	Closed	0.110	0.024	0.069	0.108	0.161
	2003	19	Parr	Closed	0.603	0.060	0.495	0.600	0.731
	2009	41	Fry	Closed	0.090	0.023	0.054	0.088	0.142
	2009	41	Parr	Closed	0.189	0.044	0.128	0.182	0.297
	2011	54	Fry	Closed	0.006	0.006	0.000	0.004	0.021
	2011	54	Parr	Closed	0.089	0.022	0.051	0.087	0.139
North Broad Cove	2005	33	Fry	Closed	1.533	0.123	1.315	1.526	1.785
	2005	33	Parr	Closed	0.001	0.002	0.000	0.000	0.006
	2009	42	Fry	Closed	1.448	0.181	1.211	1.420	1.848
	2009	42	Parr	Closed	0.134	0.040	0.077	0.128	0.228
	2011	56	Fry	Closed	0.104	0.025	0.061	0.102	0.160
	2011	56	Parr	Closed	0.212	0.036	0.148	0.209	0.289
Park Harbour	2005	32	Fry	Closed	0.477	0.059	0.371	0.474	0.602
	2005	32	Parr	Closed	0.027	0.012	0.009	0.025	0.057
	2008	36	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2008	36	Parr	Closed	0.006	0.007	0.000	0.004	0.024
	2010	48	Fry	Closed	0.151	0.033	0.094	0.148	0.220
	2010	48	Parr	Closed	0.416	0.054	0.320	0.414	0.531
	2011	53	Fry	Closed	0.376	0.058	0.274	0.372	0.501
	2011	53	Parr	Closed	0.073	0.022	0.036	0.071	0.124
	2012	60	Fry	Closed	0.601	0.081	0.457	0.595	0.774
	2012	60	Parr	Closed	0.095	0.028	0.048	0.092	0.158
Saltons	2005	30	Fry	Closed	0.001	0.002	0.000	0.000	0.006
	2005	30	Parr	Closed	0.010	0.007	0.001	0.009	0.029
South Broad Cove	2010	49	Fry	Closed	0.024	0.014	0.005	0.021	0.059
	2010	49	Parr	Closed	0.001	0.003	0.000	0.000	0.009
Southwest	2005	29	Fry	Closed	0.006	0.005	0.001	0.005	0.018
	2005	29	Parr	Closed	0.193	0.026	0.146	0.191	0.248
	2008	38	Fry	Closed	0.008	0.005	0.001	0.006	0.021
	2008	38	Parr	Closed	0.107	0.020	0.072	0.106	0.151
	2010	46	Fry	Closed	0.022	0.009	0.009	0.021	0.042
	2010	46	Parr	Closed	0.164	0.023	0.122	0.163	0.211
	2011	57	Fry	Closed	0.008	0.005	0.001	0.006	0.021
	2011	57	Parr	Closed	0.146	0.024	0.102	0.144	0.197
Terra Nova	2002	2	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2002	2	Parr	Closed	0.075	0.023	0.038	0.073	0.127
	2002	3	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2002	3	Parr	Closed	0.138	0.029	0.086	0.135	0.200
	2002	4	Fry	Closed	0.001	0.002	0.000	0.000	0.006
	2002	4	Parr	Closed	0.258	0.038	0.190	0.256	0.338

	2002	5	Fry	Closed	0.001	0.003	0.000	0.000	0.008
	2002	5	Parr	Closed	0.023	0.014	0.005	0.020	0.057
	2002	6	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2002	6	Parr	Closed	0.013	0.010	0.002	0.011	0.038
	2002	7	Fry	Closed	0.001	0.003	0.000	0.000	0.008
	2002	7	Parr	Closed	0.029	0.015	0.008	0.026	0.066
	2002	9	Fry	Closed	0.001	0.002	0.000	0.000	0.006
	2002	9	Parr	Closed	0.006	0.006	0.000	0.004	0.022
	2003	15	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2003	15	Parr	Closed	0.047	0.017	0.020	0.045	0.085
	2003	18	Fry	Closed	0.001	0.003	0.000	0.000	0.008
	2003	18	Parr	Closed	0.015	0.011	0.002	0.012	0.042
	2003	24	Fry	Closed	0.001	0.003	0.000	0.000	0.008
	2003	24	Parr	Closed	0.081	0.025	0.041	0.079	0.137
	2003	25	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2003	25	Parr	Closed	0.013	0.009	0.002	0.011	0.037
	2003	26	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2003	26	Parr	Closed	0.044	0.017	0.018	0.042	0.083
	2006	35	Fry	Closed	0.006	0.006	0.000	0.005	0.023
	2006	35	Parr	Closed	0.137	0.029	0.086	0.135	0.200
	2010	45	Fry	Closed	0.001	0.002	0.000	0.000	0.006
	2010	45	Parr	Closed	0.030	0.013	0.010	0.028	0.061
	2011	51	Fry	Closed	0.001	0.002	0.000	0.000	0.007
	2011	51	Parr	Closed	0.185	0.036	0.120	0.182	0.264
	2012	59	Fry	Closed	0.001	0.003	0.000	0.000	0.008
	2012	59	Parr	Closed	0.077	0.033	0.033	0.071	0.155
Wings	2003	20	Fry	Closed	0.183	0.036	0.121	0.181	0.261
	2003	20	Parr	Closed	0.260	0.040	0.188	0.258	0.344
	2003	21	Fry	Closed	0.317	0.052	0.225	0.313	0.430
	2003	21	Parr	Closed	0.430	0.059	0.326	0.427	0.552
	2006	34	Fry	Closed	0.767	0.076	0.628	0.764	0.923
	2006	34	Parr	Closed	0.262	0.041	0.189	0.260	0.349
	2008	39	Fry	Closed	0.427	0.059	0.318	0.424	0.551
	2008	39	Parr	Closed	0.387	0.054	0.289	0.385	0.501
	2009	43	Fry	Closed	0.782	0.125	0.602	0.765	1.072
	2009	43	Parr	Closed	0.369	0.077	0.261	0.356	0.552
	2010	47	Fry	Closed	2.054	0.166	1.762	2.042	2.407
	2010	47	Parr	Closed	0.479	0.058	0.375	0.476	0.601
	2011	58	Fry	Closed	0.028	0.014	0.008	0.026	0.061
	2011	58	Parr	Closed	0.152	0.033	0.095	0.149	0.224
	2012	61	Fry	Closed	1.334	0.135	1.092	1.325	1.620
	2012	61	Parr	Closed	0.612	0.073	0.478	0.608	0.765

Appendix II – Posterior distributions of annual densities

Park	River	Year	Life Stage	Mean	SD	2.5%	Median	97.5%
KNP	Kouchibouguac River	1974	Fry	0.002	0.003	0.000	0.001	0.011
			Parr	0.164	0.066	0.072	0.152	0.323
		1977	Fry	0.075	0.046	0.022	0.064	0.195
			Parr	0.114	0.041	0.055	0.108	0.212
		1978	Fry	1.195	0.880	0.261	0.975	3.432
			Parr	0.086	0.038	0.034	0.079	0.180
		1982	Fry	0.044	0.042	0.006	0.032	0.151
			Parr	0.064	0.032	0.022	0.056	0.144
		1999	Fry	0.959	1.124	0.091	0.614	3.866
			Parr	0.540	0.373	0.136	0.445	1.508
		2000	Fry	0.299	0.208	0.071	0.246	0.840
			Parr	0.208	0.086	0.087	0.192	0.416
		2001	Fry	0.183	0.122	0.048	0.153	0.497
			Parr	0.182	0.073	0.079	0.169	0.356
		2002	Fry	0.568	0.555	0.090	0.418	1.931
			Parr	0.247	0.133	0.081	0.217	0.593
		2008	Fry	0.435	0.575	0.037	0.264	1.913
			Parr	0.302	0.218	0.071	0.246	0.854
		2009	Fry	0.479	0.610	0.045	0.298	2.033
			Parr	0.147	0.110	0.031	0.118	0.439
		2010	Fry	0.644	0.617	0.100	0.472	2.182
			Parr	0.406	0.221	0.131	0.357	0.948
		2011	Fry	0.370	0.363	0.059	0.271	1.261
			Parr	0.267	0.148	0.085	0.233	0.631
		2012	Fry	0.596	0.407	0.141	0.491	1.659
			Parr	0.215	0.093	0.087	0.197	0.440
		2013	Fry	0.379	0.348	0.060	0.278	1.277
			Parr	0.099	0.058	0.029	0.086	0.243
		2016	Fry	1.011	0.713	0.245	0.836	2.782
			Parr	0.169	0.080	0.061	0.154	0.366
		2017	Fry	0.493	0.338	0.118	0.410	1.359
			Parr	0.162	0.073	0.065	0.148	0.347
		2018	Fry	0.250	0.181	0.055	0.203	0.724
			Parr	0.452	0.188	0.192	0.416	0.926
		2019	Fry	0.126	0.073	0.038	0.109	0.316
			Parr	0.059	0.022	0.026	0.055	0.112
		2021	Fry	0.463	0.560	0.048	0.299	1.885
			Parr	0.160	0.102	0.043	0.136	0.427
		2022	Fry	0.458	0.562	0.048	0.298	1.812
			Parr	0.059	0.041	0.015	0.049	0.162
	Kouchibouguacis	1974	Fry	0.007	0.012	0.000	0.003	0.037

			Parr	0.021	0.018	0.003	0.016	0.068
		2000	Fry	0.080	0.103	0.007	0.049	0.343
			Parr	0.085	0.061	0.020	0.070	0.246
		2001	Fry	0.041	0.044	0.006	0.028	0.149
			Parr	0.076	0.043	0.024	0.066	0.184
		2002	Fry	0.140	0.135	0.021	0.102	0.491
			Parr	0.027	0.016	0.007	0.023	0.069
		2003	Fry	0.101	0.097	0.015	0.073	0.359
			Parr	0.035	0.021	0.010	0.031	0.090
		2008	Fry	0.039	0.044	0.004	0.026	0.154
			Parr	0.120	0.069	0.036	0.104	0.295
		2009	Fry	0.305	0.672	0.007	0.118	1.716
			Parr	0.004	0.016	0.000	0.000	0.033
		2010	Fry	0.446	0.417	0.065	0.329	1.516
			Parr	0.051	0.034	0.012	0.042	0.137
		2011	Fry	0.024	0.017	0.005	0.019	0.069
			Parr	0.060	0.026	0.024	0.055	0.122
		2013	Fry	0.002	0.025	0.000	0.000	0.016
			Parr	0.026	0.034	0.002	0.016	0.114
		2014	Fry	0.079	0.104	0.007	0.047	0.342
			Parr	0.020	0.016	0.003	0.016	0.064
		2015	Fry	0.035	0.022	0.009	0.030	0.091
			Parr	0.014	0.007	0.005	0.013	0.031
		2016	Fry	0.036	0.022	0.009	0.030	0.094
			Parr	0.009	0.005	0.002	0.008	0.021
		2017	Fry	0.509	0.312	0.136	0.439	1.287
			Parr	0.102	0.045	0.039	0.093	0.213
		2018	Fry	0.073	0.081	0.008	0.050	0.288
			Parr	0.158	0.092	0.046	0.137	0.395
		2019	Fry	0.168	0.168	0.025	0.120	0.590
			Parr	0.033	0.022	0.008	0.028	0.092
	Black River	2021	Fry	0.107	0.082	0.020	0.084	0.322
			Parr	0.036	0.017	0.013	0.033	0.078
		2022	Fry	0.082	0.065	0.017	0.065	0.248
			Parr	0.095	0.043	0.036	0.087	0.201
FNP	Point Wolfe River	1984	Fry	0.000	0.001	0.000	0.000	0.001
			Parr	0.036	0.022	0.010	0.031	0.091
		1985	Fry	0.016	0.037	0.001	0.007	0.082
			Parr	0.026	0.019	0.006	0.022	0.076
		1987	Fry	0.439	1.340	0.011	0.162	2.503
			Parr	0.228	0.194	0.039	0.173	0.723
		1988	Fry	0.170	0.187	0.020	0.116	0.657
			Parr	0.087	0.042	0.033	0.079	0.191
		1989	Fry	0.026	0.031	0.003	0.017	0.107
			Parr	0.041	0.020	0.015	0.037	0.090

1990	Fry	0.041	0.049	0.005	0.028	0.158
	Parr	0.023	0.012	0.008	0.021	0.054
1991	Fry	0.089	0.169	0.005	0.047	0.425
	Parr	0.126	0.080	0.034	0.106	0.332
1992	Fry	0.147	0.258	0.009	0.076	0.725
	Parr	0.055	0.036	0.015	0.046	0.151
1993	Fry	0.031	0.036	0.003	0.021	0.127
	Parr	0.093	0.044	0.034	0.084	0.202
1994	Fry	0.007	0.010	0.001	0.005	0.031
	Parr	0.058	0.028	0.022	0.053	0.127
1995	Fry	0.009	0.012	0.001	0.006	0.036
	Parr	0.029	0.015	0.010	0.026	0.066
1996	Fry	0.002	0.002	0.000	0.001	0.007
	Parr	0.014	0.008	0.005	0.013	0.034
1997	Fry	0.011	0.013	0.001	0.007	0.043
	Parr	0.005	0.003	0.001	0.004	0.013
1999	Fry	0.001	0.002	0.000	0.000	0.005
	Parr	0.001	0.001	0.000	0.001	0.004
2000	Fry	0.000	0.000	0.000	0.000	0.001
	Parr	0.006	0.004	0.002	0.006	0.016
2001	Fry	0.008	0.009	0.001	0.006	0.033
	Parr	0.003	0.002	0.001	0.002	0.007
2002	Fry	0.001	0.002	0.000	0.001	0.006
	Parr	0.007	0.004	0.002	0.007	0.017
2003	Fry	0.001	0.001	0.000	0.000	0.004
	Parr	0.013	0.006	0.005	0.012	0.029
2004	Fry	0.027	0.033	0.003	0.017	0.107
	Parr	0.003	0.002	0.001	0.003	0.009
2005	Fry	0.002	0.003	0.000	0.001	0.009
	Parr	0.029	0.014	0.011	0.026	0.062
2006	Fry	0.004	0.004	0.000	0.002	0.014
	Parr	0.003	0.002	0.001	0.002	0.007
2015	Fry	0.000	0.001	0.000	0.000	0.003
	Parr	0.000	0.001	0.000	0.000	0.003
2016	Fry	0.000	0.001	0.000	0.000	0.003
	Parr	0.003	0.003	0.000	0.002	0.010
2017	Fry	0.014	0.009	0.003	0.012	0.038
	Parr	0.000	0.001	0.000	0.000	0.004
2018	Fry	0.000	0.000	0.000	0.000	0.001
	Parr	0.018	0.008	0.007	0.017	0.037
2019	Fry	0.008	0.004	0.003	0.007	0.018
	Parr	0.003	0.002	0.001	0.003	0.008
2020	Fry	0.010	0.004	0.004	0.009	0.021
	Parr	0.012	0.004	0.005	0.011	0.021
2021	Fry	0.089	0.029	0.046	0.085	0.159

		Parr	0.025	0.007	0.014	0.024	0.042
	2022	Fry	0.124	0.041	0.064	0.118	0.221
		Parr	0.059	0.016	0.033	0.058	0.096
Upper Salmon River	1984	Fry	0.347	0.536	0.020	0.184	1.675
		Parr	0.096	0.067	0.023	0.079	0.267
	1985	Fry	0.317	0.350	0.039	0.216	1.202
		Parr	0.178	0.084	0.066	0.162	0.388
	1987	Fry	0.071	0.085	0.009	0.050	0.255
		Parr	0.351	0.148	0.146	0.322	0.713
	1993	Fry	0.076	0.174	0.002	0.030	0.434
		Parr	0.160	0.131	0.031	0.126	0.495
	1994	Fry	0.036	0.134	0.001	0.013	0.209
		Parr	0.137	0.111	0.027	0.108	0.421
	1995	Fry	0.041	0.123	0.001	0.015	0.221
		Parr	0.023	0.019	0.004	0.017	0.073
	1996	Fry	0.026	0.081	0.001	0.009	0.152
		Parr	0.022	0.020	0.004	0.016	0.072
	1997	Fry	0.036	0.097	0.001	0.012	0.216
		Parr	0.019	0.018	0.003	0.014	0.063
	1999	Fry	0.009	0.045	0.000	0.003	0.056
		Parr	0.007	0.007	0.001	0.005	0.026
	2000	Fry	0.000	0.002	0.000	0.000	0.002
		Parr	0.014	0.013	0.002	0.010	0.047
	2001	Fry	0.003	0.004	0.000	0.002	0.012
		Parr	0.012	0.006	0.004	0.011	0.028
	2002	Fry	0.000	0.000	0.000	0.000	0.001
		Parr	0.003	0.002	0.001	0.003	0.010
	2003	Fry	0.000	0.000	0.000	0.000	0.000
		Parr	0.001	0.001	0.000	0.001	0.003
	2005	Fry	0.001	0.001	0.000	0.000	0.005
		Parr	0.000	0.000	0.000	0.000	0.001
	2006	Fry	0.059	0.061	0.009	0.042	0.215
		Parr	0.000	0.000	0.000	0.000	0.001
	2015	Fry	0.000	0.000	0.000	0.000	0.001
		Parr	0.004	0.002	0.001	0.004	0.010
	2016	Fry	0.000	0.000	0.000	0.000	0.001
		Parr	0.000	0.000	0.000	0.000	0.002
	2017	Fry	0.064	0.011	0.044	0.063	0.088
		Parr	0.000	0.000	0.000	0.000	0.001
	2018	Fry	0.006	0.003	0.002	0.006	0.012
		Parr	0.025	0.007	0.015	0.025	0.040
	2019	Fry	0.084	0.027	0.042	0.080	0.149
		Parr	0.010	0.004	0.004	0.009	0.019
	2020	Fry	0.001	0.001	0.000	0.001	0.004
		Parr	0.123	0.032	0.072	0.120	0.196

		2021	Fry	0.009	0.004	0.003	0.008	0.018
			Parr	0.011	0.004	0.005	0.010	0.019
		2022	Fry	0.103	0.034	0.052	0.098	0.183
			Parr	0.014	0.005	0.006	0.013	0.026
CBHNP	Clyburn	2012	Fry	0.370	9.329	0.004	0.070	1.430
			Parr	0.023	0.147	0.000	0.009	0.109
		2015	Fry	0.398	18.952	0.002	0.047	1.067
			Parr	0.004	0.016	0.000	0.000	0.034
		2016	Fry	0.197	0.992	0.009	0.092	0.881
			Parr	0.028	0.037	0.002	0.018	0.114
		2017	Fry	0.248	0.372	0.027	0.160	0.982
			Parr	0.051	0.047	0.010	0.039	0.163
		2018	Fry	0.160	0.514	0.009	0.079	0.787
			Parr	0.038	0.048	0.004	0.026	0.141
		2019	Fry	0.078	0.315	0.004	0.039	0.354
			Parr	0.015	0.021	0.002	0.010	0.056
		2020	Fry	0.134	0.279	0.014	0.088	0.516
			Parr	0.126	0.102	0.040	0.106	0.336
		2021	Fry	0.068	0.088	0.009	0.049	0.249
			Parr	0.154	0.102	0.054	0.133	0.385
		2022	Fry	0.002	0.004	0.000	0.001	0.010
			Parr	0.043	0.030	0.012	0.037	0.110
GMNP	Trout River	2002	Fry	0.004	0.006	0.000	0.002	0.020
			Parr	0.109	0.061	0.036	0.096	0.262
		2011	Fry	0.128	0.119	0.021	0.095	0.428
			Parr	0.104	0.056	0.035	0.093	0.247
		2013	Fry	0.066	0.064	0.010	0.049	0.227
			Parr	0.070	0.039	0.023	0.062	0.167
		2014	Fry	0.064	0.125	0.004	0.034	0.300
			Parr	0.022	0.022	0.003	0.016	0.077
		2015	Fry	0.113	0.053	0.040	0.103	0.240
			Parr	0.085	0.026	0.045	0.082	0.143
		2017	Fry	0.140	0.053	0.064	0.131	0.267
			Parr	0.036	0.010	0.020	0.035	0.058
		2019	Fry	0.064	0.025	0.028	0.060	0.125
			Parr	0.013	0.004	0.006	0.013	0.023
		2021	Fry	0.170	0.086	0.060	0.151	0.388
			Parr	0.017	0.007	0.007	0.016	0.034
		2022	Fry	0.065	0.048	0.015	0.052	0.184
			Parr	0.037	0.018	0.014	0.034	0.081
	Western Brook	2020	Fry	1.732	2.837	0.130	1.001	7.780
			Parr	0.205	0.201	0.042	0.160	0.626

Curriculum Vitae

John Robinson

Bachelor of Wildlife Conservation, University of Prince Edward Island, 2019

Diploma in Environmental Sciences: Wildlife and Fisheries Conservation, Lakeland College, 2017

Conference Presentations:

“River Restoration in Action: Recovering Endangered Atlantic Salmon and Aquatic Ecosystems One National Park at a Time” - Fisheries Society of the British Iles 2023 International Annual Symposium. Fish Habitat Ecology in a Changing Climate. University of Essex, UK.

“Does One Size Fit All?: Assessing Salmon Restoration Strategies Across Five National Parks” - Society of Canadian Aquatic Sciences 2023. Montreal, Quebec.