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OUR FILE NO. M6387-000

November 21, 2024

VIA EMAIL: DFO.MINISTER-MINISTRE.MPO@DFO-MPO.GC.CA

The Honourable Diane Lebouthillier
Minister of Fisheries, Oceans and the Canadian Coast Guard

Minister's Office
200 Kent St
Station 15N100
Ottawa ON K1A 0E6

Dear Minister:

Re: Request for Immediate Action – Science Based Plan to Save Miramichi Atlantic Salmon

We are the solicitors for Save Miramichi Salmon Inc., a new association based on the Miramichi River in New Brunswick.

Our client was formed to advance the cause of preserving and protecting the native Miramichi Atlantic salmon. Our client’s members are a collection of stakeholders on the Miramichi River, including owners of riparian land, private fisheries, and sport fishing guiding and outfitting businesses all of which have direct interests, both personal and pecuniary, in the well-being of Atlantic salmon. Their individual experiences all stretch back three or four decades with one member/directors family’s involvement in the salmon fishery spanning more than a century.

We write to you about a matter of utmost urgency and to demand your immediate action. The Miramichi Atlantic salmon are at a crisis point which now requires your immediate, vigorous and timely intervention to prevent the inevitable collapse of the species within the near future. This crisis has been precipitated by an explosion of striped bass which is a voracious predatory species of fish. The striped bass population must be brought under control without delay.

To assist, we have enclosed with this letter a scientific briefing prepared by our clients with assistance from noted fisheries biologists, former eminent senior DFO scientist, Dr. John Ritter and Dr. R. Allen Curry, a senior scientist with the Canadian Rivers Institute and University of New Brunswick professor. The briefing is based on literature, much of which is peer reviewed, and evidence collected from the river, both of which we have also enclosed.

The Miramichi Atlantic salmon, or “plamu” in Mi’kmaq, are an iconic species of Canadian fish, central to the cultures of both the indigenous and non-indigenous people on the Miramichi. They have been present in the Miramichi River since the end of the last ice age. While we are not experts on Mi’kmaq culture and history, we understand that Atlantic salmon has similar significance to the Mi’kmaq people as the various Pacific salmon species has to the various First Nations in British Columbia.

Like most other anadromous salmon species, mature Atlantic salmon spawn in freshwater from the headwaters of the Miramichi to the lower non-tidal end of the river. After hatching, juvenile salmon spend 2-3 years in the river; then as smolts they migrate in the months of May and June from freshwater to the ocean, where they spend one, two, or three years before returning to the watershed to spawn. As you know, unlike Pacific salmon they can survive spawning and some return to the rivers to spawn multiple times. This smolt outward migration must run the gauntlet of spawning and aggressively feeding striped bass which electronic tagging has proved have reduced the percentage of salmon smolts that make it out of the river and into the ocean from 70% 15 years ago to less than 10% today in the NW Miramichi and 30% in the SW Miramichi. This extraordinary level of predation is completely unsustainable and is on the verge of extirpating Atlantic salmon from what has long been considered the greatest salmon river in North America.

Striped bass, while historically native to the Gulf of St. Lawrence and the Miramichi watershed, were never a dominant species in the overall ecosystem, nor were they established in any significant numbers beyond the head of tide. There thus existed a natural separation within the ecosystem between adult striped bass, which generally frequented tidal waters, and juvenile salmon, which were primarily found in non-tidal waters.

More recently, in the last 20 years, from the observations of my clients, striped bass have spread throughout the river into the habitat of juvenile salmon and are now aggressively preying on them well before their smolt migration.

The collapse of Atlantic salmon stocks of the Miramichi River now appears imminent without decisive science-based action. This emergency has been precipitated, in large part, by your department’s patent failure to correctly interpret the scientific data and consequent failure to properly manage the populations of striped bass, Atlantic salmon, and other species in an ecologically balanced proactive manner.

In the 1980s, the populations of striped bass on the Miramichi were significantly lower than today but generally viewed as nonetheless stable and healthy. One of our client’s members advises us, for example, that New England striped bass conservationists in the early 1980s viewed the Miramichi Striped Bass populations to be the only healthy stock on the North American eastern seaboard; while at the same time their numbers were such that they were seldom observed above the head of tide and were never reported as being a significant threat to migrating smolts or to juvenile salmon in their upriver habitat.

This balance, however, was disrupted by your department in the 1990s, when it determined that the Miramichi striped bass were at risk. What followed was a comprehensive recovery effort to “rebuild” striped bass populations primarily through closing the historic annual commercial net and sport fisheries.

This effort led to a decades long monotonic growth of the striped bass population, which in turn elevated populations to what is now, a destructive level that is disrupting the delicate balance of the Miramichi ecosystem.

Striped bass are now prevalent throughout much of the Miramichi watershed and, as our client's submission makes clear, are devouring migrating smolts at unprecedented levels. We also understand that striped bass have also now infiltrated freshwater habitat and are attacking Atlantic salmon at all levels of their juvenile development.

This has put significant and untenable pressure on Atlantic salmon stocks in the Miramichi. Both the scientific and anecdotal evidence assembled by our client is clear and persuasive that:

- Striped bass populations are now at unsustainably high levels;
- DFO's population targets and limits for striped bass are incorrectly and artificially high; and
- The large numbers of striped bass throughout the Miramichi watershed have caused a precipitous and continuing decline of the Miramichi's Atlantic salmon population.

For reasons which defy good and sound science, your department has taken no meaningful steps to bring the striped bass population back into check. This inaction appears to be based on an elevated limit reference point and population targets which are not supported by common sense or the weight of science, which is explained in the enclosed submission from our client. The result is that Atlantic salmon in the Miramichi are now at risk of extirpation if immediate and decisive action is not taken.

You have been entrusted by Parliament, under the *Fisheries Act* to "manage, conserve, and develop "the fishery" on behalf of Canadians in the public interest."[1] It bears repeating that the fishery does not belong to the King; you are merely the steward of common property. Canada's fishery is a "common property resource" which is a source of "national or provincial wealth". All Canadians possess rights with varying degrees of priority to access this resource. The *Fisheries Act* provides you with both the tools to regulate the exercise of those rights,[2] together with a duty to "manage, conserve, and develop" the object of those rights.

The scope and application of your duty over this most important resource is necessarily conditioned and informed by the precautionary principle, which requires that:

a) Policies must be based on the precautionary principle to ensure sustainable development;

b) Environmental measures must anticipate, prevent and attack the causes of environmental degradation; and

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[1] *Comeau's Sea Foods Ltd. v. Canada (Minister of Fisheries and Oceans)*, [1997] 1 S.C.R. 12, at para. 37
[2] See *R v. Sparrow*, [1990] 1 S.C.R. 1075

c) Where there are threats of serious or irreversible damage, lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation.[3]

Management of the striped bass in a manner which puts other fisheries at risk is a clear breach of your duty as described above. This breach is compounded by a pattern of further inaction and inertia. Your current approach of drawn-out public consultation and lengthy implementation timelines, focus on abstracts, and combined with further restrictions on uses which have minimal-to-no impact on striped bass mortalities, is inadequate to combat the problem at hand and does not respect the rights and interests you have been entrusted stewardship over.

Recent measures announced by DFO to reduce the population do not go nearly far enough and are exceedingly unlikely to achieve the necessary result given the current gross ecological imbalance. The recreational creel limit increase is modest at best, and even with the increased indigenous commercial quota, it is hard to see how it will result in material reductions to the striped bass populations when the existing quota is not being fully utilized. The 2025 increased harvest levels seem designed to maintain the bass stock at or near its current level, which as both science and experience clearly show, is incompatible with a sustainable Miramichi Atlantic salmon population and a balanced Gulf ecosystem.

The current striped bass population is patently too high and can be sustained at much lower levels. At current numbers, the striped bass are a clear threat to the viability of Atlantic salmon. The precautionary principle demands that the striped bass population be rapidly and significantly brought down to restore the balance in the ecosystem.

Our client therefore demands that you direct DFO to take immediate action to drastically curtail the striped bass population. The following is a list of measures which will ensure the future survival of Atlantic salmon:

- Immediately, but in any event before January 15, establish a senior departmental emergency response committee including the assistant deputy minister for science with indigenous and non-indigenous Miramichi stakeholders, properly funded and mandated to address this crisis in the immediate term, armed with clear qualitative and temporal mile posts/deadlines and accountability and charged with assessing and implementing immediately, based on proper, transparent, responsive and vetted science and accepted Ministerial obligations and management principles, the following measures:
    - Allow striped bass harvest levels that achieve a bass population of 100,000 bass within four years (2028), i.e., commercial harvests of 300,000 to 400,000 fish in 2025, 2026, and 2027, understanding that levels will be adjusted and may in fact be increased via an adaptive management analysis each year with First Nations continuing to have primary and potential full access to this fishery, but with fair and equitable secondary access to the fishery for the non- indigenous commercial interests with funding for gear and market

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[3] See *Morton v. Canada (Fisheries and Oceans)*, 2015 FC 575, at para. 41 to 42

development for both the Indigenous and non-indigenous commercial net fisheries and the sport fisheries.

- In addition to the commercial harvest, the recreational striped bass fishery should continue to be a component of the total harvest.

- The upper limit of the retention slot for both the commercial and recreational fisheries should be eliminated.

- In the recreational fishery upstream of the heads-of-tide in all scheduled[4] salmon rivers a daily retention limit without a body size restriction should be implemented. The required retention limits under the Maritimes Provinces Fisheries Regulations (MFPR), must reflect the ecosystem imbalance and destructive impact of Striped Bass on juvenile salmon in their nursery habitats.

- Funding should be provided to the Miramichi Salmon Conservation Centre that will bring that facility to the optimal production level for rapidly increasing salmon stocking to overcome current disastrously low levels and to sustain the population into the future. Support by DFO must include:

    A. Capital funding for required facility improvements that are identified by a hatchery committee.

    B. Funding to allow the facility to operate with adequate staffing and required materials, and funds for hatchery operation including activities associated with broodstock acquisition and fish distribution. The operational funds must be provided for the foreseeable future until salmon sustainability has been re-established based on data over a 5-year period.

    C. DFO must guarantee the timely provision of required permits and other approvals to allow fish procurement (juveniles and/or adults) and for stocking programs to be implemented.

- Provide capacity funding to Indigenous commercial fishing enterprises on the Miramichi to develop and expand the Striped Bass fishery.

- Provide financial and market support to the sport fishery.

- Elevate Atlantic salmon to “Major Fish Stock” status under the *Fisheries Act*.

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[4] Scheduled” is a term in New Brunswick fishing regulations and in common use among anglers. It denotes a designated Atlantic salmon river where fly fishing is the only method of angling that is permitted.

  - Examine and implement improved management models for Miramichi Fisheries recognizing rights, interest and socio economic factors.

It is your role and legal duty as minister, and DFO's role as your supporting agency, to use the tools provided under the *Fisheries Act* to manage "the fishery" in a way which balances all of these competing interests and preserves this common property resource, to the best of your ability, together with all of its constituent parts, for future generations. Your department's persistent failure to so manage the balance between Atlantic salmon and the striped bass of the Miramichi river is a patent of breach of that legal duty. On account of all of the interests in the fishery described above, your department *must* take immediate action to bring the striped bass populations in check.

If you do not act on this demand within 30 days of receipt of this letter, our clients have instructed us to bring proceedings in the Federal Court, without further notice to you, to compel your action in this most important and urgent matter.

Yours truly,

MACKENZIE FUJISAWA LLP

Per:

IAN M. KNAPP

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SAVE MIRAMICHI SALMON INC.
Sauvons les saumon de la Miramichi / Plamu 1st

Executive Summary of a Scientific / Technical Presentation to DFO from the Save Miramichi Salmon Organization on the Effects of Striped Bass on Miramichi Atlantic Salmon, plus Required Mitigation

Date: November 7, 2024
By: John Bagnall, Chair Science Committee

Save Miramichi Salmon is a group of people who are interested in stopping and reversing the steep decline in the salmon population of the greater Miramichi River system. A major component of arresting the decline involves bring the striped bass population (essentially the spawning stock) of the southern Gulf of St. Lawrence (sGSL) to a level that will allow a fish community to exist that is representative of a healthy ecosystem. The sGSL's striped bass spawn in the spring in the upper estuary of (primarily) the Northwest/Little Southwest composite branch (the NW composite) of the Miramichi system. Bass spawning in the upper estuary of the Southwest/Renous (the SW composite) estuary is suspected as well. Salmon smolts moving from fresh water to the ocean in the spring encounter striped bass and a portion of the smolt run is consumed by these bass, which at the same time feed primarily on other more populous species, the gaspereau (blueback herring and alewives), and rainbow smelt. Bass are now moving into upland reaches of the river where they eat juvenile salmon and smaller individuals of other fish species.

The sGSL's bass stock has increased from a level of fewer than 5,000 spawners in the late 1990s to what is now suspected, as of 2023, to be approximately 500,000, and perhaps more. Judging from returns to the NW Miramichi salmon protection barrier, coincident with the two orders-of-magnitude increase in the number of bass, the greater Miramichi's salmon population has plunged by 96% (1,136 to 43) between 2010 and 2024.

In the case of important commercial species, DFO manages their populations individually according to the "Precautionary Approach" framework for Maximum Sustainable Yield (MSY) of each. However, if it can be proven the population of one species has a negative effect on that of another "*such as in the case of rebuilding a predator species that could result in a decline of a prey species, rebuilding objectives need to be carefully developed through a balanced approach to ensure neither is depleted to* ***a point of serious harm****. This DFO 2019 policy acknowledges that it is not possible to simultaneously achieve yields corresponding to MSY predicted from a single species when there are multiple, interacting species and in such a case rebuilding efforts should be approached within an ecosystem context to the extent possible*".

It seems obvious to most people that the 2019 policy is tailor-made for the southern Gulf bass / greater Miramichi salmon situation. Evidence of serious harm on salmon caused by bass was sought in Research Document 2022/030. Four tests were presented in that document that if passed would allow the multi-species policy to be implemented:

1. That bass were having a significant same-year effect on SW composite salmon smolts (as indicated by survival of tagged samples) during their passage through the estuary;
2. That bass were having a significant same-year effect on NW composite salmon smolts during their passage through the estuary;

3. That bass were having a significant effect on the returns of grilse (1 sea-winter aged salmon) one year later to the SW composite; and
4. That bass were having a significant effect on the returns of grilse (1 sea-winter aged salmon) to the NW composite.

The author confirmed that serious harm had resulted in cases 2 and 3, but not in 1 and 4. Therefore the 2018 policy on multi-species management was not implemented, and bass are being managed to provide MSY. The population is protected by a lower limit (the Limit Reference Point, or LRP) below which the bass population will not be allowed to fall for fear of significant population harm. This number is 330,000 bass.

Test 1 included data only to 2018. Four more years of data (2019 to 2023 minus the Covid year of 2020) overturned the "no-effect" decision in that case leaving Test 4 as being the lone impediment to multi-species policy implementation. We believe Test 4 should also provide evidence for implementing the multi-species policy, or that the test itself should be considered irrelevant. For example:

a. We feel this test was not conducted properly. As a first step, the author of Research Document 2022/030 re-calculated historic annual smolt outputs using "cohort analysis". The resulting smolt outputs were compared with total annual grilse returns as extrapolated from captures at the Cassilis trap in the estuary of the NW composite. Many of the calculated return rate numbers do not agree with and have absolutely no correlation with numbers published for the same years in Research Document 2016/029. Considering the 2016 uses the commonly accepted and employed mark-recapture method to estimate smolt numbers, this method would seem to be more credible than the cohort analysis method of calculation. This brings into serious doubt the "no bass effect" conclusion.

b. In addition, the finding of no significant effect of bass numbers on the following years' grilse returns implies it doesn't matter how many smolts are produced in the Northwest composite, that grilse returns are essentially random, or is an assumption of density dependence. Most salmon fisheries theory assumes that the mortality of salmon in the ocean is density-independent, a rationale based on the idea that the population density is far below the assumed carrying capacity for salmon in that habitat. Derivative from this theory is an assumption that, over the long haul, the more post-smolts that enter the high seas, the more adult salmon can be expected to return. Gibson (2006) included an assessment of the NW Miramichi River where it was identified as density independent for both grilse and two sea-winter salmon. We found 25 cases where density dependence / density independence was examined in eastern Canadian rivers, 16 for grilse, and 9 for MSW salmon. 24 were judged to be density independent. This further brings into question the conclusion of Res. Doc. 2022/030 that bass were having no significant effect on the returns of grilse (1 sea-winter aged salmon) to the NW composite.

The preceding demonstrates only that there is a negative correlation between bass numbers and a reduction of Miramichi salmon to a point where the salmon population has been seriously harmed. Maybe it is simple coincidence, but other possible reasons have been eliminated. These include:

- During “high bass” years, the sublethal effects of aluminum toxicity due to effluent from an abandoned mine causing elevated mortality rates of Northwest Miramichi tagged smolts once they reach salt water;
- A size discrepancy between “high-mortality-rate” tagged smolts in “high bass” years and the lower rates in “low bass” years;
- A “tag effect” caused by high mortality from recently tagged smolts in comparison with untagged smolts;
- That a population increase in some other predator besides bass is causing the elevated post-smolt mortality rates; and
- That the Miramichi salmon population’s decline is simply part of a general decline in the populations of all salmon rivers draining to the Gulf of St. Lawrence.

The preceding confirms the Occam’s Razor conclusion that it is the over-abundance of striped bass that has caused the precipitous decline in the Atlantic salmon population of the greater Miramichi system to the point that the population has been **seriously harmed**.

We have developed a simple population model that shows the Miramichi’s salmon population is sustainable only up to a population level of approximately 100,000 bass, a number that is alluded to as a potential experimental fisheries target in Research Document 2022/030. The bass population must be rapidly decreased to this level. Since the population now is very near or at the MSY, the proposed 175,000 commercial harvest (125,000 new removals) and the increase in the recreational fishery harvest will be massively insufficient to bring about a population that is compatible with a sustainable Miramichi salmon population. In fact, even if it is achieved, the 2025 harvest allocation will probably be insufficient to decrease the bass population at all. We demonstrated this with a simple back-calculated Beverton-Holt model and resulting calculations. The additional 125,000 bass harvest in 2025 will not bring the stock to a level lower than the existing 330,000 bass LRP. We know this because were told by DFO **after** the announcement of the increase that without the multi-species policy in place, the LRP level of 330,000 bass is inviolate. With the policy in place, a new LRP at a level lower than 100,000 experimental fisheries target is required.

We believe the sGSL striped bass / greater Miramichi salmon population situation has been grossly mismanaged by DFO, and in compensation, we strongly request immediate action be taken to rapidly decrease the bass numbers through a large increase in the commercial fishery of up to 400,000 bass per year, the dropping of the upper slot limit in both the commercial and recreational fisheries, the liberalization of creel limits for bass in waters above the heads-of-tide of the two Miramichi composites, and a salmon stocking program based out of the Miramichi Salmon Conservation Centre. The specific methods used for the hatchery-based enhancement should be the responsibility of experts in salmon genetics, hatchery-based salmon enhancement products that would achieve best results, salmon nutrition, and salmon hatchery / grow-out methods. DFO should fund the capital improvements and the increased operational expenditures that are urgently required to help save the Atlantic salmon of the greater Miramichi River system.

Spring 2024: Miramichi striped bass with partially digested smolts in its stomach

Not only the Miramichi. Bass with consumed parr August 17 Little Main Restigouche, New Brunswick. [Approximately 175 km from tidewater].

SAVE MIRAMICHI SALMON INC.

Sauvons les saumon de la Miramichi / Plamu 1st

Save Miramichi Salmon – Scientific/Technical Supporting Document

Prepared by:

Save Miramichi Salmon Science Committee

6-Nov-24

Abstract from the Save Miramichi Salmon – Scientific Technical Support Document

The Miramichi River in New Brunswick, Canada is one of the world's most famous Atlantic salmon rivers. The Miramichi historically hosted annual runs of 1,000,000 or more adult salmon due to its unparalleled, large area of excellent spawning habitat. As recently as 2011, the Miramichi had a run approaching 100,000 adults and the river supported about 90% of the Atlantic salmon fishing effort in the Province. In 2000 the Miramichi was home to 50% of all the Atlantic salmon in the western Atlantic. This fishery has had a significant socio-economic benefit for the Province of New Brunswick, and the salmon are very important for food and ceremonial purposes to the First Nations.

The Miramichi also has a native population of striped bass. Over the last hundred plus years a small commercial fishery plus a recreational fishery kept the population of striped bass at a balanced level. In the late 1980's, the population of striped bass seemed to drop below 5,000 adults and DFO put in place a complete moratorium on the harvest of striped bass. A target population of approximately 30,000 striped bass was DFO's stated goal. Bass are prolific spawners and responded so well to the harvest moratorium that the population reached 100,000 adults in approximately 2011. Since 2011, the population has increased rapidly, and in 2024 it has reached a level of at least 500,000 adult spawners.

Striped bass spawn near the head of tide in both the NW and SW branches of the Miramichi River. At the time of spawning a great density of striped bass inhabit not only these areas, but are also found in considerable numbers downriver throughout the tidal estuary. During the same time, small Atlantic salmon called smolts that have grown for up to three years throughout the freshwater habitat of the Miramichi system migrate down all the branches of the Miramichi on their way to the ocean. There they will live from one to three years before returning to spawn in their natal river. All of these young salmon must pass through the constricted head-of-tide zones where the striped bass are aggregating to spawn. Striped bass are voracious predators and, in their massive aggregations of recent years, they exact a significant mortality on the outgoing salmon smolts.

Smolt tagging experiments by the Atlantic Salmon Federation and the Miramichi Salmon Association have been carried out for many years, and conclusively show that the striped bass now consume a devastating 95% percent of the outgoing smolts in the NW Miramichi and 65% percent in the larger SW Miramichi. This means that only 5% of the smolt run in the NW and 35% in the SW Miramichi are making it to the ocean. The tagging experiments track the smolt migration down the river and show that nearly all the mortality is taking place when the young salmon approach the striped bass spawning areas and continues throughout the tidal estuary. Data were collected for many years prior to the striped bass population explosion, and it showed that smolt survival to the ocean from the Miramichi freshwater habitat had previously been about 75% which is similar to other nearby Canadian rivers.

Once the remaining salmon smolts reach the ocean there is again a very significant natural mortality exacted on them. Because the ocean environment is so vast the percentage of smolts that survive to adulthood is no different regardless of the size of the outgoing smolt migration, a concept called density independence. The fewer smolts that make it safely to the ocean, the fewer adults that return to the river to spawn. Some adult salmon return from the ocean after two years at sea and are called multi-sea-winter salmon, and some come back after one

year as grilse. The adult salmon population of the Miramichi has decreased by more than 90% since 2011, and scientists believe that it is trending rapidly towards extinction.

According to documented Canadian Department of Fisheries and Oceans (DFO) internal policies no species is supposed to be managed so that it causes serious damage to another. DFO has avoided reducing the striped bass population, so that the salmon and the bass can coexist in the Miramichi by denying that the striped bass are the major problem. In the detailed documentation to follow, John Bagnall and the scientific team he has assembled – including Dr. John Ritter, retired head of anadromous fish science for the Maritimes division of the DFO - will show that this was simply an incorrect position for DFO to have taken. Due to multiyear efforts from a coalition of ENGO's advocating for salmon, DFO has increased the commercial harvest allowed by FN to 175,000 individual adult striped bass. The problem is that a population of adult striped bass will see approximately 50% of the spawning stock being replaced annually by new maturing year classes – a concept called recruitment. 50% of 600,000 – and it could be more – adult striped bass is 300,000 or more new recruits are added to the Miramichi striped bass spawning stock annually.

In addition to the commercial harvest there is a recreational fishing harvest. The exact extent of the recreational harvest is unknown, but it was assumed by DFO to be a component of total mortality (commercial, recreational, plus natural) which they pegged at approximately 20% of the population annually. This level of mortality plus the projected additional commercial fishing mortality of 125,000 bass (175,000 proposed minus the current level of 50,000) is not expected to decrease the bass population much if at all. The bass population will not contract to the level of 100,000 that is needed for the bass and salmon populations to coexist in the river.

The document to follow provides full scientific justification for DFO to rapidly and aggressively reduce the populations of striped bass in the Miramichi watershed and to provide funding for a modern stocking program to restore a healthy balance between the Atlantic salmon and striped bass in the Miramichi River.

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1.0 Introduction

- This submission was prepared on behalf of the non-governmental organization (NGO), Save Miramichi Salmon (SMS), an *ad hoc* Atlantic salmon defence group of committed individuals who are focused on restoring the salmon population of the greater Miramichi River system, which is comprised of four rivers that flow to a common estuary. These rivers are the Southwest Miramichi (SW), the Renous, the Little Southwest Miramichi (LSW), and the Northwest Miramichi (NW). Our organization includes owners of riparian property with private water rights, property owners on public water, and fishing outfitters and guides. Our primary focus is the ever-increasing and excessive numbers of striped bass and their devastating effects on Atlantic salmon (salmon) and other species.
- Our foundational platform is:
    - With an urgency reflective of the dire situation the population faces today, our focus is on restoring the Atlantic salmon population of the greater Miramichi system, a population which has been rapidly declining in recent years. This situation cannot wait for more discussion or a National Strategy because the ecological imbalance in the Miramichi River is too great and has persisted for too long;
- We feel that the situation has been allowed to develop due to the Department of Fisheries and Oceans' (DFO) ignoring and/or misinterpreting data resulting in the failure to take a reasonable and balanced management approach as required by a policy under DFO's own fisheries management framework, the "Precautionary Approach". The result has been an explosion of the southern Gulf of St. Lawrence's (sGSL's) striped bass population, which has reached its highest abundance in recent history;
- This has caused the decimation of the Miramichi salmon with a very real danger of their extirpation;
- Our group's proposal for remediation of the situation is balanced and grounded in good science, real data, and respects the ecosystem as well as stakeholders and Rightsholders. It is imperative that the proposed actions be taken today; and
- While we could spend significant energies relaying the historical failures to effectively manage the fish and fisheries of the Miramichi River ecosystem, our commentary focuses on the current striped bass threat and its devastating impact on Atlantic salmon. Steps to better manage striped bass are likely to concurrently benefit the sea-run brook trout, rainbow smelt, and gaspereau that cohabit the estuary, and may benefit the lobster population of the southwestern Gulf of St. Lawrence as well.

2.0 Miramichi Salmon History

The importance of Atlantic salmon to Indigenous people and recent settlers cannot be overstated. The annual return of adult salmon to the river to spawn, the run of salmon, supported Indigenous peoples since the last glacial retreat. The run supported settlement along the river as early as the 1600s. Nicolas Denys, an early settler who had a trading post on the Miramichi in 1648, wrote that there were so many salmon, "...that one is unable to sleep, so great is the noise they make in falling upon the water after having thrown or darted themselves into the air". This abundance of salmon has supported many people with food, both Indigenous and settlers, as well as creating an economy with its many jobs related to the now, world-renown sport fishery. This included a commercial salmon fishery. This fishery closed when DFO realized a pound of salmon caught translated to $1/lb, whereas the recreational fishery value was returning an astounding $40/lb. The non-Indigenous commercial net fishery was permanently closed.

The Miramichi has supported the largest salmon run (population) in eastern North America, at one point accounting for 50% of the western North Atlantic salmon abundance. It is world-famous and has supported as much as 90% of all salmon fishing trips in the Province of New Brunswick. The famous and infamous have come from all over the world to fish: His Majesty, the King (when he was the Prince of Wales) , Ted Williams, Chuck Yeager, Benny Goodman, Tom Selleck, and Dick Cheney, to name just a few. Plus, the persistent existence of the Miramichi River salmon to local people fishing Atlantic salmon, cannot not be overstated.

The greater Miramichi system once supported >1M adult salmon returning to the river annually. Today, the run hovers around 8,000 adults (extrapolated from 15 October 2024 trap data), both multi-sea-winter salmon (MSW) and 1-sea-winter grilse.

2.1 2019 Strategy Document

DFO produced a strategic management plan in 2019, "The Wild Atlantic Salmon Conservation: Implementation Plan 2019 to 2021". The 2019 plan set commitments to address critical threats to effective conservation such as predation by striped bass and protection from invasive species, e.g., smallmouth bass. It included no timelines or details. Rather it portrayed an optimistic note that matters would be addressed in a timely way. The 2019 document was not ambiguous; DFO understood the critical timelines and paths necessary to effectively address the decline of the Miramichi salmon. At that time, the Minister of Fisheries and Oceans clearly stated that DFO should not manage the rebuilding of one stock to the detriment of other stocks, that DFO will maintain the balance among the managed stocks, and that multi-species management will occur using a "Precautionary Approach" framework.

2.2 2024 Draft Atlantic Salmon Strategy Document

Five years later and after decades of data collection, peer-reviewed science, and the creation of many management plans, the 2024 "Draft Strategic Plan" was released. The Plan is a collection of principles and processes. It does not address the structure of DFO's leadership role or the requirement for immediate hands-on and properly financed action to save the Miramichi's Atlantic salmon. It has taken approximately 15 years to craft this Plan, and it proposes another 12-year implementation timeframe. It is our contention that the Miramichi salmon population will be near to extinct before this Plan is implemented.

The Plan itself is a blank slate for action with no timeline for its execution. Neither does it provide any financial commitment or leadership for implementation. It ignores the reality of the current critical situation for the survival of the Miramichi salmon and offers none of the urgent prescriptive actions that are abundantly apparent and absolutely necessary today to save this important resource. All of the critical and imminent threats to the Miramichi salmon and their habitat have been well-studied. While threats were identified in the 2019 Strategy Document, no action has taken place to mitigate them, and most importantly, no action has been taken to produce a balanced ecosystem as called for in the 2019 document.

To ensure the survival of the Miramichi salmon population, which includes a balanced state of the entire ecosystem, we propose an addendum be included to the "24 Draft Strategy" that provides for immediate, effective, and funded action to save this valuable ecosystem and resource. The 2024 Draft Strategy fails to acknowledge the urgency of the situation. While Projects are discussed, these must be triaged such that the most serious challenges are addressed rapidly and vigorously to reflect the seriousness of the current crisis state. There may be others, but the Atlantic salmon/striped bass situation on the Miramichi system is one that cannot be subjected to the paralysis of lengthy consultations suggested in the "Draft Strategy". DFO needs to invoke immediate action on situations that exists today on rivers such as the Miramichi, a process that is unencumbered by the delaying inertia that permeates under the "2024 Draft Strategy.

2.3 The Immediacy of Action Required for the Miramichi River's Salmon and Its Ecosystem

While common sense, the data, and the science clearly point to a connection between the soaring population of striped bass and the precipitous decline of salmon, there has been a failure by DFO to recognize that reality and more importantly, DFO has seemingly invested considerable effort to deny it. Our position is supported by the data and the correct and approrirate scientific analyses missing from DFO's efforts to date.

2.4 Striped Bass Management

2.4.1 Bass Movements and Interaction with Salmon

The population of the native, sGSL's striped bass (bass) has exploded since late 1990s, and this fact, by scientific measure and anecdotal observation, is irrefutable. Bass are voracious predators feeding on a variety of prey species including rainbow smelt, gaspereau, brook trout, and juvenile salmon. Atlantic salmon smolts are moving from their upstream freshwater nursery habitat towards their high seas rearing areas at the same time as bass are congregated in the estuary to spawn. In the Miramichi, a portion of the bass population moves into river reaches above the head-of-tide prior to the spawn and then moves downstream to spawn. Bass spawn in the Miramichi estuary and can migrate on feeding forays that take them well above the tidal zones that extend into all four rivers of the Miramichi system. They can stay in upland pools for much of the summer, feeding on Atlantic salmon parr (see Exhibit A) and other freshwater resident fish species. Bass do not discriminate among the potential food items available in the estuary and river, be they trout, smelt, gaspereau, or salmon smolts.

2.4.2 The "No-Effects" Decision by DFO

In Research Document 2022/30 (Chaput 2022), it is acknowledged that striped bass eat salmon smolts, but, the author concludes there is no population-level effect of bass on Atlantic salmon. He does this despite documenting (1) a significant negative correlation between bass abundance and the survival through the Miramichi estuary of NW Miramichi tagged smolts, plus (2) showing a significant negative correlation between bass abundance and grilse returns to the SW Miramichi one year later, i.e., reflecting the loss of smolt output from the year before. (Please note that in this document we may refer to the NW composite that includes the Northwest Miramichi plus the Little Southwest (LSW) Miramichi, and the SW composite that includes the Southwest Miramichi and the Renous rivers. Each of these composites discharge into common bays of the greater Miramichi estuary. In addition, the term smolt,or smolts is used to also apply to post-smolts – i.e. smolts that have recently entered the marine environment.)

Figures 1 and 2 (below) depict annual bass stock number estimates and and the NB Dept. of Natural Resources and Energy Development's data on annual grilse and MSW salmon returns to the Northwest and Dungarvon headwater protection barriers. Figure 3 and 4 again depict bass numbers and in these cases, total returns of grilse and MSW salmon to the Northwest and Southwest Miramichi composites, estimates that are extrapolated from returns to the Cassilis and Millerton estuary traps, respectively. Estimates of bass numbers and salmon numbers extrapolated from trap data were taken from DFO assessment documents (i.e., Research or Advisory Documents). The bass numbers in the Figures 1 to 4 are for the year of their estimate. Estimates of bass numbers were not available for 2010, 2012, and 2020, and likewise salmon return estimates were not available for 2020. The decline in the grilse returns began after the very obvious spike in bass numbers to over 200,000 spawners in 2011 and the onset of the massive growth in the bass spawner population. The decline in MSW salmon followed thereafter.

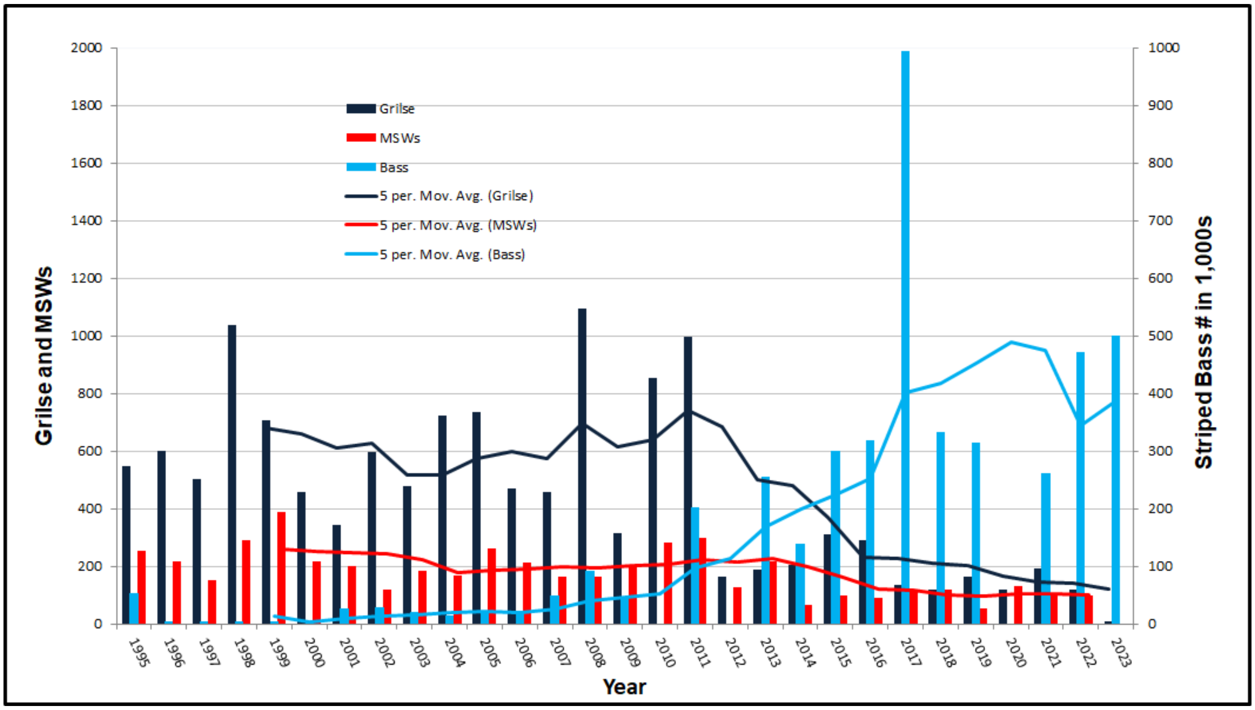


Figure 1 Annual Northwest Barrier Counts of Grilse and MSW Salmon, Numbers of Striped Bass Spawners, and Trend Lines.

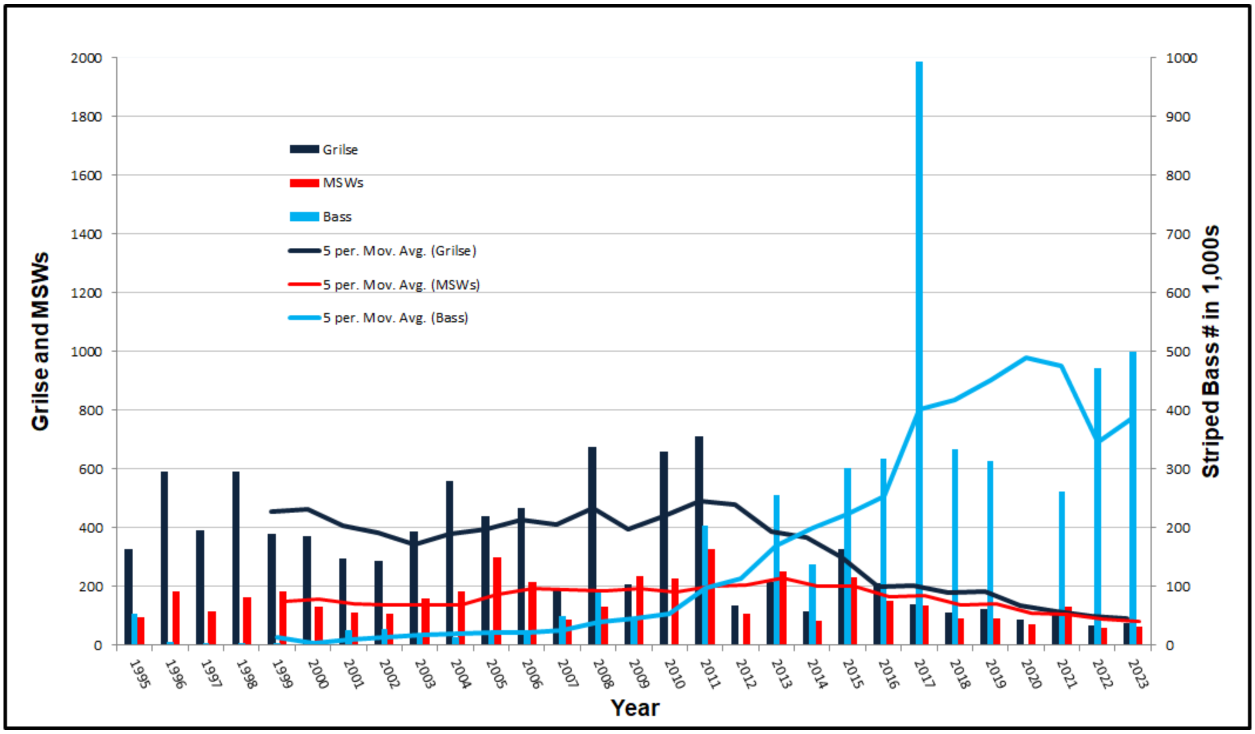


Figure 2 Annual Dungarvon Barrier Counts of Grilse and MSW Salmon, Numbers of Striped Bass Spawners, and Trend Lines

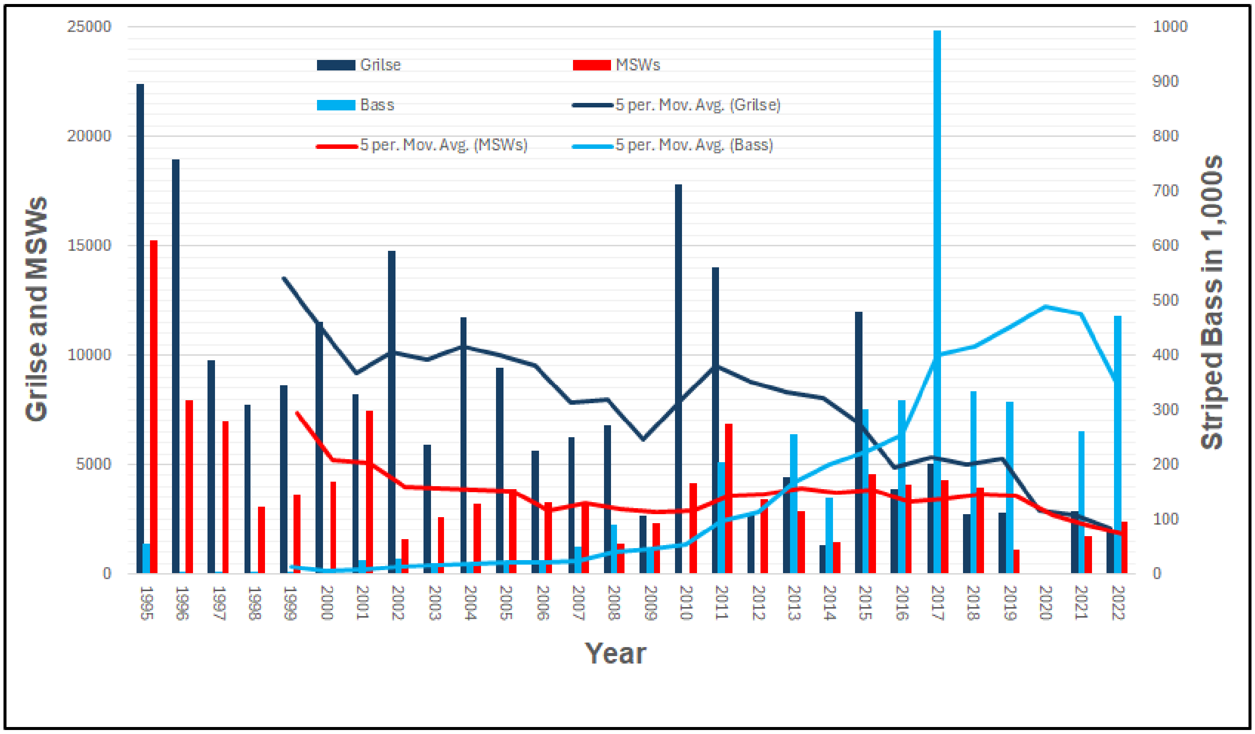


Figure 3 Annual Returns of Grilse (Small Salmon) and MSW Salmon to the Northwest Miramichi, Numbers of Striped Bass Spawners, and Trend Lines.

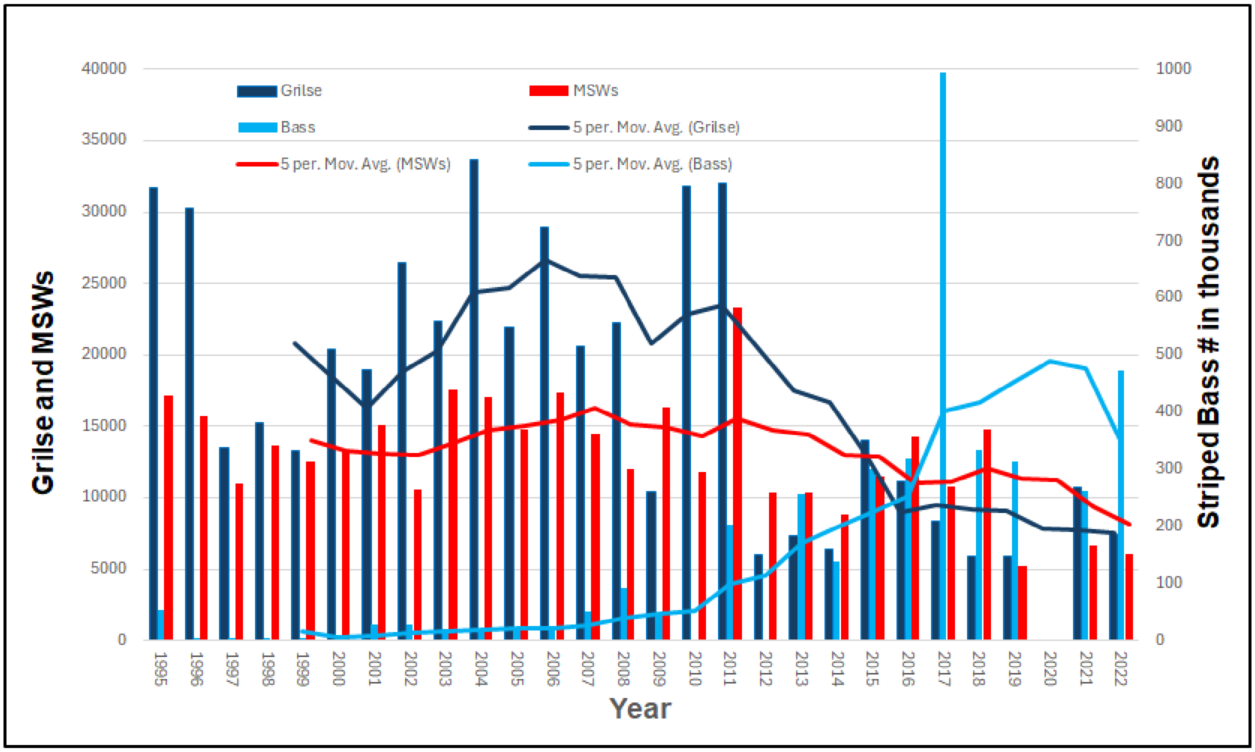


Figure 4 Annual Returns of Grilse (Small Salmon) and MSW Salmon to the Southwest Miramichi, Numbers of Striped Bass Spawners, and Trend Lines.

2.4.3 Apparent Effects of Bass on Salmon

Estimates of the Striped bass spawning population have approached 500,000 for several recent years and may have exceeded this number in 2023 and probably have in 2024. According to data from the Atlantic Salmon Federation, the recent historically high annual bass numbers are concurrent with tagged smolt mortality rates of approximately 95% in the NW Miramichi (Atlantic Salmon Federation data supplied by Neville Crabbe, 2023). It is a well-accepted rule-of-thumb target in Atlantic salmon management that approximately 5% of adult returns are necessary to sustain a population. To achieve the 5% adult recruitment target for the past several years, NW Miramichi salmon, after leaving the Miramichi estuary, would have to travel thorough the Gulf of St. Lawrence and Strait of Belle Isle to Labrador and perhaps Greenland, and return to the Miramichi with zero mortality. It is incomprehensible that this could be considered as being possible.

Concurrently, DFO's river specific data for Bay of Chaleur rivers, the Restigouche and Cascapedia, where no substantial striped bass numbers are present in their estuaries, have tagged smolt mortality rates through the estuaries and the Bay of Chaleur of approximately 10% vs. the ~95% with NW Miramichi tagged smolts. A quick and simple look at salmon returns in Table 1 provides a visual of the dramatic reductions in grilse and MSW salmon returns to the salmon protection Barrier on the NW Miramichi since 2010. For comparison, DFO's estimates of the Numbers of Striped Bass Spawners for the year previous to the grilse return years (2009 and 2023). The decrease in the percentage total salmon and grilse returns to the Dungarvon barrier on the Southwest composite are similar over the same period.

Table 1 Comparison of Salmon Returns to the Northwest Miramichi Salmon Protection Barrier, 2010 versus 2024, plus Striped Bass Spawner Numbers in the Previous Year (yr-1).

Final Date	Year	Total Grilse to Date	Total MSW to Date	Total Atlantic Salmon to Date	Striped Bass (yr-1)	
Oct. 22	2010	852	284	1136	48040	
Oct. 11	2024	8	39	47	500000	*
Change	14 years	-844	-245	-1089	451960	
Change Magnitude	-	-99.1%	-86.3%	-95.9%	940.8%	

*** 500,000 striped bass in 2023 is the estimated size of the spawning population alluded to by DFO at a Consultation Meeting between Stakeholders and DFO in January 2024.**

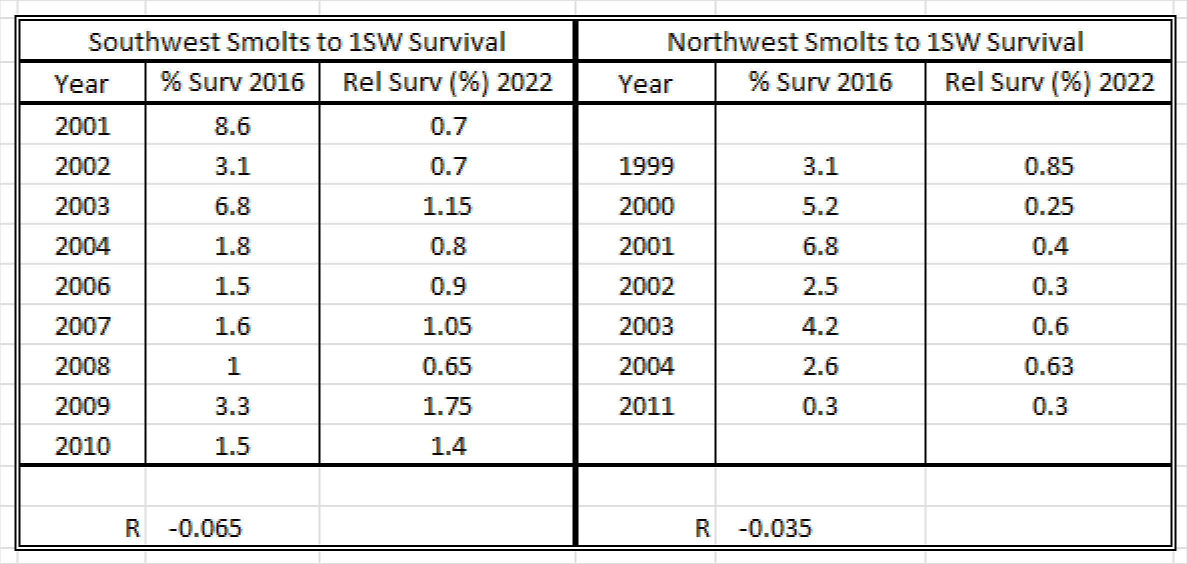
2.4.4 Rebutting DFO's Conclusion of Striped Bass having "No-Effects" on Miramichi Salmon

Research Document 2022/030 (Chaput 2022) provides two reasons to conclude that the large numbers of striped bass are having no harmful effect on the Miramichi salmon population (i.e., "no-effects'). The first reason for the author's rejection of a harmful effect was the lack of significant correlation between estimated annual striped bass spawner numbers and the same-year mortality rates for the SW Miramichi acoustically-tagged smolts. This conclusion was reported despite a significant correlation between bass numbers and mortality rates for NW Miramichi tagged smolts - an analysis reported in the same document, and given that the vast majority of striped bass are in the NW Miramichi. When including data for years since 2018 (the final year of data used in the analyses reported in Research Document 2022/030), a significant correlation between annual bass numbers and mortality rates for tagged SW Miramichi smolts is also apparent (see Appendix A-1).

The second reason for DFO's "no-effects" conclusion was based on a regression analysis of Smolt-to-1SW salmon survival rates for the NW Miramichi versus the numbers of Striped Bass Spawners in the year of smolt emigration (Chaput 2022). Both the Striped Bass Numbers and 1SW Salmon Returns, used in the analyses, were DFO's annual estimates of their abundances, determined through mark-and-recapture programs carried out by DFO. The numbers of Smolts emigrating from both the NW and SW Miramichi systems respectively, were determined through a "cohort analysis" with 2-year old smolts derived from small parr density measures and 3-year old smolts from large parr abundances. Total smolt outputs were estimated through extrapolation of smolt production estimates to the total wetted habitat areas potentially utilized by salmon (these areal estimates of salmon abundance are the metric used by DFO). The author states that the Smolt-to-1SW salmon survival rates determined are "Relative Survival Rates", and that "The Term Relative Survival Rate is used because the estimated smolt production is raised using the total habitat area of the rivers. This exaggerates the smolt production because the juvenile indices are derived for specific components of the habitat, classic juvenile rearing habitat." The cohort analysis is DFO's choice of abundance metric despite many external reports of its shortcomings.

The Smolt-to-1SW survival rates utilized in Chaput (2022) are very different from those reported for many of the same years by Chaput et al. (2016) because of the two different methods by which smolt numbers were determined, i.e., via cohort analyses versus mark-and-recapture. Differences between Smolt-to-1SW salmon survival rates determined by the two methods are detailed in Table 2

Table 2 Comparison of Annual Smolt to1SW Salmon Survival Rates for the Same Years as Published in Res. Docs. 2016/029 (Chaput et al. 2016) and 2022/030 (Chaput 2022).



Southwest Smolts to 1SW Survival			Northwest Smolts to 1SW Survival		
Year	% Surv 2016	Rel Surv (%) 2022	Year	% Surv 2016	Rel Surv (%) 2022
2001	8.6	0.7			
2002	3.1	0.7	1999	3.1	0.85
2003	6.8	1.15	2000	5.2	0.25
2004	1.8	0.8	2001	6.8	0.4
2006	1.5	0.9	2002	2.5	0.3
2007	1.6	1.05	2003	4.2	0.6
2008	1	0.65	2004	2.6	0.63
2009	3.3	1.75	2011	0.3	0.3
2010	1.5	1.4			
R	-0.065		R	-0.035	

Correlation analyses between both sets of survival rates show no similarity for both the Northwest and Southwest systems ($p > 0.05$). The lack of similarity is related to the difference in smolt number estimated by the two different methods because like numbers of 1SW salmon would have been used in both sets of analyses. It seems reasonable to accept that the smolt number estimates by mark-and-recapture (Chaput et al. 2016) as being the more accurate given this is a well know and proven method of estimating population numbers in fisheries science. The above invalidates DFO's Regression Analysis and as evidence in support of its conclusion that the massive striped bass spawner population is having no effect on adult salmon returns to the Miramichi River.

A simple correlation of historic bass numbers versus grilse returns one year later since 1995 for each drainage composite are negative and significant. Please refer to Appendix A-2.

In addition to the incorrect no-effect reasoning of Chaput (2022) that resulted from use of the cohort analysis, the author seems to imply that the number of smolts entering the marine environment has no bearing on subsequent adult salmon returns, that returns are independent of smolt output or perhaps they actually decline with increasing output, so-called density dependence. There is consensus among Atlantic salmon biologists and fisheries managers that high seas rearing area for salmon is a density independent environment (e.g., Hansen and Quinn, 1998; Gibson, 2006 [Res. Doc. 2016/029]). From Gibson (2006): "As outlined by Jonsson and Jonsson (2004), most salmon fisheries theory assumes that the mortality of salmon in the ocean is density-independent, a rationale based on the idea that the population density is far below the assumed carrying capacity for salmon in that habitat."

Density independence for salmon in the marine environment implies that a relationship between smolt output and adult salmon returns described by an ascending straight line leading essentially to infinity best fits the data. The instantaneous rate of return at any point on the line is positive and constant. Density dependence such as the Beverton-Holt relationship commonly used in fisheries science and management, normally indicates that, above a smolt output point, there are progressively smaller rate increases of adult returns. After some point of smolt output , the instantaneous return rate starts to decrease, but never goes to or falls below zero. An increase in returns followed at a certain smolt output point by a downturn of returns or a random rate of return (a Ricker relationship) with increasing smolt output is however possible (Gibson, 2006). Please refer to the following examples in Figure 5:

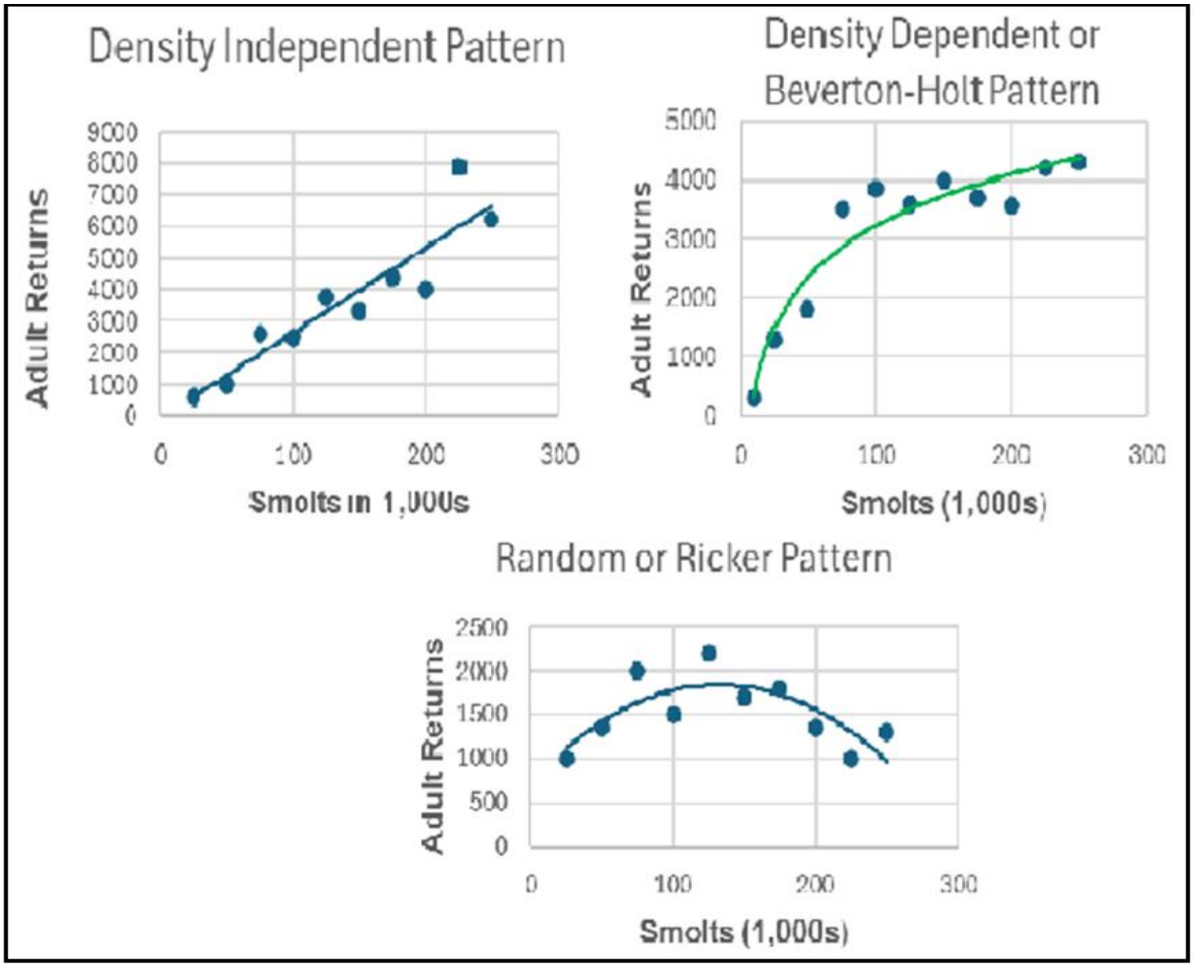


Figure 5 **Generalized Types of Adult Return versus Smolt Output Patterns.**

Gibson (2006) examined 15 eastern North American rivers for evidence of potential marine density dependence in grilse returns. Twelve were density independent. A straight-line function best fit the data. One (Quebec's St. Jean) exhibited a mild density dependent fit, and grilse return patterns from two Newfoundland rivers (the Campbellton and Trepassey) seemed to be random fits. Subsequent figures provided by by Dr. Brian Dempson, DFO Newfoundland (retired - pers. comm. to J. Bagnall, 2024) included data from 1992 to 2018. These data for Trepassey now seem to conform to a density independent pattern. Gibson (2006) also looked at two sea-winter maiden returns to nine rivers. All nine patterns were typical of density independence.

The following (Figure 6) are Gibson's (2006) charts for the Northwest Miramichi.

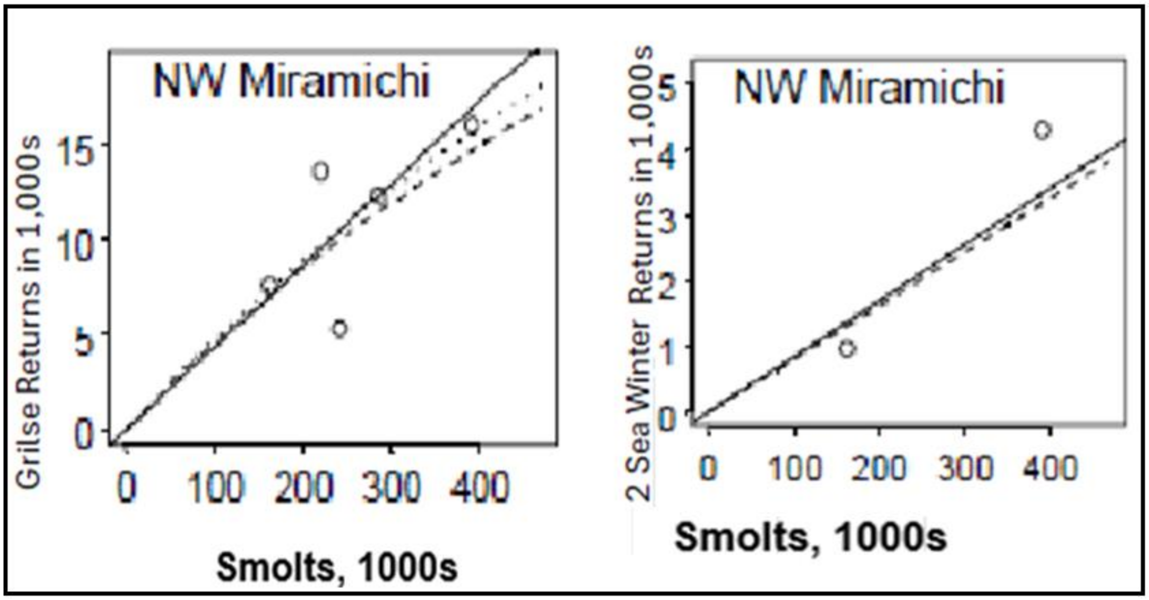


Figure 6 **Grilse Returns versus Smolt Outputs for the Northwest Miramichi from Gibson (2006).**

These figures demonstrate density independence, i.e., adult salmon returns being highly correlated with smolt output for both NW grilse and two sea-winter salmon returns. Given that the author of Res. Doc. 2022/030 is using these same data generated by DFO and is very well entrenched in the international community of Atlantic salmon fisheries management, it is surprising that a conclusion suggesting density dependence is implied in Res. Doc. 2022/030..

2.4.5 DFO's Alternative Explanations for the Decline of Miramichi Salmon

The "no-effects" decision expressed in Res. Doc. 2022/030 follows a pattern of a seemingly conscious disregard for DFO's "precautionary approach" and "ecosystem perspective". To begin with, while the effect of predation on native fishes by striped bass is a very well-established fact and all the available data for the Miramichi point to the predation impact on smolts, DFO chose to propose three alternatives that lack any science basis. The following bullet points describe these conjectures:

- From Chaput et. al. (2018) - "*The fish tagged during 2003–2008 were taken from a different branch of the Northwest Miramichi than those tagged in 2013–2016 with the smolts from the latter period captured downstream and released again above a tributary (Tomogonops River) impacted by acid and metal runoff* *from a decommissioned base metal mine. There are concerns for Atlantic salmon smolt vulnerability to episodic acidification and elevated concentrations of bioavailable aluminum during spring snow melt and increased. Thorstad et al. (2013) reported on delayed mortality in the early period of marine migration of smolts exposed to aluminum and moderate acidification in freshwater. This cannot be excluded as a factor contributing to the lower apparent survival rates in the Northwest Miramichi smolts in the latter part of the time series…"*

DFO's hypothesis is based on the implication that the long-mothballed, Heath Steele Mine with its well treated and monitored effluent could have caused delayed mortality among Northwest Miramichi smolts, mortality which was not experienced by the earlier Little Southwest tagged smolts that did not travel past the mine site. We are not surprised that no results of any study have been reported to stakeholders, and we assume DFO has now rejected its hypothesis.

- From Chaput et. al. (2018): - "*We cannot exclude the possibility that the differences in estimated survival rates between bays and over years in this study are also in part due to differences in the size distributions of acoustically-tagged smolts among years and rivers.*"

  In other words, the increased mortality of smolts in the "high bass" years as indicated by a lower rate of tag detections at the end of the estuary may have been because the tagged smolts used in the "high bass" years were smaller (shorter). We have heard nothing more about it.

- The effects of tagging were brought up in Chaput et. al. (2018) with references deleted and our annotations in parentheses [ ]:

  "*An important concern regarding the use of marked animals to make inferences on behaviour and survival of unmarked/ unhandled animals is the consequence of tagging and handling effects on the estimates of survival or migration dynamics. It is extremely difficult to make the case that a tagged smolt would behave and have the same mean probability of survival as an untagged smolt. In terms of absolute levels, it is unlikely that the estimates derived from marked animals correspond to those of unmarked animals. There can be important growth and survival effects of handling and tagging even when animals are held in captivity post tagging and monitoring tagged fish in captivity does not provide much insight into the conditions encountered by fish released to the wild. The capture, handling, tagging procedures in addition to introducing stress and injury to individual animals also interrupt the migration phenology [behaviour related to natural cycles] of wild smolts during a particularly sensitive period. Removal from schooling with conspecifics, release back to the river during the day or even near dusk when wild conspecifics are sheltering and not in active migration phase, can result in increased vulnerability to predation*."

  The science of tagging fish, including salmon smolts is very well-established and a well-defined discipline. There is zero evidence that a Miramichi River salmon smolt behaves any different than the 10,000+ studies of tagged salmon smolts in the literature. Nonetheless, the Canadian Rivers Institute went further by examining predator tags in Miramichi River smolts. In the pending CRI publication, tags were inserted into pre-smolt groups in the fall as well as into other groups of smolts in the spring. There was no difference in mortality/survival rates through the estuary between the fall-tagged group and those tagged in the spring. The conclusions are: 1) there is no tag effect; and 2) that a cold-blooded predator like the striped bass, present at the time of the smolt migration is the pre-eminent consumer of seaward-migrating, juvenile salmon in the Miramichi estuary in the Spring.

Another potential reason that has been advanced, but to our knowledge has not been formally proposed, is that the decline in the greater Miramichi's salmon population is simply typical of a general decline that is common to all rivers draining to the Gulf of St. Lawrence, a decline perhaps related to global warming. For example, from the minutes of FOPO (2019), Mr. Serge Doucet (Regional Director General, Gulf Region, Department of Fisheries and Oceans):

"Now, some have suggested that there may be a link between that increase in striped bass populations and the decreased populations of Atlantic salmon. However, studies by DFO have not been able to establish such a direct causality. In fact, ***a decline in the abundance of Atlantic salmon has taken place in all areas of eastern North America, including places where there are no striped bass, or the phenomenon of striped bass increase has not taken place****. Similar declines in the population of Atlantic salmon are also being recorded in the European range. A variety of factors could explain the decrease in salmon populations. For instance,* ***we can no longer ignore the warming climate, which led to unprecedented angling closures in 2018 as rivers reached temperatures lethal to Atlantic salmon****, a species best suited to colder water."*

Here, Mr. Doucet's implies that the universal decline he referred to, and its attribution to global warming was settled science, but it appears to have been speculation on his part. Dr. John Ritter (2024, unpublished) has rebutted this hypothesis, and further, has identified striped bass as being the cause of the decline. Dr. Ritter's report has been shared with DFO's, Gulf Region.

2.4.5.1 Summary – Evidence for Striped Bass Causing Population-Level Damage to Miramichi Atlantic Salmon

1. Atlantic salmon of the Northwest and Southwest composites of the Miramichi River system have been in decline since the steep drop-off in returns in 2012. The decline has been greatest on the Northwest composite where returns are at a critically low level.
2. The survival of tagged smolts from both the NW and SW composites during their travel through the estuary are significantly negatively correlated with bass numbers for the same year when the most recent years data are included in the analyses.
3. The grilse returns to both NW and SW Miramichi systems are significantly negatively correlated with bass numbers for the previous year when the most recent years data are included in the analyses.
4. The absence of a similar decline pattern in salmon returns to others rivers in the Gulf similar to the downturn in Miramichi salmon since 2011 rules out any density dependent effects in the marine environment, and such effects would be inconsistent with results reported in the literature.
5. The decline in Miramichi salmon is inconsistent with the stability of other Gulf salmon populations which, in general, have not been declining since 2011 (there have been some dips in some populations in most recent years but not long term declines).
6. The potential for the salmon downturn to be attributed to factors other than striped bass, factors such as delayed mortality from exposure to mine effluent, the result of the use of different sized smolts in pre versus post bass population explosion year tagging experiments, and tagging effects are not credible or have been invalidated.

We have therefore refuted the arguments put forth in Res Doc 2022/030, used by the author to justify the conclusion that the decline in and low salmon returns to the Miramichi system were not attributed to the growing bass population.

The decision of "no-effect" in Res. Doc. 2022/030 has had substantial consequences, specifically leading to **<u>NO</u>** action on the part of DFO to appropriately manage the Miramichi bass and salmon populations.

2.4.6 House of Commons Standing Committee on Fisheries and Oceans (FOPO) and DFO's Policy on Multi-Species Management

In February of 2019, the House of Commons Standing Committee on Fisheries and Oceans, commonly referred to as FOPO, held hearings on "Striped bass in the southern Gulf of St. Lawrence and Miramichi River: striking a delicate balance", also the title of their final report. The following relevant recommendation came from the report:

- Recommendation 2 - That Fisheries and Oceans Canada's restoration framework prioritize the long-term balance of fish species in the Southern Gulf of Saint Lawrence and Miramichi River (FOPO, 2019).

Chaput (2022) acknowledged this recommendation by referring to an "existing DFO policy". From Chaput (2022): "*DFO developed a policy to support rebuilding plans under the precautionary approach framework for stocks that are in the critical zone. DFO (2019) states that* ***in cases where rebuilding a stock has the potential to negatively impact the status of another, as in the case of rebuilding a predator species that could result in a decline of a prey species, rebuilding objectives need to be carefully developed through a balanced approach to ensure neither is depleted to a point of serious harm [our emphasis]****. Most importantly DFO (2019) acknowledged that it is not possible to simultaneously achieve yields corresponding to Maximum Sustainable Yield (MSY) predicted from single-species assessments for a system of multiple, interacting species and rebuilding efforts should be approached within an ecosystem context to the extent possible.*" (Note "MSY" is the maximum annual yield or harvest of a species that is possible without decreasing the population size.)

DFO has stated that in any future striped bass fisheries, the harvest, will not be great enough to allow the striped bass population to drop below the Limit Reference Point (LRP –DFO, 2023) of 330,000 spawners (A. Gagne, DFO Minister's assistant, personal communication to J. Bagnall, 2024). The LRP is a point below which a further population decline should be avoided to "prevent serious harm to the stock". For the Miramichi striped bass population, Res. Doc. 2022/029 states: "*Based on the trajectory of the population over the relatively short period of assessment, maintaining a spawner abundance that exceeds 330 thousand spawners should be more than sufficient to avoid serious harm to the population.*" To begin with, the established LRP is high given the knowledge that the bass population has successfully recovered from a level of ~4,500 spawners which was the mean estimated abundance for the period 1996 to 2000 (Chaput et. al., 2022). If 300,000 striped bass is "more than sufficient", then what would be considered a science-based LRP, or in other words, "sufficient" and therefore an actual science-based number reflecting the true LRP? Importantly, bass were surviving at numbers closer to 5,000 through the 1990s which is very clear evidence that the actual LRP is not anywhere close to the elevated 330,000 accepted by DFO today. Interestingly, a fisheries target or Optimum Sustainable Yield stock level of 100,000 bass was put forward as a "trial balloon" in Chaput (2022). We propose that this fisheries target level and an associated lower LRP value that has actual value in protecting the bass stock be implemented immediately so that DFO can assess the impact on the striped bass and other species of the Miramichi impacted by the bass.

2.4.7 2025 New Quota Announcement – Evidence of its Insufficiency

In July of 2024, DFO announced an increase in the allowable harvest of striped bass for the First Nations' commercial fishery. Much ado is being made of the total 175,000 bass quota. However, a portion of the original 50,000 quota (the actual harvested portion) of this number is already incorporated into the recent bass population trajectory, which is suspected to have reached an observed population of ~500,000 striped bass, a number hinted at during the Easten NB Coastal and Inland Recreational Fisheries Advisory Committee meeting in January – 2024. From Chaput and Douglas (2022): "The abundance trajectory of this population indicates that to date, the exploitation rate has been less than the surplus production of the population." In other words, the population in 2022 was stable, or perhaps (and probably) increasing despite the commercial harvest, and (at the time) a 3-fish retension limit in the daily recreational fishery. The increase in removals proposed to start in 2025 would involve the increased catch from achieving the original 50,000 bass quota, plus the announced potential harvest of an additional 125,000 bass, an increase in recreational retention from three to four fish daily, and the newly announced allowable by-catch in existing commercial fisheries. A critical point to understand is that the inviolable nature of a 330,000 bass LRP was communicated to us (A. Gagne, pers. Comm. to J. Bagnall – mentioned previously) after the July-2024 announcement of the increase in the allowable bass harvest for 2025 to a level of 175,000. The timing of these statements by DFO implies that they are confident the additional annual harvest allocation will not reduce the bass stock to a level fewer than 330,000, i.e. retaining the status quo of the illogically high LRP.

At maximum sustainable yield (MSY), a 200,000 to 400,000 allowable harvest would be possible under the selected model DFO is presumably working from (drawn from Chaput and Douglas , 2022). Assuming a more reasonable estimate of the very high 2017 stock estimate, which we and others view as an outlier, the bulk of the currently assumed stock level of ~500,000 bass was propped up by the progeny of (recruitment from) an average of approximately 330,000 spawners from 2015 to 2018, this despite commercial and recreational fishery removals. Fish from previous generations reproduce to replace those removed. As it is, the stock is increasing rapidly, and it appears to be approaching the level that will support MSY in a single species context. Thus, the net effect of the announced 2025 (and future) removal increases is the high potential for a negligible-to-non-existent change in the spawning stock. In fact, despite the potential increased removals, the stock may continue to increase and may have already increased substantially since the last publish stock value of 471,000 for 2022. (As an aside, why has the 2023 striped bass stock level not been announced yet? It was known in January – 2024. What is the big secret?)

2.4.8 The 100,000 Bass Target – Confirmation through a Population Model

Despite the conclusions of Chaput (2022), our subsequent work referred to above has demonstrated that past bass numbers and smolt mortality through the estuary are highly correlated for both drainage composites of the Miramichi, and the relationships are statistically significant. The CRI predator tag study has identified striped bass as the reason for the increased smolt mortality over the past 13 or 14 years.

The correlation between bass numbers and smolt mortality has predictive value. We have developed a model that estimates Atlantic salmon population abundance associated with various striped bass population levels plus, salmon metrics for initial egg deposition, survival rate to the smolt stage, survival rates in the high seas, and repeat spawner numbers. We explain its components in the next sub-section, but what the model does is compare the number of eggs initially deposited with the number of eggs that are deposited in the next generation by the survivors. The popular conclusion discussed previously is that a striped bass population of 100,000 allows the Miramichi salmon populations in each major composite drainage (rivers identified previously) to be sustainable. This will be confirmed, at least partially in the sub-sections below.

2.4.9 Egg-to-smolt / Smolt-to-Egg Model

The following is an explanation of how we arrived at the 100,000 bass stock number. The model, called an "Egg-to-smolt / Smolt-to-Egg" exercise judges population sustainability according to bass population numbers primarily by varying the survival rates of smolt salmon through the estuary. Variability in the survival rates is produced according to the regression of the historic annual mortality (1 minus survival) rates of tagged smolts as measured by the Atlantic Salmon Federation and the bass number as published by DFO. Other model variables or calculated values assumed or included are:

a. Initial salmon Egg Deposition. 2.4 eggs per m^2 is the classic Elson's normal that was used as a the bench mark for a sustainable salmon population in Atlantic Canadian rivers, until the Precautionary Approach method replaced it with different lower and upper reference points that are unique to individual rivers. The actual deposition rate is now desperately low on the NW composite, assumed to be approximately 0.5 eggs per m^2, and 1.2 eggs per m^2 on the SW composite. The deposition rate would increase in the next generation, if more adults return to spawn than what produced the eggs in the first place. The calculated surplus fish value and subsequent egg deposition would decrease if returning runs plus repeat spawners are lower than the input values or if the returning pre-spawning recruits are over-harvested.

b. The deposition rate is multiplied by the wetted area of the drainage to get total eggs deposited; also the standard metric from DFO.

c. MSA Southesk Hatchery eggs. This can be any number up to 11,500,000, perhaps more. This maximum assumes 6 tanks of MSW equivalents, tanks with 2,908 ft^3 volume, 1.25 lb/ft^3, 12 lb/ female, 95% female from the culling of males, and a fecundity of 7,387 eggs per female.

d. Total egg deposition. SAS eggs and naturally spawned eggs are added.

e. Egg-to-smolt Survival. An assumption. The model is very sensitive to this number. It is an index of habitat quality and density dependence at high egg depositions. It also depends on the percentage of 2-year-old smolts vs. 3-year-old smolts, with a higher survival if 2-year-olds dominate. The observed range is from 0.5% to 3%. Judging from recently published survival rates from the adjacent Nashwaak and Tobique drainages (Gibson et. al., 2016) and from Figure 7 (from O'Connell et. al., 2006), a value of 1% was used. This rate is conservatively high as both the published Nashwaak and Tobique rates are <1%.

f. Bass numbers. This determines the striped bass induced smolt mortality rate. Historically, bass numbers have ranged from <5,000 to what is now suspected to be ~600,000.

g. Smolt-to-High-Sea's survival. As alluded to previously, the smolt-to-high-seas survival value in the next row is calculated from a correlation between known estimates of bass numbers and mortality of smolts travelling though the estuary. (Several notes: First, the 2017 outlier year of 2017 was eliminated from the regressions. Second, a bass value of 500,000 was used for 2023, a number that was based on statements made by DFO at a consultation meeting in January. The bass number for 2010 was underestimated by DFO and therefore that year's estimate was not used. Bass data for 2012 and 2020 <Covid> were not available.) There is a different correlation equation for each major sub-drainage, the SW Composite, and the NW Composite. The variation is due to different annual rates of survival through the estuary for smolts from the two drainage composites, with SW smolts experiencing historically slightly lower mortality rates than those from the NW composite. As can be judged from the very high significance value, the excel-provided trend lines and formulas for them fit the empirical data very well. For the NW composite, the log trend, exponential trend, and power trend each have coefficients of determination of >84% with the power trend offering the best at 86%, so it was chosen. On the SW, a simple linear trend was acceptable. The equations for each of these trend lines were used to calculate the mortality, and by subtraction (1 minus mortality= survival), the survival rates of smolts from each drainage composites during their transits through the estuary.

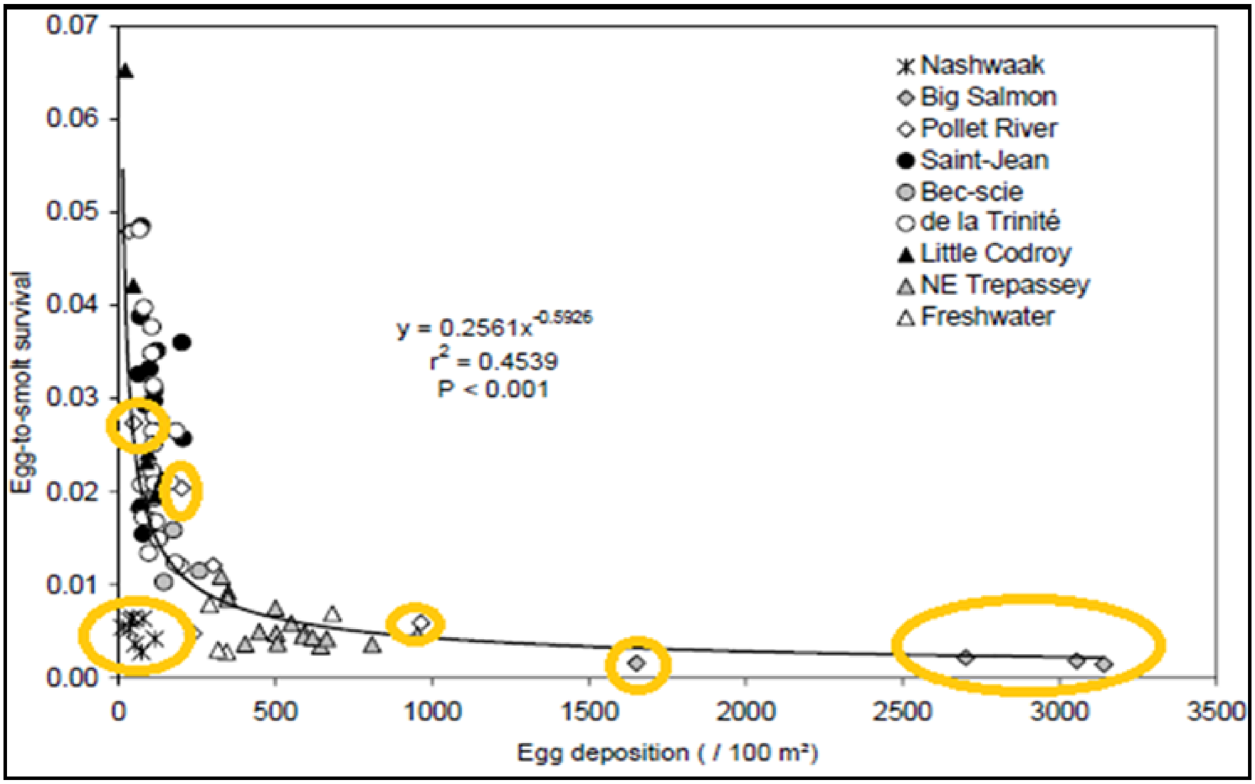


Figure 7 Egg-to-Smolt Survival rates for Western Atlantic Rivers (yellow circled values are from NB rivers)

h. <u>Subsequent high seas survival</u>. The next calculation is to estimate a subsequent high seas survival rate for the smolts that survive from Portage Island to the high seas and back. Based on advice from DFO (M. Hardy to J. Bagnall, 2024) the survival is variable, to account for a potentially higher rate of survival for smolts surviving high bass numbers – i.e., bass select for the larger, stronger smolts. The value used is calculated from the regression of predicted compensatory survival values (low when estuary survival is high / high when estuary survival is low). The logic for the regression, as per Chaput et. al. (2018), is the observation that larger smolts have 1.5 to 1.7 times the survival advantage of shorter smolts. A classic rule-of-thumb is that acceptable survival rates from smolts to adults is 5%, which was used as the baseline for very low bass populations. 1.7 times this is 8.5%, and this value was the upper limit used for very high bass populations associated with high smolt mortality. A linear regression was used to calculate high seas survival rates associated with moderate smolt mortality rates associated with moderate bass population numbers.

i. % Repeat Spawners. Salmon are potential multi-time spawners. Between 8% and 25% of grilse and 10% to 40% of MSWs survive to spawn again (from Chaput et.al., 2016). The NW/LSW has fewer MSWs and more grilse spawners than the SW/Renous (34/66 vs. 45/55 respectively). Therefore, the total percent repeat spawners varies with which major sub-drainage is chosen. We chose a conservatively high 20% survival for grilse and 30% for MSWs. A separate calculation can be used to determine the number of grilse and MSWs contribute to the repeat spawning class – i.e., to simulate recovery or decline. For simplicity purposes, the initial maiden egg deposition rate has been duplicated and used in calculations below to determine the repeat spawner numbers.

j. Repeat spawners calculated from percentages of the maiden spawner returns (although, as indicated, this can vary).

k. Repeat spawners plus pre-spawning maiden adults = "Total spawners".

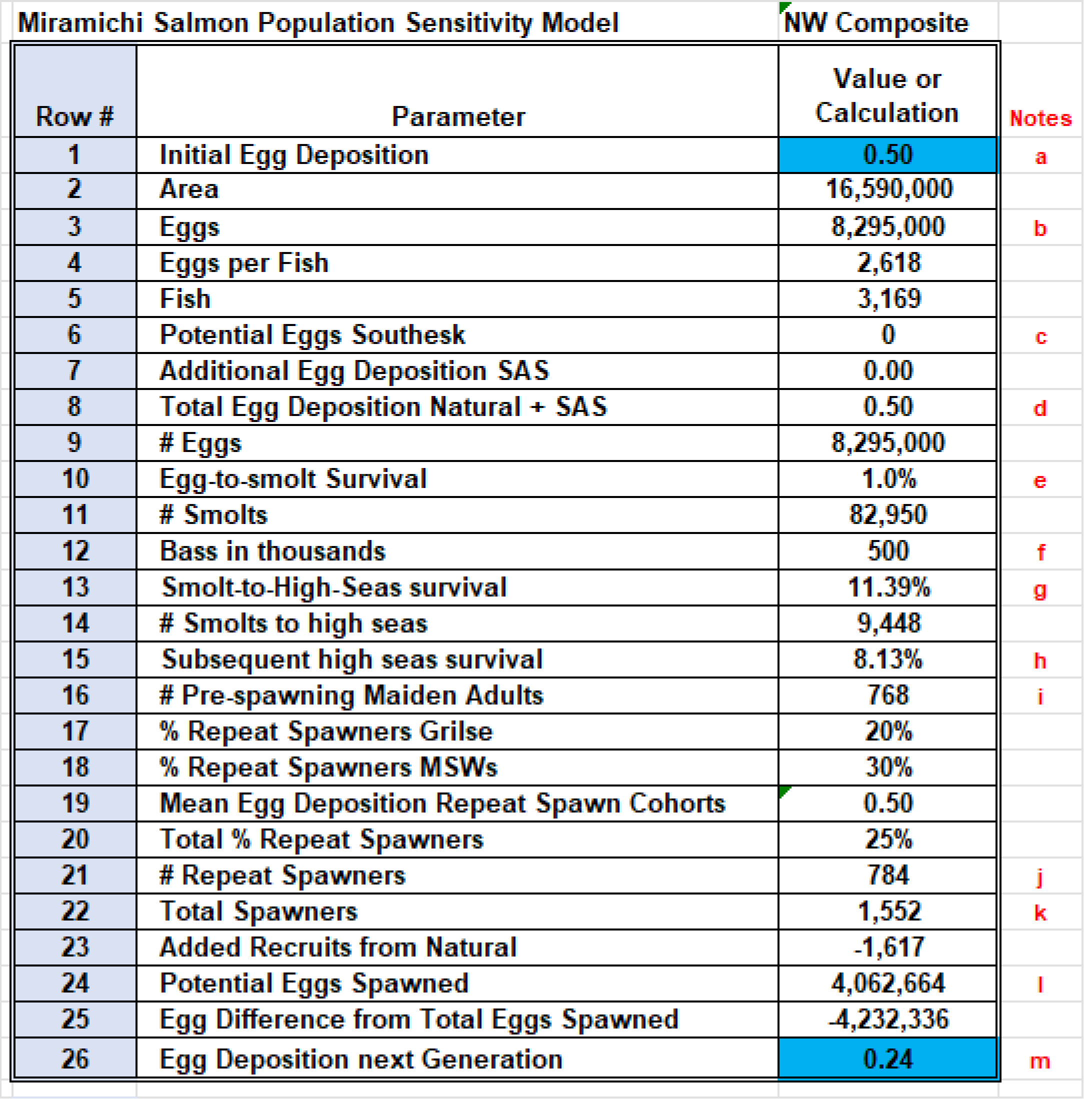
l. The number of eggs potentially spawned by these fish calculated according to fecundity demographics.

m. The value in "l" is converted to a deposition rate next generation according to the drainage wetted area. If the value in Row 26 is greater than that of Row 1, the population is poptentially sustainable.

As implied previously, the model provides a salmon and grilse return number that is converted to a potential egg deposition rate that in turn determines whether the population is potentially sustainable as long as fishery removals are not too great. The bottom-line, zero-fishery-removal, egg deposition rate that is highlighted in blue (Row 26) is compared with the initial rate, also highlighted in blue (Row 1), and to judge sustainability, the bottom line should exceed the top line, and to account for unforeseen survival decreases, by a substantial amount. Following are several model runs to demonstrate the necessity of a maximum of 100,000 bass to ensure Miramichi salmon sustainability. The first two (Tables 3 and 4) are approximately where we are now for the NW composite drainage and for the SW drainage to predict the effect on sustainability. No hatchery supplementation is assumed.

Our egg depositions (Row 1) are based on extrapolations from 2024 barrier captures. The results (Row 26) are approximately what we have experienced and will continue to experience with an excessively large bass population. The Northwest population has crashed and that of the SW has slowly diminished. With 500,000 bass, even supplementation with the maximum 11.5 million eggs from the Southesk hatchery would not bring the NW anywhere near sustainability. This model run is not shown, but the result of 500,000 bass and supplementation with 11.5 million hatchery eggs is a deposition rate of 0.41 eggs per m^2 from the initial deposition of 0.5 eggs per m^2, up from a "no supplementation" rate of 0.24 eggs per m^2.

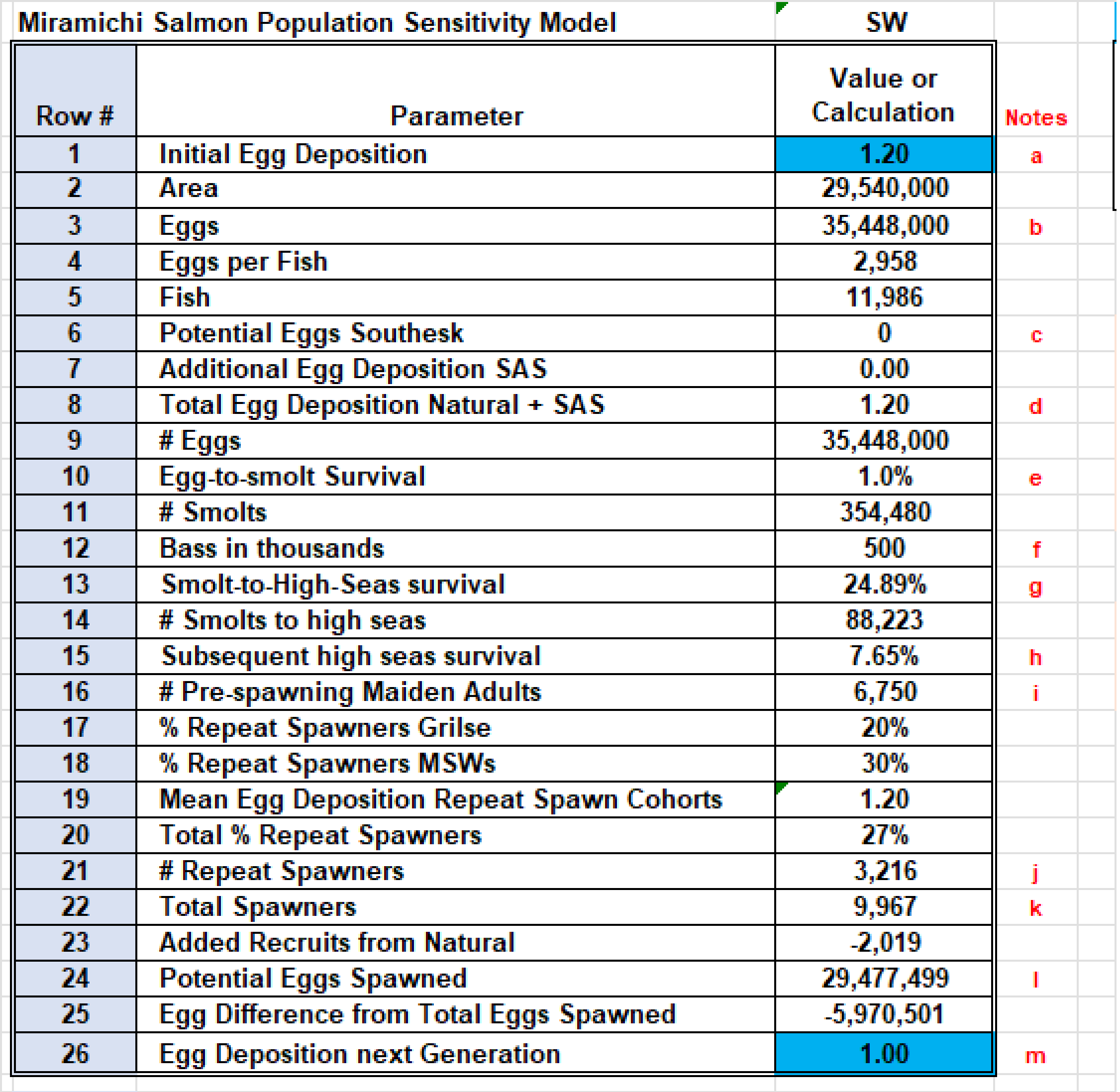
Table 3 Northwest Composite Run Assumed 0.5 eggs/ m^2. Bass population= 500K, probably a reasonable value as of 2024. The notes in red correspond to the explanations above.)



Miramichi Salmon Population Sensitivity Model — NW Composite

Row #	Parameter	Value or Calculation	Notes
1	Initial Egg Deposition	0.50	a
2	Area	16,590,000	
3	Eggs	8,295,000	b
4	Eggs per Fish	2,618	
5	Fish	3,169	
6	Potential Eggs Southesk	0	c
7	Additional Egg Deposition SAS	0.00	
8	Total Egg Deposition Natural + SAS	0.50	d
9	# Eggs	8,295,000	
10	Egg-to-smolt Survival	1.0%	e
11	# Smolts	82,950	
12	Bass in thousands	500	f
13	Smolt-to-High-Seas survival	11.39%	g
14	# Smolts to high seas	9,448	
15	Subsequent high seas survival	8.13%	h
16	# Pre-spawning Maiden Adults	768	i
17	% Repeat Spawners Grilse	20%	
18	% Repeat Spawners MSWs	30%	
19	Mean Egg Deposition Repeat Spawn Cohorts	0.50	
20	Total % Repeat Spawners	25%	
21	# Repeat Spawners	784	j
22	Total Spawners	1,552	k
23	Added Recruits from Natural	-1,617	
24	Potential Eggs Spawned	4,062,664	l
25	Egg Difference from Total Eggs Spawned	-4,232,336	
26	Egg Deposition next Generation	0.24	m

Table 4 Southwest Composite Run (1.2 eggs per m^2 and a 500K bass population)



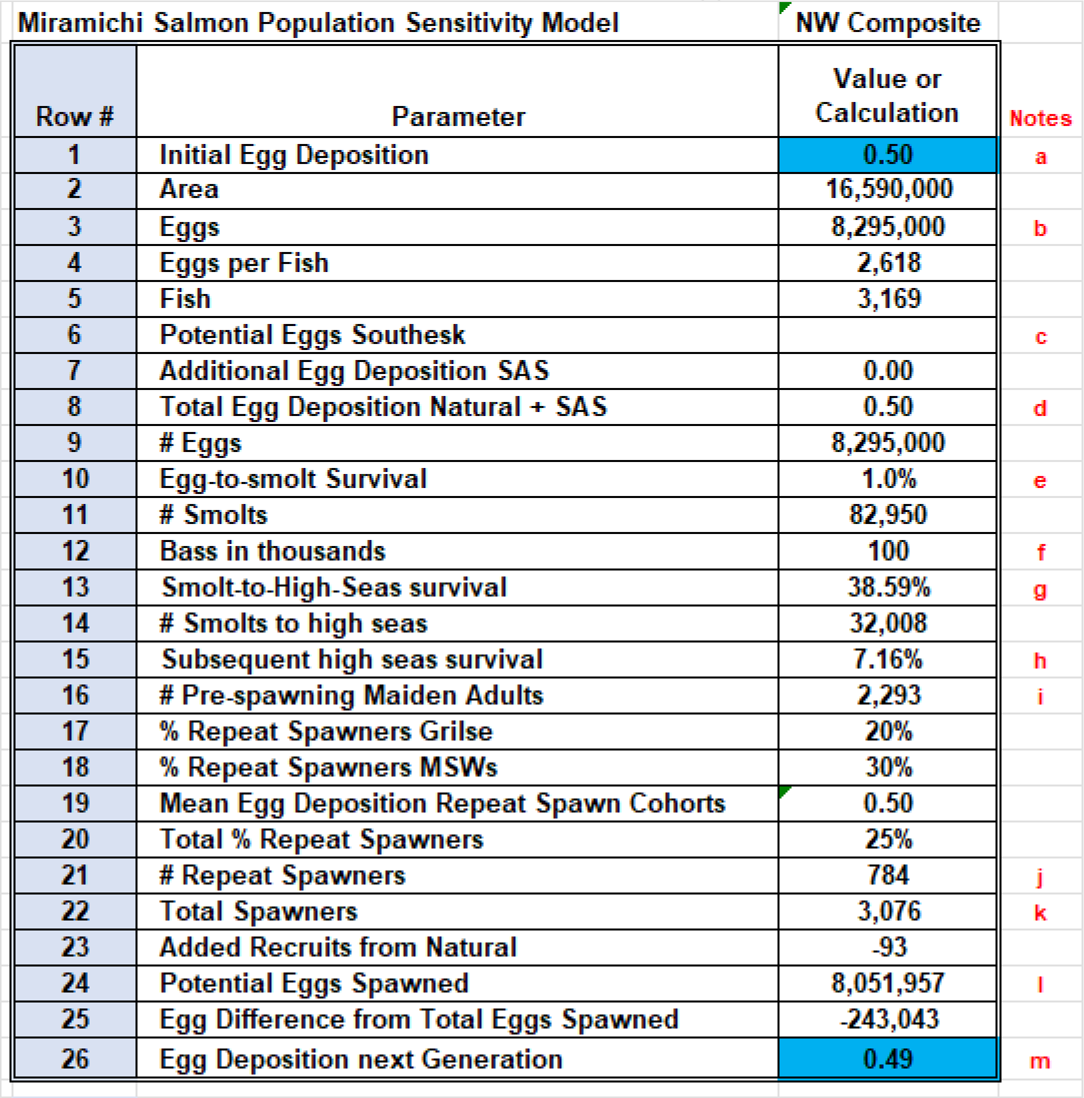
Miramichi Salmon Population Sensitivity Model — SW

Row #	Parameter	Value or Calculation	Notes
1	Initial Egg Deposition	1.20	a
2	Area	29,540,000	
3	Eggs	35,448,000	b
4	Eggs per Fish	2,958	
5	Fish	11,986	
6	Potential Eggs Southesk	0	c
7	Additional Egg Deposition SAS	0.00	
8	Total Egg Deposition Natural + SAS	1.20	d
9	# Eggs	35,448,000	
10	Egg-to-smolt Survival	1.0%	e
11	# Smolts	354,480	
12	Bass in thousands	500	f
13	Smolt-to-High-Seas survival	24.89%	g
14	# Smolts to high seas	88,223	
15	Subsequent high seas survival	7.65%	h
16	# Pre-spawning Maiden Adults	6,750	i
17	% Repeat Spawners Grilse	20%	
18	% Repeat Spawners MSWs	30%	
19	Mean Egg Deposition Repeat Spawn Cohorts	1.20	
20	Total % Repeat Spawners	27%	
21	# Repeat Spawners	3,216	j
22	Total Spawners	9,967	k
23	Added Recruits from Natural	-2,019	
24	Potential Eggs Spawned	29,477,499	l
25	Egg Difference from Total Eggs Spawned	-5,970,501	
26	Egg Deposition next Generation	1.00	m

A caveat to be aware of with these runs and as alluded to In Point “i” above, is the populations that determine repeat spawning may be higher than those assumed here, and therefore egg depositions from repeat spawning may be higher than calculated and bottom line egg deposition rates also higher. When the previous populations were very high in comparison with that of the assumed maiden population, the population decline would be lower than shown above. Our model runs here simply duplicate the applied maiden population to calculate repeat spawning numbers and egg deposition from repeat spawners. Of course the reverse is true, if the past populations are lower than those of the maiden population (a recovery scenario), the bottom line deposition would be lower because of fewer repeat spawners contributing to it.

The following two runs (Tables 5 and 6) demonstrate what would happen with a 100,000 bass stock number.

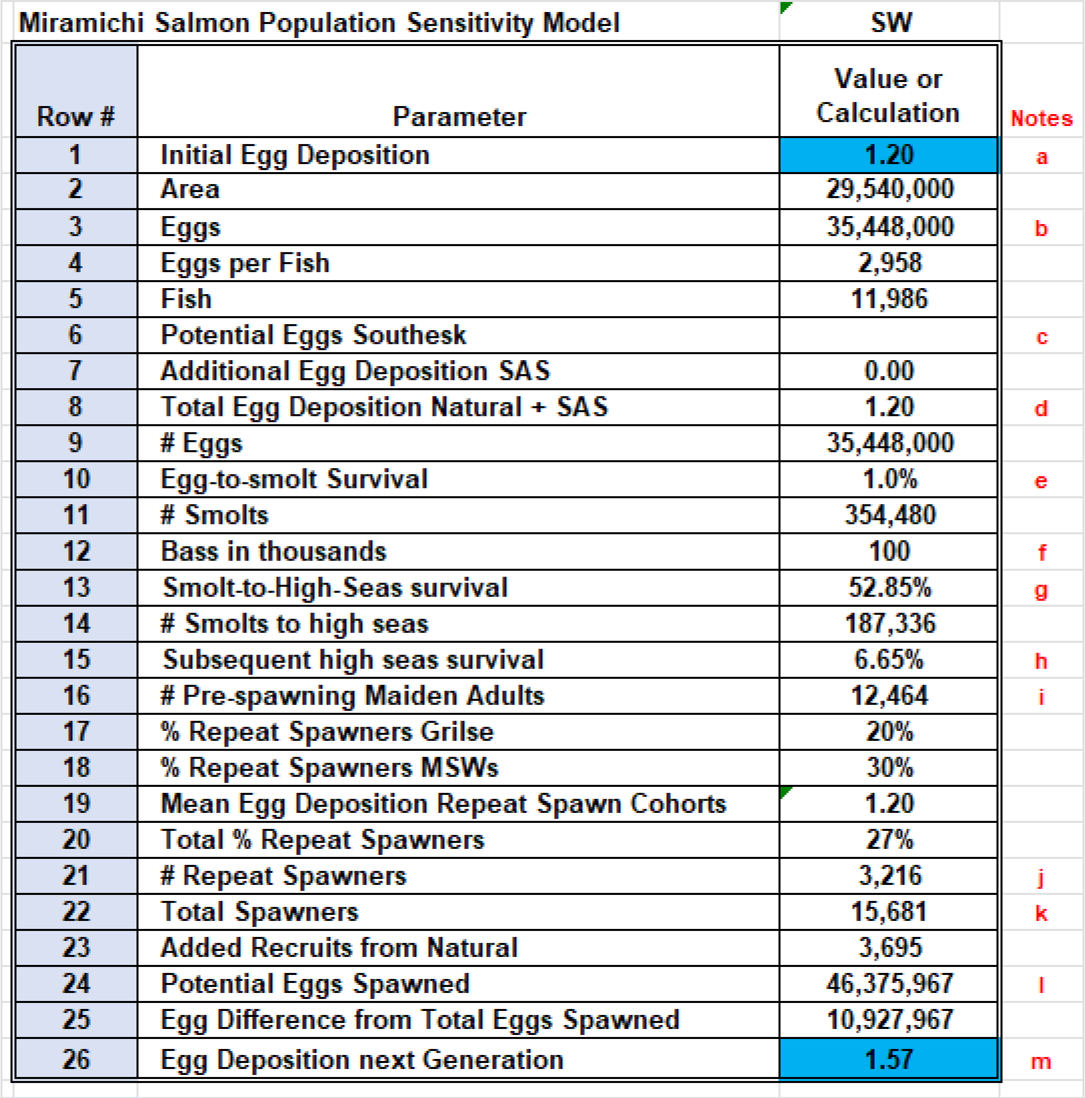
Table 5 NW Composite Run (100,000 Bass and a 0.5 Eggs per m^2 Deposition Rate)



Miramichi Salmon Population Sensitivity Model — NW Composite

Row #	Parameter	Value or Calculation	Notes
1	Initial Egg Deposition	0.50	a
2	Area	16,590,000	
3	Eggs	8,295,000	b
4	Eggs per Fish	2,618	
5	Fish	3,169	
6	Potential Eggs Southesk		c
7	Additional Egg Deposition SAS	0.00	
8	Total Egg Deposition Natural + SAS	0.50	d
9	# Eggs	8,295,000	
10	Egg-to-smolt Survival	1.0%	e
11	# Smolts	82,950	
12	Bass in thousands	100	f
13	Smolt-to-High-Seas survival	38.59%	g
14	# Smolts to high seas	32,008	
15	Subsequent high seas survival	7.16%	h
16	# Pre-spawning Maiden Adults	2,293	i
17	% Repeat Spawners Grilse	20%	
18	% Repeat Spawners MSWs	30%	
19	Mean Egg Deposition Repeat Spawn Cohorts	0.50	
20	Total % Repeat Spawners	25%	
21	# Repeat Spawners	784	j
22	Total Spawners	3,076	k
23	Added Recruits from Natural	-93	
24	Potential Eggs Spawned	8,051,957	l
25	Egg Difference from Total Eggs Spawned	-243,043	
26	Egg Deposition next Generation	0.49	m

Table 6 Southwest Miramichi Run (100,000 Bass and a 1.2 Eggs per m^2 Deposition Rate)



Miramichi Salmon Population Sensitivity Model		SW	
Row #	Parameter	Value or Calculation	Notes
1	Initial Egg Deposition	1.20	a
2	Area	29,540,000	
3	Eggs	35,448,000	b
4	Eggs per Fish	2,958	
5	Fish	11,986	
6	Potential Eggs Southesk		c
7	Additional Egg Deposition SAS	0.00	
8	Total Egg Deposition Natural + SAS	1.20	d
9	# Eggs	35,448,000	
10	Egg-to-smolt Survival	1.0%	e
11	# Smolts	354,480	
12	Bass in thousands	100	f
13	Smolt-to-High-Seas survival	52.85%	g
14	# Smolts to high seas	187,336	
15	Subsequent high seas survival	6.65%	h
16	# Pre-spawning Maiden Adults	12,464	i
17	% Repeat Spawners Grilse	20%	
18	% Repeat Spawners MSWs	30%	
19	Mean Egg Deposition Repeat Spawn Cohorts	1.20	
20	Total % Repeat Spawners	27%	
21	# Repeat Spawners	3,216	j
22	Total Spawners	15,681	k
23	Added Recruits from Natural	3,695	
24	Potential Eggs Spawned	46,375,967	l
25	Egg Difference from Total Eggs Spawned	10,927,967	
26	Egg Deposition next Generation	1.57	m

These runs demonstrate the NW composite's salmon population becomes almost sustainable with a bass population of 100,000. A decline in any of the survival rates would plunge the salmon spawning population into decline. Besides bass population reduction, large initial annual hatchery supplementation is needed to boost recovery.

With 100,000 bass, the Southwest's salmon population would become sustainable, although at a bottom-line lower-than-acceptable egg deposition rate for rapid recovery. The SW composite's drainage area is so large that blanket stocking with Smolt-to-Adult Supplementation fish or 0+ fry would be impractical. Introductions into SW habitat that is extremely underpopulated with juveniles would be the best use of hatchery production, of which the great majority should be targeted to the more vulnerable NW composite.

This model, like any others comes with the caveat that personal bias could skew results. However, as indicated in the explanations of variables, conservative values were used that favoured lower calculated mortality rates and therefore higher bottom-line deposition rates. The results indicate a bass stock level of greater than 100,000 is incompatible with Miramichi salmon sustainability. This agrees with observations and speculation in Chaput (2022):

- *"For both the Southwest Miramichi and Northwest Miramichi tagged smolts, the lowest survival rates from head of tide to bay exit were estimated in the recent period (2013 to 2016) when the estimated abundance of Striped Bass was greater than 100 thousand spawners.*
- *For the Southwest Miramichi, there have been equally low relative survival rates at very low and very high Striped Bass spawner abundances, with higher relative survivals of salmon at Striped Bass abundances of 20 to 100 thousand spawners.*
- *Based on acoustic tagging estimates of survivals through Miramichi Bay, the years when Striped Bass spawner abundances exceeded approximately 100 thousand spawners corresponded to year with visibly lower estimated survival rates*
- *Striped Bass abundances in the range of 100 thousand spawners in the past corresponded to high landings of gaspereau and smelt, and the highest survival rates of acoustically tagged smolts through Miramichi Bay. Setting a management objective for Striped Bass at approx. 100 thousand spawners, perhaps calling this a target reference point (rather than upper stock reference), will result in large reductions of the potential fisheries yield of Striped Bass."*

The final mention of 100,000 bass in Res. Doc. 2022/030 is in the following:

- *"It is not clear from these time series of data, that reducing Striped Bass spawner abundances to the level of the early 2000s, i.e., less than 100 thousand spawners, would improve the acoustic tagged smolt survival estimates, the population level relative survival rates derived from the cohort model, or the landings trends of gaspereau and Rainbow Smelt in the commercial fisheries."*

The conclusion in the last bulleted paragraph was annulled during previous discussions when the cohort analysis method of determining smolt output was invalidated and density independence of smolts in the high seas was asserted.

2.4.10 Getting to a Stock Level of 100,000 Bass

This begs the question as to how the sGSL's striped bass population can be reduced to achieve the 100,000-bass target. The obvious answer is via an increased commercial fisheries' harvest. What magnitude this should take is a vital question to answer, and we have attempted to do so in the following.

Various stock-recruitment models are discussed in research DFO (2021), and in Chaput and Douglas (2022). These documents are complex with many scenarios being examined that seem to understandably confound the authors as much they did our group when we read them. That being said, what we do know is the Limit Reference Point (LRP) below which population harm may occur was established in these documents at 330,000 spawners, and that this number recruits (new fish added to the "stock" <comprised of fish large enough for the fishery and to spawn>) to a population that is one half the maximum that the environment will support. Although other models mentioned in the previously referenced documents can produce stocks of considerably more, the maximum stock level that seems to have been settled on is a value of just over one million.

We assumed a maximum 1.2 million spawners, a conservative value. The range in ½ "K" (carrying capacity) from Table 6.4(a), Model 5 (Res. Doc. 2022.029) is 500 to 570 spawners, which is doubled to one million to 1.14 million. This closely agrees with our assumed value. Figure 8, reproduced from Res. Doc. 2022/029 gives credence to our approximation.

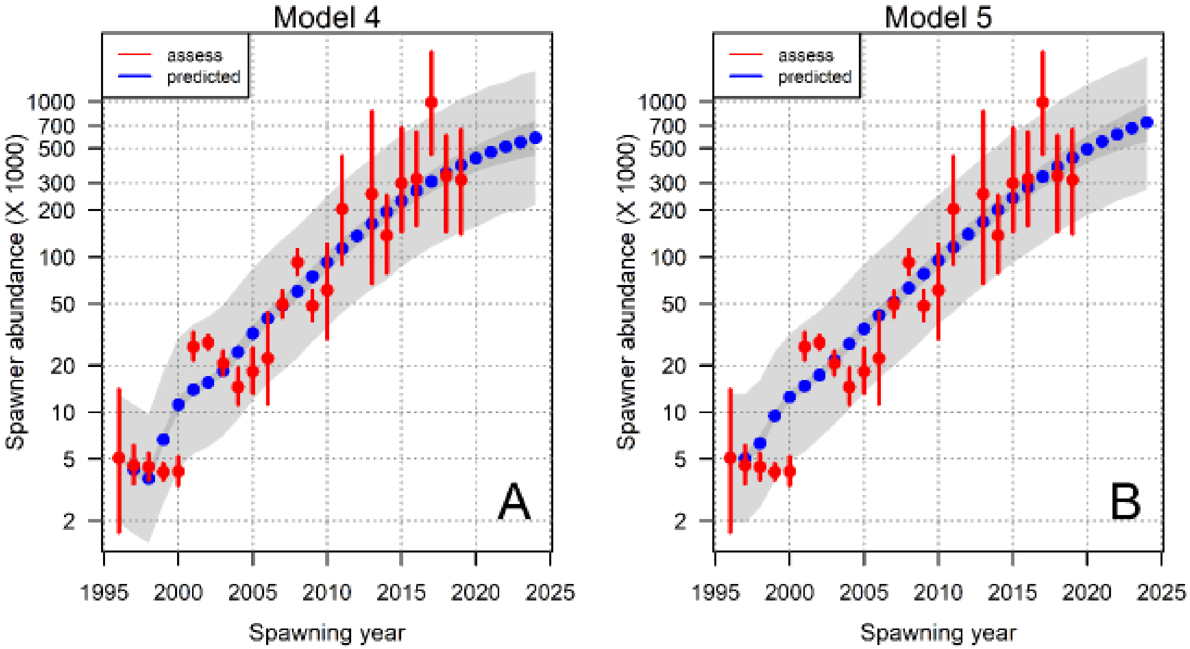


Figure 8 **Southern Gulf of St. Lawrence Bass Population Trajectory and Projection (Chaput and Douglas, 2022)**

This means we have three known values, zero or very close to it, which recruits to zero fish, 330,000 that recruits to one-half of 1.2 million, or 600,000 fish, and 1.2 million, the maximum stock level, which recruits to itself, 1.2 million. This all that is needed for Excel to produce a regression and calculate an exponential trend line. This line resembles a Beverton-Holt stock-recruitment curve, which allows prediction of the effects of various harvest levels. The MSY occurs at a stock (spawning population) level of 635,000, which generates a recruited population of 983,000 and 358,000 recruits. Therefore, it is more conservative than the assumptions in Res. Doc. 22/29. Figure 9 is the "Beverton-Holt" Curve that was generated.

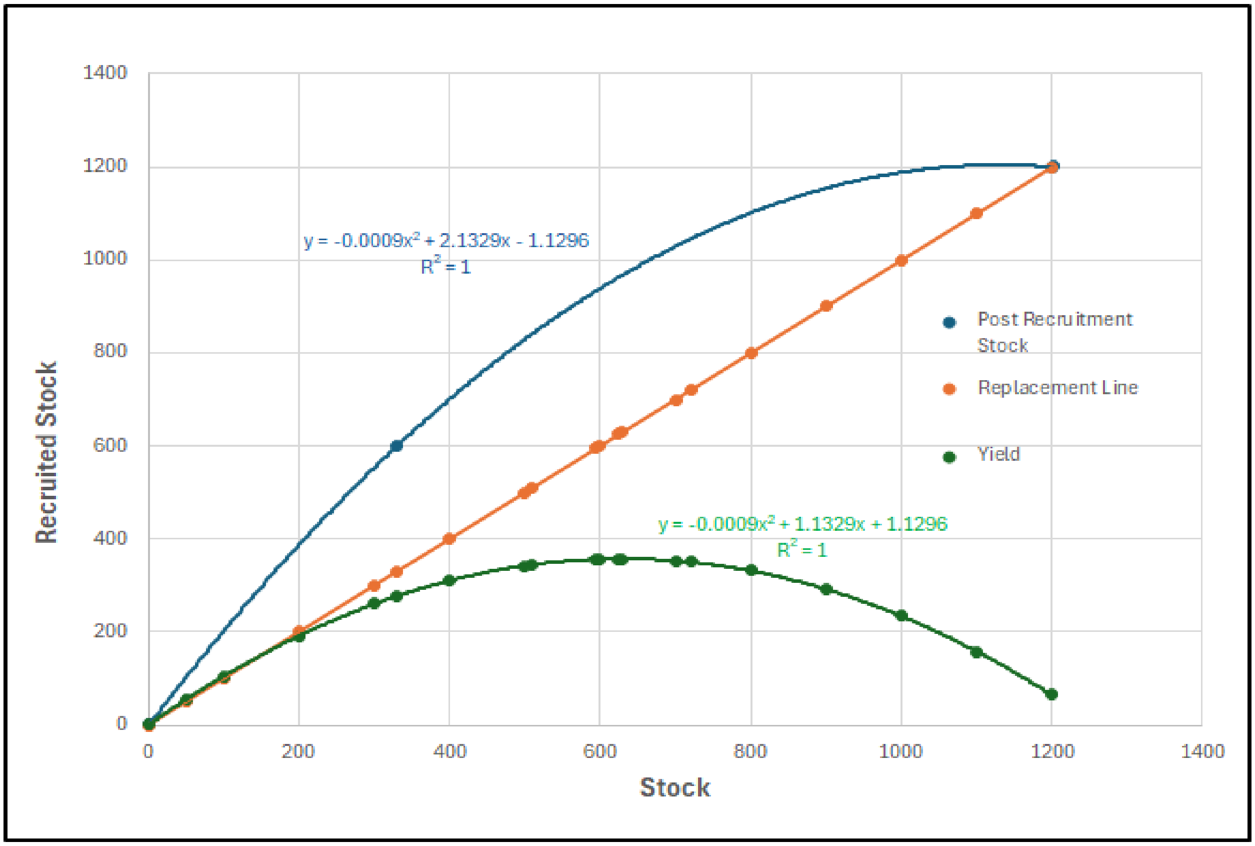
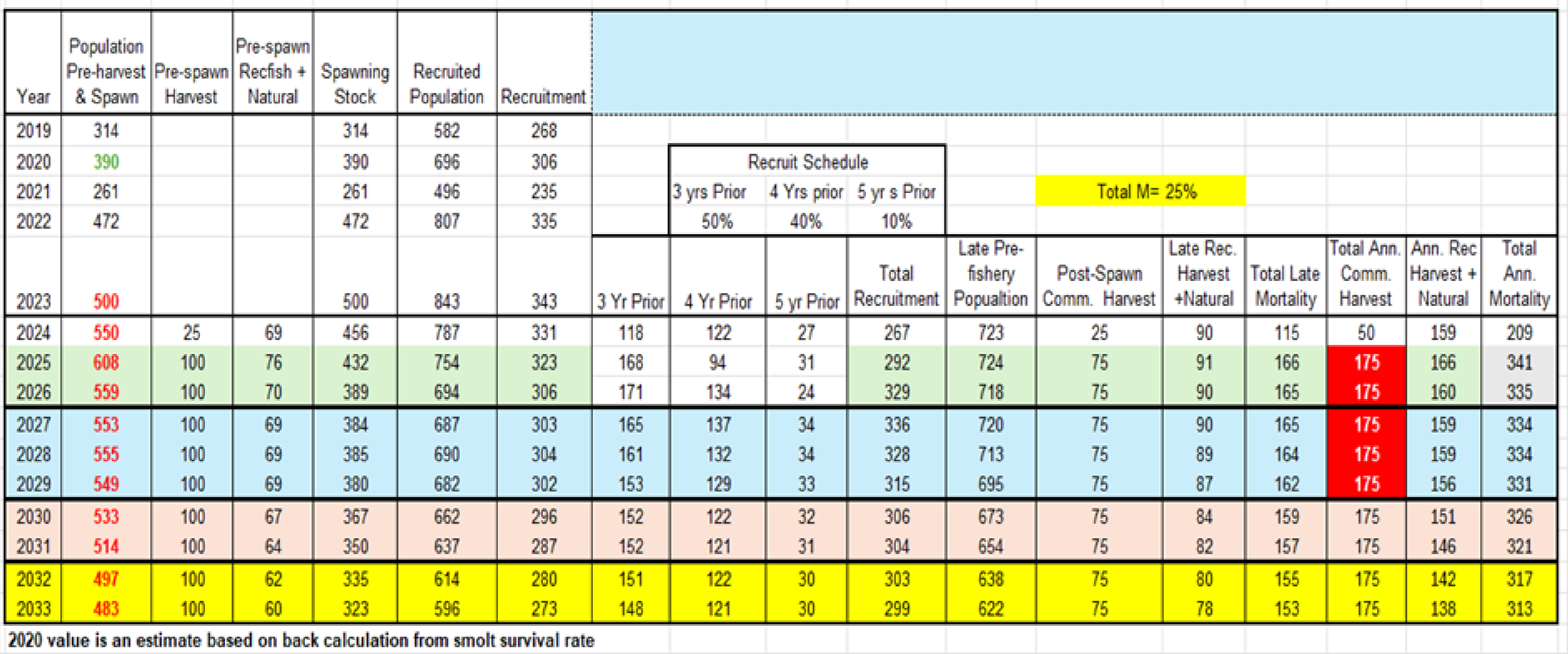


Figure 9 Beverton-Holt Striped Bass Stock-Recruitment Model Developed from First Principles

Using the predictive potential of the trend line formula, the following table (Table 7), employs reasonable assumptions for commercial and recreational fishery + natural mortality values before and after spawning and a recruitment schedule of 50% of age 3s (recruitment from 3 years prior), 40% of age 4s, and 10% of age 5s, to predict the next year's initial stock to which salmon smolts and smolts would be exposed. Natural and fishing mortality are combined into a rate of 25%, 12.5% for the six months after fishing the previous year and before spawning and 25% after spawning until the end of fishing in the year in question. This approximately agrees with the "M" rate of 22% that is discussed in such documents as DFO (2021), and Chaput and Douglas (2022). An extra three percentage points were added to account for an increase in recreational fishing because of the increase in the creel limit from three to four fish per day. DFO might question the validity of our assumptions, and we acknowledge they are unsupported estimates. But we think the removal rates are conservatively high, and we do know that the sum of the proposed new commercial harvest, the natural mortality values, and the recfish removals will not bring the bass stock to a point that is lower than the 330,000 LRP. We know this because the DFO Minister's assistant told us so (again) **after** the announcement of the allotted increase. (Refer to Section 2.5.7 - above.)

Table 7 demonstrates how the recently announced commercial harvest level of 175,000 will barely budge the initial stock level that is assumed to have the greatest effect on salmon smolt and smolt survival in the lower rivers and estuaries. Table 8 shows how a much greater commercial harvest is required to rapidly bring the striped bass stock to an environmentally acceptable level of approximately100,000.

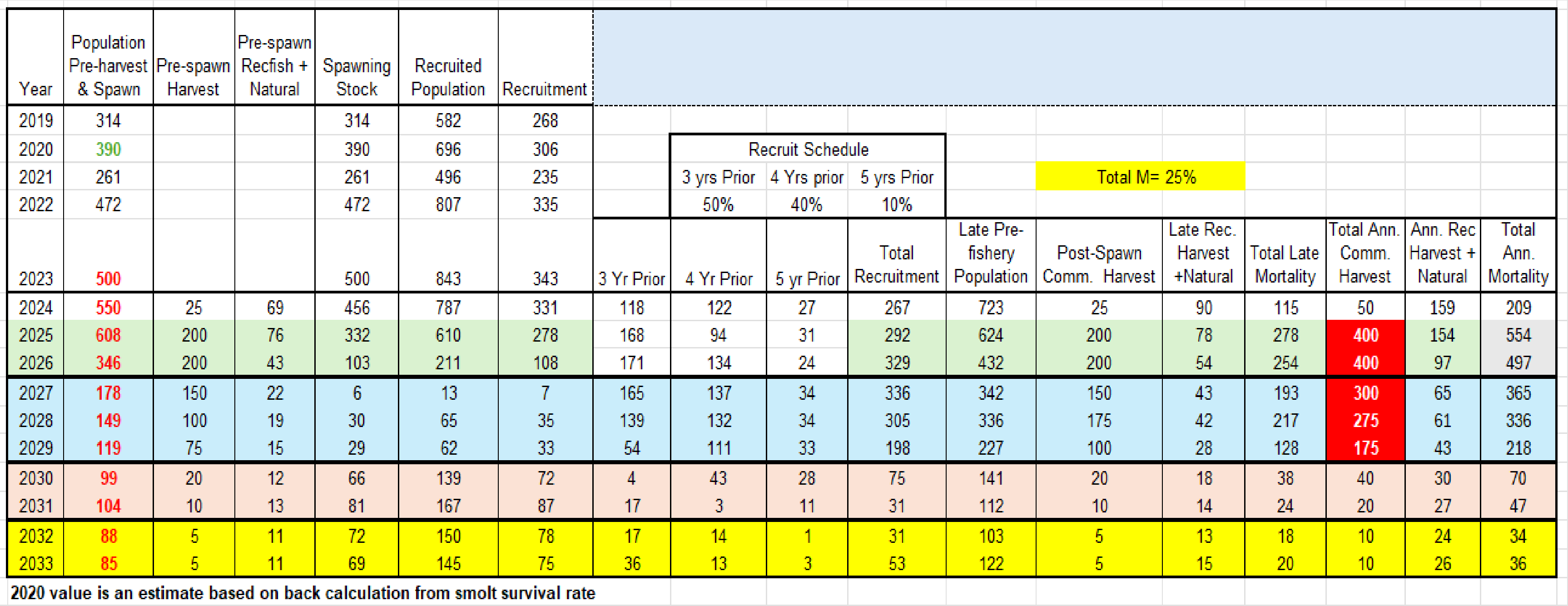
Table 7 Model Demonstrating the Inadequacy of the Recently Announced 175,000 Bass Harvest Level to Achieve a Bass Stock Level of 100,000



Year	Population Pre-harvest & Spawn	Pre-spawn Harvest	Pre-spawn Recfish + Natural	Spawning Stock	Recruited Population	Recruitment	3 Yr Prior	4 Yr Prior	5 yr Prior	Total Recruitment	Late Pre-fishery Popualtion	Post-Spawn Comm. Harvest	Late Rec. Harvest +Natural	Total Late Mortality	Total Ann. Comm. Harvest	Ann. Rec Harvest + Natural	Total Ann. Mortality
2019	314			314	582	268											
2020	390			390	696	306		Recruit Schedule									
2021	261			261	496	235		3 yrs Prior	4 Yrs prior	5 yr s Prior		Total M= 25%					
2022	472			472	807	335		50%	40%	10%							
2023	500			500	843	343											
2024	550	25	69	456	787	331	118	122	27	267	723	25	90	115	50	159	209
2025	608	100	76	432	754	323	168	94	31	292	724	75	91	166	175	166	341
2026	559	100	70	389	694	306	171	134	24	329	718	75	90	165	175	160	335
2027	553	100	69	384	687	303	165	137	34	336	720	75	90	165	175	159	334
2028	555	100	69	385	690	304	161	132	34	328	713	75	89	164	175	159	334
2029	549	100	69	380	682	302	153	129	33	315	695	75	87	162	175	156	331
2030	533	100	67	367	662	296	152	122	32	306	673	75	84	159	175	151	326
2031	514	100	64	350	637	287	152	121	31	304	654	75	82	157	175	146	321
2032	497	100	62	335	614	280	151	122	30	303	638	75	80	155	175	142	317
2033	483	100	60	323	596	273	148	121	30	299	622	75	78	153	175	138	313

2020 value is an estimate based on back calculation from smolt survival rate

Table 8 Proposed Model for Rapidly Achieving a Bass Stock Level of 100,000



Year	Population Pre-harvest & Spawn	Pre-spawn Harvest	Pre-spawn Recfish + Natural	Spawning Stock	Recruited Population	Recruitment	3 Yr Prior	4 Yr Prior	5 yr Prior	Total Recruitment	Late Pre-fishery Population	Post-Spawn Comm. Harvest	Late Rec. Harvest +Natural	Total Late Mortality	Total Ann. Comm. Harvest	Ann. Rec Harvest + Natural	Total Ann. Mortality
2019	314			314	582	268											
2020	390			390	696	306		Recruit Schedule									
2021	261			261	496	235		3 yrs Prior	4 Yrs prior	5 yrs Prior		Total M= 25%					
2022	472			472	807	335		50%	40%	10%							
2023	500			500	843	343											
2024	550	25	69	456	787	331	118	122	27	267	723	25	90	115	50	159	209
2025	608	200	76	332	610	278	168	94	31	292	624	200	78	278	400	154	554
2026	346	200	43	103	211	108	171	134	24	329	432	200	54	254	400	97	497
2027	178	150	22	6	13	7	165	137	34	336	342	150	43	193	300	65	365
2028	149	100	19	30	65	35	139	132	34	305	336	175	42	217	275	61	336
2029	119	75	15	29	62	33	54	111	33	198	227	100	28	128	175	43	218
2030	99	20	12	66	139	72	4	43	28	75	141	20	18	38	40	30	70
2031	104	10	13	81	167	87	17	3	11	31	112	10	14	24	20	27	47
2032	88	5	11	72	150	78	17	14	1	31	103	5	13	18	10	24	34
2033	85	5	11	69	145	75	36	13	3	53	122	5	15	20	10	26	36

2020 value is an estimate based on back calculation from smolt survival rate

In both tables, total harvests to achieve the target are highlighted in red with white font for the first five years. The starting bass stock to which the salmon smolts and smolts are exposed are in red font in the second column entitled "Population Pre-harvest & Spawn". Of course, the calculations in both tables are probably not precise, and because of unpredictable annual bass survival variability, maybe not overly accurate. However, it demonstrates that very large bass removals for several years are required to achieve the Optimal Sustainable Yield fisheries target of 100,000 alluded to in Chaput (2022) and confirmed by our egg-to smolt / smolt-to-egg model.

The bass "stock" level can be envisioned as an investment (the principal) and the recruitment as the rate of return (the accrued increase in annual value). We need to ignore the rule of investing to never touch the principal and take big initial portions out of it. Assuming a past stable population, referring to Figure 9, if we don't want to touch the principal, every year we would harvest only from the stock level defined by the blue line down to the level defined by the orange "replacement" line. With the current population of 500,000 or more, to achieve our environmental goal, which requires a maximum bass stock of 100,000 we will have to harvest to a level well below the replacement line.

Since the striped bass LRP is, by definition, one-half the maximum stock level, a higher maximum population level, entirely possible according to Chaput and Douglas (2022), would result in a new higher LRP defined. The resulting Beverton-Holt model would reflect a higher recruitment level and greater initial harvests would be required to decrease to stock to the fishery target level of the 100,000 bass. The model applied in Table 8 is therefore conservative and its implementation would still protect the bass population from over-harvest. Annual adaptive management effort would be required in years subsequent to the first to determine harvests that will achieve, maintain, and not greatly undercut the 100,000-target bass stock level.

2.5 On Striped Bass Population Protection

We feel it is important to not only discuss the bass stock level that would protect salmon, but also the minimum level to which the bass population should be allowed to fall, essentially an LRP value. The ~4,500 value for bass from which the population recovered provides a potential LRP consistent with a recognized method for its determination known as "Brecover". The authors of Sci. Advis. Rep. 2021/018 and Res. Doc. 2022/029 rejected this method, and 4,500 spawners is obviously a level far too low to which to allow the population to descend. However, the passing mention in Res. Doc. 2022/030 of an experimental fisheries target level, implied to be 100,000 bass, indicates this level is not one that would imperil the population. Further, DFO. (2011) states the following:

The RPA (Recovery Potential Assessment) *for Striped Bass in the sGSL* (southern Gulf of St. Lawrence) *proposed a recovery limit and compliance rule of* ***21,600 spawners*** *in 5 of 6 years (DFO 2006). Douglas et al. (2006) further proposed that once the recovery limit was met, achieving an increased level of* ***31,200 spawners*** *in 3 of 6 years could be a recovery target to consider for managing access to the resource.*

The passage from the 2011 DFO publication is a strong indication that a level of 100,000 represents virtually no risk of population failure. Either of the 21,600 or 31,200 numbers would be appropriate LRP levels for sGSL's bass. The 31,200 number was also suggested by Bill Taylor, then President of the Atlantic Salmon Federation. From the minutes of testimony for FOPO (2019):

- *"I would suggest that the recovery target the DFO set of 31,200 is your bottom. That's the floor:"*

Additional Corrective Actions Required

In addition to rapidly bringing the bass population down to a spawning stock level of 100,000, the residual effects of years of mismanagement must be corrected. For example, to address bass predation on salmon juveniles in nursery habitat above the heads of tide, the daily creel limits on scheduled salmon rivers such as those of the Miramichi system should be set to a level that would not seriously affect the chance of an angler "limiting out". The slot range for retaining a bass in upland scheduled waters should also be eliminated.

With 100,000 bass, the Southwest's salmon population would become sustainable, although at a bottom-line lower-than-acceptable egg deposition rate for rapid recovery. The Miramichi Salmon Conservation Centre should be used to supplement the current dismally low egg deposition rates on both river composites. It is designed for Smolt-to-Adult supplementation and has a capacity to produce 3,000 MSW equivalent fish. The egg production capacity is therefore in the 10 to 15 million range. A majority of female smolts would be selected. It is envisioned that a straight release of pre-spawning adult fish would be the primary method of supplementation. A secondary method would involve the holding back of female salmon to be spawned at the facility and produce first-feeding fry. As of now the hatchery has capacity for 3,000 fish large MSW equivalent fish. We have been advised by an expert advisor that 25% should be males just in case there are an insufficient number of wild males to induce spawning. (There is uncertainty of the science saying precocious parr will accomplish this.) Holding back females to spawn in the hatchery and produce ~1.5 million feeding fry is currently envisioned. These fry would be stocked into previously identified severely underpopulated habitat in each drainage composite. To meet genetic guidelines, it is proposed to conduct alternate year stocking of the NW composite one year and the SW the next.

We have demonstrated that even after achieving and maintaining the 100,000 bass fisheries target level, the Miramichi's salmon population will recover very slowly, and in the case of the Northwest Miramichi imperceptibly or not at all unless hatchery supplementation occurs. With the required permits, upgrades to the facility and ancillary equipment, and improved staffing levels, the required supplementation that is envisioned could be accomplished at the Miramichi Salmon Conservation Centre. Because of its negligent management of the striped bass population, we consider DFO to be at fault for allowing the Miramichi's salmon populations from each composite to fall to their present levels – i.e., well below their LRP limits. For this reason, we demand that DFO pay for the aforementioned facility upgrades, additional personnel costs, and provide permits for the yet-to-be fully identified stocking programs.

Summary and Conclusion

Atlantic salmon of the Northwest and Southwest composites of the Miramichi River system have been in decline since the steep drop-off in returns in 2012. The decline has been greatest on the Northwest composite where returns are at a critically low level. The cause is the concurrent steep increase in the striped bass population. This cause and effect has been denied by DFO because of mis-interpretation of existing data. DFO should now acknowledge this effect, and under existing policy, a balanced management approach should be implemented "to ensure neither (the bass nor the salmon population) is depleted to a point of serious harm". Bass numbers could be reduced substantially before serious population-level harm results, however salmon numbers are now dangerously low. Immediate action is required to decrease the bass population to no more than 100,000 spawners and to decrease the associated Precautionary Approach Limit Reference Point for bass to a level than lower than this.

To address the residual effects of mismanagement of the bass population to the detriment of salmon, regulations for retention of bass in scheduled salmon rivers above the heads-of-tides should be greatly liberalized as described above.

In addition, the salmon populatuions from each Miramichi composite will recover very slowly or imperceptibly unless their populations are supplemented with hatchery fish. This could be accomplished by the Miramichi Salmon Conservation Centre. Because of its mismanagement of the striped bass population, we consider DFO to be at fault for allowing the Miramichi's salmon populations from each composite to fall to their present unacceptable levels. For this reason, we demand that DFO pay for facility upgrades, additional personnel costs, and provide permits for the required stocking programs.

3.0 References

Chaput, G., Douglas, S.G., and Hayward, J. 2016. Biological Characteristics and Population Dynamics of Atlantic Salmon (Salmo salar) from the Miramichi River, New Brunswick, Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2016/029. v + 53 p.

Chaput, G., J. Carr, J. Daniels, S. Tinker, I. Jonsen, and F. Whoriskey. 2018. Atlantic salmon (Salmo salar) smolt and early smolt migration and survival inferred from multi-year and multi-stock acoustic telemetry studies in the Gulf of St. Lawrence, northwest Atlantic. – ICES Journal of Marine Science, doi:10.1093/icesjms/fsy156.

Chaput, G. and Douglas, S. 2022. Fisheries Reference Points for Striped Bass (Morone saxatilis) from the Southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2022/029. xiv + 153 p.

Chaput, G. 2022. Multi-species Considerations for Defining Fisheries Reference Points for Striped Bass (Morone saxatilis) from the Southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2022/030. vii + 50 p.

Chaput, G., Carr, J., Daniels, J., Tinker, S., Jonsen, I., and Whoriskey, F. 2018. Atlantic salmon (Salmo salar) smolt and early smolt migration and survival inferred from multi-year and multi-stock acoustic telemetry studies in the Gulf of St. Lawrence, northwest Atlantic. – ICES Journal of Marine Science, doi:10.1093/icesjms/fsy156.

Douglas, S.G., G. Chaput, and D. Caissie. 2006. Assessment of Status and Recovery Potential for Striped bass (Morone saxatilis) in the Southern Gulf of St. Lawrence. CSAS Res. Doc 2006/041. 95 p.

DFO. 2019. Guidance for the Development of rebuilding plans under the Precautionary Approach framework: Growing stocks out of the Critical Zone. Sustainable fisheries framework (SFF): A fishery decision-making framework Incorporating the Precautionary Approach. 32 p.

DFO. 2021. Reference points for Striped Bass (Morone saxatilis) for the southern Gulf of St. Lawrence population. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2021/018. (Erratum: December 2022).

DFO. 2011. Allowable harm assessment of Striped Bass (Morone saxatilis) in the southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2011/014.

DFO. 2013. Update to 2012 on spawner abundance and biological characteristics for striped bass (Morone saxatilis) in the southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Sci. Resp. 2013/010.

FOPO. 2019. Striped bass in the southern Gulf of St. Lawrence and Miramichi River: striking a delicate balance. Report of the Standing Committee on Fisheries and Oceans. 25 p.

Gibson, A.J.F. 2006. Population Regulation in Eastern Canadian Atlantic salmon (Salmo salar) populations. Res. Doc. 2006/016.

Gibson, A.J.F., R.A. Jones, and G.J. MacAskill. 2016. Recovery Potential Assessment for Outer Bay of Fundy Atlantic Salmon (Salmo Salar): Population Dynamics and Viability. DFO Can. Sci. Advis. Sec. Res. Doc. 2016/032. v + 87 p.

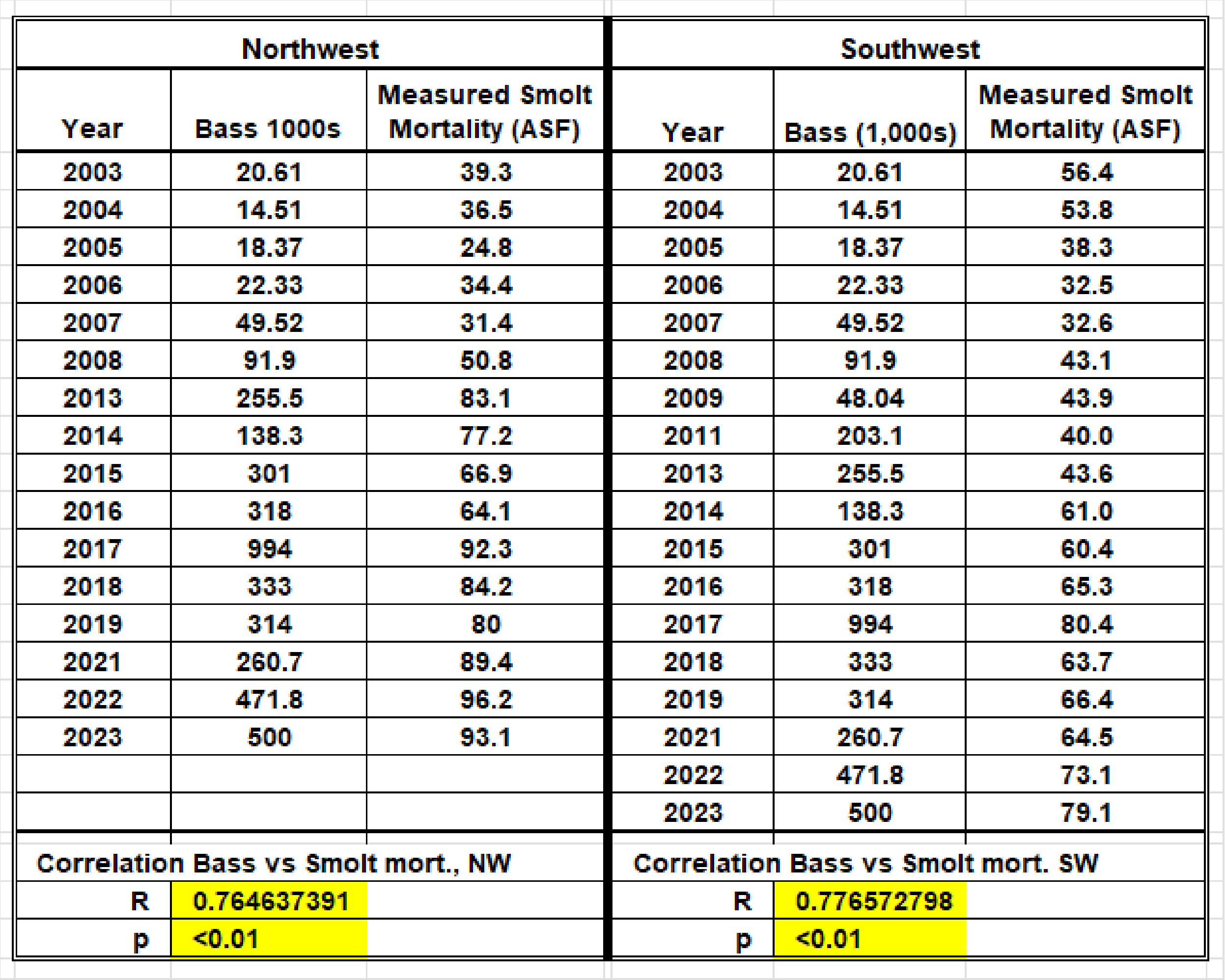
Jonsson, N., B. Jonsson, And L.P. Hansen. 1998. The relative role of density-dependent and density-independent survival in the life cycle of Atlantic salmon Salmo salar. Journal of Animal Ecology 0887\56\ 640_651_ 0887 British Ecological Society 640.

O'Connell, M.F., J. B. Dempson, and G. Chaput. 2006. Aspects of the life history, biology, and population dynamics of Atlantic salmon (Salmo salar L.) in Eastern Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2006/014. 28 p. + 19 figures.

Ritter, J. 2024. Unpublished. Is striped bass (Morone saxatilis) predation the cause of the decline in and low Atlantic salmon (Salmo salar) returns to the Miramichi river?. 16p.

Appendix A-1 - Regressions of Bass Numbers vs. Atlantic Salmon Smolt Mortality in the Miramichi Estuary

Table A-1 Regression of Bass Population on Tagged Smolt Mortality



Northwest			Southwest		
Year	Bass 1000s	Measured Smolt Mortality (ASF)	Year	Bass (1,000s)	Measured Smolt Mortality (ASF)
2003	20.61	39.3	2003	20.61	56.4
2004	14.51	36.5	2004	14.51	53.8
2005	18.37	24.8	2005	18.37	38.3
2006	22.33	34.4	2006	22.33	32.5
2007	49.52	31.4	2007	49.52	32.6
2008	91.9	50.8	2008	91.9	43.1
2013	255.5	83.1	2009	48.04	43.9
2014	138.3	77.2	2011	203.1	40.0
2015	301	66.9	2013	255.5	43.6
2016	318	64.1	2014	138.3	61.0
2017	994	92.3	2015	301	60.4
2018	333	84.2	2016	318	65.3
2019	314	80	2017	994	80.4
2021	260.7	89.4	2018	333	63.7
2022	471.8	96.2	2019	314	66.4
2023	500	93.1	2021	260.7	64.5
			2022	471.8	73.1
			2023	500	79.1
Correlation Bass vs Smolt mort., NW			Correlation Bass vs Smolt mort. SW		
R	0.764637391		R	0.776572798	
p	<0.01		p	<0.01	

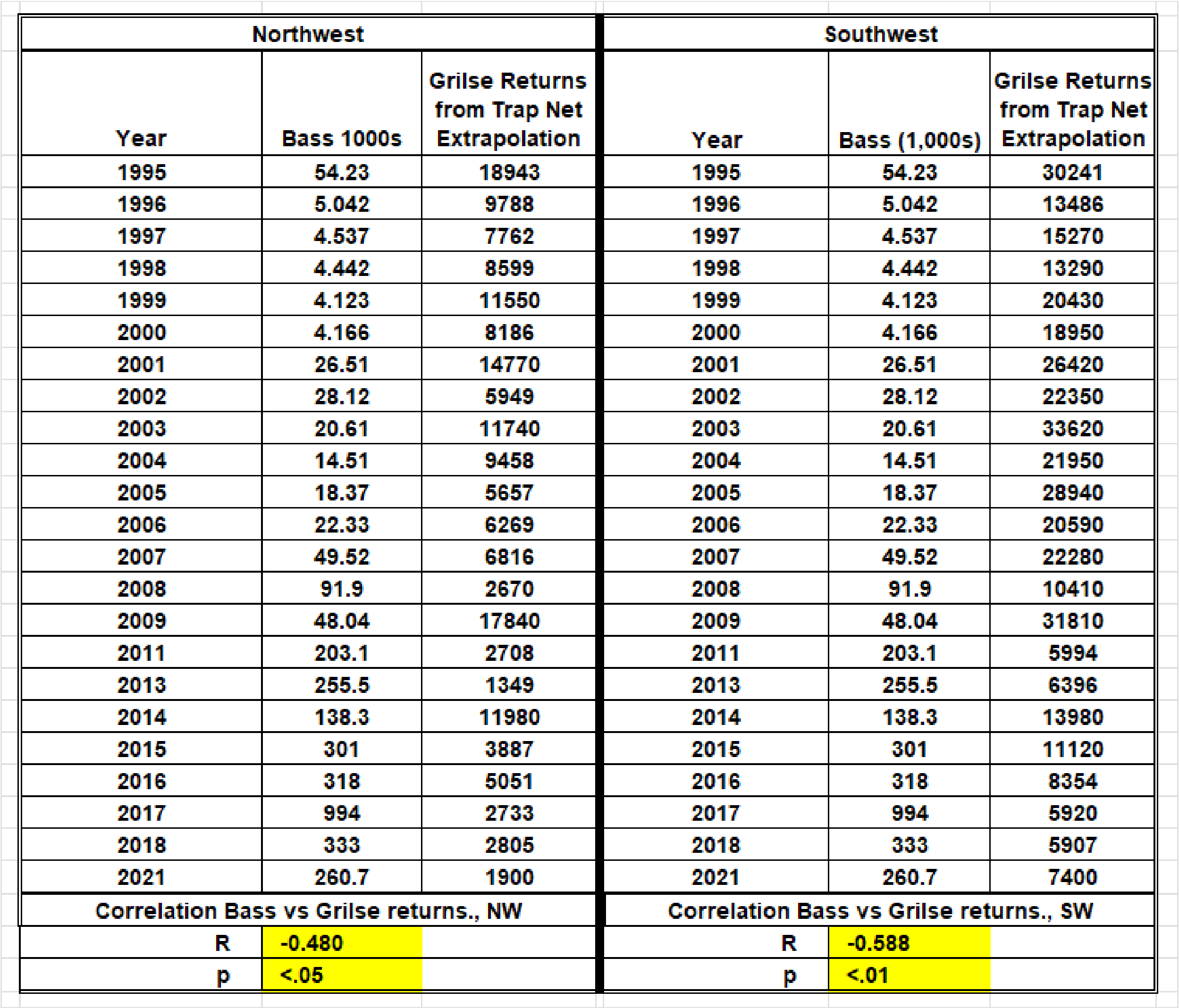
NW v B= Survival NW post smolts vs. Bass

SW v. B= Survival NW smolts vs. Bass

NOTE: The 500,000 value for bass in 2023 was an estimate based on information provided at the Eastern NB Coastal and Inland Fisheries Meeting.

Appendix A-2 - Regressions of Bass Numbers vs. Atlantic Salmon Grilse Returns to the Miramichi Composite Drainages

Table A-2 Regression of Bass Population on Grilse (1 sea-winter) returns



Northwest			Southwest		
Year	Bass 1000s	Grilse Returns from Trap Net Extrapolation	Year	Bass (1,000s)	Grilse Returns from Trap Net Extrapolation
1995	54.23	18943	1995	54.23	30241
1996	5.042	9788	1996	5.042	13486
1997	4.537	7762	1997	4.537	15270
1998	4.442	8599	1998	4.442	13290
1999	4.123	11550	1999	4.123	20430
2000	4.166	8186	2000	4.166	18950
2001	26.51	14770	2001	26.51	26420
2002	28.12	5949	2002	28.12	22350
2003	20.61	11740	2003	20.61	33620
2004	14.51	9458	2004	14.51	21950
2005	18.37	5657	2005	18.37	28940
2006	22.33	6269	2006	22.33	20590
2007	49.52	6816	2007	49.52	22280
2008	91.9	2670	2008	91.9	10410
2009	48.04	17840	2009	48.04	31810
2011	203.1	2708	2011	203.1	5994
2013	255.5	1349	2013	255.5	6396
2014	138.3	11980	2014	138.3	13980
2015	301	3887	2015	301	11120
2016	318	5051	2016	318	8354
2017	994	2733	2017	994	5920
2018	333	2805	2018	333	5907
2021	260.7	1900	2021	260.7	7400
Correlation Bass vs Grilse returns., NW			Correlation Bass vs Grilse returns., SW		
R	-0.480		R	-0.588	
p	<.05		p	<.01	

NOTE: The 500,000 value for bass in 2023 was an estimate based on information provided at the Eastern NB Coastal and Inland Fisheries Meeting.

Exhibit A

Recent photos representing devastating effects of striped bass predation on Atlantic salmon

PHOTO #1 [Spring 2024]: Miramichi striped bass with partially digested smolts in its stomach

PHOTO #2 [July 27, 2024]: First striped bass catch ever recorded at Dudley Bogan, McNamee, New Brunswick

PHOTO #3 [July 27, 2024]: Summer striped bass catch recorded at Dudley Bogan, McNamee, New Brunswick [unheard of and unprecedented in 100 + years of Wilson's Sporting Camps records]

PHOTO #4 [August 17th, 2024 Weekend]: Stomach contents of striped bass, Little Main Restigouche, New Brunswick. [Approximately 175 km from tidewater]. This is evidence of the spread of striped bass invasion throughout the rivers on the Gulf Region of New Brunswick's coastline.

ICES Journal of Marine Science

International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

ICES Journal of Marine Science (2018), doi:10.1093/icesjms/fsy156

Atlantic salmon (*Salmo salar*) smolt and early post-smolt migration and survival inferred from multi-year and multi-stock acoustic telemetry studies in the Gulf of St. Lawrence, northwest Atlantic

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Chaput, G., Carr, J., Daniels, J., Tinker, S., Jonsen, I., and Whoriskey, F. Atlantic salmon (Salmo salar) smolt and early post-smolt migration and survival inferred from multi-year and multi-stock acoustic telemetry studies in the Gulf of St. Lawrence, northwest Atlantic. – ICES Journal of Marine Science, doi:10.1093/icesjms/fsy156.

Received 8 June 2018; revised 17 September 2018; accepted 26 September 2018.

The migration dynamics and inter-annual variation in early at-sea survival of Atlantic salmon (*Salmo salar*) smolts over 14 years of study are reported for four river populations located in the Gulf of St. Lawrence (Canada). Acoustically tagged smolts were monitored at three points along their migration from freshwater to the Labrador Sea, a migration extending more than 800 km at sea and a period of 2 months. A hierarchical state-space version of the Cormack–Jolly–Seber model was used to estimate apparent survival rates from incomplete acoustic detections at key points. There was a positive size-dependent probability of survival through the freshwater and estuary areas; the odds of survival of a 16 cm smolt were 1.5–1.7 times higher than for a 13.5 cm smolt, length at tagging. Length adjusted (centred to the mean fork length of smolts during the study of 14.6 cm) survivals through the estuary and nearshore waters were estimated to range between 67 and 90% for the two river populations migrating through Chaleur Bay in contrast to lower survival estimates of 28–82% for the two populations from the neighbouring Miramichi Bay. Across the 14 years of study, survival estimates varied without trend for the populations of Chaleur Bay, but declined for the populations migrating through Miramichi Bay. Survival through the Gulf of St. Lawrence was variable but generally high among years and rivers, ranging from 96% day^{-1} to 99% day^{-1}. Long term, replicated studies at multiple sites using acoustically tagged smolts can provide empirical data to examine hypotheses of the location and timing of factors contributing to smolt and post-smolt mortality of salmon at sea.

Keywords: Keywords: acoustic telemetry, Atlantic salmon, hierarchical CJS model, smolt survival.

Introduction

Many Atlantic salmon (*Salmo salar*) populations in the western North Atlantic portion of the species' range are currently at or near record low abundances (ICES, 2017). Since the 1990s, sharp declines in population abundance estimates or fisheries landings, as proxies for abundance, have been noted (Beaugrand and Reid, 2003, 2012; Chaput *et al.*, 2005). Historically, multiple causes in fresh water (dams, poor land-use patterns, etc.) diminished the salmon's natural range and reduced population productivity (Parrish *et al.*, 1998) but declines in the past two decades cannot be directly or exclusively attributed to freshwater factors. In the past decade, return rates of smolts to one-sea-winter salmon for populations in the North Atlantic have been low, averaging less than 3.5% (an instantaneous mortality rate >3.35) across monitored rivers (ICES, 2017). Favourable oceanographic conditions have been associated with higher abundances of Atlantic salmon in some populations and in some years, but the same factors do not appear to be acting on all populations equally (Friedland *et al.*, 2000; Peyronnet *et al.*, 2007).

Anadromous salmonid population abundances are most sensitive to factors affecting marine survival (Otero *et al.*, 2011; Kilduff *et al.*, 2015; Nieland *et al.*, 2015), suggesting that the causes of the most recent declines of Atlantic salmon are due to increased mortality at sea. Increasingly variable inter-year marine survival for several Pacific salmon species is correlated with changes in oceanographic conditions (Kilduff *et al.*, 2015). Local effects, such as fish passage and the nearshore ecosystem, and offshore factors including variations in the physical, chemical, and biological components of the ecosystem are involved in the mortality of Atlantic salmon but the location, timing, and the proportional contribution of various factors to total mortality remain elusive (Thorstad *et al.*, 2012). If an important component of the annual marine mortality of anadromous salmon takes place in the initial phase of seaward migration and can be documented, then further studies can be defined to understand the causal mechanisms and advise on mitigation options. However, if the early marine phase is not an important survival period/area, then local mitigation may not be useful and factors further afield need to be studied.

It is now possible to implant electronic (acoustic) transmitters in small fish and track their movements over increasingly long periods of time. Such studies can provide information on individual fish distribution, migration rates, marine residency patterns, as well as population-level survival rates and to identify critical marine habitats and periods (Lacroix, 2008; Drenner *et al.*, 2012; Thorstad *et al.*, 2012; Goulette *et al.*, 2014; Hussey *et al.*, 2015). In eastern North America, acoustic tracking studies have been undertaken on a geographically diverse number of wild Atlantic salmon populations ranging from the southern areas in Maine (USA; Lat. 44.67°N; Kocik *et al.*, 2009) to northeastern populations in Newfoundland (Lat. 47.9°N; Dempson *et al.*, 2011) and mid-latitude populations in the Gulf of St. Lawrence (Lat. 50.28°N; Lefèvre *et al.*, 2013; Daniels *et al.*, 2018). With few exceptions (Lacroix, 2008; Kocik *et al.*, 2009; Stich *et al.*, 2015), the studies reported on movements and survival of Atlantic salmon smolts in the home river estuary within 50 km from the head of tide or to nearshore environments in the vicinity of the river's confluence with the sea, and rarely included more than 2 years of smolt tagging from the same river. At many sites, predation on smolts during the initial period and area of migration is considered to be important, and local conditions that either enhance or reduce predation risk may determine initial survival (Kocik *et al.*, 2009; Halfyard *et al.*, 2013; Daniels *et al.*, 2018). Few of the published studies provide sufficient annual replication to characterize the annual variation in survival rates thus precluding the testing of hypotheses of factors which may be responsible for the early marine-phase mortality of salmon smolts.

We report on data collected from a 14-year and multi-population acoustic telemetry study to quantify survival rates of acoustically tagged wild Atlantic salmon at pre-defined geographic locations during the freshwater (smolt stage), estuarine (smolt stage), and open ocean (post-smolt stage) migratory phases. The study considers populations of Atlantic salmon from four unimpacted (free fish passage) rivers from the southern Gulf of St. Lawrence (Canada) over the period 2003–2016. Acoustically tagged salmon smolts are tracked over a period extending approximately 2 months at sea and over 800 km offshore.

Material and methods

Description of study area

The Miramichi River (47.2°N 65.0°W) has a basin area of approximately 14 000 km^2 with two major tributaries that converge in tidal waters; the Southwest Miramichi River and the Northwest Miramichi River (Chiasson, 1995). Salmon from the Miramichi River pass through Miramichi Bay on their migration to the Gulf of St. Lawrence. Miramichi Bay, a shallow natural barrier-built estuary, is seasonally stratified with average salinities in the outer portion of less than 25 parts per thousand (ppt) and a maximum depth that rarely exceeds 10 m (Chiasson, 1995; St-Hilaire *et al.*, 1995). The Restigouche River (48.0°N 66.3°W; basin area of 12 820 km^2) and the Cascapedia River (48.2°N 65.9°W; basin area of 3 147 km^2) both flow into Chaleur Bay, an open bay that enters directly into the Gulf of St. Lawrence. It is deep compared to Miramichi Bay, with a central trough of maximum depth of approximately 100 m and surface salinities generally less than 27 ppt during the open water period (Koutitonsky and Bugden, 1991). The Chaleur and Miramichi Bays are located in the southwest portion of the Gulf of St. Lawrence (GSL), a stratified semi-enclosed sea with an approximate surface area of 226 000 km^2 (Koutitonsky and Bugden, 1991). Surface layers (<30 m) are of low salinity (27–32 ppt) and sea surface temperatures generally approximate or exceed 20°C in summer (DFO, 2017). The Gulf of St. Lawrence has two connections to the North Atlantic Ocean; Cabot Strait to the east at a width of 104 km and a maximum depth of 480 m and the Strait of Belle Isle to the north at a width of 15 km and depth less than 60 m (Koutitonsky and Bugden, 1991) (Figure 1). The head of tide locations of the four study rivers are approximately 900 km from the Strait of Belle Isle (Figure 1).

Smolt collection and tagging

Atlantic salmon smolts were captured in rotary screw traps (Chaput and Jones, 2004) set at the same locations over the study period for the Southwest Miramichi (127 km above the head of tide), Restigouche River (115 km above the head of tide), and the Cascapedia River (8 km above the head of tide) (Figure 1). For the Northwest Miramichi, smolts were captured in the Little Southwest Miramichi River (30 km above the head of tide) during 2003–2008 and in the Northwest Miramichi River (24 km above the head of tide) during 2013–2016. The distances from the head of tide to the outer bays ranged from just under 70 km for the Miramichi River locations to between 47 and 106 km for the two rivers in Chaleur Bay. The outlets of the two bays are approximately 800 km south of the Strait of Belle Isle (Figure 1).

The dates of tagging and release of Atlantic salmon smolts varied by river and year (Supplementary Figure S1). Generally, smolts were first captured and tagged in the Southwest Miramichi, then the Northwest Miramichi, followed by the Restigouche and finally the Cascapedia. Among years, the dates of release varied by as much

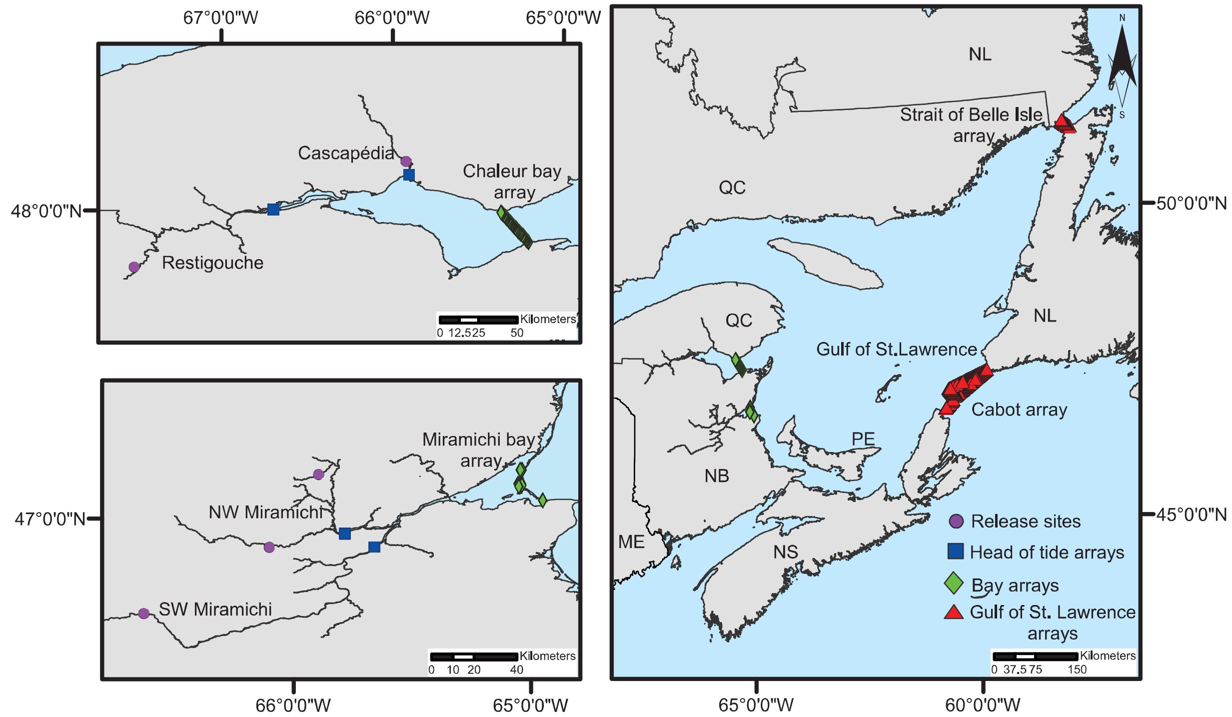

Figure 1. Map of study area. Left sub-panels show the release locations by study river, the head of tide receiver locations, and the respective bay receiver lines. Right panel depicts bay receiver lines as well as the receiver line locations at exits from the Gulf of St. Lawrence.

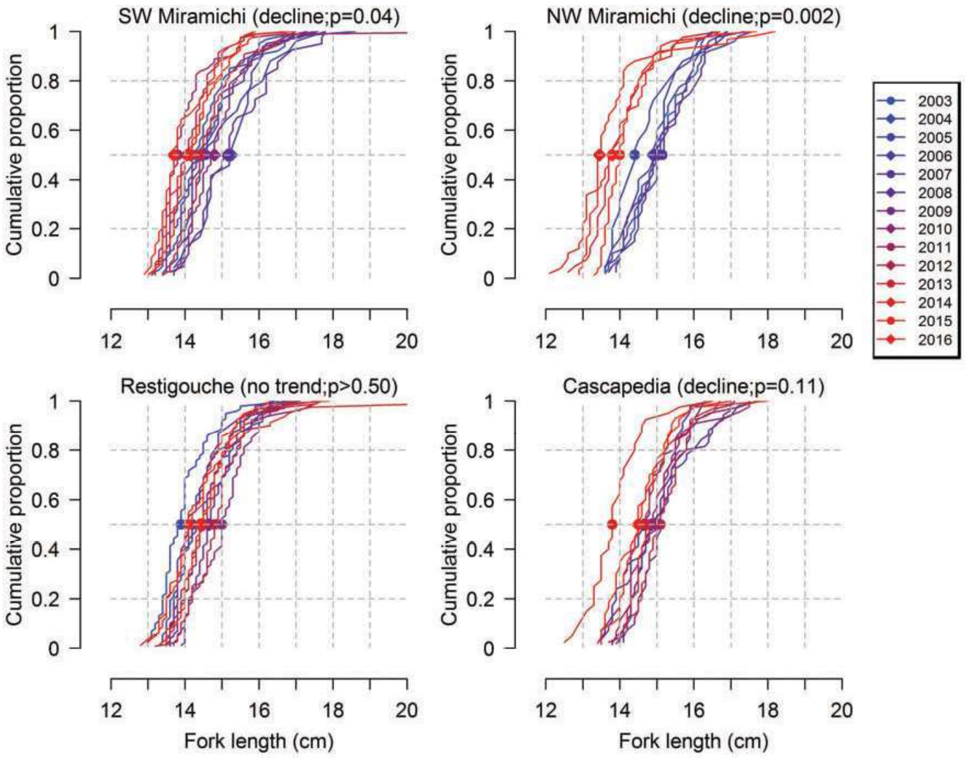

Figure 2. Cumulative frequency distributions by fork length (cm) of Atlantic salmon smolts tagged and released in the four rivers over the period 2003–2016. Also shown in each river panel is the trend and associated *p*-value of the linear regression of median length vs. year.

as 3 weeks, the earliest dates of tagging were in 2010, 2012, and 2013 while the latest dates of tagging occurred in 2008 and 2014.

Smolts were selected for tagging on the basis of length with efforts to select smolts generally greater than 13 cm fork length (FL). The realized length range was 12.1–23.3 cm FL with median lengths among rivers and years ranging from 13.5 to 15.0 cm FL (Figure 2). Size distributions of tagged smolts in 2014 in three of the four rivers were among the smallest of the time series (Figure 2).

Acoustic transmitters, models V8SC and V9 (9 mm diameter, 2.9–3.3 g in air, hereafter referred to as V9) and the smaller V8 (8 mm diameter, 2.0 g in air) pinging at 69 kHz (Innovasea Marine Systems Canada, Inc., Halifax, NS) with a unique identification code, were surgically implanted into the peritoneal cavity of selected smolts (Daniels *et al*., 2018). Smolts were generally tagged and released at the site of capture. The exception was in 2014–2016, when smolts from the Northwest Miramichi River were released upstream of their capture location, approximately 52 km above the head of tide. Fish recovered post-surgery in a holding pen in the river for a few hours prior to being released. Each tag was programmed to ping at random delays of either 20–60 s or 25–55 s and had an estimated minimum battery life of 74 days (Supplementary Table S1). The whole weight of tagged smolts was not reliably measured. The tag burden, expressed as the ratio of tag length to fork length of fish, ranged from 9.0 to 16.9%, with a mean of 14.2%. Based on general length to weight relationships of smolts from these rivers (Fisheries and Oceans Canada, unpublished data), the tag burden ratio in terms of weight was estimated to be very similar to the tag burden ratio based on length.

Receiver deployments and monitoring

Lines, in some cases staggered, of acoustic receivers (VR2, VR2W, VR2AR, or VR4 models, Innovasea Marine Systems Canada, Inc.) were deployed at the head of tide of each of the four rivers and at the outer bay exits to the Gulf of St. Lawrence (Miramichi Bay and Chaleur Bay) (Figure 1). Effective detection range of the receivers is considered to encompass a radius of 0.5 km. Since 2007, a receiver line has been installed annually at the Strait of Belle Isle (Figure 1). A second receiver line was installed 3.5 km north of the primary Strait of Belle Isle line during 2015 and 2016 to provide empirical data for estimating the detection efficiency of the primary Strait of Belle Isle line. A partial receiver line was installed across the Cabot Strait beginning in 2010 and complete coverage closing off the strait was established in 2012 (Figure 1). Spacing distance among receivers at the Strait of Belle Isle line was a maximum of 800 m. All receiver lines were seasonally deployed in spring and removed in the fall, with the exception of Cabot Strait receiver line which operates year-round. The structure (number of receivers, placement) of each receiver line was generally similar among years, however, the physical environment and other anthropogenic factors affected the detection range of the individual receivers and the overall array.

Description of data

There are 14 years of tracking data for the Southwest Miramichi River (2003–2016), 10 years for the Northwest Miramichi River (2003–2008; 2013–2016), 13 years for the Restigouche River (2004–2016), and 11 years for the Cascapedia River (2006–2016) (Figure 3; Supplementary Table S2). A total of 2 862 Atlantic salmon smolts had complete tagging and release information over the 48 years and river combinations. The number of smolts tagged and released annually by river ranged from 25 to 105 fish. When presented, migration characteristics are based on the times and dates of the first detections of individual smolts at any receiver in each array.

Modelling detection and survival probabilities

A Bayesian state-space implementation of the Cormack–Jolly–Seber (CJS) model (Gimenez *et al*., 2007; Royle, 2008) is used to disentangle the imperfect detection (p) of tagged smolts on the receiver arrays from apparent survival (ϕ) during their out migration from freshwater to the Gulf of St. Lawrence and to the Strait of Belle Isle. The state-space parameterization of the CJS model constructs distinct models for the unobserved survival (ϕ) process and the observed detection process (p). For simplification, we refer to apparent survival or simply survival as the joint probability of a tagged fish surviving to pass a receiver array and of the tag being detected at that array (see discussion for implications on estimates of apparent survival of tagged smolts).

The unobserved survival process model (Equation 1) assumes that if a fish (i) carrying an acoustic tag is alive at the observation point j-1 then its survival state at point j is a realization from a Bernoulli process with parameter ϕ_j. The state process (survival) is represented by a binary variable $z(i, j)$, which takes the value 1 if individual i is alive at the detection point j and 0 otherwise. This process is modelled as random draws from a Bernoulli distribution where $z(i, j)$ is conditional on $z(i, j$-1), whether fish i is alive (1) or dead (0) at the previous detection point:

$$z(i,j) \mid z(i,j-1),\ \O_j \ \sim \ \text{Bernoulli}(z(i,j-1)\O_j) \tag{1}$$

with $j = 1$–4 corresponding to the three to four post-release detection points where a fish, which is alive may be observed after initial tagging and release ($j = 0$). If a fish is not alive at j-1 then $z(i, j) = 0$ with probability 1. The initial state at release, i.e. $z(i, 0)$, is set equal to 1.

The re-observations $y(i, j)$ are modelled as independent Bernoulli random variables, conditional on the $z(i, j)$'s and the probability of detection (p):

$$y(i,j) \mid z(i,j),\ p_j \ \sim \ \text{Bernoulli}(z(i,j)p_j), \tag{2}$$

where $y(i, j) = 0$ with probability 1 if $z(i, j) = 0$, otherwise $y(i, j)$ is a Bernoulli random variable with parameter p_j the probability of detection at array j.

The parameters p and ϕ are proportions bounded on the range [0, 1] but are logit-transformed to improve the model's convergence properties. A hierarchical structure assuming exchangeability is considered for the detection and survival parameters, conditional on individual effects of tag type and smolt size, reflecting the multi-year (t) and multi-population (r) design of the study (Gelman *et al*., 2004; Bonner and Schwarz, 2006). The exchangeability assumption considers that the year and population specific parameters at the arrays ($p_{r,t,j}$; $\phi_{r,t,j}$) are drawn from common prior distributions with unknown hyperparameters for the corresponding groups (river r and array j) (Gelman *et al*., 2004).

Individual effects

Individual effects on p and ϕ are examined for the acoustic tag type used and the size of smolts at tagging (Royle, 2008). The V9

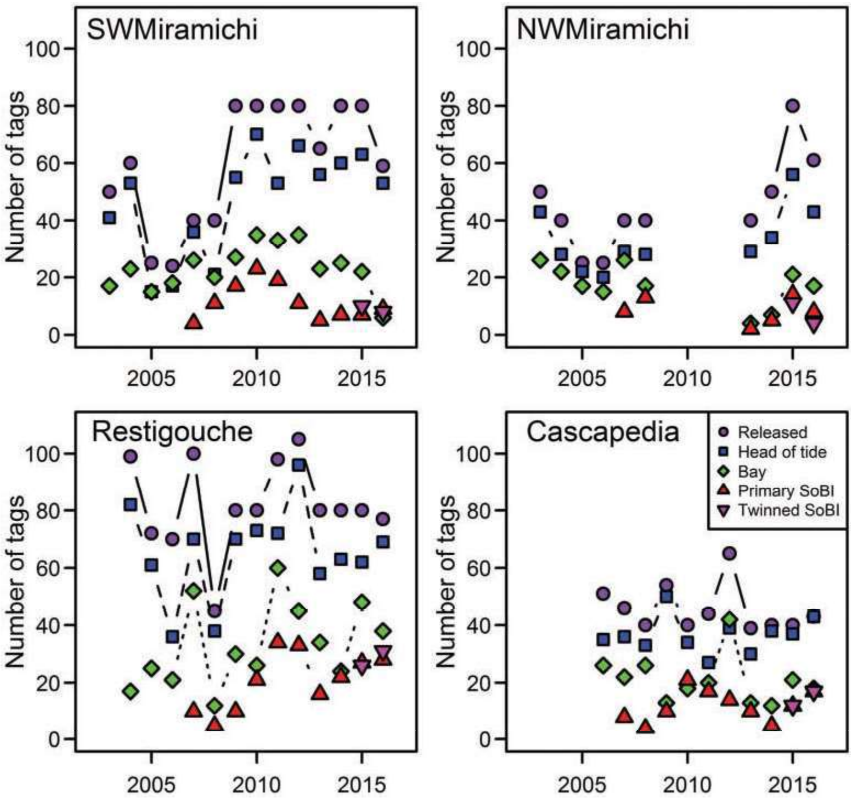

Figure 3. Summary of number of tagged fish released and number of tags detected at the respective receiver lines for four rivers during 2003–2016, for tag types (V9, V8) combined.

version of the tag was used during 2003–2015. The V8 version was used in 2014–2016 (Supplementary Table S2). The tag model is considered to have potential effects on both the probability of detection, due to differences in output, and on the probability of survival, due to differences in relative tag burden.

Tag type (V9; V8) is modelled as an offset on the logit scale of the average probability of detection at the arrays:

$$y(i[r,t],j) \mid z(i[r,t],j),\ p(i[r,t],j) \sim \text{Bernoulli}(z(i[r,t],j)p(i[r,t],j)) \tag{3}$$

with

$$\text{logit}\Big(p(i[r,t],j)\Big) = \mu^{p}(r,t,j) + \beta_p * v9_i - \beta_p * v8_i \tag{4}$$

for $j = 1{:}3$; $r = 1{:}4$; $t = 2003{:}2016$; $i =$ individual fish 1:2 862; $p(i[r,t],j)$ the probability of detecting fish i within river r and year t at array j; $\mu^{p}(r,t,j)$ the mean logit probability of detection within river r, year t, at array j; β_p the offset in the probability of detection for tag type; $v9_i = 1$ if tag type was V9 for fish i, 0 otherwise; $v8_i = 1$ if tag type was V8 for fish i, 0 otherwise.

Preliminary analyses of the proportions of tags detected at the arrays suggested a positive association with the fork length of the fish at tagging (Figure 4). There is also a decreasing temporal trend (linear regression of median length vs. year) in the size distributions of smolts tagged in the Southwest Miramichi and Northwest Miramichi rivers over the period of study (Figure 2). Survival probabilities relative to the length of fish at tagging by river and tag type overall are modelled as linear effects on the logit scale with tag type included as an interaction term with fork length (i.e. differing slopes for tag type) as:

$$z(i[r,t],j) \mid z(i[r,t],j-1),\ \text{Ø}(i[r,t],j) \sim \text{Bernoulli}(z(i[r,t],j-1)\text{Ø}(i[r,t],j)) \tag{5}$$

With

$$\text{logit}\Big(\text{Ø}(i[r,t],j)\Big) = \mu^{\text{Ø}}(r,t,j) + (\alpha_{fl} + \beta_{\text{Ø}} * v9_i - \beta_{\text{Ø}} * v8_i) * fl'_i \tag{6}$$

for $j = 1{:}2$; $r = 1{:}4$; $t = 2003{:}2016$; $i = 1{:}\ 2\ 862$; $\text{Ø}(i[r,t],j)$ the probability of survival of fish i within river r and year t through transition zone j; $\mu^{\text{Ø}}(r,t,j)$ the mean logit probability of survival within river r, year t through transition zone j; α_{fl} the average slope over tag type to fish length relationship, logit scale; $\beta_{\text{Ø}}$ the offset in the slope of fork length due to tag type; $v9_i = 1$ if tag type was V9 for fish i, 0 otherwise; $v8_i = 1$ if tag type was V8 for fish i, 0 otherwise; $fl'_i = fl_i - \overline{fl_{.}}$ the centred fork length (cm) of individual i; $\overline{fl_{.}}$ the mean fork length (cm) of smolts across all rivers and years.

The effects for smolt length and tag type on the probabilities of survival are considered for the release to head of tide transition and the head of tide to bay array transitions ($j = 1$, 2) but not for the Gulf of St. Lawrence transition ($j = 3$). It is assumed that the tagging and handling effects associated with the size of smolt tagged and tag type are negligible for the surviving smolts migrating through the Gulf of St. Lawrence given the time required for the tagged smolts to reach the Gulf and the increased body size of the surviving smolts which would result in reduced tag burdens on the survivors.

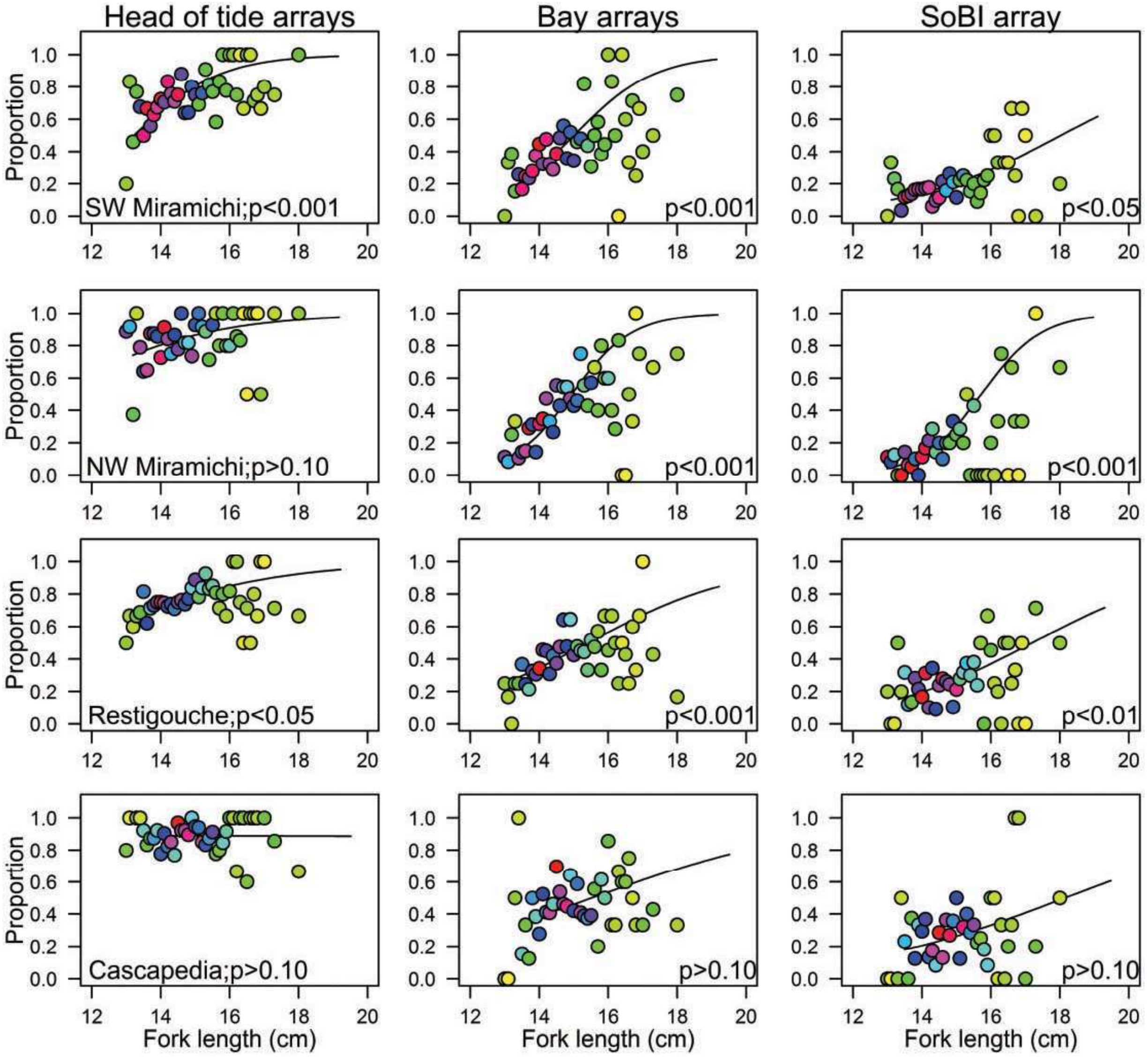

Figure 4. Proportion by fork length bin (0.1 cm) of tagged smolts detected at the head of tide lines (left column), at the bay lines (middle column), and at the primary Strait of Belle Isle line (right column) by river over all years 2003–2016. The colour of the symbol represents the relative sample size by year for each length bin with yellow (light grey) lowest and magenta (dark grey) the largest. Fork length bin 13.0 cm includes all fish of length $<=$ 13.0 cm. Fork length bin 17.3 includes lengths of 17.1–17.5, and fork length bin 18.0 includes fish of lengths $>$17.5 cm. The *p*-values for the null hypothesis (Ho: slope = 0) of the logistic regressions of detected tag (binary 0, 1) vs. fork length (centred to the mean fork length of 14.6 cm over all years) are also shown in each panel.

Hierarchical structure for probability of detection (*p*)

For the head of tide arrays (p_1), the number of receivers and their placement was annually similar (exchangeable) for a location but differed among locations. The annual probabilities of detection [$\mu^p(r,t,j)$ in Equation (3)] are modelled conditional on a prior hyperdistribution for each array ($r = 1{:}4$) (Table 1). For the two bay receiver arrays (p_2), the number of receivers and their placement was annually similar but differed between the deployments at the outlet of Miramichi Bay and the outlet of Chaleur Bay to the Gulf of St. Lawrence. The annual river-specific probabilities of detection at the bay arrays are modelled as exchangeable among years and rivers within each bay; Miramichi Bay array ($b = 1$) for the Southwest Miramichi ($r = 1$) and Northwest Miramichi ($r = 2$) rivers and Chaleur Bay array ($b = 2$) for the Restigouche River ($r = 3$) and Cascapedia River ($r = 4$) (Table 1). During 2003–2006, the primary Strait of Belle Isle array was not operational and the bay arrays were the last detection point. For those years, the prior distribution for the probability of detection at each bay array is set at the respective bay-specific hyper-distribution of the probability of detections inferred from the 2007 to 2016 monitoring years.

The number of receivers and their placement at the primary Strait of Belle Isle line ($j = 3$) was generally similar during 2007–2016. The probability of detection at the primary Strait of Belle Isle is assumed to be identical among rivers ($r = 4$) and exchangeable across years (t) (Table 2). An informative prior for the mean probability of detection of the primary array is derived from an analysis of a sentinel tagging experiment conducted at the primary Strait of

Table 1. Variables, likelihoods, and priors for the observation (p; probability of detection) and the process (ϕ; probability of survival) dynamics of the hierarchical state space formulation of the Cormack–Jolly–Seber capture and recapture model.⊗ ⊗ ⊗ ⊗ ⊗

Parameter	Likelihood	Priors
Probability of detection (p)	Head of tide lines ($j = 1$) $\text{logit}(p_{1,\ r,\ t})\ \sim\ N(\mu.p_{1,r},\ \tau.p_{1,r})$ $r = 1:\ 4;\ t = 2003:\ 2016$	$\mu.p_{1,r} \sim\ N(0,\ \tau.\varepsilon_{p;\ 1,r})$ $\tau.\varepsilon_{p;\ 1,r} = 1/(\sigma.\varepsilon^2_{p;\ 1,r});\ \sigma.\varepsilon_{p;\ 1,r} = 10$ $\tau.p_{1,r} =\ 1/(\sigma.p_{1,r})^2; \sigma.p_{1,r}\ \sim\ \text{Uniform}(0,\ 10)$
	Bay lines ($j = 2$) $\text{logit}(p_{2,r\|b,\ t}) \sim N(\mu.p_{2,b},\ \tau.p_{2,b})$ $r = 1:\ 4;\ b = 1:\ 2;\ t = 2003:\ 2016$	$\mu.p_{2,b} \sim\ N(0,\ \tau.\varepsilon_{p;\ 2,b})$ $\tau.\varepsilon_{p;\ 2,b} = 1/(\sigma.\varepsilon^2_{p;\ 2,b})\ ;\ \sigma.\varepsilon_{p;\ 2,b} = 10$ $\tau.p_{2,b} =\ 1/(\sigma.p_{2,b})^2; \sigma.p_{2,b}\ \sim\ \text{Uniform}(0,\ 10)$
	At primary Strait of Belle Isle line ($j = 3$) $\text{logit}(p_{3,\ t})\ \sim\ N(\mu.p_3,\ \tau.p_3)$ $t = 2007:\ 2016$	$\mu.p_3 = \log\left(\frac{p'_3}{1-\ p'_3}\right); p'_3\ \sim\ \text{Beta}(4.4,\ 5.6)$ $\tau.p_3 =\ 1/(\sigma.p_3)^2; \sigma.p_3\ \sim\ \text{Uniform}(0,\ 10)$
	Tag type $\beta_p\ \sim\ N(\mu.\beta_p,\ \tau.\beta_p)$	$\beta_p = 0$ $\tau.\beta_p =\ 1/(\sigma.\beta_p)^2; \sigma.\beta_p\ \sim\ \text{Uniform}(0,\ 10)$
Product (δ_4)	At secondary Strait of Belle Isle line ($j = 4$) $\delta_{4,y}\ \sim\ \text{Beta}(a, b)$	$a = b = 1$
Probability of survival (Ø)	To head of tide line ($j = 1$) $\text{logit}(\varnothing_{1,\ r,t})\ \sim\ N(\mu.\varnothing_{1,r},\ \tau.\varnothing_{1,r})$ $r = 1:4;\ t = 2003:\ 2016$	$\mu.\varnothing_{1,r} \sim\ N(0,\ \tau.\varepsilon_{\varnothing;\ 1,r})$ $\tau.\varepsilon_{\varnothing;\ 1,r} = 1/(\sigma.\varepsilon^2_{\varnothing;\ 1,r});\ \sigma.\varepsilon_{\varnothing;\ 1,r} = 10$ $\tau.\varnothing_{1,r} =\ 1/(\sigma_{1,r})^2; \sigma_{1,r}\ \sim\ \text{Uniform}(0,\ 10)$
	From head of tide to bay line ($j = 2$) $\text{logit}(\varnothing_{2,r,\ t})\ \sim\ N(\mu.\varnothing_{2,r},\ \tau.\varnothing_{2,r})$ $r = 1:4;\ t = 2003:\ 2016$	$\mu.\varnothing_{2,r} \sim\ N(0,\ \tau.\varepsilon_{\varnothing;\ 2,r})$ $\tau.\varepsilon_{\varnothing;\ 2,r} = 1/(\sigma.\varepsilon^2_{\varnothing;\ 2,r});\ \sigma.\varepsilon_{\varnothing;\ 2,r} = 10$ $\tau.\varnothing_{2,r} =\ 1/(\sigma_{2,r})^2; \sigma_{2,r}\ \sim\ \text{Uniform}(0,\ 10)$
	From bay line to primary Strait of Belle Isle line ($j = 3$) $\text{logit}(\varnothing_{3,\ r\|b,t})\ \sim\ N(\mu.\varnothing_{3,b},\ \tau.\varnothing_3)$ $r = 1:4;\ b = 1:2; t = 2007:\ 2016$	$\mu.\varnothing_{3,b} \sim\ N(0,\ \tau.\varepsilon_{\varnothing;\ 3})$ $\tau.\varepsilon_{\varnothing;\ 3} = 1/(\sigma.\varepsilon^2_{\varnothing;\ 3});\ \sigma.\varepsilon_{\varnothing;\ 3} = 10$ $\tau.\varnothing_3 =\ 1/(\sigma_3)^2; \sigma_3\ \sim\ \text{Uniform}(0,\ 10)$
	Fork length $\alpha_r\ \sim\ N(\mu.\alpha_r,\ \tau.\alpha)$ $r = 1:4$	$\mu.\alpha_r = 0$ $\tau.\alpha =\ 1/(\sigma.\alpha)^2; \sigma.\alpha\ \sim\ \text{Uniform}(0,\ 10)$
	Tag type $\beta_\varnothing\ \sim\ N(\mu.\beta_\varnothing,\ \tau.\beta_\varnothing)$	$\mu.\beta_\varnothing = 0$ $\tau.\beta_\varnothing =\ 1/(\sigma.\beta_\varnothing)^2; \sigma.\beta_\varnothing\ \sim\ \text{Uniform}(0,\ 10)$

The superscript $r|b$ refers to the river (r) within a bay (b) with 1|1 the Southwest Miramichi within Miramichi Bay, 2|1 the Northwest Miramichi within Miramichi Bay, 3|2 the Restigouche River within Chaleur Bay, and 4|2 the Cascapedia River within Chaleur Bay. In all cases, σ refers to the standard deviation. For all normal distributions, the variance is expressed as precision (inverse of variance).

Belle Isle array to provide independent information on detection probabilities (Supplementary). The average detection probability to a radius of 0.5 km was estimated to be 44% and this is used to parameterize an informative but uncertain prior for the mean detection probability of the primary Strait of Belle Isle array (Table 1).

The secondary Strait of Belle Isle line ($j = 4$) was installed in 2015 and 2016 to provide empirical data to estimate the detection efficiency of the primary Strait of Belle Isle array. This is the last point of detection and there is no auxiliary information on the expected value of the probability of detection. The detections at this array are modelled as conditional on the product of p_4 and ϕ_4 (δ; on the logit scale) and on a fish being alive at the primary Strait of Belle Isle array ($z(i, 3)$);

$$y(i,\ 4)\ |\ z(i,\ 3), \delta_4\ \sim\ \text{Bernoulli}(z(i,\ 3), \delta_4). \qquad (7)$$

It is assumed that δ_4 differs between years but is similar for the river origins of the smolts and an annually uninformative prior for the product is used (Table 1).

Hierarchical structure for probability of survival (ϕ)

The first transition stage (ϕ_1) encompasses the point of release ($j = 0$) to the head of tide array ($j = 1$) and the distance as well as the physical and biological environment of this zone differs among the four rivers. Thus, the $\mu^{\phi}(r, t, 1)$ [Equation (6)] are modelled exchangeably among years (t) for each river group ($r = 1{:}4$) (Table 1). The second transition stage (ϕ_2) encompasses the geographic region from the head of tide array ($j = 1$) to the bay arrays ($j = 2$) in Miramichi Bay and Chaleur Bay. The distance from the head of tide arrays and the physical and biological characteristics of the estuary zones also differ among the four rivers and the survival probabilities are modelled exchangeably among years (t) for each river group (r) (Table 1).

The third transition stage (Gulf of St. Lawrence) encompasses the geographic region from the exit of Miramichi and Chaleur Bays ($j = 2$) to the Strait of Belle Isle array ($j = 3$). There is a minimal difference in the straight-line migration distances between the bay arrays and the Strait of Belle Isle array but there are important differences in the observed migration durations with tagged smolts from

Table 2. Summary of migration characteristics (median and 5th to 95th percentile range for all smolts and years) and survival rates (range of median values as % and based on median migration days expressed as % day^{-1}) of acoustically tagged Atlantic salmon smolts from four rivers in the Gulf of St. Lawrence.

Transition	River	Distance	Migration (days)	Migration (km day^{-1})	Survival (%)	Survival (% day^{-1})
Release to head of tide array	Southwest Miramichi	127	5.2; 2.1–11.3	27; 12–65	62–92	93–98
	Northwest Miramichi	30; 52	2.9; 0.8–7.7	10; 4–37	90–91	93–98
	Restigouche	115	4.4; 2.3–10.6	26; 11–50	73–93	96–98
	Cascapedia	8	1.2; 0.2–4.2	6; 2–36	93–97	82–97
Head of tide to bay array	Southwest Miramichi	68	3.6; 2.2–7.9	19; 9–31	42–82	83–97
	Northwest Miramichi	67	4.6; 2.6–10.2	15; 7–26	28–74	78–93
	Restigouche	106	7.4; 4.2–13.7	14; 8–25	67–95	96–99
	Cascapedia	47	8.0; 3.7–18.5	6; 3–13	68–90	93–99
Bay array to primary Strait of Belle Isle array	Southwest Miramichi	~800	48.4; 31.7–53.7	17; 15–25	54–64	98–99
	Northwest Miramichi		46.2; 35.2–51.5	17; 16–23	60–68	99–99
	Restigouche		40.2; 29.8–49.3	20; 16–27	35–74	96–99
	Cascapedia		36.1; 21.9–47.9	22; 17–37	27–78	97–99

the Miramichi River locations taking longer to transit the Gulf of St. Lawrence than smolts from the Chaleur Bay rivers (Table 2; Supplementary Table S3). The probabilities of survival are modelled exchangeably among years by bay specific group, Miramichi Bay rivers and Chaleur Bay rivers (Table 1).

Odds ratios of survival probabilities associated with the size of smolt at tagging are calculated as the predicted survival of a 16 cm smolt relative to the predicted survival of a 13.5 cm smolt. Survival rates by transition zone for a smolt of fork length corresponding to the mean length smolts in the study (14.6 cm), are presented as well as the survival rates per median day at large in each zone ($\phi^{1/d}$). Uncertainties in the estimates of the probabilities of detection and survival are described by the coefficient of variation, corrected for the logit transformation.

Model fitting and assessing convergence

The hierarchical state-space CJS model was fit to smolt tagging data from the four rivers for the years 2003–2016, representing 2 862 observations, using the freely available software package OpenBUGS (Lunn *et al.*, 2013). A total of 60 000 Markov chain Monte Carlo (MCMC) simulations with two chains was used, the first 50 000 were discarded and the remaining 10 000 samples were thinned by 10 to produce 2 000 MCMC values to summarize the posterior distributions. We assessed whether there was evidence of non-convergence by examining trace plots of the MCMC chains, by checking that the Gelman–Rubin r-hat statistics were < 1.1, and by examining for unimodal distributions of the model parameters (Brooks and Gelman, 1998). The diagnostics examined did not suggest any non-convergence.

We examined a number but not all possible combinations of parameters and model structures for detection and survival. Adequacy of the model was assessed by predicting detections at the three array locations based on the posterior distributions of p and ϕ (Supplementary Table S4 and Figures 3S–S10). By design, we favoured a hierarchical structure for the probabilities of detection to make use of the data from the entire time series of the study. The alternative of assuming a constant probability of detection over years at each of the arrays was not realistic given the empirical observations to the contrary, and all the model variants that assumed a constant probability of detection over years resulted in higher residual deviances and poor fits. Independent and hierarchical structures were examined for the probability of survival with and without individual effects associated with size of smolts and tag type (Supplementary Table S4).

When discussed, statistical significance corresponds to a p-value $< = 0.05$. For the individual effects parameters, the p-values are calculated as the smallest proportion of the MCMC values drawn from the marginal posterior distribution that overlap zero.

Results

During 2003–2016, a total of 2 862 Atlantic salmon smolts from four rivers were tagged with acoustic transmitters. A total of 2 243 of these tags, 78% of releases, were subsequently detected at receiver arrays located at or near the head of tide (Figure 3; Supplementary Table S2). A total of 1 160 tags, 41% of released fish, were subsequently detected at the bay receiver arrays. Finally, 487 tags, 17% of released fish for the corresponding years, were detected at the primary Strait of Belle Isle array, almost 2 months and more than 800 km away from their release locations. In 2015 and 2016, 119 tags were detected at the secondary Strait of Belle Isle array, representing 23% of the tagged smolts released, similar to 122 tags detected at the primary Strait of Belle Isle array for the same years. During 2010–2016, only two tags placed in smolts were detected at the Cabot Strait line (Ocean Tracking Network, unpublished data) suggesting that the Strait of Belle Isle is the primary migration route for smolts from the rivers in this study leaving the Gulf of St. Lawrence.

Migration summaries

Tagged smolts from the Cascapedia River had the shortest freshwater distance from release to the head of tide (8 km) and were generally detected within 2 days post-release (Table 2; Supplementary Table S3). In contrast, smolts from the Southwest Miramichi had the longest migration distance to the head of tide (127 km) and most fish were detected at the head of tide 2–11 days post release. Tagged smolts from the Southwest Miramichi River and the Restigouche River had the fastest migration rates in freshwater at a median over years of 27 km day^{-1} and 26 km day^{-1}, respectively (Table 2). The median migration rate of tagged smolts in the Northwest Miramichi was 10 km day^{-1} and the slowest migration rate was estimated for the Cascapedia smolts at 6 km day^{-1} (Table 2).

Migration rates of tagged smolts through the bays were highest for the Southwest Miramichi (19 km day^{-1}), relatively similar for

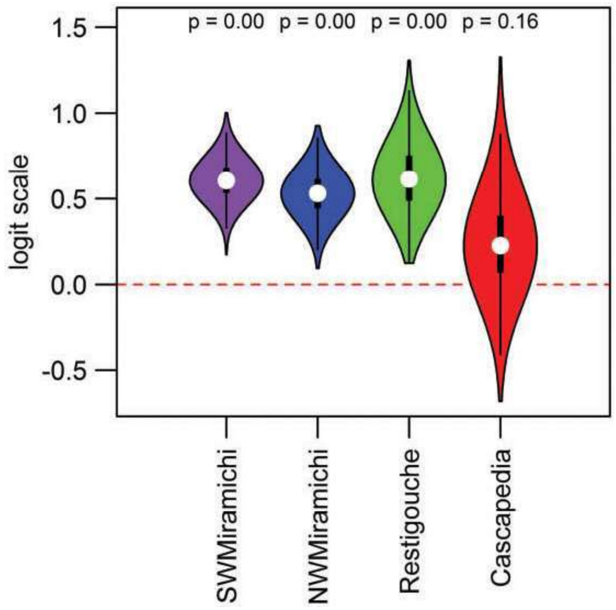

Figure 5. Posterior distributions of the fork length individual effects (logit scale) on the probability of survival through the freshwater and bay zones for each river. Violin plots summarize the kernel densities of the distributions and show the median (open symbol), interquartile range (thick line), and the range of 95% of the observations (thin line).

the Northwest Miramichi and Restigouche River groups (15 km day^{-1} and 14 km day^{-1}, respectively), and slowest for the Cascapedia River smolts (6 km day^{-1}; Table 2). In contrast to the differences in migration rates through Chaleur Bay, the days from release to detection at the Chaleur Bay arrays were more similar for the Restigouche (7 days) and Cascapedia (8 days) smolts, indicating that the Cascapedia smolts which had a very short freshwater phase post-release lingered longer in brackish and saltwater compared to the Restigouche smolts before exiting the bay (Table 2). These differences in migration duration between rivers within the same bay may reflect an acclimation period associated with tagging and handling.

The migration duration through the Gulf of St. Lawrence varied from just over 20 days to just under 65 days (Table 2; Supplementary Table S3). Migration duration was generally the longest for the Southwest Miramichi smolts at 48 days (median), and the shortest but most variable for the Cascapedia smolts at 36 days (Table 2). Detections of acoustic tags at the Strait of Belle Isle from the four rivers were synchronized among the four rivers with detections across all years (2007–2016) and rivers occurring during a relatively narrow 4-week period of late June to late July (Supplementary Figure S2). Migration rates through the Gulf of St. Lawrence were in the range of 17–22 km day^{-1} (median over all rivers and years) with the fastest median migration rate for the Cascapedia and Restigouche smolts (22 and 20 km day^{-1}, respectively) and the slower rates for the Northwest Miramichi and Southwest Miramichi smolts (17 km day^{-1}; Table 2).

Individual—tag type and fork length

There was no statistically significant ($p = 0.36$) difference in the probabilities of detection associated with the tag type used. Tag type was also not a statistically significant covariate ($p = 0.12$) for survival when included as an interaction term with fork length (Supplementary Figure S8c). Statistically significant ($p < 0.05$) positive associations of fork length on probability of survival were estimated for three of the four rivers; the exception being the Cascapedia River ($p = 0.16$) (Figure 5). Based on the median of the marginal posterior distribution of the fork length coefficient, the odds ratio of survival for a smolt of 16 cm fork relative to a smolt of 13.5 fork length was in the range of 1.51–1.74.

Estimated probabilities of detection

The estimated probabilities of detection at the head of tide receiver lines were generally high (often >90%) with few exceptions such as for the Northwest Miramichi in 2006 (<40%) and for the Restigouche in 2011 (~50%) (Figure 6). The uncertainties (coefficient of variation on the inverse logit scale) in the annual probabilities of detection at the head of tide array were most consistent for the Southwest Miramichi, Cascapedia, and Restigouche Rivers (0.1–11.1%) and most variable for the Northwest Miramichi River (0.1–32.4%).

The estimated probabilities of detection were higher at the Miramichi Bay line compared to the Chaleur Bay line, with median posterior values across years of 83 vs. 56%, respectively (Figure 6). The uncertainties in the annual probabilities of detection were higher at the bay arrays compared to the head of tide arrays, with annual CVs by river ranging from 7.3 to 31.9%.

The probabilities of detection of the primary Strait of Belle Isle line varied annually from a low of 42% in 2006 (median value; very similar to the prior) to a high of just over 67% in 2010 and in 2015 (Figure 6). The median of the estimates for 2015 and 2016 derived from the detections at the twinned line were 67 and 64%, respectively. The uncertainties in the annual estimates of the probability of detection ranged from 8.6 to 30.4% with CV values in 2015 and 2016 of 8.7 and 8.6%, respectively.

Estimated probabilities of survival

The posterior distributions of the estimated probabilities of survival in freshwater, in the estuary, and in the Gulf of St. Lawrence, standardized to the mean fork length (14.6 cm) of smolts from all rivers and years, are shown in Figure 7. The probabilities of survival in freshwater were highest (median 96%) and relatively similar (median range 93–97%) over years for the Cascapedia River, slightly lower (90%) for the Northwest Miramichi River and lowest for the Restigouche (median values 73–93%) and Southwest Miramichi (range of medians 62–92%) rivers which had the longest distance and migration duration to head of tide (Figure 7; Table 2). Survival estimates from release to the head of tide were generally high and greater than 90% day^{-1} for all rivers and years (Table 2). The uncertainties in the annual estimates of survival were lowest for the Cascapedia and the Northwest Miramichi Rivers (annual CV range of 2.3–4.4%), in contrast to the Southwest Miramichi and Restigouche rivers with annual CV ranges of 2.7–10.6%, respectively.

The estimated probabilities of survival of tagged smolts transiting Chaleur Bay were higher (annual medians ranging from 67 to 95%, 93 to 99% day^{-1}) than for smolts transiting Miramichi Bay (annual medians ranging from 28 to 82%, 78 to 97% day^{-1}) (Figure 7; Table 2). The lowest estimated survivals of any rivers and years were for smolts from the Northwest Miramichi River during 2013–2016 (medians ranging from 28 to 45%; Figure 7). Survival rates of smolts migrating through Miramichi Bay were higher during 2006–2008

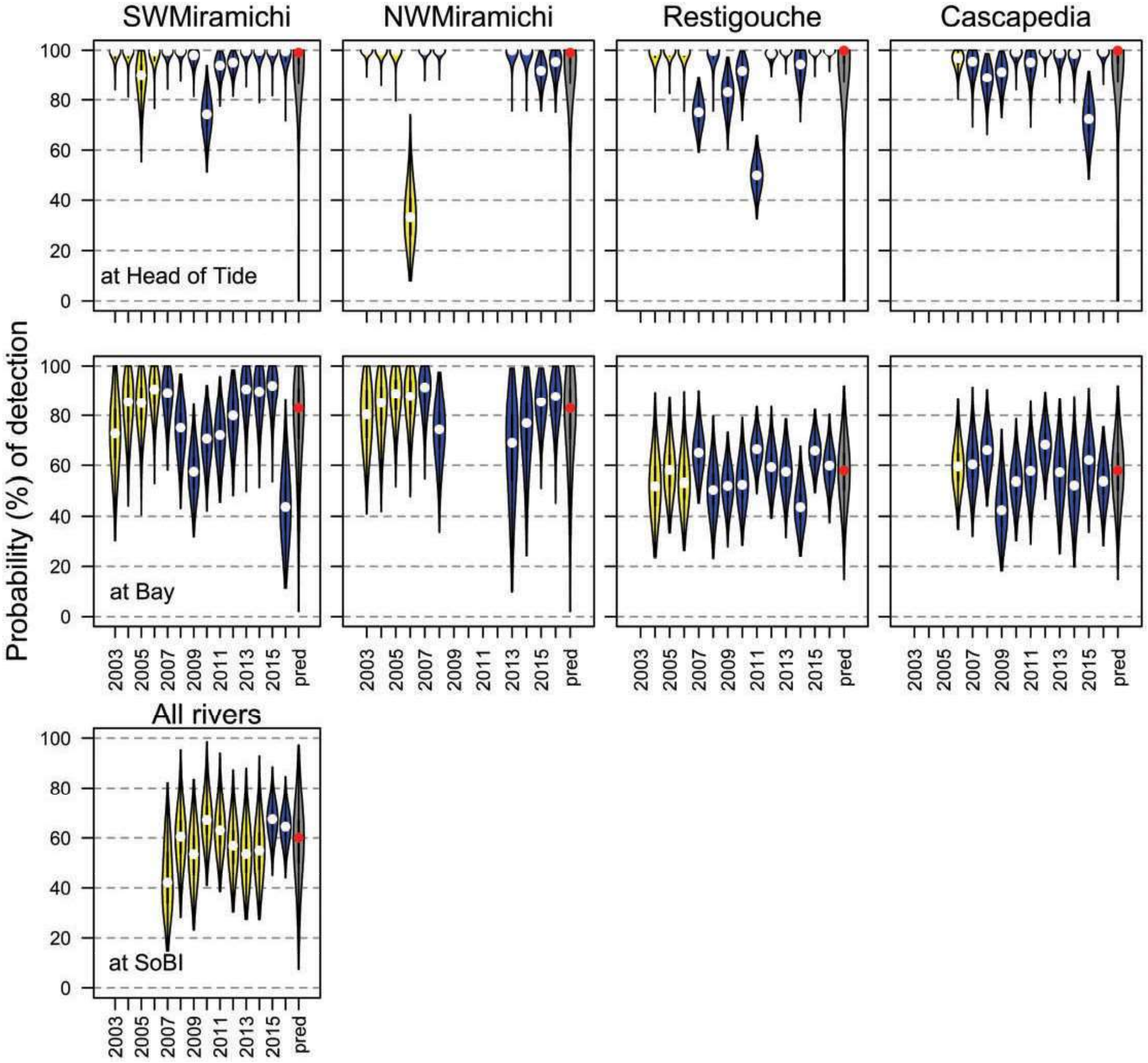

Figure 6. Violin plots of posterior distributions of the probability of detection by river at the head of tide lines (upper row), at the bay lines (middle row), and by year (all rivers) at the primary Strait of Belle Isle line (lower row), 2003–2016. The open symbol is the median of the posterior distribution. For the bay lines, the violin plots in blue are for the years (2007–2016) when the primary Strait of Belle Isle line was operational. For the Strait of Belle Isle line, the violin plots in blue correspond to the years when the secondary twinned line was operational (2015 and 2016). The results are for Model variant 11 g in Supplementary. In all panels, the grey shadings are the posterior distributions of the predicted probabilities of detection over all years for the corresponding spatial hierarchical structure (Table 1).

(91–94% day^{-1} for Southwest Miramichi; 90–91% day^{-1} for Northwest Miramichi) than during the last 4 years of the study (2013–2016; 83–90% day^{-1} for Southwest Miramichi; 78–91% day^{-1} for Northwest Miramichi). This contrasts with survival rates through Chaleur Bay that remained high with no evidence of declines over the same periods (Figure 7). Uncertainties (CV) in the annual estimated probabilities of survival were greatest for the Northwest Miramichi River, ranging from 17 to 43%.

With few exceptions (Restigouche River in 2007–2009, Cascapedia River in 2008), the median estimated survival rates of tagged smolts through the Gulf of St. Lawrence were between 45 and 78% (Figure 7), 96–99% day^{-1} (Table 2). Estimates of survival rates through the Gulf of St. Lawrence were the most uncertain of all the transition zones, with annual CVs ranging from 13 to 45%.

Discussion

The objectives of this study were to characterize the early phase migration and to gain insights into the location and timing of smolt and post-smolt mortality of wild Atlantic salmon smolts and post-smolts from unimpacted (free fish passage) rivers in eastern Canada. Atlantic salmon smolts and post-smolts were successfully detected using acoustic telemetry during the initial 50+ days post migration from freshwater, through estuaries and nearshore bays and to distances exceeding 900 km at sea from the point of release in freshwater.

In this study, we refer for convenience to the estimation of survival of smolts and post-smolts when what is in fact being estimated is the probability of detecting a tag that has been deployed

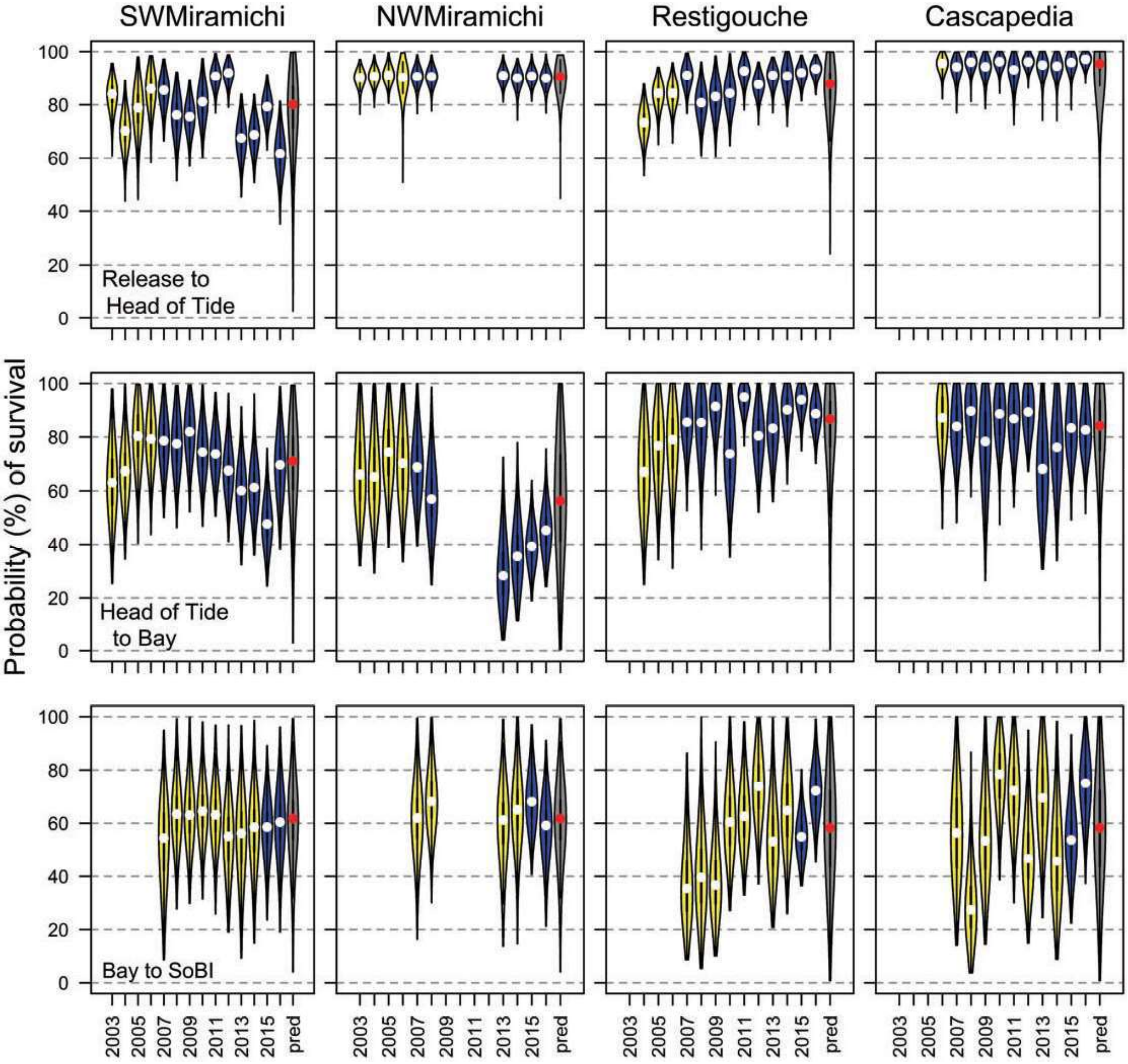

Figure 7. Violin plots of posterior distributions of probability of survival for a smolt of centred length 14.6 cm to the head of tide lines (upper row), from the head of tide to the bay lines (middle row), and from the bay lines to the primary Strait of Belle Isle line (lower row), for the four rivers, 2003–2016. For the survivals through the bays, the violin plots in blue are the probabilities of survival estimated for the years when the primary Strait of Belle Isle line was operational (2007–2016). For the survivals through the Gulf of St. Lawrence, the violin plots in blue are the years when the secondary twinned line was operational (2015 and 2016). In all panels, the grey shadings are the posterior distributions of the predicted probabilities of survival over all years for the corresponding spatial hierarchical structure (Table 1).

in a fish, conditional on the tag being retained in the fish, the fish with the tag moving within range of the receivers and a tag transmission being detected by a receiver. Within the model in this study, the observations represent the last component of this dynamic (probability of a tag transmission being detected by a receiver) whereas what we referred to as survival represents the other components; that an implanted tag is retained in a fish and the tag within the fish migrates downstream within the range of receivers, more correctly termed apparent survival.

Location and timing of mortality

The main interest of this study was to estimate population level smolt and post-smolt survivals. Survival rates through the freshwater stage were generally high (>90%) for the fish released less than 30 km from the head of tide but lower (60–90%), annually variable and negatively associated with the migration distance or the time from release to detection for the two rivers (Southwest Miramichi and Restigouche) with a longer freshwater migration distance. Variable but generally high survival rates (ranging from 70 to 100%) of acoustically tagged smolts over variable migration distances in freshwater zones (2–53 km) have been reported in other studies (Lacroix, 2008; Halfyard *et al.*, 2012; Lefèvre *et al.*, 2013; Gibson *et al.*, 2015; Crossin *et al.*, 2016; Hawkes *et al.*, 2017). The exception to this was reported for smolts in the Penobscot River for which survival rates through freshwater zones were much lower (mean cumulative survival of 47%) and attributed to the effects of passage at hydro facilities (Stich *et al.*, 2015).

Survival rates through estuaries and coastal bays were lower than in freshwater areas, highly variable among years and contrasted among bays. Comparisons across studies of survival estimates through estuaries, bays, or nearshore coastal environments are more difficult in large part because of physical differences in the dynamic and geographically diverse habitats transited by salmon in the species range in eastern North America. In this study, the two areas monitored differ in coastal structure, water chemistry and passage routes, with the Miramichi being a semi-enclosed bay with constrained passage into the Gulf of St. Lawrence in contrast to the wide and open Chaleur Bay. Dempson *et al.* (2011) monitored tagged smolt movements in a geographically complex fjord, island and channel habitat, similar to the complex nearshore area traversed by tagged smolts in the study by Stich *et al.* (2015) and Hawkes *et al.* (2017). The study area in Halfyard *et al.* (2013) consisted of geographically complex areas with diverse estuaries and bays and extended estuary areas whereas in the study area of Lefèvre *et al.* (2013) the river opened almost directly into the Gulf of St. Lawrence with no estuary or inner bay component. Survival rates through these diverse areas are highly variable, with values of 54–87% for the Conne River study (Dempson *et al.*, 2011), 39–74% for smolts for the Atlantic coast of Nova Scotia (Halfyard *et al.*, 2012), and much lower survivals, <50%, for two southern stocks (Kocik *et al.*, 2009; Hawkes *et al.*, 2017). Survival rates per day in these near coastal areas are highly variable but compared to the freshwater phase of the migration, the survival rates are lower in the estuary/bay areas (see studies above).

Once the smolts leave the coastal bays, inferred apparent survival rates as post-smolts through the Gulf of St. Lawrence were in the range of 28–78% with survivals rates exceeding 96–99% day^{-1} for all rivers and years. Survival rates of tagged smolts, expressed as rates per day, are lowest in the estuary portions and highest during the migration through the Gulf of St. Lawrence, which is consistent with mortality being highest on small fish at first entry to the sea and declining as fish grow and move offshore (Thorstad *et al.*, 2012).

The spatial and temporal differences in apparent survival rates from our study in two neighbouring coastal areas, and between two rivers within one basin over two time periods, may in part be related to both physico-chemical and biotic differences. Although we present the temporal trend in survival rates of Northwest Miramichi smolts as a contiguous series, the fish tagged during 2003–2008 were taken from a different branch of the Northwest Miramichi than those tagged in 2013–2016 with the smolts from the latter period captured downstream and released again above a tributary (Tomogonops River) impacted by acid and metal runoff from a decommissioned base metal mine (St-Hilaire and Caissie, 2001). There are concerns for Atlantic salmon smolt vulnerability to episodic acidification and elevated concentrations of bioavailable aluminum during spring snow melt and increased runoff (Kroglund *et al.*, 2008; Kelly *et al.*, 2015). Thorstad *et al.* (2013) reported on delayed mortality in the early period of marine migration of smolts exposed to aluminum and moderate acidification in freshwater. This cannot be excluded as a factor contributing to the lower apparent survival rates in the Northwest Miramichi smolts in the latter part of the time series as well as a factor contributing to differences between the Miramichi Bay and the Chaleur Bay rivers.

There is also an important biotic difference in the estuarine environments of Miramichi Bay and Chaleur Bay. The upper portion of the Northwest Miramichi estuary is the only confirmed spawning location of striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence and the spawning period overlaps in timing with the downstream smolt migration. Atlantic salmon smolts have been identified in stomachs of striped bass sampled from the Miramichi (DFO, 2016). Furthermore, Daniels *et al.* (2018) report on inferred predation rates of striped bass on salmon smolts from the Miramichi based on contrasting movement patterns of acoustically tagged animals. The inferred predation rates ranged from 2 to 18%, between stocks and years, with annual variations in the spatial and temporal overlap of the two species likely contributing to the differences in the inferred predation rates.

Finally, we cannot exclude the possibility that the differences in estimated survival rates between bays and over years in this study are also in part due to differences in the size distributions of acoustically tagged smolts among years and rivers. A weight of evidence analysis of factors contributing to variations in apparent survival rates is beyond the scope of this paper however the data from this 14-year study would be appropriate for testing these hypotheses.

Limitations of study and results

Modelled estimates of apparent survival of acoustically tagged and tracked smolts can be biased. Survival estimates can be biased downward if the tag is not retained by the fish and the fish survives and migrates past the receiver arrays. There is evidence from literature that implanted tags can be expelled from the body cavity without resulting in death of the fish, and the probability of expulsion was related to the ratio of tag size to fish size (Lacroix *et al.*, 2004; Welch *et al.*, 2007; Sandstrom *et al.*, 2013). Expulsion of larger tags (24 mm length by 8 mm diameter) was noted but Lacroix *et al.* (2004) indicated that the 24 mm tags were the only tags, which were retained by some fish during the 316-day duration of the experiment. In the study by Welch *et al.* (2007), tag expulsion of 24 mm by 8 mm tags generally occurred after 4 weeks post-surgery. Based on these studies, tag shedding in this study was not considered to be a factor that would bias the estimation of apparent survivals since the smolts had migrated through the bays within 2 weeks or less in most cases.

However, it is assumed that a tag detection at a receiver line is from a tag in a salmon smolt rather than in the stomach of a predator swimming by the receiver. If the predation event occurred upstream of the bay array, then some of the detections at the bay arrays could be of tags in predator stomachs rather than smolts and in such cases, the inferred survival rate of tagged smolts to the bay arrays would be overestimated; consequently the survival rate through the Gulf of St. Lawrence would be underestimated.

One important factor that can affect the exchangeability assumption of survival in the hierarchical model used in this study is the size of the smolts tagged. Sizes of smolts used in the experiments varied annually and differed among rivers. Other studies have reported on correlations between tagging effects (survival) and smolt size (Lacroix *et al.*, 2004; Welch *et al.*, 2007; Halfyard *et al.*, 2013). Lacroix *et al.* (2004) recommended a transmitter length of 16% or less of fish length for telemetry studies. The V9 tags used in this study measured 21 mm in length, and based on criteria of Lacroix *et al.* (2004) could be placed in smolts 13.1 cm or longer. There were very few smolts in this study that did not meet this minimum size, representing <2% of smolts over all rivers and years and less than 6% of smolts from the Northwest Miramichi.

Relevance of inferences from tagged smolts to untagged smolts

An important concern regarding the use of marked animals to make inferences on behaviour and survival of unmarked/unhandled animals is the consequence of tagging and handling effects on the estimates of survival or migration dynamics. It is extremely difficult to make the case that a tagged smolt would behave and have the same mean probability of survival as an untagged smolt. In terms of absolute levels, it is unlikely that the estimates derived from marked animals correspond to those of unmarked animals (Riley *et al.*, 2018). There can be important growth and survival effects of handling and tagging even when animals are held in captivity post tagging (Moore *et al.*, 1990; Lacroix *et al.*, 2004; Welch *et al.*, 2007; Ammann *et al.*, 2013) and monitoring tagged fish in captivity does not provide much insight into the conditions encountered by fish released to the wild. The capture, handling, tagging procedures in addition to introducing stress and injury to individual animals (Ammann *et al.*, 2013) also interrupt the migration phenology of wild smolts during a particularly sensitive period (Riley *et al.*, 2007). Removal from schooling with conspecifics, release back to the river during the day or even near dusk when wild conspecifics are sheltering and not in active migration phase, can result in increased vulnerability to predation (Furey *et al.*, 2016). There is evidence from this study that acoustically tagged Atlantic salmon smolts less than 14 cm fork length suffered a higher mortality than smolts of greater size and this could be an effect of stress from tagging and handling and correlated with the tag/body size ratio. In two rivers (Southwest Miramichi, Restigouche), estimated apparent survival rates to the head of tide receivers after correcting for size, are negatively associated with the migration duration, which can be interpreted as a delayed mortality from handling and tagging and an increased vulnerability to predation. For the Cascapedia smolts for which there is a very short freshwater migration distance and time from release to the head of tide, the smolts had a prolonged migration duration through Chaleur Bay compared to Restigouche River smolts suggesting that there may have been a period of acclimation in the bay specifically for the Cascapedia smolts.

Estimating apparent survival rates at the further migration points and times is also challenging. As fish die over time, there are fewer tagged fish available with which to estimate detection and survival probabilities. This has consequences on the uncertainty of the estimates, as evidenced from the higher coefficient of variations of the estimates of the detection and apparent survival probabilities at the bay and the Strait of Belle Isle arrays. Increased sample sizes of tagged fish from a single stock could be considered, as was the case for the Restigouche River, by tagging multiple stocks that share a common bay exit, or multiple stocks that share a common exit to the Labrador Sea. The probability of detection at receiver lines can only be inferred if there are tags, which are detected at a "downstream" array (along the migration route, or temporally). At the last detection array only the product of the survival and detection can be inferred (Gimenez *et al.*, 2007; Royle, 2008). The use of auxiliary data such as sentinel tags to independently inform on detection rates is required if survival rates to the last array are to be estimated. Auxiliary data from sentinel tags are best incorporated in the model as prior information. When the last array is twinned, as was done for the Strait of Belle Isle line in 2015 and 2016, the detection probabilities of the next to last array can be estimated from observations, i.e. the prior for the primary Strait of Belle Isle line is updated with observations. In this study, the prior probability of detection based on sentinel tags was strongly updated by the empirical observations of the secondary twinned line at the exit to the Labrador Sea.

Insights into the factors that modify the variation in survival rates within particular areas of the smolt and post-smolt migration require experiments to be conducted over multiple years and populations (Thorstad *et al.*, 2012). The multi-year and multi-river aspects of this study provided particular advantages to describing and modelling smolt migrations and estimating survival rates that otherwise would not be possible from single year and single river experiments. The observations in this study can be effectively modelled using a hierarchical structure and such a model provides a means of using all the information even in years when the full monitoring infrastructure is not in place.

In long-term studies, it is imperative that the methods and experimental design be standardized to ensure that the empirical observations reflect to the extent possible, the variations in the phenomenon of interest, rather than a consequence of differences in methodologies, experimental design, or technologies. In the study reported here on estimating survival rates of Atlantic salmon smolts from four rivers over 14 years, factors that could be standardized include the tag type, the size distribution of smolts being tagged, the tag implantation procedures and the placement of the receiver arrays. By standardizing these elements of the study, the individual river experiments are more likely to be exchangeable and by using hierarchical models, the inferences on the parameters of interest less uncertain.

Acknowledgements

The authors acknowledge the field work contributions over the years of study of staff from the Atlantic Salmon Federation, in particular Graham Chafe, who were involved in the tagging operations, receiver deployments, and data retrievals. Technical and financial support was provided by a number of local organizations, including International Paper, the Miramichi Salmon Association, Fisheries and Oceans Canada, the Restigouche River Watershed Management Committee, the Cascapedia River Society, the Gesgepegiac First Nation, the Listuguuj First Nation, the Ministère des Forêts, de la Faune et des Parcs (Québec), and the Oceans Tracking Network. This study would not have been possible without the financial contributions of a large number of individuals, foundations, and organizations that sponsored this research through donations to the Atlantic Salmon Federation. A complete list of contributors can be found at www.asf.ca. We also acknowledge the editorial contributions of the two anonymous reviewers.

Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

References

Ammann, A. J., Michel, C. J., and MacFarlane, R. B. 2013. The effects of surgically implanted acoustic transmitters on laboratory growth, survival and tag retention in hatchery yearling Chinook salmon. Environmental Biology of Fishes, 96: 135–143.

Beaugrand, G., and Reid, P. C. 2003. Long term changes in phytoplankton, zooplankton and salmon related to climate. Global Change Biology, 9: 801–817.

Beaugrand, G., and Reid, P. C. 2012. Relationships between North Atlantic salmon, plankton, and hydroclimatic change in the Northeast Atlantic. ICES Journal of Marine Science, 69: 1549–1562.

Bonner, S. J., and Schwarz, C. J. 2006. An extension of the Cormack–Jolly–Seber model for continuous covariates with application to *Microtus pennsylvanicus*. Biometrics, 62: 142–149.

Brooks, S. P., and Gelman, A. 1998. General methods for monitoring convergence of iterative simulations. Journal of Computational and Graphical Statistics, 7: 434–455.

Chaput, G. J., and Jones, R. A. 2004. Catches of downstream migrating fish in fast-flowing rivers using rotary screw traps. Canadian Manuscript Report of Fisheries and Aquatic Sciences, 2688: v + 14 pp.

Chaput, G., Legault, C. M., Reddin, D. G., Caron, F., and Amiro, P. G. 2005. Provision of catch advice taking account of non-stationarity in productivity of Atlantic salmon (*Salmo salar* L.) in the Northwest Atlantic. ICES Journal of Marine Science, 62: 131–143.

Chiasson, A. 1995. The Miramichi bay and estuary: an overview. *In* Water, Science and the Public: The Miramichi Ecosystem, pp. 11–28. Ed. by E. M. P. Chadwick. Canadian Special Publication of Fisheries and Aquatic Sciences, 123: 300.

Crossin, G. T., Hatcher, B. G., Denny, S., Whoriskey, K., Orr, M., Penney, A., and Whoriskey, F. G. 2016. Condition-dependent migratory behaviour of endangered Atlantic salmon smolts moving through an inland sea. Conservation Physiology, 4: cow018.

Daniels, J., Chaput, G., and Carr, J. 2018. Estimating consumption rate of Atlantic Salmon smolts (*Salmo salar*) by Striped Bass (*Morone saxatilis*) in the Miramichi River estuary using acoustic telemetry. Canadian Journal of Fisheries and Aquatic Sciences, 1. https://doi.org/10.1139/cjfas-2017–0373.

Dempson, J. B., Robertson, M. J., Pennell, C. J., Furey, G., Bloom, M., Shears, M., Ollerhead, L. M. N., *et al.* 2011. Residency time, migration route and survival of Atlantic salmon *Salmo salar* smolts in a Canadian fjord. Journal of Fish Biology, 78: 1976–1992.

DFO. 2016. Spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 2015. DFO Canadian Science Advisory Secretariat Science Response, 2016/017.

DFO. 2017. Oceanographic conditions in the Atlantic zone in 2016. DFO Canadian Science Advisory Secretariat Science Advisory Report, 2017/031.

Drenner, S. M., Clark, T. D., Whitney, C. K., Martins, E., Cooke, S. J., and Hinch, S. G. 2012. A synthesis of tagging studies examining the behaviour and survival of anadromous salmonids in marine environments. PLoS One, 7: e31311.

Friedland, K. D., Hansen, L. P., Dunkley, D. A., and MacLean, J. C. 2000. Linkage between ocean climate, post-smolt growth, and survival of Atlantic salmon (*Salmo salar* L.) in the North Sea area. ICES Journal of Marine Science, 57: 419–429.

Furey, N. B., Hinch, S. G., Bass, A. L., Middleton, C. T., Minke-Martin, V., and Lotto, A. G. 2016. Predator swamping reduces predation risk during nocturnal migration of juvenile salmon in a high-mortality landscape. Journal of Animal Ecology, 85: 948–959.

Gelman, A., Carlin, J. B., Stern, H. S., and Rubin, D. B. 2004. Bayesian Data Analysis, 2nd edn. Chapman & Hall/CRC, Boca Raton, Florida. 668 pp.

Gibson, A. J. F., Halfyard, E. A., Bradford, R. G., Stokesbury, M. J. W., and Redden, A. M. 2015. Effects of predation on telemetry-based survival estimates: insights from a study on endangered Atlantic salmon smolts. Canadian Journal of Fisheries and Aquatic Sciences, 72: 728–741.

Gimenez, O., Rossi, V., Choquet, R., Dehais, C., Doris, B., Varella, H., Vila, J.-P., *et al.* 2007. State-space modelling of data on marked individuals. Ecological Modelling, 206: 431–438.

Goulette, G. S., Hawkes, J. P., Kocik, J. K., Manning, J. P., Music, P. A., Walling, J. P., and Zydlewski, G. B. 2014. Opportunistic acoustic telemetry platforms: benefits of Collaboration in the Gulf of Maine. Fisheries, 39: 441–450.

Halfyard, E. A., Gibson, A. J. F., Ruzzante, D. E., Stokesbury, M. J. W., and Whoriskey, F. W. 2012. Estuarine survival and migratory behaviour of Atlantic salmon *Salmo salar* smolts. Journal of Fish Biology, 81: 1626–1645.

Halfyard, E. A., Gibson, A. J. F., Stokesbury, M. J. W., Ruzzante, D. E., and Whoriskey, F. W. 2013. Correlates of estuarine survival of Atlantic salmon postsmolts from the Southern Upland, Nova Scotia, Canada. Canadian Journal of Fisheries and Aquatic Sciences, 70: 452–460.

Hawkes, J. P., Sheehan, T. F., and Stich, D. S. 2017. Assessment of early migration dynamics of river-specific hatchery Atlantic salmon smolts. Transactions of the American Fisheries Society, 146: 1279–1290.

Hussey, N. E., Kessel, S. T., Aarestrup, K., Cooke, S. J., Cowley, P. D., Fisk, A. T., Harcourt, R. G., *et al.* 2015. Aquatic animal telemetry: a panoramic window into the underwater world. Science, 348: 1255642.

ICES. 2017. Report of the Working Group on North Atlantic Salmon (WGNAS), 29 March–7 April 2017, Copenhagen, Denmark. ICES CM 2017/ACOM: 20: 296 pp.

Kelly, J. T., Lerner, D. T., O'Dea, M. F., Regish, A. M., Monette, M. Y., Hawkes, J. P., Nislow, K. H., *et al.* 2015. Evidence for episodic acidification effects on migrating Atlantic salmon *Salmo salar* smolts. Journal of Fish Biology, 87: 1129–1146.

Kilduff, D. P., Di Lorenzo, E., Botsford, L. W., and Teo, S. L. H. 2015. Changing central Pacific El Niños reduce stability of North American salmon survival rates. Proceedings of the National Academy of Sciences of the United States of America, 112: 10962–10966.

Kocik, J. F., Hawkes, J. P., Sheehan, T. F., Music, P. A., and Beland, K. F. 2009. Assessing estuarine and coastal migration and survival of wild Atlantic salmon smolts from the Narraguagus River, Maine using ultrasonic telemetry. American Fisheries Society Symposium, 69: 293–310.

Koutitonsky, V. G., and Bugden, G. L. 1991. The physical oceanography of the Gulf of St. Lawrence: a review with emphasis on the synoptic variability of the motion, *In* The Gulf of St. Lawrence: Small Ocean or Big Estuary? pp. 57–90. Ed. by J.-C. Therriault. Canadian Special Publication of Fisheries and Aquatic Sciences, 113: 359.

Kroglund, F., Rosseland, B. O., Teien, H.-C., Salbu, B., Kristensen, T., and Finstad, B. 2008. Water quality limits for Atlantic salmon (*Salmo salar* L.) exposed to short term reductions in pH and increased aluminum simulating episodes. Hydrology and Earth System Sciences, 12: 491–507.

Lacroix, G. L., Knox, D., and McCurdy, P. 2004. Effects of implanted dummy acoustic transmitters on juvenile Atlantic salmon. Transactions of the American Fisheries Society, 133: 211–220.

Lacroix, G. L. 2008. Influence of origin on migration and survival of Atlantic Salmon (*Salmo salar*) in the Bay of Fundy, Canada. Canadian Journal of Fisheries and Aquatic Sciences, 65: 2063–2079.

Lefèvre, M. A., Stokesbury, M. J. W., Whoriskey, F. G., and Dadswell, M. J. 2013. Migration of Atlantic salmon smolts and post-smolts in the Rivière Saint-Jean, QC north shore from riverine to marine ecosystems. Environmental Biology of Fishes, 96: 1017–1028.

Lunn, D., Jackson, C., Best, N., Thomas, A., and Spiegelhalter, D. 2013. The BUGS Book a Practical Introduction to Bayesian Analysis. CRC Press Taylor & Francis Group, LLC, Boca Raton, Florida. 381 pp.

Moore, A., Russell, I. C., and Potter, E. C. E. 1990. The effects of intraperitoneally implanted dummy acoustic transmitters on the behaviour and physiology of juvenile Atlantic salmon, *Salmo salar* L. Journal of Fish Biology, 37: 713–721.

Nieland, J. L., Sheehan, T. F., and Saunders, R. 2015. Assessing demographic effects of dams on diadromous dish: a case study for Atlantic salmon in the Penobscot River. ICES Journal of Marine Science, 72: 2423–2437.

Otero, J., Jensen, A. J., L'Abée-Lund, J. H., Stenseth, N. C., Storvik, G. O., and Vøllestad, L. A. 2011. Quantifying the ocean, freshwater and human effects on year-to-year variability of one-sea-winter Atlantic salmon angled in multiple Norwegian rivers. PLoS One, 6: e24005.

Parrish, D. L., Behnke, R. J., Gephard, S. R., McCormick, S. D., and Reeves, G. H. 1998. Why aren't there more Atlantic salmon (*Salmo salar*)? Canadian Journal of Fisheries and Aquatic Sciences, 55: 281–287.

Peyronnet, A., Friedland, K. D., Poole, R., O'Maoiléidigh, N., and Manning, M. 2007. Links between marine growth and survival of Irish Atlantic salmon (*Salmo salar*, L.) Journal of Fish Biology, 71: 691–700.

Riley, W. D., Moore, A., Russell, I. C., Davidson, I. C., Cove, R. J., and Ives, M. J. 2007. Impact of trapping and tagging on the timing of continued seaward migration of wild salmon, *Salmo salar*, smolts. Fisheries Management and Ecology, 14: 287–290.

Riley, W. D., Ibbotson, A. T., Gregory, S. D., Russell, I. C., Lauridsen, R. B., Beaumont, W. R. C., Cook, A. C. , *et al.* 2018. Under what circumstances does the capture and tagging of wild Atlantic salmon *Salmo salar* smolts impact return probability as adults? Journal of Fish Biology, 93: 477–489.

Royle, J. A. 2008. Modeling individual effects in the Cormack–Jolly–Seber model: a state-space formulation. Biometrics, 64: 364–370.

Sandstrom, P. T., Ammann, A. J., Michel, C., Singer, G., Chapman, E. D., Lindley, S., MacFarlane, R. B., *et al.* 2013. Growth, survival, and tag retention of steelhead trout (*Oncorhynchus mykiss*) and its application to survival estimates. Environmental Biology of Fishes, 96: 145–164.

St-Hilaire, A., Booth, D., Bettignies, C., Chadwick, E. M. P., and Courtenay, S. C. 1995. Is the Miramichi a stratified estuary? *In* Water, Science and the Public: The Miramichi Ecosystem, pp. 73–82. Ed. by E. M. P. Chadwick. Canadian Special Publication of Fisheries and Aquatic Sciences, 123: 300.

St-Hilaire, A., and Caissie, D. 2001. River mixing characteristics in the Northwest Miramichi River (NB) and associated metal concentration following a discharge by Heath Steele Mines in 1991. Canadian Technical Report of Fisheries Aquatic Sciences, 2367: vi + 23 pp.

Stich, D. S., Bailey, M. M., Holbrook, C. M., Kinnison, M. T., and Zydlewski, J. D. 2015. Catchment-wide survival of wild- and hatchery-reared Atlantic salmon smolts in a changing system. Canadian Journal of Fisheries and Aquatic Sciences, 72: 1352–1365.

Thorstad, E. B., Whoriskey, F. W., Uglem, I., Moore, A., Rikardsen, A., and Finstad, B. 2012. A critical life stage of the Atlantic salmon *Salmo salar*: behaviour and survival during the smolt and initial post-smolt migration. Journal of Fish Biology, 81: 500–542.

Thorstad, E. B., Uglem, I., Finstad, B., Kroglund, B., Einarsdottir, I. E., Kristensen, T., Diserud, O. , *et al.* 2013. Reduced marine survival of hatchery-reared Atlantic salmon post-smolts exposed to aluminium and moderate acidification in freshwater. Estuarine, Coastal and Shelf Science, 124: 34–43.

Welch, D. W., Batten, S. D., and Ward, B. R. 2007. Growth, survival and tag retention of steelhead trout (*O. mykiss*) surgically implanted with dummy acoustic tags. Hydrobiologia, 582: 289–299.

Handling editor: Caroline Durif

Fisheries and Oceans Canada
Canada

Pêches et Océans
Canada

Ecosystems and
Oceans Science

Sciences des écosystèmes
et des océans

Canadian Science Advisory Secretariat (CSAS)

Research Document 2022/029

Gulf Region

Fisheries Reference Points for Striped Bass (*Morone saxatilis*) from the Southern Gulf of St. Lawrence

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May 2022

Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

Published by:

Fisheries and Oceans Canada
Canadian Science Advisory Secretariat
200 Kent Street
Ottawa ON K1A 0E6

http://www.dfo-mpo.gc.ca/csas-sccs/
csas-sccs@dfo-mpo.gc.ca

ISSN 1919-5044
ISBN 978-0-660-43300-4 Cat. No. Fs70-5/2022-029E-PDF

Correct citation for this publication:

Chaput, G. and Douglas, S. 2022. Fisheries Reference Points for Striped Bass (*Morone saxatilis*) from the Southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2022/029. xiv + 153 p.

Aussi disponible en français :

*Chaput, G. et Douglas, S. 2022. Points de Référence de la Pêche du Bar Rayé (*Morone saxatilis*) du Sud du Golfe du Saint-Laurent. Secr. can. des avis sci. du MPO. Doc. de rech. 2022/029. xv + 159 p.*

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ABSTRACT

The Striped Bass (*Morone saxatilis*) population of the southern Gulf of St. Lawrence, the most northern spawning population of the species distribution in eastern North America, is widely distributed in estuaries and coastal waters of the southern Gulf of St. Lawrence from the north shore of the Gaspe Peninsula in Quebec to the northern tip of Cape Breton Island, Nova Scotia. For purposes of assessment and development of fisheries reference points, the southern Gulf Striped Bass population distribution comprises the Gulf of St. Lawrence region. Following on the sustained rebuilding of the spawner abundances from the lows of the late 1990s to the current high abundances that exceed 300 thousand spawners, DFO Gulf Ecosystems and Fisheries Management requested the development of fisheries based reference points that conform to the Precautionary Approach (PA) to guide further management decisions on the development of the Striped Bass fisheries. The extensive information on the abundance and biological characteristics of the Striped Bass population of the southern Gulf of St. Lawrence is presented. An age structured population model is used to estimate stock and recruitment parameters and associated mortality rates at age based on assessed abundances of spawners for the years 1996 to 2019. Equilibrium modelling is used to define candidate Limit Reference Point (LRP), Upper Stock Reference (USR), and removal rate references that would conform to the Precautionary Approach. Despite model uncertainties, a LRP value of just over 330 thousand spawners is consistent with one of the population model results as well as with the history of the management decisions for re-opening of fisheries access since 2013. The USR value of 720 thousand spawners would represent a healthy condition for this population, based on the assessed spawner abundances to 2019 and on the potential productive capacity of this population. A number of knowledge gaps and uncertainties remain. The most important assessment and management gap is the incomplete to non-existent catch statistics for any of the Striped Bass fisheries in the southern Gulf of St. Lawrence, including Indigenous Food Social and Ceremonial fisheries and the larger recreational fisheries. In the absence of these catch and harvest data, it is not possible to provide fisheries management advice in terms of total allowable catches nor can the status of the population relative to removal rates be assessed. Striped Bass is a predator of other valued anadromous fisheries species in the southern Gulf of St. Lawrence. The reference points presented are derived based on optimizing value functions specific to Striped Bass. No multi-species reference points or management options are discussed.

1. INTRODUCTION

Striped Bass (*Morone saxatilis* Walbaum, 1792; Order Perciformes; Family Percichthyidae) is widely distributed throughout the estuaries and coastal waters of the southern Gulf of St. Lawrence (southern Gulf), from the north shore of the Gaspe Peninsula in Quebec to the northern tip of Cape Breton Island, Nova Scotia. The spawning population in the southern Gulf of St. Lawrence is at the northern extent of the species distribution (Figure 1.1).

Genetic analyses and conventional tagging studies have indicated that this population is geographically isolated within the southern Gulf of St. Lawrence and distinct from any other Striped Bass population, including the only other remaining Canadian population which spawns in the Shubenacadie River, Nova Scotia (Bradford et al. 2001a; COSEWIC 2004; Wirgin et al. 1993, 2020).

Previous to 2017, the extent of occurrence of the southern Gulf of St. Lawrence Striped Bass population was assumed to have been restricted to the southern portion of the Gulf of St. Lawrence (COSEWIC 2012). In 2017, an extraordinary expansion of Striped Bass into previously undocumented areas along the north shore of the St. Lawrence and into southern Labrador was noted (DFO 2018; Valiquette et al. 2018; Figure 1.2). The potential distribution of the southern Gulf Striped Bass population is now considered to occasionally extend into those northern areas and the estuary of the St. Lawrence River. Striped Bass sampled from the Bras d'Or Lake and Mira River areas of eastern Cape Breton have been shown to be genetically similar to Striped Bass from the southern Gulf of St. Lawrence (Bentzen, P., Mcbride, M., and Paterson, I.G. 2014. Report: Genetic analysis of Striped Bass collected in Bras d'Or Lake. Report to the Eskasoni Fish and Wildlife Commission; referenced in LeBlanc et al. 2020), however it is unknown if this is due to the contemporary migration of southern Gulf of St. Lawrence Striped Bass or due to other speculated factors that would have isolated the two groups of fish (Andrews et al. 2019a).

Striped Bass juveniles (age-0) originating from the Miramichi River were used in a re-introduction program in the St. Lawrence River beginning in the late 1990s. Successful spawning and recruitment from this program has been confirmed (DFO 2017). Tracking studies of acoustically tagged Striped Bass from the St. Lawrence group and from the southern Gulf of St. Lawrence group as well as differences in elemental composition of the otoliths of bass spawned in Miramichi and in the St. Lawrence River have indicated a general geographic isolation of the two groups. The St. Lawrence progeny are generally restricted to the St. Lawrence River itself (at least to date) whereas the Miramichi origin fish have a broader distribution, that extends into the estuary of the St. Lawrence and to the lower north shore of the St. Lawrence (Valiquette et al. 2017; Valiquette et al. 2018).

For purposes of assessment and development of fisheries reference points, the southern Gulf Striped Bass population distribution comprises the Gulf of St. Lawrence region, from the western tip of Cape Breton Island to the north shore of the Gaspe Peninsula in the St. Lawrence River and it is managed as a single biological unit.

Descriptions of Striped Bass biology and life history abound (COSEWIC 2004) and the following summary for the population of the southern Gulf is primarily taken from Douglas et al. (2003) and Douglas and Chaput (2011b).

- Striped Bass is a relatively long-lived iteroparous spawner.
- The Northwest Miramichi River estuary is the only confirmed spawning location that is annually predictable in time and space (Bradford and Chaput 1996; Robichaud-LeBlanc et al. 1996) and that has produced annual recruitment in the southern Gulf of St. Lawrence.

The Northwest Miramichi estuary possesses features that are seemingly unique and important for successful Striped Bass spawning in the southern Gulf of St. Lawrence but these are not well understood. The favourable conditions may be related to the Northwest Miramichi estuary's specific hydrology and conditions that permit the retention and successful egg and larval development.

- Spawning occurs in late May to early June in the upper estuary, at the upper extent of the salt wedge within tidal waters, of the Northwest Miramichi River, (Robichaud-LeBlanc et al. 1996; Douglas et al. 2009). Spawning activities are motivated by warming temperatures (Douglas et al 2009; Figure 1.3).
- Striped Bass is a pelagic spawner, the eggs and milt are broadcast simultaneously into the water column.
- The eggs float freely, are generally neutrally buoyant in slight saline water, and hatch in a few days depending on water temperature.
- The yolk of young larvae is exhausted within 5 to 10 days post-hatch, also conditional on temperature.
- The larvae feed on planktonic organisms (Robichaud-LeBlanc et al. 1997) and move to the near shore shallow areas of the rivers shortly after the onset of exogenous feeding.
- Young of the year Striped Bass gradually migrate downstream to Miramichi Bay in the summer and diffuse in a northwest and easterly direction from the Miramichi (Robinson et al. 2004). The confirmed coastal distribution of young of the year by the first autumn can extend from Miscou Island (NB) in the north to Pictou (NS) in the east (Douglas and Chaput 2011b).
- Growth of young of the year is quite fast, with individuals reaching of 8 to 15 cm fork length and whole weights of 10 to 50 g, by the end of the first summer (Bradford et al. 1997; Robichaud-LeBlanc et al. 1998).
- Post-spawned adults return to marine waters and undertake coastal feeding migrations through the summer and autumn, extending in some exceptional years such as in 2017 to the north shore of the St. Lawrence and to southern Labrador (DFO 2018).
- Striped Bass are generalist feeders with shifts in prey composition occurring with age and size. Larger bass are known piscivores, and consume a wide range of invertebrate and vertebrate prey. Striped Bass sampled from the spawning areas in the Northwest Miramichi consume anadromous species (Rainbow Smelt, gaspereau, Atlantic Salmon smolts) based on availability determined by timing of migrations into and out of the Miramichi (DFO 2016; Hanson 2020).
- At the onset of winter, beginning in late September to October, Striped Bass of all age and size groups re-ascend into estuaries and river mouths throughout the southern Gulf to overwinter.
- The southern Gulf of St. Lawrence population is the only population where avoidance of lethal marine conditions (sub-zero water temperatures) during winter is an obligate element of its life history and this can only be attained by overwintering in upper estuaries and river mouths (Cook et al. 2006). A literature review of locations and characteristics of overwintering habitat for Striped Bass is provided in Andrews et al. (2019b).
- In 2004, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) recognized the Striped Bass of the southern Gulf as a designatable unit (DU) and evaluated its status as 'Threatened' (COSEWIC 2004).

- Efforts to rebuild from the low spawner abundances of the mid 1990s included the introduction of restrictive fisheries management measures, most notably the closure of directed commercial fishing in 1996, and the closure of recreational and Aboriginal food, social, and ceremonial (FSC) fisheries in 2000.
- The modest increase in spawner abundance since then suggested that the management interventions had been positive for the population. In its re-evaluation in 2012, COSEWIC concluded that although it had increased strongly in abundance, it was known from only a single spawning location and the population continued to be susceptible to high rates of poaching as well as bycatch in legal fisheries, and consequently was given the status of Special Concern (COSEWIC 2012).

1.1. OBJECTIVES OF THE SCIENCE PEER REVIEW

The Striped Bass population of the southern Gulf of St. Lawrence had declined to less than 5,000 spawners in the late 1990s. Following on the prohibition of retention of bycaught Striped Bass in several commercial fisheries targeting other diadromous species in 1996, the closure of the recreational fisheries and the suspension of Indigenous Food, Social, and Ceremonial (FSC) fisheries allocations for Striped Bass in 2000, the estimated abundance of Striped Bass subsequently increased to over 200 thousand spawners in 2011 with peak abundance estimated at over 900,000 spawners in 2017 (DFO 2020). A small number of FSC fisheries were reinstated in 2012. The recreational fishery reopened in 2013 with increasing annual access for retention and a pilot Indigenous commercial fishery was licenced in 2018 and 2019.

With continued requests for additional fisheries access to southern Gulf Striped Bass, Fisheries and Oceans Canada (DFO) Gulf Ecosystems and Fisheries Management Branch requested the development of fisheries based reference points that conform to the Precautionary Approach (PA) to guide further management decisions on the development of the Striped Bass fisheries.

Striped Bass is large bodied and a piscivorous predator through most of its life. Concerns have been expressed by Atlantic Salmon fishery advocates and some gaspereau and Rainbow Smelt commercial fishery interests that the rebuilding of Striped Bass stock in the southern Gulf has contributed to declines in abundances of Atlantic Salmon and other diadromous species because of high levels of predation on these species by Striped Bass. Considering the interactions of Striped Bass with other valued fisheries species, DFO Fisheries management also requested a review of approaches and potential reference points for Striped Bass that take account of these ecosystem considerations.

The specific objectives of the science peer review are to:

- Review the available information on the abundance and biological characteristics (size at age, mortality rate estimates, size structure) of the Striped Bass population of the southern Gulf of St. Lawrence relevant for the definition of reference points;
- Review candidate fishery reference points for Striped Bass and provide estimates of these based on the available information from the southern Gulf population;
- Review and advise on the consequences of fishery management measures on the derivation of fishery reference point values;
- Consider options for incorporating species interactions considerations in the definition of reference points for Striped Bass; and
- Consider uncertainties in the definition of the reference points and management approaches for Striped Bass.

1.2. ORGANISATION OF THE DOCUMENT TO ADDRESS THE TERMS OF REFERENCE

This document is organized to sequentially to address the terms of reference.

Section 2 provides an overview of the history of fisheries for Striped Bass in the southern Gulf, with an emphasis on the management measures and fisheries situation since the re-opening of access to the resource in 2013. Particular challenges to the compilation of fisheries catch and effort data are described. Additional details on the fisheries are provided in Appendix 1.

Section 3 summarizes the assessment program and the estimates of total spawner abundances and abundances at age of spawners on the Northwest Miramichi River spawning area for the period 1994 to 2019. Information on the biological characteristics of the population are provided, including size-at-age, weight-length relationship and weight-at-age, estimated abundance of spawners at age, maturity-at-age and proportion female at age on the spawning grounds, as well as estimates of mortality-at-age and overall. Details on the size-at-age analyses and derivation of an age-length key to convert abundance of spawners at length to abundance of spawners at age are provided in Appendix 2. The biological characteristics information is used in the population modelling in section 4.

Section 4 describes the age-structured population model which was used to estimate important population dynamics parameters which are required to derive candidate reference points. The population model uses as input the estimated abundances at age from the assessments in 1996 to 2019 to make inferences on stock and recruitment parameters, mortality rates at age, and proportion of recruits that become spawners. Seven variants of the basic age-structured model are examined, with differing informative assumptions on the life history parameters and exploring different stock and recruitment functions. The input data are presented in Appendix 3, the model codes for three of the seven models are in Appendix 4, and the detailed diagnostics of the retained models are provided in Appendix 5.

Section 5 reviews some candidate reference points and describes the methods used to define these candidate reference points based on the outputs from the population model in section 4. Equilibrium approaches, which simulate population abundance trajectories based on estimated and fixed life history parameters, are used to compare abundance, age structure, and fisheries yields for different levels of fishery exploitation. Concepts of maximum sustainable yield and spawner per recruit and their associated reference points are described. Empirical driven methods that rely exclusively on past observations are also described as alternatives to model dependent approaches for defining reference points.

Section 6 describes the results of the equilibrium modelling and the corresponding values for the candidate reference points. This section also addresses the question of how the values of the reference points are modified by the assumptions on natural mortality, on the fishing management strategy when these include length based limits on retention, and the inclusion or exclusion of catch and release mortality considerations when estimating yield based reference points. The section also provides a summary of the conclusions on reference points and introduces the issue of management reference points that account for species interactions. Details on this latter point are provided in a separate document (Chaput 2022).

Section 7 addresses the uncertainties associated with the derivation of reference points for the Striped Bass population of the southern Gulf of St. Lawrence. The uncertainties discussion includes aspects of life history including size-at-age, maturation and in particular mortality rates. For mortality rates, we consider the evidence for the causes of mortality of Striped Bass, including fisheries derived, anthropogenic, and other sources of natural mortality. Other uncertainties discussed include the assumptions on the density dependent stock and recruitment relationship and considerations on the choice of models and the time series of abundance estimates that are available for characterizing the productive potential of this population.

The references cited in this report are provided in section 8.

2. FISHERIES ON STRIPED BASS

Striped Bass have been exploited in numerous fisheries of the southern Gulf of St. Lawrence for over a century of records. Catches of Striped Bass dating to 1868 and onward are available in annual reports of the Department of Marine and Fisheries but these have not been compiled for this report. Compiled annual commercial catch records for Striped Bass date from 1917 (LeBlanc and Chaput 1991) but these only account for reported commercial catches. There is an absence of reported landings from the southern Gulf of St. Lawrence during the period 1933 to 1968. This is not interpreted to be a period without harvests however, as numerous regulatory changes were made during that period to Striped Bass fisheries that likely impacted the fishing activities (Appendix 1); for example in 1949, an amendment was made to the Special Fishery Regulations for the province of New Brunswick effectively closing the commercial fishery by authorizing the retention of Striped Bass only in angling fisheries. This was followed by an amendment in 1960 that authorized the sale of Striped Bass incidentally captured in nets , traps, or weirs set for catching fish other than Striped Bass.

In 1993, the Nova Scotia Fishery Regulations, the New Brunswick Fishery Regulations, and the Prince Edward Island Fishery Regulations were revoked and replaced with the Maritime Provinces Fishery Regulations that specified regulations specific to fishing in the three Maritime provinces and in adjacent tidal waters. Of note in this amendment are the regulations specific to fishing for Striped Bass in the waters of DFO Gulf Region (Tables 2.1, 2.2; Appendix 1).

In 1996, Paragraph 4(2)b of the Maritime Provinces Fisheries Regulations which permitted the retention of unlimited bycatch of Striped Bass in commercial fishing gears for gaspereau, Rainbow Smelt, American Shad, and American Eel was repealed (Canada Gazette Part II, Vol. 130, No. 5; SOR/96-125).

Subsequent modifications to the Striped Bass fisheries management of the southern Gulf were made via licence conditions (for commercial fisheries) and variation orders for recreational fisheries. Additional restrictions to various fisheries interacting with Striped Bass were introduced from 1996 to 2000 which culminated in the closure of all legal Striped Bass fisheries (Table 2.1).

In addition to the directed fishery management measures, short-term closures to directed recreational fisheries in the spawning area of the Northwest Miramichi to preclude harm to spawning fish were instituted since 2017 (Table 2.3). The temporary closure to all recreational fisheries of the spawning area in the Northwest Miramichi during the peak spawning period was previously identified as one of several management measures that would enhance the protection of Striped Bass and promote its recovery (Appendix 1).

Although the fisheries on Striped Bass were essentially closed in 2000, Striped Bass of various life stages continued to be intercepted in a variety of illegal, commercial, and Indigenous FSC fisheries although the extent of these losses to the population is unknown (Chiasson et al. 2002; Douglas et al. 2006; DFO 2011). DFO (2011) indicated that Striped Bass of various life stages continued to be intercepted in a variety of illegal fisheries, commercial fisheries, and aboriginal FSC fisheries, with a total estimated loss of medium and large sized Striped Bass in all southern Gulf of St. Lawrence fisheries in the range of 60,000 fish per year. The total number of bass handled in the fisheries was estimated to be 152,000 fish, of which 41% were estimated to have died or been killed (DFO 2011). The activity with the greatest contribution to the total loss of Striped Bass is considered to be the illegal fishery, accounting for over 50% of the estimated adult losses, followed by the recreational fishery (illegal retention and bycatch) at about 15% (DFO 2011).

As abundance was estimated to have increased almost monotonically since the late 1990s, a number of food, social, and ceremonial (FSC) fisheries were reinstated in 2012 (Table 2.1). The recreational fishery reopened in 2013 and a pilot Indigenous commercial fishery was licenced in 2018 and 2019 (Table 2.2).

Striped Bass originating from the southern Gulf are also exploited in fisheries along the coast of Chaleur Bay and around the Gaspe Peninsula in Quebec. Fisheries management measures for the recreational Striped Bass fishery in Quebec, similar to the fisheries management measures in DFO Gulf Region, were introduced in 2013 (Table 2.2). Based on elemental composition analyses of otoliths and different characterizations of these signatures in Striped Bass originating from the Miramichi River and from the St. Lawrence River spawning areas, Valiquette et al. (2018) indicated that the southern Gulf of St. Lawrence Striped Bass distribution extended around Chaleur Bay and upstream along the Gaspe peninsula to Rivière du Loup. Occasionally, as noted in the samples of Striped Bass from 2017, southern Gulf bass were also distributed along the lower north shore of the St. Lawrence River (Valiquette et al. 2018). Tag returns of bass marked in the southern Gulf and reports of the presence of Striped Bass in southern Labrador in late summer and into the winter (DFO 2018) as well as detections of acoustically tagged Striped Bass on the receiver line at Port Hope (Labrador; Figure 1.2) confirmed the broader excursion of southern Gulf Striped Bass outside its historic range in 2017 and its exploitation in various fisheries in and outside (north) of the Gulf of St. Lawrence.

2.1. FISHERIES EFFORT AND CATCH STATISTICS

There are no complete fishery catch data for Striped Bass in the southern Gulf of St. Lawrence. Historically, fisheries statistics included only commercial harvests, exclusive of recreational and Indigenous peoples fisheries harvests. LeBlanc and Chaput (1991) summarize the reported landings of Striped Bass from the southern Gulf of St. Lawrence for the period 1917 to 1988 (Table 2.4). Peak recorded harvest was 61.4 t in 1917. There were no recorded landings for the years 1935 to 1967. Peak recorded landings in the second period of records after 1967 was 47.8 t in 1981 with 15.25 t recorded in the last year (1996) of authorized commercial landings. Detailed reported commercial harvests by statistical districts in DFO Gulf NB as well as by season and regions for the contemporary period of the fishery are provided in Bradford et al. (1995a) and Douglas et al. (2003).

Striped Bass are particularly vulnerable to capture in several fisheries in estuaries of the southern Gulf of St. Lawrence. Unregulated and directed commercial fishing up to March 1996 was attributed to have been the principal factor for the reduction in spawner abundance between May 1995 and May 1996. An estimated 14.5 t of Striped Bass were recorded harvested during January and February 1996 from the Richibucto district of New Brunswick, most likely taken in bow-net and gillnet fisheries under the ice (Bradford and Chaput 1998). Within the Miramichi system 12,300 bass were estimated to have been removed, and added to an estimated 18,800 bass (17.3 t) reported as landed and sold in districts other than the Miramichi River, the total removals were estimated to have been in excess of 40,000 fish representing 80% of the estimated spawning stock of Striped Bass in 1995 (Bradford and Chaput 1998).

The Indigenous pilot commercial fishery for Striped Bass in the Miramichi River was conducted in 2018 and 2019. The total allowable catch (TAC) was set at 50,000 fish (50-65 cm TL limit) in

2018 and 50,000 fish (50-85 cm TL limit) in 2019. Privacy rules preclude the reporting of harvests from this fishery in this report but DFO Fisheries Management indicated that the harvests were substantially below the TAC in both years.

There are no compiled reports of catches and harvests of Striped Bass in the Indigenous FSC fisheries in the southern Gulf.

In addition, young of the year (YOY) Striped Bass are susceptible to capture in the openwater fall fishing gears (boxnets and gillnets) set for Rainbow Smelt (Bradford et al. 1995b, 1997). The bycatch in the Miramichi fisheries was most important in the last half of October. Interceptions of YOY bass were estimated to have been in the hundreds of thousands annually, in the Miramichi River alone, most of which would be dead given the difficulty to sort and release them alive from the large quantities of fish captured in these fisheries (Bradford et al. 1995b, 1997). Bycatch of YOY striped bass were also reported in the Tabusintac and Richibucto River fisheries. The opening of the fall openwater smelt fishery in the Miramichi was delayed from Oct. 15 to Nov.1 in 1999.

2.2. RECREATIONAL FISHERY CATCH AND HARVEST ESTIMATES

Since the re-opening of the recreational fisheries in 2013, partial catch data from the recreational fishery for some geographic areas of the southern Gulf and in some years have been collated but they are very incomplete.

2.2.1. Year 2013

Estimates of caught and retained Striped Bass in the Miramichi River and in the southern Gulf of St. Lawrence during the two retention periods of 2013 are reported by DFO (2014) and summarized in Table 2.5. The creel survey was conducted exclusively in the Miramichi River area during the May 1-15 retention period. The estimates are considered incomplete because interviews were from incomplete fishing trips, the survey only covered a portion of the 15-day season, and not all Miramichi fishing locations nor all times of the day were surveyed (DFO 2014). Of note, DFO (2014) indicated that individual anglers reported single trip catches of Striped Bass ranging from 0 to as high as 120 fish per trip, highlighting the potential for high catch rates realized in May in the Miramichi and the extensive catch and release activities in the recreational fishery.

The estimates for the second retention period in August 2013 are also considered to be underestimates of catch and retained bass (Table 2.4). Only a few (8) of the large number of access points (bridges, wharves, public beaches etc.) along the shore of the southern Gulf were surveyed, the survey only covered the retention period in August at obvious access points and during the daily open period (two hours before sunrise, two hours after sunset) and little to none of the effort from shoreline or boats was measured in the survey. Based on the available information, and assuming a 10% hook and release mortality, there were more losses attributed to catch and release mortality then retentions although the catch and release losses occur over the entire size range of bass angled whereas the retention losses were for a slot size (DFO 2014).

2.2.2. Year 2014

In 2014, a survey was again attempted in the Miramichi River area during the May retention period. Catches of Striped Bass were again considered underestimated (Table 2.4) given that interviews only covered a portion of the 25-day season (DFO 2015a). As was the case during the 2013 fishery, catches of Striped Bass in single trips by individual anglers ranged from 0 to 111 fish per trip, with large variation in catches and success rates (DFO 2015a). During the

August and September 2015 retention periods, DFO Conservation and Protection officers conducted 434 individual interviews and documented a total harvest of 58 Striped Bass and 455 released fish. Insufficient coverage precluded the extrapolation of interviewed catches to a total for these retention periods. Angling data was also obtained from mail-in cards and a self-reporting website in 2014 (DFO 2015a). There were a low number of overall returns. For the 91 self-reporting web entries, it was indicated that 1,560 Striped Bass were released and 40 fish were retained. The data cannot be used to estimate the total catches and retentions however it does illustrate the extent of fishing activity that occurred in 2014, with a point estimate of 16 fish released per angler and with less than half the anglers retaining one Striped Bass.

The province of Quebec conducted creel surveys in 2014 at fisheries access points along the north shore (Quebec portion) of Chaleur Bay. A total of 766 interviews were completed in 2014 (DFO 2015a) resulting in an estimated total catch (released and retained fish) of 9,010 fish (5,370 to 12,650 95% confidence interval) and an estimated retention of 554 fish (299 to 809; Table 2.5). Data also included the proportion of the retained catch by size group and the proportion of the estimated released fish by size group (Table 2.6).

2.2.3. Year 2015

No creel surveys of the recreational fishery for Striped Bass in the southern Gulf of St. Lawrence were conducted in 2015.

The province of Quebec conducted creel surveys in 2015 at fisheries access points along the north shore (Quebec portion) of Chaleur Bay (Table 2.5). The estimated catches from fishing effort at the survey points in 2015 were 1,172 fish retained, 20,797 fish released with a point estimate of total losses (including catch and release mortalities) of 3,252 fish.

2.2.4. Year 2016+

No creel surveys of the recreational fishery for Striped Bass in the southern Gulf of St. Lawrence have been conducted since 2014.

Since 2016, the province of Quebec has conducted a limited survey of angling activities at four sites within two sectors during an eight week period, beginning on 1 July. Indicators of angling activity included the number of anglers per sampling unit (time, site), fishing trip duration, rate of success, probability of retention of at least one fish, and distribution of catches within length categories. The indicators of fishing success and distribution of sizes in the catches are summarized in Table 2.6.

3. ASSESSMENT AND BIOLOGICAL CHARACTERISTICS OF STRIPED BASS OF THE SOUTHERN GULF OF ST. LAWRENCE

Since 1994, monitoring of the bycatch in the commercial gaspereau trapnets of the Miramichi River has been the principal source of information for the estimation of the Striped Bass spawning population of the southern Gulf of St. Lawrence (DFO 2020). Selected biological characteristics (e.g. fork length, age, sex, and spawning stage) were recorded from fish captured in commercial gaspereau trapnets (May and June) and at index trapnet monitoring facilities operated by DFO Science (May-October). Ages are interpreted from scales.

The spawner abundance was usually estimated from mark and recapture experiments in which adult Striped Bass were tagged early in May and monitored throughout June as they were captured and released as bycatch in the gaspereau fishery of the Northwest Miramichi Estuary (Bradford and Chaput 1996; Douglas and Chaput 2011a). Catch per unit effort (CPUE) from this fishery has been used as an index of abundance for Striped Bass (Douglas and Chaput 2011a;

Figure 3.1) and estimates of catchability of the gear are used to derive the estimates of abundance. Since 2014, an adjustment to the estimation model has been made to account for the observed spawning and post-spawning behaviour of Striped Bass, using movement data of Striped Bass implanted with internal acoustic tags. The tracking of acoustically tagged Striped Bass provided information on the daily distribution of spawners in the Miramichi system and therefore their availability to the gaspereau trapnets of the Northwest Miramichi (DFO 2020).

Estimated abundances of bass spawners in the Northwest Miramichi were at or under 5,000 spawners (median) during 1996 to 2000 (DFO 2020; Figure 3.2). The decreased abundance from 60 thousand fish in 1995 to the 1996 estimate of just over 5,000 fish was largely explained by estimated removals of about 30,000 adults through unregulated and direct commercial fishing activities between May 1995 and March 1996 (Bradford and Chaput 1997). Abundance increased to between 16,000 and 26,000 during 2001 to 2006 and again to between 50,000 and 100,000 fish during 2007 to 2010. Abundances of 150 thousand to 300 thousand were estimated during 2011 to 2016 with a peak abundance in 2017 at just under 1 million fish (Figure 3.2). Striped Bass spawner abundance in 2018 and 2019 was estimated to have fallen back to approximately 300 thousand spawners.

Coincident with the high level of abundance in 2017, evidence from tag returns indicates that a component of the southern Gulf Striped Bass population migrated further north in 2017 than previously known, extending into southern Labrador (DFO 2018). In 2017, nine acoustic tag detections at the Port Hope (southern Labrador) acoustic receiver line were attributed to Striped Bass (Table 3.1). Of these, seven Striped Bass had a previous overwintering and / or spawning history in the Miramichi. Exposure to new sources of fishing mortality occurred for southern Gulf Striped Bass that migrated north in 2017 as reported by interceptions of several tens of thousands of pounds of Striped Bass in commercial gear set for cod, in herring nets and halibut trawls along the south coast of Labrador (DFO 2018). Only 3 of the 7 acoustically tagged bass detected in Labrador with a previous recorded affinity to the Miramichi were detected in the Miramichi in the winter of 2017/18, a loss of 57% of the original detections off Labrador. Losses of Striped Bass that had migrated outside the historic range to the Quebec north shore and Labrador in summer and fall 2017 may in part explain the reduced estimated abundance of Striped Bass on the spawning grounds in 2018 and 2019 relative to 2017 (DFO 2020).

3.1. AGE AND SIZE AT AGE

Ages of Striped Bass are interpreted from scales. Size-at-age has been reported previously by Chaput and Robichaud (1995) and in Douglas et al. (2006). Sampling and age determination has occurred opportunistically. There has not been any age validation nor is a reference scale set available for doing reader tests. Tagging and subsequent recaptures of tagged fish provide some information on changes in fork length over multiple years, but these are not reported here.

Striped Bass grow during the open-water season in the southern Gulf (May to October). No growth occurs through the winter when bass are overwintering and they do not feed under the ice in the upper areas of estuaries; this is evident from an examination of size distributions of bass sampled in the fall in the Miramichi at DFO index trapnets which are identical to those of bass sampled the following spring in the Miramichi (for example, see DFO 2020).

A total of 8,497 age and length data are available from sampling in the southern Gulf of St. Lawrence over all years between 1975 and 2013. From the samples available, maximum age interpreted is 15 years and maximum fork length recorded is 116 cm.

3.1.1. Von Bertalanffy Growth Model

A von Bertalanffy growth function was adjusted to the selected age and length data over all years:

$$L_a = L_\infty \left(1 - e^{-K(a - a_0)}\right) e^{\varepsilon}$$

with
L_a = length (cm) at interpreted age a,
L_∞ = predicted asymptotic length (cm),
K = predicted metabolic parameter,
a_0 = predicted apparent age at time of hatching, and
$n\varepsilon \sim N(0, \sigma^2)$.

Samples used for the von Bertalanffy model were restricted to those collected in May and June (n = 8,376), corresponding to the size at spawning time, and the start of the biological year (Table 3.2). No distinction is made between males and females.

The von Bertalanffy model parameters were estimated with OpenBugs using non-informative priors for the parameters (L_∞, K, a_0, σ) to be estimated (Lunn et al. 2013; Appendix 2). The posterior distributions of the parameters are summarized in Table 3.3 and a visualization of the data, model fits and predicted length distributions at age are presented in Figure 3.3.

3.2. SPAWNER ABUNDANCE AT AGE

Scale sampling and age interpretations are not available for all assessment years, nor are there sufficient samples of older and larger fish in any year to adequately estimate their relative abundances. There is information on the length distribution of spawners based on directed sampling by DFO Science from bycatches in the commercial gaspereau fishery and catches in dedicated science trapnets for Striped Bass assessment in the Northwest Miramichi (Figure 3.4). Consequently, the von Bertalanffy model predicted length at age distributions were used to derive an age length key which was then used to estimate the annual abundance at age of spawners (Figure 3.5) based on the assessed annual length distributions of the spawners (Figure 3.4) and the assessed total abundance of spawners (see Appendix 2 for details).

3.3. WEIGHT AT LENGTH RELATIONSHIP

A weight from length relationship was derived using data specific to the Striped Bass population of the southern Gulf of St. Lawrence. The most extensive data (N = 1,839) for whole weight (kg) and fork length (cm) were obtained from sampling during May and June, 2013 to 2015, related to the diet study of Striped Bass of the Miramichi River (Figure 3.6).

For purposes of the stock and recruitment equilibrium modelling, the coefficients of the relationship for sexes combined were used (Table 3.4).

3.4. FECUNDITY TO SIZE RELATIONSHIP

There is no southern Gulf specific fecundity to weight relationship. Data presented in Douglas et al. (2003) indicated that fecundity of Shubenacadie bass varied from 53,000 to 1.4 million eggs for bass ranging from 44.9 to 91.0 cm fork length. Goodyear (1985) presented fecundity at weight data for Striped Bass which translates to about 83,000 eggs per kg (see Figure 2 in Douglas et al. 2006). For purposes of modelling, a value of 83,000 eggs per kg was used (Douglas et al. 2006 used 83,177 eggs per kg). Based on the predicted mean length at age of bass from the Miramichi and the weight (kg) to length (cm) relationship, fecundity of an age 4 female bass (mean weight = 1.2 kg) would be 100,000 eggs whereas fecundity of age 15+ bass (mean weight = 7.1 kg) would be just under 600 thousand eggs.

3.5. MATURITY AT AGE, PROPORTION OF MATURE FISH ON SPAWNING GROUNDS

Three aspects of maturation and spawning of Striped Bass were considered by Douglas et al. (2006):

- There are no data with which to directly estimate the age or size at 50% maturity because no representative sampling of bass at age and maturation assessment is available. Based on studies elsewhere, the maturation schedule of male and female bass was assumed to differ, with males maturing earlier than females. Based on available samples of sex at age during May and June, there is evidence of higher proportions of males at ages 2 to 4 and more balanced sex ratios at ages 6 and older (Table 3.5). It was assumed that male bass first mature at age 3 years and female bass first mature at age 4 years, and all bass are mature by age 6 years (Douglas et al. 2006). This is supported by the observations of increased estimated abundances at ages 3 to 5 of spawners when following cohorts.
- Not all mature Striped Bass are considered to be on the spawning grounds in the Northwest Miramichi. This inference is based on reports of adult sized Striped Bass, some in ripe condition (males and females), in other estuaries of New Brunswick and Nova Scotia in May and June.
- There is also the possibility of skipped spawning in Striped Bass, particularly of larger fish. Rideout and Tomkiewicz (2011) review the evidence for and causes of skip spawning in fish, in which fish forego egg production until the subsequent year, as a potential plastic response of individual fish to low levels of stored energy or unsuitable environmental conditions. Secor (2008) and Gahagan et al. (2015) report on non-annual spawning of Striped Bass. Secor et al. (2020), using tracking of acoustically tagged Striped Bass, reported skip spawning percentages of 14-15%, with a higher percentage for bass in the year of tagging. The authors indicated that skip spawning could occur due to energetic constraints and seasonal movements and attributed the higher non-spawning behaviour in the year of tagging as the result of a residual tagging and handling effect.
- In 2017, nine acoustic tag detections at the Port Hope (southern Labrador) acoustic receiver line were attributed to Striped Bass (Table 3.1). Of these, seven Striped Bass had a previous overwintering and / or spawning history in the Miramichi. Of note, are the three Striped Bass acoustic tags detected in Labrador which were subsequently detected in the Miramichi (i.e., returned from Labrador) in the winter of 2017/18 and 2018/19 and the spawning that had occurred in the Miramichi in 2017, not in 2018, but spawning again in 2019, providing evidence of skipped spawning for those three fish.

Insights into the proportion female at age on the spawning grounds is available from the directed sampling as part of a diet study of Striped Bass in the estuary of the Miramichi River conducted during May and June of 2013 to 2015. Figure 3.7 shows the proportion female by cm fork length bin. Overlain on the plot are the 95% confidence interval range of the predicted fork length at age from the von Bertalanffy model. For bass less than 32 cm fork length, there is a varying but generally equal proportion of males and females in the samples; we interpret this as representative of immature fish. There is a low proportion female for bass ranging from 33 to 48 cm, roughly equivalent to age 3, increasing proportion female in the size range of age 4 bass with the proportion female levelling off at around 0.5 for size ranges of bass aged 5 and older (Figure 3.7).

The assumptions regarding the proportions mature at age and the proportions of mature bass on the spawning grounds result in estimates of the proportions of recruits by age, sexes combined, that are on the spawning grounds in the Miramichi. If the proportion of mature recruits present on the spawning grounds is the same for male and female bass at all ages, then the proportion female at age of spawners depends only on the ratio of the maturation schedules (Table 3.6).

3.6. MORTALITY

We assumed similar mortality at age for male and female bass.

3.6.1. Estimate of Natural Mortality of Ages 0 to 3

Estimates of natural mortality (M) for age-0, and ages 1 to 3 were derived using the empirical relationship published in Gislason et al. (2010) that relates instantaneous natural mortality rate to von Bertalanffy growth characteristics of the species. The equation derived by Gislason et al. (2010) is:

$$ln(M) = 0.55 - 1.61 * ln(L) + 1.44 * ln(L_\infty) + ln(K)$$

with
M = instantaneous natural mortality rate,
L = length of fish (mm),
L_∞ = predicted asymptotic length (mm) from von Bertalanffy growth function, and
K = metabolic parameter from von Bertalanffy growth function.

Based on the point estimates of L_∞ (907 mm), and *K* (0.1685) from the von Bertalanffy fit to the Striped Bass data (Table 3.2), estimated size at age of age 0 bass at the end of the growing season, and predicted mean sizes at ages 1 to 3 in mid-year (mean of $L_{a,t}$ and $L_{a+1,t+1}$), the model derived values of M for these age groups are summarized in Table 3.7.

Douglas et al. (2006) assumed an instantaneous rate (M) of 1.5 (survival = 0.22) for YOY in the first winter. Derivation of M based on the empirical relationship of Gislason et al. (2010) gives an M of 1.9. Mortality of young of the year bass in the first winter is expected to be high for this northern population. Size distribution of YOY bass in the fall, at the end of their first growing season, is annually variable with modal fork lengths varying between 9 and 15 cm (Bradford et al. 1997; Douglas et al. 2006; Figure 3.8). Chaput and Robichaud (1995) backcalculated fork lengths at age 1 (after the first winter) ranging from 10 to 15 cm depending on year class. Like adults, juveniles do not feed in the winter and no food items have been found in stomachs of juvenile bass sampled from the open water smelt fishery in November at low water temperatures (R. Bradford pers. comm.). The period of fasting likely extends from late October to late April in most years. There is limited empirical evidence that small bodied Striped Bass have a lower fitness than large bodied juveniles during the first winter. Some juvenile bass have been found frozen in surface ice in the Miramichi (Douglas pers. comm. or previous section). Variations in quantity of optimal habitat in the winter has been suggested as a possible factor contributing to variations in recruitment of the Hudson River striped bass population (Hurst and Conover 1998).

Douglas et al. (2006) had assumed that M for age 1 bass was 1.0, less than the overwinter mortality rate of YOY (1.5) but higher than the assumed value of 0.8 for age 2 bass. Values for M based on the empirical relationship of Gislason et al. (2010) and the mean size at age mid-season, are 0.82 for age 1 bass and 0.45 for age-2 bass.

Based on these values, the predicted cumulative survival rate from age-0 in the summer to age 3 is 0.039 ($\exp^{-(1.97+0.82+0.45)}$).

3.6.2. Mortality of Age 4 and Older Bass

3.6.2.1. Cohort Decline Analysis

Estimates of total mortality (Z) over age were calculated as the change in natural log of the assessed abundance at age of spawners by cohort:

$$\log(N_{y,a}) = \beta + Z * Age_a + \varepsilon;\ \varepsilon \sim N(0, \sigma_\varepsilon^2)$$

with
y the cohort,
a the age,
Z the slope of the natural log of the assessed abundance at age by cohort, and
β the intercept (log of abundance for the first age in the regression).

Z was calculated over ages 5 to 12 because it is assumed that Striped Bass from the southern Gulf of St. Lawrence are not fully mature until age 5 for males, age 6 for females and we wanted a sufficient number of cohorts in the time series to derive estimates of Z. Cohorts were retained for which there was a minimum of six available estimated abundances over the age range 5 to 12 years.

The estimated abundances at age and the estimates of Z for the 1989 to 2009 cohorts are shown in Figure 3.9. The absolute values of Z range from a low of 0.16 for the 2005 cohort to a high of 0.58 for the 1989 cohort. The 1993 cohort is the first fully assessed cohort for this population. For the fully assessed cohorts (cohorts 1993 to 2007 covering the full age range 5 to 12), the absolute values of Z ranged from 0.16 to 0.43, with a median value of 0.33.

Catch curve analyses reported in Douglas and Chaput (2011a) indicated that the total instantaneous mortality values (Z) ranged from a low of 0.08 to a high of 2.86 and corresponded to annual mortality rates of 7% to 94%. Year on year negative estimates of Z were frequent at age 3 and were not unexpected given the presumed maturity schedules for male and female bass at ages 3 to 5 resulting in partial recruitment to the spawning population of age-3 and age-4 bass. Based on the average abundance at ages 3 to 9 years over the period 1997 to 2010, the total mortality rate of adult Striped Bass was estimated at 0.47 (Z = 0.63; Douglas and Chaput 2011a), marginally lower than estimates of Z (0.8-0.9) and A (0.5-0.6) previously calculated for southern Gulf Striped Bass between the ages of 3 and 7 (Douglas et al. 2006).

Cohort decline analysis indicates variable but relatively high total mortality for Striped Bass aged 5 to 12; for the fully assessed cohorts (cohorts 1993 to 2007 covering the full age range 5 to 12), the absolute value of Z ranged from 0.16 to 0.43, with a median value of 0.33. The high mortality rate for the southern Gulf was considered consistent with the relative rarity of Striped Bass older than 10 years of age in the southern Gulf (Douglas et al. 2006).

3.6.2.2. Mortality inferred from tagging data

Acoustic tagging and tracking programs of Striped Bass conducted in 2003 to 2004, 2008 to 2009, and during 2013 to 2017 provide independent data to estimate annual mortality (converse survival) rates of adult Striped Bass to the Miramichi River. Striped Bass, ranging in size from 40.4 to 88.0 cm fork length (size data were not available for all tagged fish in all years) were tagged with acoustic transmitters and released from three locations: the Gaspe area (Quebec side of Chaleur Bay; MFFP Quebec), the Miramichi River, and a small effort from Pictou (Nova Scotia; C.F. Buhariwalla, pers. comm.). Both Vemco V13 and V16 acoustic tags were used with the majority of fish tagged with V16 tags. Anticipated battery life of the tags varied with tag type over years, and tag detections included in the survival estimates account for the expected battery life of the tags. Acoustic receivers were deployed throughout the Miramichi River and estuary year round (see Douglas et al. 2009 for details).

In this analysis, only sequential detections of tagged bass from acoustic receivers in the Miramichi River are used. It is assumed that fish detected in the Miramichi one year would be expected to return to the Miramichi the following year. Generally, survival rates are provided for the years after the year of tagging and corresponding to the open water period, i.e. survivals for the year 2017 are derived from fish tagged in 2016, that were detected in the Miramichi over the winter 2016/17 and again in the Miramichi over the winter 2017/18. The exception is for the bass tagged in 2003, 2004, 2008 and 2009; these fish were tagged and released in the spring and the survival estimates are derived from detections in the winter and spring of the following year, hence survivals correspond to the year of tagging and release.

Details on the number of bass tagged and subsequent detections, by location, tag type and size group at time of tagging are provided in Table 3.8.

3.6.2.3. Estimating the probability of survival

Over all tags available for detection, the probabilities of survival were estimated independently by tag group assuming a binomial distribution with a non-informative beta prior for the probability of survival:

$$N.tags(j,t) \mid N.tags(j,t-1), \emptyset_{j,t}, p \sim Binomial(\emptyset_{j,t} * p, N.tags(j,t-1))$$

$$\emptyset_{j,t} \sim Beta(a_{j,t}, b_{j,t});\ prior\ a_{j,t} = b_{j,t} = 1$$

with

parameter $\emptyset_{j,t}$ the probability of survival of tag group j over the period t-1 to t, and

p the probability of detection of acoustically tagged fish in the Miramichi.

Striped Bass return and overwinter in the upper portion of the Miramich River estuary and the probability of detection of these acoustic tags is considered to be 100%; total detections of individual tags generally totaled in the 100s or more.

These survival rate estimates include both natural and fishing mortality because these fish would have been vulnerable to legal and illegal fisheries over those years. To determine the extent to which survival rates in recent years may be size dependent and affected by the introduction of the retention size limit in the recreational fishery, we estimated and compared survival rates by size group for the year immediately after tagging, when the length of the fish would be expected to be most similar to their size relative to the size limits for the fishery. We also examined the survival rates over sequential years of fish in each size group, with the expectation that fish below the size limit would grow into the size limit and fish within the size limit at tagging would grow out of the size limit over time. Based on predicted fork length mid-season from von Bertalanffy model fits, the current retention size limit of 47 to 61 cm fork length in the recreational fishery results in selectivities to the recreational fishery of 0.12 for age 3 years, peaking at 0.76 to 0.79 at ages 4 and 5 years old and falling to 0.1 or less by age 10 years (Figure 6.3). Bass would be strongly selected by the fishery for two years but expected to grow through the slot over a period of 4 to 5 years.

3.6.2.4. Estimates of survival rate

Posterior distributions of the estimated probabilities of survival by tagging group (location, year of release, tag type) for sizes combined are shown in Figure 3.10. With few exceptions, annual survival rates are greater than 0.6. The estimated probabilities of survival (pooled values) were lowest during 2003 to 2009 and higher since 2014 (Figure 3.10, bottom panel).

The extent to which the estimated survival rates from tagged bass include fishing mortality is considered by examining survival rates by size group and sequential changes in survival rates for these groups (Figures 3.11, 3.12). Few bass of fork length less than the minimum retention limit were tagged in the recent years. There is an impression that survival rates of tagged bass within the retention size limit at time of tagging were lower than for bass which were outside the retention size slot, although there are notable exceptions such as the bass tagged in Gaspe for which estimated survival of bass in the slot size was better than for bass larger than the slot size for the 2014 year (Figure 3.11).

Estimates of instantaneous mortality rates (Z) ranged from 0.41 (median) during the period 2003 to 2009 to 0.22 for the period 2014 to 2018 (Figure 3.10). It is not possible to partition the natural mortality rates from fishing mortality rates with these data however considering that fishery removals would have in part contributed to the estimated mortalities, natural mortality of adult sized (> 47 cm) Striped Bass should be less than 0.2.

4. POPULATION MODELS

Estimates of key life history and population dynamics parameters are required to derive Maximum Sustainable Yield and other reference points. An age structured population model, as described in Walters and Martell (2004) and Walters et al. (2008) with an underlying stock and recruitment relationship (Beverton-Holt, power) is used to model the population dynamics of Striped Bass.

The time series of assessed abundance of spawners in the Miramichi and estimated abundances at age for the period 1996 to 2019 are used (Appendix 3; Figure 3.2). The data series begins in 1996 because prior to 1996, there was an active harvest of Striped Bass on the spawning grounds in the gaspereau fishery that was removing fish concurrent with the assessment program; the assessed population estimates for 1994 and 1995 are considered to be potential spawners rather than realized spawners. The same situation may apply since 2013 concurrent with the reopening of the Indigenous FSC fisheries and recreational fisheries, however, the harvest of Striped Bass during the assessment period (mid-May to mid-June) for those years is considered to be substantially less than what occurred prior to 1996.

4.1. MODEL SPECIFICATIONS

4.1.1. Model Equations

The life cycle population dynamic equations account for the estimated and/or assumed life history characteristics of the Striped Bass population of the southern Gulf. The beginning of the year is the spawning period, mid-May to mid-June, corresponding to the assessment period. The model assumes similar life history characteristics for male and female Striped Bass in terms of fork length-at-age, weight-at-age, and mortality-at-age.

The general model equations are described below. Modifications to these are made according to the model considered; those details are described specific to the model.

Recruitment (number) at age is calculated as:

$$N.0_y = \frac{\alpha * Eggs_y}{1+\frac{\alpha * Eggs_y}{K}} \text{ (Beverton-Holt) or}$$

$$N.0_y = \gamma * Eggs_y^{\beta} \text{ (Power function)}$$

with
$N.0_y$ = recruitment abundance (number) at age 0 in the summer in year y,
$Eggs_y$ = total eggs spawned in year y
α = Beverton-Holt density independent mortality rate (0,1),

K = Beverton-Holt asymptotic abundance of age 0 in the summer,
γ = survival rate (0,1) at the origin of the power stock and recruitment function, and
β = the density dependent compensatory survival rate of the power stock and recruitment function (if $\beta = 1$, recruitment is a proportion of eggs; if $\beta < 1$, recruitment is a decreasing proportion of increasing eggs; if $\beta > 1$, recruitment is an increasing proportion of increasing eggs).

$$N_{y+1,1} = N.0_y * e^{-Z.0}$$

with
$N_{y+1,1}$ = recruitment abundance at beginning of year y at age 1,
$N.0_y$ as defined above, and
$Z.0$ = instantaneous overwinter mortality rate of age 0,

$$N_{y+1,a+1} = N_{y,a} * e^{-(Z_a)} \text{ for a = 1 to 13}$$

with
Z_a = instantaneous mortality rate at age a

Age 15 is the oldest age and included as a plus group. Abundances of the plus group are calculated as:

$$N_{y,a} = N_{y-1,a-1} * e^{-(Z_{a-1})} + N_{y-1,a} * e^{-(Z_a)} \text{ for a = 15+.}$$

Spawner abundances (number) at age and total eggs are calculated as:

$$Sp_{y,a} = N_{y,a} * p.rec.sp_a$$

with
$Sp_{y,a}$ = abundance (number) of spawners of age a at beginning of year y,
$N_{y,a}$ = recruitment abundance of fish of age a at beginning of year y,
$p.rec.sp_a$ = proportion of mature recruitment at age a present on the spawning grounds.

$$Eggs_y = \sum_{a=3}^{A} Sp_{y,a} * p.fem_a * fec * u.Wt_a$$

with
$Eggs_y$ = total eggs spawned in year y calculated as the sum of eggs at age a, a = 3 to A (15+ group)
$Sp_{y,a}$ = abundance (number) of spawners of age a in year y,
$p.fem_a$ = proportion female of spawners at age a,
fec = 83,000 eggs per kg of female bass
$u.Wt_a$ = mean weight (kg) at age a (Figure 3.6; Appendix 3).

4.2. EGG TO YOY FUNCTIONAL RELATIONSHIP

We assumed that there is a density dependent compensatory function between eggs spawned and production of young-of-the-year (YOY) in the first summer (Goodyear 1985). We modeled this dynamic as a Beverton-Holt function (Hilborn and Walters 1992) or as an alternate power function.

The combination of high fecundity and iteroparity of Striped Bass are indicative of a species with high mortality in the early stages. Inter-year class variability in Striped Bass has been observed to be high, largely determined during the egg and larval stages and influenced by environmental factors (see references within Richards and Rago 1999; Uphoff 1989; Rutherford et al. 2003). Instantaneous daily rates of mortality (M d^{-1}) between the egg and the 8 mm larval stage have been estimated to vary between 0.11 and 0.34, with overall survival after 20 days varying between 0.03% and 11% (Rutherford et al. 1997). Increased juvenile production is not guaranteed by increased spawning stock but the chances of producing a strong year class are improved at high spawner abundances.

For the southern Gulf of St. Lawrence Striped Bass population, the life stage at which the carrying capacity limit is defined is assumed to be during the early juvenile (age-0, summer) stage as the habitat and food base for the larvae and post-metamorphosis juveniles is constrained to a relatively small tidal spawning and rearing area in the Northwest Miramichi (Robichaud-LeBlanc et al. 1996, 1997; Douglas et al. 2009). Cowan et al. (1993) contend that the year-class strength of Striped Bass is determined prior to metamorphosis (larval stage) as a combination of factors including maternal effects (larger females spawn more and larger eggs which contribute to larger larvae at hatch and better survival), prey abundance and quality.

Douglas et al. (2006) used a rate of 0.1% for survival to the end of the growing season for this population at the northern limit of the species distribution. Although there are no measures of absolute abundance of age-0 bass at the end of the first summer, the mean asymptotic abundance (K) was assumed to be in the order of a few million fish with 10s of millions of individuals possible for strong year classes (Douglas et al. 2006). Estimates of bycatch in the fall open water fishery of the Miramichi were over half a million fish in a year when spawner abundance was low (Bradford et al. 1997).

4.3. DATA

The data (observations) for model fitting are provided in Appendix 3. The observations include the assessed estimates of total spawner abundances and estimates of the number of spawners at age calculated from the assessed size distribution and an age-length key. Empirical data on weight-at-age and assumptions of maturation schedules by age for males and females are also shown in Appendix 3. Specifically, the observations for model fitting are:

- Assessed (median) total spawners (number) 1996 to 2019 (excluding 2012)
- Estimated abundance at age of spawners 1996 to 2019 (excluding 2012) based on:
    - Fork length distribution of spawners by year, 1996 to 2019 (excluding 2012), and
    - Age and length data to develop an age-length key based on von Bertalanffy growth model.

4.4. LIKELIHOODS

Lognormal likelihoods for abundance (number of fish) included:

- Median spawner abundance at ages 3 to 8 by year ($Sp.obs_{a,y}$) as

$$Sp.obs_{a,y+a} \sim LogN\left(log.\mu.Sp_{a,y+a}, log.\sigma_a\right) \text{ for a = 3 to 8, y = 1996 to 2019-a.}$$

with
$log.\mu.Sp_{a,y+a}$ the predicted mean (natural log scale) abundance of spawners age *a* in year *y+a*.

The sequence *y+a* is used for the appropriate cohort link; the 1996 cohort (1996 spawning) is first observed as 3-year olds in 1999, 4-year olds in 2000, etc. By age 8, the cohorts included in the model are 1996 to 2011. In all cases, the 2012 data are missing (but the missing data are included in the likelihood).

- Median total spawner abundance (age 3 to 15+) by year ($Sp.tot.obs_y$) as

$$Sp.tot.obs_{y+12} \sim LogN\left(log.\mu.Sp.tot_{y+12}, log.\sigma_{sp.tot}\right) \text{ for y = 1996 to 2008.}$$

with
$log.\mu.Sp.tot_{y+12}$ the predicted mean (natural log scale) total abundance of spawners, ages combined, year *y+12*. In this case, the sequence y+12 corresponds to the predicted spawners for the 2008 to 2019 assessment years. Although the 2008 (1996+12) to 2010 assessment years include spawners at ages 13 to 15+ from the 1993 to 1995 cohorts for which there are no originating spawner abundances (hence resulting from sequential survivals from initial abundances at age 3 in 1996 to 1998 and unrelated to the stock and recruitment function), the percentage of these age groups to total spawners in any of those years is small (< 1%) and considered to have minimal consequence on the likelihood.

4.5. INITIAL YEAR 1996

Estimated recruitment at age and spawners at age for the first year, 1996, are derived directly from the assessed and estimated spawner abundances at age in 1996.

Recruitment at age was estimated as:

$$N_{1996,1} = \frac{Obs.sp_{1996,3}}{p.rec.sp_3} * e^{(Z_1 + Z_2)}$$

$$N_{1996,2} = \frac{Obs.sp_{1996,3}}{p.rec.sp_3} * e^{(Z_2)} \text{ and}$$

$$N_{1996,a} = \frac{Obs.sp_{1996,a}}{p.rec.sp_a} \text{ for a = 3 to 15+.}$$

Total spawners, total eggs, and recruitment at age 0 are as defined above.

Depending on the model, Z_1 and Z_2 above are either given informative priors or are not used because the life cycle transition goes directly from age-0 to age-3 (Model 5) or from eggs to age-3 (Model 6, 7).

For models 5, 6 and 7 described below, the predicted recruitment abundance at age 3 is derived from either eggs or age-0 recruitment in year-3. Therefore, initial values for age-3 recruitment for 1997 and 1998 are derived from the assessed spawner abundances at age 3 for those years adjusted by the proportion of recruitment that become spawners at age 3 (as was the case for age 3 in 1996).

$$N_{y,3} = \frac{Obs.sp_{y,3}}{p.rec.sp_3} \text{ for y = 1997 and 1998.}$$

4.6. MODEL VARIANTS

Seven age-structured life cycle models with differing assumptions and parameters to be estimated were examined. Some life history characteristics (mean weight-at-age, proportion female at age of spawners, eggs per kg of spawner) were set at fixed values in all models. For the other life history parameters (Beverton-Holt stock and recruitment parameters, survival, proportion of recruits that are spawners), prior distributions were used for the parameters (Table 4.1). Time varying parameters were not considered in the models.

The model predictions of abundances at age and total spawner abundance were fitted to the point estimates of abundances of spawners at age and estimated total spawners from the assessments conducted in the Miramichi over the period 1996 to 2019.

The models were coded in OpenBugs with posterior distributions derived from Monte Carlo Markov Chain simulations with Gibbs sampling (Lunn et al. 2013; Appendix 4).

4.6.1. Model 1

The initial model assumed informative prior information for most of the life history parameters with the exception of the parameters of the stock and recruitment Beverton-Holt function and the precision parameters of the likelihoods (Table 4.1).

Parameters in the model to be estimated are:

- α (survival rate at the origin);
- K, asymptotic carrying capacity of age 0 in the first summer;
- σ, for ages 3 to 8 and for total spawners;
- Z for ages 0 (overwinter survival), 1, and 2 from the Z to length relationship of Gislason et al. (2010; informative priors);
- Z at age assumed similar for ages 3 to 15+ at median value (0.33) of the cohort decline analysis of estimated spawners at ages 5 to 12 (informative prior); and
- Proportion of recruits at age that are spawners (sexes combined), based on assumed maturation schedule of males and females (informative priors).

4.6.2. Model 2

In the second model, the mortality rates at ages 3 to 8 were estimated independently but with informative priors with the same rates over years; the mortality rate for ages 9 to 15 was set at the mortality rate at age 8 (Table 4.1).

Parameters in the model to be estimated are:

- α (survival rate at the origin);
- K, asymptotic carrying capacity of age 0 in the first summer;
- σ, for ages 3 to 8 and for total spawners;
- Z for ages 0 (overwinter survival), 1, and 2 from Z to length relationship of Gislason et al. (2010; informative priors);
- Z for ages 3 to 8; Z for ages 9 to 15+ = Z at age 8 (informative prior); and
- Proportion of recruits at age that are spawners (sexes combined), based on assumed maturation schedule of males and females (informative priors).

4.6.3. Model 3

In the third model, the mortality rates at ages 3 to 8 were given independent and weakly informative priors (Table 4.1).

Parameters in the model to be estimated are:

- α (survival rate at the origin);
- K, asymptotic carrying capacity of age 0 in the first summer;
- σ, for ages 3 to 8 and for total spawners;
- Weakly informative priors for Z for ages 3 to 8; Z for ages 9 to 15+ = Z at age 8;
- Z for ages 0 (overwinter survival), 1, and 2 from the Z to length relationship of Gislason et al. (2010; informative priors); and

- Proportion of recruits at age that are spawners (sexes combined) for ages 3 to 6. Proportion for ages 7 to 15 set equal to proportion at age 6.

4.6.4. Model 4

In the fourth model, the proportion of recruits that are spawners at ages 3 to 6 and the survivals at age-0, 1, and 2 are given weakly informative priors, to be estimated (Table 4.1; Appendix 4a).

Parameters in the model to be estimated are:

- α (survival rate at the origin);
- K, asymptotic carrying capacity of age 0 in the first summer;
- σ, for ages 3 to 8 and for total spawners;
- Z for ages 3 to 8; Z for ages 9 to 15+ = Z at age 8;
- Weakly informative priors for Z for ages 0 (overwinter survival), 1, and 2 centered on Z to length relationship of Gislasson et al. (2010); and
- Weakly informative priors for proportion of recruits at age that are spawners (sexes combined) for ages 3 to 6. Proportion for ages 7 to 15 set equal to proportion at age 6.

4.6.5. Model 5

In the fifth, the cumulative survival from age 0 (summer) to age 3 was estimated, excluding the need for priors on survivals at age 0, 1, and 2 (Table 4.1; Appendix 4b).

Parameters in the model to be estimated are:

- α (survival rate at the origin);
- K, asymptotic carrying capacity of age 0 in the first summer;
- σ, for ages 3 to 8 and for total spawners;
- Z for ages 3 to 8; Z for ages 9 to 15+ = Z at age 8;
- Cumulative Z for age 0 (summer) to age 3; and
- Proportion of recruits at age that are spawners (sexes combined) for ages 3 to 6. Proportion for ages 7 to 15 set equal to proportion at age 6.

4.6.6. Model 6

In this model, the Beverton-Holt stock and recruitment parameters were estimated for eggs to recruitment at age 3 (Table 4.1; Appendix 4c).

Parameters in the model to be estimated are:

- α (survival rate at the origin; cumulative survival eggs to age-3);
- K, asymptotic carrying capacity at age 3;
- σ, for ages 3 to 8 and for total spawners;
- Z for ages 3 to 8; Z for ages 9 to 15+ = Z at age 8; and
- Proportion of recruits at age that are spawners (sexes combined) for ages 3 to 6. Proportion for ages 7 to 15 set equal to proportion at age 6.

4.6.7. Model 7

In the final model, a power function for the spawner to recruitment relationship to age 3 was examined, that defines a density dependent survival but no carrying capacity limit. Given the relatively short time series of stock and recruitment data and the one way trip of increasing abundance observed, this model was used to examine the strength of evidence of a compensatory relationship with an asymptote for carrying capacity for recruitment measured at age 3.

$$N_{y+3,3} = \gamma * Eggs_y^{\beta}$$

with
$N_{y+3,3}$ = recruitment abundance (number) at age 3 in year y+3,
$Eggs_y$ as defined above,
γ = density independent mortality rate (0,1), and
β = density dependent component, expected to be < 1 if there is density dependence.

Parameters in the model to be estimated are (Table 4.1):

- γ proportional survival from eggs to age 3;
- β the density dependent compensatory parameter for age 3;
- σ, for ages 3 to 8 and for total spawners;
- Z for ages 3 to 8; Z for ages 9 to 15+ = Z at age 8; and
- Proportion of recruits at age that are spawners (sexes combined) for ages 3 to 6. Proportion for ages 7 to 15 set equal to proportion at age 6.

4.7. MODEL RESULTS

Model diagnostics for variants 4, 5, and 6 are detailed in Appendix 5 and summarized in Table 4.2.

The time series of increasing abundance of spawners for the Striped Bass population during 1996 to 2019 follows a one way trajectory and the observations provide limited information to clearly define the population dynamics. Despite this, a number of conclusions can be drawn from these analyses:

- There is sufficient evidence that survival rates at age for the time series of observations differ with the lowest estimated survival rates for ages 4 to 6 and the highest rates for ages 8 plus.
- Estimated survival rates of Striped Bass of ages 7 and older, appear to have increased over the time period 1996 to 2019 (based on positive temporal trend in residuals), although such a change was not incorporated in the model.
- The proportion of recruits at age that become spawners increases from age 3 to 6, as expected.
- There is a negative correlation between the estimated survival rate at the origin of eggs to age-0 summer abundance of the Beverton-holt relationship and the density independent survival rate estimated for other ages (age-0 and age-3 in Models 3 and 4, age 0 to 3 in Model 5). This trade-off in parameter estimates occurs because of an absence of observations allowing for the partitioning of survival for the intermediate age groups (ages 0, 1, and 2).

- There is insufficient evidence to unequivocally conclude or reject the assumption of a density-dependent compensatory stock and recruitment relationship for this population. There is little difference in the fit to observations of the power function model compared to models with assumed Beverton-Holt stock and recruitment functions. The power function of eggs to recruitment at age 3 provides the lowest deviance value of all the models but with a density dependent parameter that encompasses unity, hence a proportional relationship.

A priori, a density dependent Beverton-Holt stock and recruitment function is assumed and models with this stock and recruitment function were considered further.

4.7.1. Beverton-Holt SR Model Results

There is no difference in fits to observations of the model with a Beverton-Holt stock and recruitment function between eggs and age-0 abundance in the summer (followed by density independent survival to age-3; Models 4 and 5) and the model that fit the stock and recruitment function from eggs directly to age-3 (Model 6; Table 4.2; Appendix 5).

In terms of the models that estimate survival at the origin and carrying capacity to age-0, the following are noted:

- The first model (model 4) that incorporated an egg to age-0 stock and recruitment function considered weakly informative priors on the overwinter survival rates at age-0 and the survival rates at ages 1 and 2 to estimate the abundances at age-3, the first age of spawners with observations.
- The alternate model (model 5) directly estimated a cumulative survival rate from age-0 to age 3.
- There is a strong negative correlation in the estimates of survival at the origin from eggs to age-0 and the estimates of survival at age 0 and at age 3 in model 4 and in the estimates of survival from age-0 to 3 in model 5 (Appendix 5).
- The estimated survival at the origin (eggs to age-0) for model 4 is approximately three times higher than the estimate for model 5 (Table 4.3). The cumulative survival from age-0 to 3 in model 4 (based on priors for survival rates for overwinter survival at age-0 and survivals at age 1 and 2) is much lower (by a factor of 4) than for model 5 which directly estimates a cumulative survival from age-0 to 3.
- The cumulative survival from egg to age-3 at the origin, in the absence of density dependent compensatory survival, is quite low at 3 to 4 fish per 100,000 eggs. The scaled egg to age-3 survival for model 5 (median = 3.65 E-5) is similar to that of model 4 (median = 3.34 E-5) and with large uncertainties; consequently, there is no difference in the estimated density independent survival rates from eggs to age-3 between the models ($p = 0.26$; Table 4.3).
- The lifetime reproductive rate, expressed as the cumulative production of age-3 recruits in absence of density-dependent compensatory survival over the lifetime of a spawner (sexes combined), is approximately 5.0 to 5.5 recruits at age-3, and similar for these two models (Table 4.3).
- The estimate of K at age-0 is higher for model 4 than for model 5 ($p = 0.06$) however the age-3 asymptotic abundance estimated by correcting K at age-0 by cumulative survival between age-0 and age 3 results in a significantly higher asymptotic abundance value at age 3 for model 5 compared to model 4 ($p < 0.001$; Table 4.3).
- Beverton-Holt K at age-0 and scaled to age-3 are not attainable with the assumed and estimated life history parameter values from these models as shown by the equilibrium

asymptotic values which are lower than the theoretical asymptotic values from the Beverton-Holt model. Equilibrium modelling using the assumed life history characteristics (weight at age, maturation schedule, fecundity) and the estimated population dynamic parameters (survivals, proportion of recruits that become spawners, Beverton-Holt stock and recruitment parameters) result in asymptotic abundance values at age-0 that are 81% of Beverton-Holt K from models 4 and 5, respectively (Table 4.3).

Model 6 estimated the Beverton-Holt stock and recruitment parameters directly from eggs to age -3.

- The median estimate of eggs to age-3 survival from this model (median = 4.09 E-5) is much lower than the density independent survival at the origin (eggs to age-0) from models 4 and 5.
- The survival from eggs to age-3 for model 6 is higher than the scaled survival from eggs to age-3 from model 4 and model 5 but with large uncertainties that overlap among models resulting in no significant differences in the scaled survival rates among the models (p = 0.20, 0.42, respectively; Table 4.3).
- The lifetime reproductive rate is similar for the three models considered (Table 4.3).
- The estimated carrying capacity at age-3 from model 6 is approximately nine times and four times higher than the scaled carrying capacity to age -3 for model 4 and model 5, respectively, and despite large uncertainties, the distributions do not overlap among the models ($p < 0.001$, 0.01, respectively; Table 4.3).
- Equilibrium modelling of asymptotic abundance at age-3 for model 6 gives a value of 2.9 million recruits and 815 thousand spawners at age-3, 78% of the Beverton-Holt derived carrying capacity value for recruitment at age-3.

4.7.2. Choice of Model

The choice of model has consequences on the interpretation of population abundance and trends as well as on the derivation of the reference points.

- A priori, a density dependent Beverton-Holt stock and recruitment function with density dependence occurring between eggs and age-0 summer abundance is assumed so these models are retained (models 4 and 5). Model 6 (Beverton-Holt stock and recruitment function with density dependence occurring between eggs and age 3) is not retained; in its recruitment profile, model 6 is very close to a proportional relationship.
- There is little information to support preferentially selecting Model 4 over Model 5. Diagnostics of model fits suggest a slight improvement in the predicted to observed total spawner abundances for model 5 but the difference is very minor (Figures 4.1, 4.2; Appendix 5). Deviance values from the two models are essentially identical. There are fewer prior requirements (fewer parameters) for model 5 compared to model 4 as only cumulative survival from age-0 to age-3 is estimated but other than that, the estimates of survival at ages 3 to 8 and the proportion of recruits that are spawners are similar between models (Figure 4.3).
- Model 4 parameter estimates indicate the population has a higher survival rate at the origin and a higher carrying capacity to age-0, however, the carrying capacity at age-3 is lower for model 4 compared to model 5 due to the lower cumulative survival from age-0 to age-3 inferred from model 4 (Figure 4.4).

- The lower carrying capacity at age-3 and the higher survival rate at the origin from model 4 will in turn result in lower reference values for maximum sustainable yield and other reference points compared to model 5.
- Estimates of maximum sustained yield and candidate reference values are presented for both models 4 and model 5.
- More detailed diagnostic and summaries for models 4 and 5 and summaries for model 6 are available in Appendix 5.

5. REFERENCE POINTS FOR STRIPED BASS

5.1. RECOVERY OBJECTIVES FROM THE RECOVERY POTENTIAL ASSESSMENT

Following on the first status assessment by COSEWIC (2004) of the southern Gulf of St. Lawrence Striped Bass Designatable Unit as threatened, a Recovery Potential Assessment was conducted that included proposals for abundance recovery objectives (DFO 2006; Douglas et al. 2006). Mortality, fecundity, and stock and recruitment dynamics were modeled using general life history information of the species and observed or assumed values specific to southern Gulf Striped Bass. The choice of parameter values in the model were supported by observations on characteristics of the population and balancing of life stage abundances. The characteristics of the southern Gulf population considered included:

- prior for expected abundance of adult bass and spawners,
- relative age structure of the spawners, and
- sex ratio of spawners.

5.1.1. Prior Expectation of Striped Bass in an Exploited State

An estimate of historical maximum abundance was stated as a reasonable expectation of a recovered population. The maximum recorded annual fishery landing of southern Gulf Striped Bass since 1917 was 61.4 t (in 1917). The maximum commercial landing during 1968 to 1996 was 47.1 t. Using the historical maximum landing of 61.4 t, an assumed weight for the exploited Striped Bass population of 1.9 kg, and an assumed (without information) exploitation rate of 50%, the abundance of adult-sized (3 year and older) Striped Bass in the southern Gulf was considered to have been between 65,000 and 200,000 fish (Douglas et al. 2006).

A deterministic life history equilibrium model was run over a range of egg depositions to derive four spawning stock reference levels: spawners at equilibrium in the absence of fisheries (Seq), the spawning stock which produced the maximum gain (Sopt), and spawning stocks at a fishing rate which resulted in 50% and 30% spawning per recruit (50%SPR, 30%SPR). The mortality rate and life history parameters were assumed as:

- Beverton-Holt stock and recruitment relationship with α = 0.001 and K expressed as abundance of age-0 at the end of the summer = 1.5 million fish;
- M = 1.5 for the six months of overwintering for YOY;
- M = 1 for age 1 bass;
- M = 0.6 for age 2 and older bass;
- Maturation schedule of males and females (or proportion of bass at age on spawning grounds); and

- Fecundity based on mean weight at age.

Sopt (spawners that produce C_{MSY}) was proposed as the recovery limit for the southern Gulf Striped Bass and spawners for 50%SPR as the recovery objective for directed fisheries. Since the parameters for the Beverton-Holt compensatory function were not known, simulations under lower and higher average YOY production (1, 1.5, 2 million; K) and for lower and higher density independent survival (0.0005, 0.001, 0.002; α) were run. Based on the prior expectation of adult abundance being in the range of 65,000 and 200,000 bass, the YOY productive capacity of 1.5 million and the density independent survival rate of 0.1% were retained as suitable values for deriving the reference levels. The Seq value (spawners at replacement in terms of lifetime egg production) was estimated at 63,000 fish. The proportion female in the spawners was 0.34.

The Sopt value was calculated at 21,600 spawners and the 50%SPR value was 31,200 spawners. These were proposed as the recovery limit and the recovery target, respectively, the latter being the value for considering any directed fisheries. Compliance rules were also proposed for assessing whether the population was recovered; for 5 of 6 consecutive years for the recovery limit and once this was attained, attainment of the recovery target in 3 of 6 consecutive years. It was also indicated that the assessment of spawner abundance relative to the recovery objectives would be based on the 5th percentile of the annual abundance, keeping with the premise that there should be a low probability of the abundance indicator being below the recovery limit (Douglas et al. 2006). The expectation of reasonable abundance, i.e. adult Striped Bass of 100 thousand, and the recovery objectives were exceeded after 2010.

5.2. FISHERY REFERENCE POINTS

Striped Bass is a valued Indigenous FSC, recreational, and previously commercial fish and it was assumed that the reference points of interest to DFO Fisheries Management would be used to manage harvest fisheries. A large number of reference points have been proposed and discussed in the literature (Goodyear 1993; Mace 1994; Myers et al. 1994; Gabriel and Mace 1999). We focused on a limited number of possible reference points that could be derived from equilibrium modelling of maximum sustainable yield, from spawner potential per recruit (SPR) and reference values based on historical observations.

5.2.1. Methods

Given the iteroparous nature of Striped Bass, the concepts of Maximum Sustainable Yield (MSY) and associated metrics including B_{MSY} (biomass at MSY), C_{MSY} (catch at MSY) and F_{MSY} (fishing rate at MSY) are relevant. With carrying capacity in units of juvenile stages, B_{MSY} is calculated using the assumed life history characteristics that include a stock recruitment relationship, natural mortality-at-age, partial recruitment to the fishery at age, weight-at-age, proportion female spawners at age, and fecundity. Important population dynamics parameters, in particular the stock and recruitment parameters, were obtained from model fitting to observations. MSY reference points are derived using an equilibrium model that incorporates the joint probability distributions of these life cycle model parameters.

Reference points corresponding to Spawner per recruit (SPR) concepts were also considered. SPR is presented as a proportion of the spawner potential which remains when fished relative to a population that is not fished (Goodyear 1993). There is no spawner to recruitment function in SPR calculations. SPR reference point values discussed in literature include: 30%SPR (fishing rate that reduces the spawner production to 30% of the unfished condition) as a maximum fishing rate (Mace and Sissenswine 1993; ICES 1997) and 50%SPR (fishing rate that reduces the spawner potential to 50% of the unfished condition) as a target fishing rate, presented as F_{pa} in ICES (2001). These fishing rate reference points can be converted to abundance reference points using a stock and recruitment function. Equilibrium modelling is used to calculate the equilibrium abundance at fishing rates corresponding to 30%SPR and 50%SPR. Spawner per recruit reference points are derived using the joint probability distributions of the life cycle model parameters for spawners of ages 3 to 15+.

MSY and SPR reference point are context specific. The reference point values derived depend not only on the parameter estimates of the population dynamics (survival, prop. recruits to spawners) but also on the fisheries management scenarios, particularly those that have size limits for harvest retentions. The size limits, combined with the size distributions at age, define the partial recruitment at age to the fishery and hence the proportion of the total annual losses at age attributed to fishing.

We also considered a traffic light approach that relies exclusively on past observations without a model as a simple and naïve alternative to define potential precautionary approach status zones. The traffic light approach was proposed for the integration of multiple indicators and for simplifying the communication of information to support management decisions (Caddy 2002).

MSY and SPR abundance reference points are calculated in terms of eggs and converted to numbers of spawners on the spawning ground of the Northwest Miramichi because this is the component that is monitored and assessed (DFO 2020).

5.2.2. Upper Stock Reference (USR)

The USR points examined include:

- Spawner abundance at 80% B_{MSY};
- Spawner abundance at equilibrium when the stock is fished at F corresponding to 50%SPR; and
- Traffic light green zone that characterizes a high abundance state.

5.2.3. Limit Reference Point (LRP)

DFO (2009) provides guidance for candidate LRPs. The LRPs examined include:

- Lowest spawner abundance that resulted in recovery of the stock (Brecover);
- Spawner abundance at equilibrium corresponding to 40% B_{MSY};
- Spawner abundance at equilibrium when the stock is fished at F corresponding to 30%SPR; and
- Traffic light reference boundary that defines a zone of low abundance based on history of assessed values.

Additionally LRPs based on the abundance of spawners (or eggs) that results in 50% of K (carrying capacity) or 50% of equilibrium asymptotic abundance are also considered:

- Spawner (number) abundance or eggs that result in 50% chance of attaining 0.5 K (at age 3); and
- Spawner (number) abundance, eggs that result in 50% chance of attaining 0.5 equilibrium asymptotic abundance (at age 3).

Density dependent effects are assumed to occur during the early life stage, i.e. from eggs to early summer recruitment. Mortality at all other life stages was assumed to be density-independent hence K can be defined for any life stage of interest that is first measured. K is presented as the spawner abundance at age 3 years, the first age of maturity that is assessed on the spawning grounds.

Previously, Douglas et al. (2006) summarized the information related to an abundance index of YOY from monitoring of catches in the rainbow smelt (*Osmerus mordax*) open water fishery in the fall during 1991 to 1998 and a summer beach seine index from 2001 to 2005. The mean annual catch rate (CPUE) of YOY bass in the open-water smelt fishery was positively correlated (R = 0.66) to the female spawner estimates derived from mark and recapture and less so for the total spawner abundance. When female spawner abundance was at or above 5,000 fish, there was a high YOY index in the fall smelt fishery supporting the premise that spawner abundance is an important component of recruitment to the fall YOY stage of striped bass (Bradford and Chaput 1997; Douglas et al. 2006).

Beach seining surveys at five to six index sites of the Miramichi were conducted during 2001 to 2005. Catch per unit effort analyses were restricted to the July sampling period because:

- YOY are readily captured in nearshore habitats of the Miramichi by this time,
- most YOY have not yet extended their distribution outside of the Miramichi system, and
- catches of YOY by beach seine in the Miramichi substantially decreased by August.

Mean CPUE estimates were highly variable between years ranging from a high of 139 YOY per sweep to a low of 4 YOY per sweep in 2003 and 2004, respectively. Douglas et al. (2006) indicated that several more years of beach seine data would be required to determine the correlation between YOY and spawners. The limited data from the Miramichi indicates that environmental factors may play an important role in year-class success, as shown in several US studies that have demonstrated that recruitment is largely determined in the first few days after spawning as a result of variable environmental conditions affecting survival (Richards and Rago 1999).

5.2.4. Removal Rate Reference Point

The fishing rate reference points considered are:

- F_{MSY} from equilibrium modelling;
- F corresponding to 30%SPR as a maximum fishing rate; and
- F corresponding to 50%SPR as a target fishing rate.

6. DERIVATION OF CANDIDATE REFERENCE POINTS

6.1. TRAFFIC LIGHT APPROACH

The traffic light approach is used to coarsely assign estimates of annual abundance of Striped Bass to three status zones, or traffic light colours. A substantial amount of work was undertaken by DFO in the early 2000s to consider what kind of indicators could be used, how to integrate multiple indicators, and how to establish the thresholds that define the zones (Halliday 2001; Halliday and Mohn 2001). Halliday and Mohn (2001) discuss a number of considerations for setting boundaries including the scale of the indicator (natural scale vs log scale) and how the observations considered may change the boundary thresholds.

The 24 year time series of spawner abundance estimates for the period 1994 to 2019 is characterized by an approximately monotonic increase in abundance. We were interested in aggregating the time series of spawner abundances into three status categories roughly equivalent to critical, cautious, and healthy zones of the PA.

6.1.1. Methods

The categories, defined as the centroids for three groups of observations, were estimated using the optimization function “kmeans” in R. This R utility uses an objective function that minimizes the sum of squares of individual points to the assigned group centers.

We examined how the definition of the groups depended on three considerations:

- the scale of the observations i.e. the natural scale versus the log scale;
- the effect of excluding the exceptional 2017 observation on the estimates of the groups; and
- the variability of the attribution of status based on the time series of observations considered. The change in estimated group centroids and the attribution of the annual observations to status zones is examined beginning with the 1994 to 2008 time series and sequentially adding one year to the data series to 2019 (excluding 2012 with no data).

Proxy values equivalent to the boundaries between the critical and cautious zones (LRP) and between the cautious and healthy zones (USR) were calculated as the means of respectively the lower and middle centroids and the middle and upper centroids.

6.1.2. Results

Log transformation versus the natural scale for observations prior to optimization of three group centroids has a large effect on the assignment of status and the calculation of proxy reference values (Figure 6.1). Using the entire time series of assessment values (medians) from 1994 to 2019 (excluding 2012), the interpretation of status is as follows (Figure 6.1, upper row):

- Based on the log scale, the abundance was in the critical zone during 1996 to 2000, has been in the healthy zone since 2011, and was in the cautious zone in all other years.
- Based on the untransformed values, the abundance was in the critical zone during 1994 to 2010 as well as in 2012, and has only been in the healthy zone in 2017.
- Following on this, the proxy LRP based on the log transformed data would be 13 thousand spawners compared to 162 thousand spawners based on untransformed data.
- The proxy USR values are similarly different, at 105 thousand based on log transformed values and over 600 thousand based on the untransformed values.

There is a large effect on the interpretation of status zones for individual years with incremental additions to the time series of observations (Figure 6.1, middle rows):

- There is similar interpretation of status, based on log transformed data and untransformed data, when the status categories are defined based on the initial short time series of observations, 1996 to 2009. In both cases, the population was assigned to the critical zone during 1996 to 2000, and to the healthy zone during 1994, 1995, 2007 to 2009.
- Sequentially adding a year to the analyses has the largest effect on the interpretation of status when the observations are on the natural scale. The status of the 1994, 1995, and 2007 to 2009 assessed years declines from healthy, through cautious and into critical as observations for the 2011, 2013 to 2015 assessment years are included in the estimation of groups.

- The interpretation of status is however much more stable when the observations are log-transformed prior to assignment to groups. At most, the status for some years declines from healthy to cautious.
- In almost all cases, the status changes from healthy to cautious or cautious to critical. It is never consistently in the opposite direction. This is expected given the almost monotonic increase in assessed abundance of this population during the period 1996 to 2019.
- The proxy LRP values based on the log transformed data are in the same range based on the initial 15 years of data (10 thousand fish) compared with the entire time series (13 thousand fish). This is not the case when the untransformed data are used; an LRP proxy value of 13 thousand fish is calculated for the initial 15 year time series whereas the proxy LRP value based on the entire time series is more than a factor larger, at 162 thousand fish.
- The proxy USR values are similarly different, based on the transformation or not of the observations. Based on the initial 15 year time series, the proxy USR values are approximately similar between the data treatments (36 versus 43 thousand for log transformed and untransformed, respectively). Using the entire time series, the proxy USR values increase to 105 thousand for log transformed and over 600 thousand for the untransformed data.

The assessed median abundance of spawners of 990 thousand fish is an exceptional observation in the relatively short time series of assessment. Excluding the 2017 observation has interesting consequences on the assessment of status and the derivation of proxy reference values (Figure 6.2):

- The status zones and the interpretation of status for the initial time series are not affected by excluding the observation for the 2017 assessment year because the groups are defined based on data from 1994 to 2009.
- For the time series extending from 1994 to 2019, the interpretation of status and the calculation of the proxy reference values based on the log transformed data are essentially similar whether 2017 is included or excluded. The proxy LPR values are 10 thousand when 2017 is excluded versus 13 thousand when 2017 is included. The proxy USR values are 87 thousand when 2017 is excluded versus 105 thousand when 2017 is included.
- In contrast, for the observations on the natural scale, the interpretations of status through time and the calculation of proxy reference values are sensitive to the inclusion versus exclusion of the 2017 value. Note that when the 2017 data point is included, the upper centroid and zone are defined exclusively by the single observation of 2017. When 2017 is excluded, the upper centroid is defined by 6 observations (Figure 6.2). When 2017 is excluded, the interpretation is that the population was in the critical zone during 1996 to 2006 and has been in the healthy zone since 2011 with the exception of the assessed abundance in 2014. The proxy LRP value for the whole time series is 44 thousand fish when the 2017 observation is excluded (Figure 6.2), compared to 162 thousand fish when the 2017 observation is included (Figure 6.1). The proxy USR value is similarly strongly affected by the 2017 observation; when the 2017 observation is excluded, the proxy USR value is 181 thousand fish in contrast to a proxy USR value of 639 thousand fish when the 2017 point is included (Figures 6.1, 6.2).

6.2. EQUILIBRIUM MODELLING

Equilibrium modelling is used to simulate predicted abundances at age and overall for different fishing rates. The equilibrium model uses the same life cycle equations as in the estimation model (section 3) with modifications as described in the next sections. Values of the population dynamics and life history parameters are taken from individual MCMC draws from the joint posterior distribution from the population model. The model in its equilibrium form is coded in R with runs forward 150 years to ensure attainment of equilibrium conditions, at fixed levels of fishing and for specific management regimes.

Maximum sustainable yield (MSY) is derived by searching over a range of fully-recruited F for the fishing rate (F_{MSY}) that results in maximum yield (in weight). Biomass at MSY (B_{MSY}), spawner abundance (number of fish) at B_{MSY}, catch (C_{MSY}; in number and weight), and age structure of the catch and of the spawners at MSY are retrieved from the simulation outcomes. Management strategies based on size limits are also examined with the model.

The MSY values are provided for the abundance (number, biomass) of spawners (ages 3 to 15+) on the spawning ground, thus the values do not represent the entire population as not all Striped Bass of ages 3 to 15+ are considered to be present on the spawning ground. The spawning period (May) is considered to be the start of the year.

6.2.1. Natural Mortality (M) At Age

Estimates of M at ages 3 to 15+ are required for the equilibrium analysis to derive fishing rate and MSY reference values.

In the age structured population model, applied to the estimated spawner abundances at age for the years 1996 to 2019, Z at age is estimated for ages 3 to 8, with Z at ages 9 to 15+ being set equal to Z at age 8. These are estimates of total mortality (sum of natural mortality and fishing mortality) as there were fisheries removals of Striped Bass over the entire time series, despite the closures of all harvest fisheries between 2000 and 2012 (DFO 2011).

Based on acoustic tagging and tracking data, estimates of instantaneous mortality rates (Z) were 0.41 (median) during the period 2003 to 2009 and 0.22 (median) during the period 2014 to 2018 (Section 3.6.2.2; Figure 3.10). It is not possible to partition the natural mortality rates from fishing mortality rates with these data however considering that fishery removals would have in part contributed to the estimated mortalities, the instantaneous natural mortality rate of adult sized (> 47 cm) Striped Bass would not be greater than 0.2.

In the coastwide assessment model for Striped Bass of the eastern seaboard of the US, M for adult bass age 4 and older is set at 0.15 (NEFSC 2019).

For purposes of equilibrium modelling and to define reference points, two scenarios for M were examined:

- Assuming M = Z as derived from the population model for ages 3 to 15+ (Figure 4.3);
- M at age 3 based on Z from the population model and M for ages 4 to 15+ from acoustic tagging information (M = 0.20 with a 5th to 95th percentile range of 0.13 to 0.28 based on S ~ beta(82,18)).

6.2.2. Fishery Selectivity at Age (s_a)

Fishery selectivity at age (s_a) to fully-recruited F is determined using the predicted fork length distribution at age from the von Bertalanffy model and relative to a defined management strategy based on fork length (Table 6.1). The proportion of the age group vulnerable to the fishery was calculated as the proportion of the area under the normal density curve contained within the lower and upper size retention limits. The proportion of the area at age is calculated as (in R code):

$$s_a = pnorm(FL.max, u.fl_a, sd_a) - pnorm(FL.min, u.fl_a, sd_a)$$

with
s_a being the selectivity at age a (range 0 to1) to fully-recruited fishing rate ,
FL.max and FL.min are the fork length size limits (cm) for a specific management strategy,
$u.fl_a$ = mean fork length (cm) of bass at age a at the time of fishery taken as mid-season, and
sd_a = mean standard deviation of the mid-season size distribution at age a (Figure 6.3).

For a management strategy with no size limits, a minimum size of 30 cm was assumed to be the smallest sized bass that would be retained. If there is no maximum size limit defined, FL.max was set to 150 cm.

6.2.3. Catch Equation

The standard Baranov catch equation was used to calculate the number and weight of fish lost due to fishing activities, assuming F and M occur simultaneously, i.e. between May and October. It is assumed that a fish that is captured and within the management size limit is retained, all other fish are released.

Total loss of fish at age resultant of fishing includes fish retained and harvested and fish lost due to catch and release mortality. A catch and release mortality rate of 9% is assumed corresponding to the catch and release mortality value used in the coastwide assessment of Striped Bass of the eastern seaboard of the US (NEFSC 2019).

$$FLoss.N_a = N_a * (1 - e^{-(M_a + s'_a F)}) * \frac{s'_a F}{s'_a F + M_a}$$

with
$FLoss.N_a$ the number of bass at age a that die from fishing activities,
N_a the estimated recruitment abundance (sexes combined) of bass at age a,
M_a is the natural mortality at age a,
F is the fully recruited fishing rate,
$s'_a = s_a + (1 - s_a) * A.CR$, s_a is the vulnerability at age to fully recruited F, and *A.CR* is the catch and release mortality rate set at 9% when losses from catch and release are accounted for in the model. Setting *A.CR* = 0 is equivalent to ignoring mortality from catch and release.

Yield in terms of retained catches, to define maximum sustainable yield, is calculated as:

$$C.N_a = N_a * (1 - e^{-(M_a + s'_a F)}) * \frac{s_a F}{s'_a F + M_a}$$

with
$C.N_a$ the retained catch in number at age a, and other components as described above.

$$C.Wt_a = C.N_a * u.wt_a$$

with
$C.Wt_a$ is the retained catch weight-at-age a, and $u.wt_a$ is the mean weight at age a at the time of the fishery (mid-year) based on $u.fl_a$.

6.2.4. Equilibrium Modelling Results

An example of the equilibrium modelling results and the reference values from model 5 is presented in Figures 6.4a to 6.4d. For illustrative purposes, the management strategy corresponding to no size limits and no accounting for catch and release mortality ($A.CR = 0$) is considered the default strategy. The summaries are presented for the assumptions on M of Striped Bass aged 3 to 15+ and include:

- Plot of survival rates at age (e^{-M}) assumption;
- Plot of proportions of recruits at age that become spawners;
- Plot of selectivity at age to fully recruited F (s_a; specific to a management scenario);
- Plot of catch at age proportions at F = (F_{MSY}, 50%SPR, 30%SPR);
- Plot of age distribution of recruitment at F = (0, F_{MSY}, 50%SPR, 30%SPR); and
- Plot of age distribution of spawners at F = (F_{MSY}, 50%SPR, 30%SPR).

MSY estimation summary outputs include:

- The equilibrium total recruitment abundance (ages 3 to 15+) over a range of fully recruited fishing rates;
- The equilibrium total spawner abundance (ages 3 to 15+) over a range of fully recruited fishing rates;
- Yield in weight over a range of fully recruited fishing rates;
- Yield in number of fish over a range of fully recruited fishing rates; and
- Posterior distributions (boxplots) of C_{MSY} (weight), C_{MSY} (number), F_{MSY}, B_{MSY} (recruitment), B_{MSY} (spawners), and eggs at B_{MSY}.

Illustrative plots of abundance (number of fish) trajectories over 150 years including:

- Predicted total recruitment at F=0 and F= 0.09; and
- Predicted recruitment at age-3 at F = 30%SPR and F = 50%SPR.

6.2.4.1. Equilibrium results for model 5 and model 4

Equilibrium modelling results based on life history parameter inferences from model 5 are summarized in Table 6.2a and Figures 6.4a to 6.4d. Results for model 4 are summarized in Table 6.2b. Abundances are summarized in terms of total abundance for ages 3 to 15+, referred to as recruits, and in terms of spawners which would be the component assessed on the spawning grounds (DFO 2020). The spawner abundance values are lower than the total abundance because not all fish at ages 3 to 15+ are spawners. Fishing occurs on recruitment, or total abundance, and catch and fishing rate references refer to the removals and removal rates from the entire stock.

As expected, total equilibrium recruitment abundance (ages 3 to 15+) is higher for the equilibrium model with lower assumed values of M at age and abundance decreases with increasing fishing mortality rates (Figures 6.4b, 6.4c; Tables 6.2a, 6.2b). The yield curve is not symmetric, rising more steeply on the ascending limb at F less than F_{MSY} and declining more slowly on the decreasing side of the yield curve (Figure 6.4b). The equilibrium abundances and yields have large uncertainty, due to the combined uncertainties in the life history parameter estimates from population modelling.

Maximum lifetime reproductive rate, defined as the cumulative production of recruits at age-3 in absence of density-dependent compensatory survival over the lifetime of a spawner (sexes combined), is 15.7 fish (median; 5th to 95 percentile range 11.1 to 23.0 fish) for the lower M at age values, and 5.0 fish (median; 5th to 95 percentile range 3.7 to 7.5 fish) for M=Z from population model 5 (Table 6.2a). Approximately similar values are calculated from model 4 (Table 6.2b).

F_{MSY} values are similar for the equilibrium models with differing assumptions for M (Figure 6.4b; Tables 6.2a, 6.2b). The population crashes (N <= 100 fish) when fully recruited F exceeds 0.70 (M = Z, panel A) and 0.87 (for lower values of M, panel B; Figure 6.1a; Table 6.2a). Spawner per recruit fishing rate reference values of 30%SPR and 50%SPR are higher for the model with higher values of M (panel A, Figure 6.4d). F at 30%SPR is higher than F_{MSY} for both scenario values of M. Approximately similar values are calculated from model 4 (Table 6.2b).

The age structure of the population is modified by the fishing activity, with a strong bias towards younger ages in the total population and in the spawners:

- the higher the fishing rate, the faster fish die because mortality at age is the sum of fishing mortality and natural mortality;
- as fishing rate increases (for a constant M), the age structure of the spawner population gets younger, the mean weight of spawners decreases, and because younger fish have a lower proportion female as spawners (before age 6), the number of eggs per spawner declines.

The MSY and SPR reference values are higher for model 5 compared to model 4 (Figure 6.5). Based on M for ages 4+ inferred from observations, B_{MSY} from model 5 is approximately twice as high compared to the estimate from model 4. F_{MSY} estimates of F = 0.17 are similar between models resulting in higher C_{MSY} values, by a factor of two, from model 5 compared to model 4 (Tables 6.2a, 6.2b; Figure 6.5).

6.2.5. Reference Points From Equilibrium Modelling

6.2.5.1. Reference points dependent on assumptions for M

MSY reference values and reference points derived from equilibrium modelling are dependent upon the assumptions of natural mortality. The reference points are defined in terms of the number of spawners on the spawning grounds, the life stage and time period corresponding to the assessments (DFO 2020). The following summaries present the results from models 4 and 5 for the default fishing strategy with no size limits for retention and no accounting for catch and release mortality (Table 6.3a, 6.3b).

The USR values (80%B_{MSY}, abundance at 50%SPR) from the equilibrium model are higher for the scenario with lower assumed natural mortality rates (Tables 6.3a, 6.3b). For model 5, spawner abundances at 80%B_{MSY} are 530 thousand fish for the M = Z scenario and 1.2 million fish for M based on observations. The spawner abundances corresponding to 50%SPR are higher yet, at 620 thousand and 1.8 million fish for scenarios of M = Z and M based on observations, respectively (Table 6.3a). In all cases, the uncertainties for the reference values are large.

USR values from model 4 with M based on observations are comparatively lower than those from model 5, at 720 thousand for 80%B_{MSY} and 1 million for 50%SPR (Table 6.3b). The uncertainties for these reference points are equally high as in model 5.

The values of the respective candidate LRPs differ substantially. Brecover, the lowest spawner abundance from which the stock recovered, is calculated as the mean estimated abundance for the period 1996 to 2000, which was 4,500 spawners (Figure 3.2; Table 6.3a). This contrasts sharply with the values for spawners at 40%B_{MSY} and spawners that produce half of asymptotic equilibrium abundance. For model 5 with M inferred from observations, these candidate LRPs equal 700 thousand and 510 thousand spawners, respectively (Table 6.3a). There is large uncertainty in these estimates. LRP values from model 4 with M informed from observations are lower by just under half compared to model 5 values, 420 thousand and 300 thousand, for 40%B_{MSY} and half asymptotic abundance respectively (Table 6.3b).

Differences in reference point values in currencies of fish between the two scenario assumptions on M are consistent with the consequences to the age structure of the spawners as affected by fishing and conditioned by assumptions on M. For example, the 40%B_{MSY} spawner abundance for the scenario with M = Z in model 5 is less than half the value for the scenario with M informed from observations (Table 6.3a).

Spawner per recruit fishing rate reference values at 30%SPR and 50%SPR are higher for the model with higher values of M (Table 6.3a). F at 30%SPR is higher than F_{MSY} for both scenario values of M. For this management strategy without size limits on retention, fully-recruited F at MSY is 0.17, compared to F = 0.12 for 50%SPR, and F = 0.24 for 30%SPR (Table 6.3a). Fishing rate reference values are similar for model 5 and model 4 (Tables 6.3a, 6.3b).

6.2.5.2. Reference points dependent on fishing strategy

Fishing strategies have a consequence on the reference point outcomes in terms of numbers of fish because fishing changes the age structure of the population at equilibrium relative to the unfished condition. Reference point values based on the life history and population dynamics parameters of model 5 are summarized in Table 6.4a and values for model 4 are presented in Table 6.4b, both with the assumption on M for ages 4 to 15+ informed from observations. Three potential fishing strategies are contrasted with all three excluding catch and release mortality.

The choice of the USR can be based on objectives related to fishery outcomes, consistent with principles of the Precautionary Approach which states that the USR value would be determined by productivity objectives for the stock, broader biological considerations, and social and economic objectives for the fishery (DFO 2009). The values from model 5 corresponding to 80%B_{MSY} range from 940 thousand to 1.2 million spawners, dependent upon the fishing strategy with near complete overlap of the 5th to 95th percentile ranges among the three fishing strategies. For model 5, the USR corresponding to 80%B_{MSY} ranges from 960 thousand to 1.2 million spawners (Table 6.4a). For model 4, the USR corresponding to 80%B_{MSY} ranges from 570 to 720 thousand spawners, dependent on fishing strategy (Table 6.4b).

Other than spawners for 40%B_{MSY}, the candidate LRPs examined and corresponding to the life history characteristics of Striped Bass from the southern Gulf, are generally invariant to fishing strategy. Brecover is not affected by fishing strategy, being based upon similar years of abundances and independent of fishing strategy simulations. The eggs for half of asymptotic abundance to age 3 are unaffected by fishing strategy because it is assumed that eggs are equivalent regardless of age of spawners and fish younger than age 3 years are generally not subject to fishing mortality and are not spawners. Differences in spawner numbers for half Beverton-Holt K and half equilibrium asymptotic abundances among the fishing strategies are due to the effects of fishing that modifies the age structure of spawners toward younger ages. Spawners for half of asymptotic equilibrium abundance from model 5 are approximately 500 thousand spawners, with large uncertainty, such that there is essentially no difference in the number of spawners among the fishing strategies (Table 6.4a). For model 4, spawners for half asymptotic abundance are quite similar among fishing strategies, rounded off to 300 thousand spawners (Table 6.4b).

We cannot make a compelling argument for using spawner abundance at F corresponding to 30%SPR as a LRP. In these analyses, spawner abundances at 30%SPR are higher than spawner abundance at 80%B_{MSY}.

Fully-recruited fishing removal reference values are very dependent on the fishing strategy and any choice of a removal rate reference would be specific to the fishing strategy for the stock. The exploitation rates on total recruits, aged 3 to 15+, vary from 14% with no size restrictions, to 18% for the slot limit of 47 to 61 cm fork length to 20% for the maximum size limit of 65 cm (Tables 6.4a, 6.4b).

6.2.5.3. Reference points accounting for catch and release mortality

The effects of including or excluding catch and release mortality on MSW reference values are generally inconsequential in these analyses given the large uncertainties in population dynamics (Table 6.5). The only exception is the estimate of F_{MSY} for the management strategy with a slot size of 47 to 61 cm which is higher when catch and release mortality is excluded compared to when it is included (Table 6.5).

When catch and release mortality is included, MSY values are lower than if catch and release mortality is excluded, i.e., similar to assuming higher natural mortality on the population (Table 6.5). Of the two management scenarios examined that have catch and release implications, the scenario with a slot size of 47 to 61 cm fork length has the largest proportional loss of fish through catch and release and the largest relative decrease (14%) in the retained catch at MSY. The retained catch represents 86% of total fishery losses for the management strategy with a slot size of 47 to 61 cm, 97% for the strategy with a maximum size limit of 65 cm, and no effect for the management strategy without size limits for retention.

Catch and release effects as modelled here do not fully account for recreational fishing practices in the southern Gulf and would underestimate the consequences of the practice on the resource. The recreational fishery for Striped Bass in the southern Gulf has a large component of catch and release, in part due to the mandatory slot size restrictions for retention and the fishing practices of individual anglers that favour a lot of angling activity without intent to retain. There is a community of recreational users that practice catch and release regardless of the retention allowances; they will catch and release fish that are within the retention limits and at peak periods of aggregation during the spring and fall some anglers have reported catching and releasing upwards of 100 fish or more per daily fishing trip (see Section 2.2). The analysis of consequences of these fishing practices on MSY and other reference values would require a different model and data inputs.

6.3. CONCLUSIONS ON REFERENCE POINTS

6.3.1. MSY and SPR Based Reference Points From Population Modelling

We used equilibrium modelling to explore candidate reference points based on life history and population dynamics parameters informed from a population model for Striped Bass of the southern Gulf of St. Lawrence. A priori, two population models with a Beverton-Holt stock and recruitment function with density-dependence occurring between the egg and age-0 life stage in the summer are considered for estimation of MSY reference values. The two models differ in the prior assumptions for the density independent survival from age-0 to age-3, 3 year olds being the first age group that is monitored as spawners. The two model variants provide similar estimates of lifetime reproductive rate to age-3 in the absence of density dependent compensatory mortality, however the estimates of carrying capacity at age 3 differ by a factor of two between the models. This has consequences on the derivation of reference points and we present candidate reference point values for both models (Figure 6.8) and suggest a choice of reference points based on population trajectory over the past two decades and the risk to population sustainability and persistence.

Information on natural mortality (M) at age is crucial in the equilibrium model and reference point calculations. The expectation from life history theory is that natural mortality is inversely related to size, and hence age. Based on sequential observations of acoustically tagged and tracked Striped Bass, instantaneous natural mortality for adult bass >= 47 cm fork length is concluded to be less than 0.2, equivalent to an annual survival rate of 0.82 or higher. Population modelling also indicates a relatively high annual survival rate of 0.77 (median) for Striped Bass aged 8 and older but with large uncertainty (5th to 95th percentile range 0.44 to 0.93). A relatively high survival rate (median = 0.67; percentiles range 0.47 to 0.86) is estimated for fish at age 3 years, an age and size group that may be outside the size preference for retention in historical and contemporary fisheries. For purposes of equilibrium modelling and MSY reference calculations, M for Striped Bass aged 4 and older is assumed to be 0.18 (5th to 95th percentile range of 0.13 to 0.28) and M for younger ages are taken from population model estimates.

Fishing strategies can have a consequence on the reference point outcomes in terms of numbers of fish because fishing changes the age structure of the population relative to the unfished condition. An USR point conditional on a fishing strategy is consistent with principles of the Precautionary Approach which states that the USR value could reflect socio-economic considerations, for example reference points that consider maximizing yield, in terms of weight or in terms of number of fish harvested. Of the two USR candidates discussed above, the spawner abundance corresponding to 80%B_{MSY} has been most frequently used in fisheries management and examples from marine fish and invertebrates assessments and management abound.

To conform to the principles of the PA policy, the LRP should be determined by biological considerations and thus preferably be invariant to fisheries exploitation strategies. LRP candidates including 40%B_{MSY} and abundance at 30%SPR are not invariant to fishing strategy. Candidate LRPs that are invariant to fishing include Brecover (although not entirely) and reference points associated with egg abundances that result in half of Beverton-Holt carrying capacity or half of maximum asymptotic abundance of recruitment at age-3.

Brecover, the lowest historical spawner abundance that did not prevent rebuilding of the population, is quite clearly the low spawner abundances estimated during 1996 to 2000, at a mean value of just under 5,000 spawners (Figure 3.2; Table 6.4). The fact that the Striped Bass population of the southern Gulf was able to monotonically increase from those low abundances to several hundred thousand spawners in less than 20 years reflects the improved survival conditions of juvenile and adult Striped Bass over this period. The carrying capacity for this population, as estimated from modelling assuming a Beverton-Holt stock and recruitment relationship with $M < 0.2$ for bass aged 4 and older and no fishing is estimated to be 2.7 to 4.7 million fish aged 3 to 15+, with the abundance of spawners at 1.8 to 3.1 million fish, dependent on model (Tables 6.2a, 6.2b). A Brecover value of 4,500 spawners represents 0.1% to 0.2% of this estimated unfished equilibrium value (B_0), substantially less than proposed LRP values equivalent to 20% of the unfished abundance at equilibrium (Myers et al. 1994; DFO 2009). A reference value equivalent to 20%B_0 would be in the range of 360 to 620 thousand spawners. Despite the abundance of spawners having been as low as 5 thousand spawners in recent history, given the indications of the potential size of this unfished population Brecover does not seem appropriate.

Candidate LRPs defined in terms of spawners or eggs that result in half asymptotic abundance (Myers et al. 1994) have been applied to Atlantic Salmon populations in eastern Canada (DFO 2015b). These candidate LRPs can be invariant to fisheries management strategy if the recruitment stage being maximized is not subject to fishing mortality and if the spawning stock is expressed in terms of eggs. They are however modified by fishing strategy when expressed in terms of number of fish. This is because fishing strategies modify the age and size structure of the spawning population; regardless of strategy, fishing disproportionally reduces the relative abundance of older fish resulting in a younger mean age of spawners and consequently fewer eggs per spawner. The LRP and the assessment of attainment of the LRP could be presented in currencies of eggs. This is a trivial exercise for the most part as biological characteristics of the spawners have been obtained annually and the quantity of eggs spawned could be calculated using the same life history characteristics as were used to derive the reference points.

F_{MSY} and F at 50%SPR are potential candidate removal rate references but their values depend on the fisheries management strategy. These removal rate references are expressed in terms of fully recruited instantaneous fishing rates which are not easily understood. The fully recruited fishing rate values were converted to exploitation rates, calculated as the ratio of catch at MSY to total abundance at MSY for ages 3 to 15+. The fishing strategy without any size limits has the lowest exploitation rate at F_{MSY} of 14%, whereas the strategy with a maximum size limit of 65 cm fork length result in an exploitation rate at F_{MSY} of 20%, with an intermediate rate of 18% for the strategy with a slot limit of 47 to 61 cm fork length (Table 6.4). Exploitation rate at F_{MSY} for the three fishing strategies of this population of Striped Bass is at or less than the assumed annual natural mortality rate of 18% ($1\text{-}e^{-M}$).

6.3.2. Proxy Reference Points Based on Traffic Light Approach

The Striped Bass stock of the southern Gulf of St. Lawrence has demonstrated a monotonically increasing abundance trajectory, with an annual rate of increase during 1996 to 2019 of 25%. Candidate reference points based exclusively on past observations and independent of a population dynamic model are attractive. Note that Brecover is such a reference point. However, such reference point definitions are dependent on a number of less desirable considerations including whether the data are log transformed, the time series of observations considered, and the inferences may be sensitive to outlier / exceptional observations. Overall, reference points defined on observations which are log transformed prior to cluster identification were less dependent upon time series considerations (Figure 6.1) and less sensitive to exceptional high or low observations (Figure 6.2) than those based on the natural scale. Using the entire time series of observations, the proxy reference points derived from log-transformed data are 13 thousand and 105 thousand for the LRP and USR respectively. Based on the natural scale of observations and conditional on there being at least five observations within individual clusters, which is the case when the 2017 value is excluded, the LRP and USR are 44 thousand and 181 thousand spawners, respectively (Figures 6.1, 6.2).

This is not a good approach as the decisions on scale of data to use are subjective and there is instability in reference values as additional years are added. The approach may have more utility if the time series of observations included the full range of potential abundances of the stock to define the groups

6.3.3. Summary of Candidate Reference Points and Corresponding Stock Status

The fishery decision-making framework that incorporates the precautionary approach (DFO 2009) was developed to guide management of fisheries exploitation in order to reduce the risk of the stock falling into the critical zone and that promotes growth of the resource into the healthy zone. As the intention within the policy is to avoid the stock falling to the LRP and the critical zone, the objective is not to manage the stock to the LRP.

The proposed candidate reference points in terms of eggs and approximate spawner abundance number equivalents are summarized in Tables 6.4a, 6.4b (DFO 2020). Consistent for both model 4 and model 5, the model derived USR value is two times the LRP value. The stock status relative to these model derived reference points, over the period of assessment 1994 to 2019 is shown in Figure 6.6. The spawner abundance has been in the healthy zone only once (in 2017) and dependent on the model, the spawner abundances were either above the LRP and below USR (model 4) or at approximately the LRP (model 5) since 2013.

There is no consensus LRP value from the two retained models; whereas the modelled LRP values are 17.3 billion eggs, equivalent to 330 thousand spawners from model 4 and 30.0 billion eggs, equivalent to 560 thousand spawners from model 5. Based on the trajectory of the population over the relatively short period of assessment, maintaining a spawner abundance that exceeds 330 thousand spawners should be more than sufficient to avoid serious harm to the population.

The carrying capacity for the Striped Bass population from the southern Gulf of St. Lawrence is unknown. Modelling informed by observations from this population suggests total abundances of age-3+ Striped Bass at B_{MSY} of 1 to 2 million fish with abundances at B_{MSY} of 860 thousand to 1.5 million spawners. Potential removals when the stock is at B_{MSY} are in the range of 200 to 400 thousand fish annually.

As an alternative, the posterior distribution of the spawner assessed values could be used to assess the probabilities of the spawner abundances being below the LRP or above the USR. From looking at the distribution of boxplots relative to the point estimates of the LRP and USR in Figure 6.6, one can see that the probability of the assessed spawner abundance being below the LRP is > 75% for all years except in 2017 for the reference derived from model 5 but the probability is just under 50% since 2015 for the LRP derived from model 4. Similarly, the probability that the spawner abundance was above the USR in 2017 was just over 50% for model 4 but < 75% relative to reference points from model 5. This interpretation of status that incorporates the uncertainty in the assessed abundance relative to point estimates of reference points would conform to the directives of the Precautionary Approach policy for characterizing uncertainty and risk.

In the eastern US Striped Bass assessment, a number of reference points have been defined and used to assess the status of the stock. A spawning stock biomass reference point ($SSB_{Threshold}$) is defined as the assessed female SSB for 1995 when the stock was declared recovered with an expanded age structure. The revised value from the most recent assessment is an $SSB_{Threshold}$ value of 91,436 t (NEFSC 2019). An SSB_{Target} is also defined, equivalent to 125% of the female $SSB_{Threshold}$, equivalent to 114,295 t (NEFSC 2019). Fishing mortality threshold and target values are also defined based on the fishing rate applied to the current estimate of SSB that results in $SSB_{Threshold}$ and SSB_{Target}. These values, from the recent assessment, are $F_{Threshold} = 0.24$ and $F_{Target} = 0.20$.

6.3.4. Guidance on Choice of Reference Points and Management Strategies

The first consideration for the development of the PA framework is the definition of the LRP. The recent fisheries management history is informative of the management decision making process and provides insights into what could be a publicly acceptable LRP. Fisheries access was responsive to the rebuilding of the Striped Bass population beginning initially with the re-opening of the Indigenous FSC fisheries in 2013, the retention recreational fisheries in 2014, and a pilot commercial fishery in 2018. The re-opening of the Indigenous fishery occurred following the conclusion that the population had first met both the limit and target recovery objectives in 2011, at a median abundance of 200 thousand spawners and a 5th percentile value of 90 thousand spawners (DFO 2013), values of abundance corresponding to the LRP value from one of the models and consistent with a harvest decision rule that allows fisheries exploitation when the stock is above the LRP. A cautious recreational fisheries strategy (two short retention seasons, 1 fish per day, slot size limit of 55 to 65 cm TL) was chosen in 2014 following on the 2013 median spawner assessment value of 250 thousand fish. Further increases in abundance in 2015, to a median estimate of 300 thousand spawners, resulted in an extended retention period in the recreational fishery for 2016. The largest change in the recreational fishery occurred in 2018 with an authorization to retain 3 fish per day within a slot size of 50 – 65 cm TL; this increased access followed on the exceptional return estimate in 2017 of just under 1 million spawners. The pilot commercial fishery was also first authorized in 2018.

The risk to the Striped Bass stock of an underestimate of the LRP from either the population models (330 to 560 thousand) is considered low. The lowest spawner abundances of the late 1990s did not preclude the rebuilding of the population at an average rate of 25% per year. Curtailing fishing mortality was an important factor in this rebuilding, with assessed abundances of recent years that are almost two orders of magnitude higher than the lowest assessed values of the late 1990s. This increase in abundance was sustained even with increased fisheries access beginning again in 2013. However, Brecover is not prudent as a LRP, given that its value of < 5 thousand spawners is less than 1% spawners at B_0, regardless of the population model considered, and would certainly place it in the at risk criterion for small population size used by COSEWIC. COSEWIC (2004) assessed the Striped Bass population as threatened despite the more recent abundances at that time that exceeded 20 thousand spawners.

An USR value of 720 thousand to 1.2 million spawners is seemingly within the scope of potential spawner abundance for this population. A healthy stock would minimally be at a population abundance that exceeds 720 thousand spawners (80%B_{MSY} under model 4). This may be an underestimate of the production potential of this population, as indicated by outputs from model 5, however full exploitation to rates equivalent to F_{MSY} and potential removals at MSY (C_{MSY}) would likely only be considered once the trajectory of the population abundance had placed it in that healthy zone. When this does occur, a re-assessment of population dynamics with additional observations could be undertaken to determine the appropriateness of the defined USR. The 2017 value of just under 1 million spawners was exceptional, and the decline in 2018 and 2019 to estimated values of just over 300 thousand spawners provides a cautionary note on the variations in size of the stock under new population dynamics conditions (extensive migration of Striped Bass beyond its historic distribution range with associated mortalities) and increasing fisheries exploitation. Some of the annual variation in abundance estimates are also likely related to the difficulties and uncertainties in assessing the abundance on the spawning grounds, i.e., year effects.

At a LRP value of 330 thousand spawners and an USR value of 720 thousand spawners, we note that increasing fisheries access on Striped Bass from the southern Gulf has been provided during a period when the stock has been situated in the cautious zone (with exception of 2017) but with a trajectory of increasing abundance towards the healthy zone.

If the assessed abundance was to increase above a proposed USR value of 720 thousand fish, this may result in requests for new and alternative fisheries access. The fisheries exploitation potential on this species is high. Historically and even now, Striped Bass are readily captured in large numbers in gaspereau trapnets in the spring during the spawning aggregations in the Miramichi; catch rates (fish per trapnet per day) in 2017 exceeded several thousand fish per net haul (Figure 3.1). Striped Bass are also reportedly captured in gaspereau fishery trapnets in other estuaries of DFO Gulf New Brunswick.

The recreational fishery is increasing in popularity throughout the Gulf of St. Lawrence including into the western portion of the Gulf. The current recreational fisheries management plan for Striped Bass in the Gulf is very generous, i.e. aggressive, relative to management of the Striped Bass stocks of the eastern US. The retention season extends from mid-April to the end of October, essentially the open water season, with a daily retention and possession limit of three bass within a defined slot limit. In the eastern US, there is a diversity of management measures tailored to stock units and management sectors, with fishing area specific seasons, daily limits and size limits for retention, however, daily retention limits along the entire eastern US seaboard are either one or two fish per day (ASMFC 2019). In the southern Gulf Striped Bass fishery, the pool of recreational anglers is unknown and unrestricted since there is no licence requirement to fish in tidal or marine waters. Relaxing the slot limits may provide more opportunity for individual anglers to retain the daily limit however in the absence of catch and effort data and monitoring of the recreational fishery, it is not possible to assess the extent to which the current recreational fisheries rules are limiting the harvests of Striped Bass in the recreational fishery.

A slot size is currently used in the recreational (and pilot commercial) fishery for Striped Bass which inevitably leads to catch and release of fish that are outside the slot for retention. Catch and release fishing is likely to be practiced regardless of size limit strategies. According to creel survey data and from anecdotal reports, some anglers in the southern Gulf will release upwards of 100 fish or more in a daily fishing trip particularly when bass are aggregated prior to or at spawning time in the Miramichi River. A catch and release mortality rate of 9% is used in the coastwide Striped Bass assessment of the US but it is recognized that the mortality rate depends upon fishing gear, water temperature, maturity state and angler practices (Millard et al. 2005; NEFSC 2019). When examined in these analyses, the consequence of including or excluding catch and release mortality on the development of MSY references and reference points was inconsequential; reference point values were indistinguishable between fisheries strategies (Table 6.5) due to the large uncertainties in the estimated population dynamics parameters. That does not mean however that catch and release has no effect on survival and abundance of Striped Bass. The mortality consequences of the catch and release fishery are unknown since there are no estimates of catches or harvests in the recreational fishery for Striped Bass (DFO 2011). In addition, a large amount of catch and release fishing occurs on fish during a stressful period as they come out of a winter fast and are physiologically switched to spawning.

The intent of the slot size is to a) to provide an opportunity for the fish to spawn once before being vulnerable to retention, and b) to protect older fish with high fecundity and hence guard against successive year classes of poor recruitment. Gwinn et al. (2015) discussed fishing strategies for competing objectives of different fishery users, as for example, when the number of fish harvested, rather than total weight, is the fishery preference. This could be the case in recreational fisheries where the preference is access to a high number of acceptably sized fish rather than maximizing the weight of fish captured; the latter objective may be more relevant for commercial fisheries. Gwinn et al. (2015) concluded that a slot size was superior to a minimum size strategy as a compromise regulation for achieving these competing objectives. Ahrens et al. (2020) assessed the performance of minimum size and slot size strategies relative to competing conservation and fisheries objectives and concluded that harvest slots were the optimal harvest regulation under multiple fisheries objectives (biomass yield, yield in number, trophy catch, and catch rates). The tradeoff between yield in weight and yield in number is shown in the results of the equilibrium analysis of fisheries strategy effects (Table 6.4); the slot size of 30 to 65 cm FL results in the highest catch number but the lowest catch weight at MSY of the fishery strategies examined. Indeed the lower the minimum size, the more yield in number can be extracted. Of importance in the discussion about reference points is that, with the available data and models, there was no difference in the reference point outcomes among the three management strategies examined. The uncertainty intervals greatly overlapped among the fisheries strategies, however, this would not be the case if the population dynamics information was more precisely known.

Based on the currently available information on the proportion female by fork length (Figure 3.7), a minimum slot size of 47 cm FL provides substantial protection from harvesting for male bass but less protection for females. A minimum size of 55 cm FL would provide better protection to first spawning of female bass.

The protection of larger and older Striped Bass, achieved through a maximum size for retention, is important for several reasons. Although it was assumed in our analyses that fecundity of Striped Bass is a linear function of weight and that egg value was similar regardless of female size, it has been widely discussed in literature that maternal effects on early life stage survival and recruitment are important in fish and in particular the value of older and larger females in the spawning population may be disproportionate to their numerical egg contribution (Barneche et al. 2018). The combination of high fecundity and iteroparity of Striped Bass are indicative of a species with high mortality in the early stages. Inter-year class variability in Striped Bass has been observed to be high, largely determined during the egg and larval stages and influenced by environmental factors (see references within Richards and Rago 1999; Uphoff 1989; Rutherford et al. 2003). Hence the importance of maintaining an abundance of older and larger spawners to take advantage of intermittent favourable environmental conditions that can produce large year classes, which can be realized with a maximum size limit fishing strategy. A maximum slot size of 61 cm FL reduces the selectivity to the fisheries to values less than 10% for Striped Bass 8 years and older (Figure 6.3).

6.3.5. Multi-Species Considerations

DFO (2019) developed a policy to support rebuilding plans under the precautionary approach framework for stocks that are in the critical zone. DFO (2019) states that in cases where rebuilding of a stock has the potential to negatively impact the status of another, as in the case of rebuilding a predator species that could result in a decline of a prey species, rebuilding objectives need to be carefully developed through a balanced approach to ensure neither is depleted to a point of serious harm. Most importantly DFO (2019) acknowledge that it is not possible to simultaneously achieve yields corresponding to MSY predicted from single-species assessments for a system of multiple, interacting species and rebuilding efforts should be approached within an ecosystem context to the extent possible.

The reference points and management strategies discussed in this working paper are based on single species management approaches for the purpose of optimizing utility functions specific to Striped Bass. The Striped Bass population of the southern Gulf has increased in abundance, out of the critical zone as presently proposed. Striped Bass is large bodied and a piscivorous predator of other valued anadromous fisheries species in the southern Gulf of St. Lawrence. Concerns have been expressed by Atlantic Salmon fishery advocates as well as some gaspereau and Rainbow Smelt commercial fishery interests that the rebuilding of Striped Bass stock in the southern Gulf has contributed to declines in abundances of Atlantic Salmon and other diadromous species because of high levels of predation on these species by Striped Bass. Similar concerns were expressed about the impact of the recovered Atlantic Coast Striped Bass on its prey-base and NEFSC (2019) summarize a number of analyses that examined the potential for Striped Bass to deplete prey populations along the Atlantic Coast. To date, no multi-species reference points or management plans have been proposed for the US situation.

One of the objectives of this review was to consider approaches and potential reference points for Striped Bass that take account of these ecosystem considerations. This objective is considered by Chaput (2022). The cautionary note from DFO (2019) is worth repeating here: it is not possible to simultaneously achieve yields corresponding to MSY predicted from single-species assessments for a system of multiple, interacting species. Thus, any multi-species management approach will be a compromise of competing single species objectives.

7. UNCERTAINTIES AND KNOWLEDGE GAPS

Although there are substantial empirical observations to characterize the life history parameters of the population of Striped Bass from the southern Gulf including the weight at length relationship, the size at age relationship, and mortality rates, a number of knowledge gaps and uncertainties remain.

7.1. ASSUMPTIONS AND OBSERVATIONS OF LIFE HISTORY

7.1.1. Size at Age Information

Age of Striped Bass in this population is determined based on interpretations from scales. Age interpretations from scales are considered to be sufficient for fish that are less than 8 or 10 years old whereas otoliths are considered more reliable at estimating the age of older fish (Secor et al. 1995; Liao et al. 2013). The oldest age interpreted using scales from samples of the southern Gulf to date is 15 years. The oldest reported age of Striped Bass in eastern US seaboard is 31 years (NEFSC 2019). If scale age interpretations underestimate the ages of Striped Bass, then the growth rates from the von Bertalanffy model would be overestimated, which would have the consequence of underestimating the abundance of older fish in the population and underestimating the fishery selectivity at age profiles. The consequences of this bias on modelled estimates of total mortality and subsequently on derivation of reference points has not been examined. There is limited information from tagging and recaptures of Striped Bass that validates the relatively slow growth rate of fish after age 7: for example a Striped Bass tagged in 2006 measuring 67.6 cm with an age interpretation of 7 years was recaptured and sampled in 2013 and was measured as 83.7 cm, an increase of 16 cm over 7 years (DFO 2014). Other tag and recapture data can provide validation for the growth rate of bass of different sizes and ages from the Miramichi.

The longest recorded Striped Bass from sampling in the Northwest Miramichi is 116 cm fork length. There are anecdotal reports of catches of very large bass in the southern Gulf of St. Lawrence. In the eastern US populations of Striped Bass, fish exceeding 180 cm total length are not considered exceptional (NEFSC 2019). Size distributions of spawners are described from sampling of bycatches of Striped Bass from commercial gaspereau fishery trapnets and at DFO index trapnets in the Northwest Miramichi (DFO 2020). The commercial gaspereau and DFO trapnets are not considered size restrictive; catches of large bodied Atlantic salmon exceeding 100 cm fork length are frequent and there are a few recorded catches of Atlantic Sturgeon in the 4 foot (120 cm) length range. The trapnets are set from shore and do not cover the deeper channel areas of the Northwest Miramichi. If larger and older Striped Bass preferentially use these deeper areas, then they would not be available for capture in the trapnets. The extent of the potential undersampling of larger fish is not known but there is some evidence that this not an important issue. Since 2015, there has a Striped Bass fishing derby in the Miramichi River in late May that targets the pre-spawning and spawning aggregations of Striped Bass. Tournament participants are allowed to high grade the catches before submitting them for registration and so those catches would be biased to the larger fish angled by parties. Extensive fishing effort by recreational fishing parties in 2019 recorded some catches of relatively large bodied fish. The data provided by the tournament organizers was aggregated by fishing party into total weight and number of fish. Based on these data, the highest mean weight per fish recorded was from an aggregate of two fish weighing 22 kg giving a mean weight of 11 kg which would be equivalent to an average fork length of 97 cm. Of the 262 fish submitted by parties, approximately 10 were estimated at mean lengths exceeding 90 cm and 70% of the average weights of fish were less than 6 kg (equivalent to 79 cm fork length).

There is evidence from literature that growth rate and size at age profiles differ for males and females, particularly after the attainment of maturity for which males are comparatively smaller at age than females (Chaput and Robichaud 1995; NEFSC 2019). The population model used in this study does not track abundance at age by sex nor does the assessment model for Striped Bass for the eastern seaboard of the US. At least in the context of estimating egg production at age, it is the mean size at age of females which would be important and using a growth function that ignores sex would result in an underestimate of size at age for females, and therefore eggs at age if females are larger at age than males. Selectivity at age to the fishery, used in equilibrium modelling to derive MSY reference values, would also be affected by differences between the sexes in growth rate and size at age. The amount of bias introduced to the estimates of spawner abundances at age and to the equilibrium model assumptions of ignoring differences in size at age by sex is not known. Incorporating differences in size at age by sex would require a different model structure from one used in this analysis.

Estimation of the von Bertalanffy growth parameters was based on samples of length and age of Striped Bass collected in May and June with the majority sampled from the spawning area in the Northwest Miramichi. Within an age group, if the probability of maturing is size dependent, with faster growing fish maturing earlier, then the use of size and age data from samples of spawners may result in an overestimation of size at younger ages, particularly ages 3 and 4. The consequences of this sampling bias on von Bertalanffy model growth parameters has not been examined.

7.1.2. Age at Maturity and Proportions of Recruits on the Spawning Grounds

There are no data with which to directly estimate the age or size at 50% maturity because there are no representative samples of bass at all states of maturity in the spring. The maturation schedule of male and female bass was assumed with males maturing earlier than females. The earlier maturation at age of male bass is supported by observations of the sex ratio of fish on the spawning grounds which indicate a predominance of males at age 3 and age 4 and equal male to female proportions for fish age 6 and older. In the population model, the parameter that is estimated is the proportion of the recruits at age that are on the spawning grounds. This parameter is a combination of proportion at age that are mature by sex and the proportion of mature individuals by sex that are spawners in the Miramichi. If such information was available and there was evidence of differences between males and females, then this could be considered but it would require a different age structured model than the one considered here.

The proportion female at age is assumed known in the model and is calculated directly from the assumed maturation profiles of male and female bass. This proportion seems appropriate as it corresponds to the proportion female at length from sampling of fish in May and June of 2013 to 2015.

It is assumed and modelled that not all mature Striped Bass are on the spawning grounds in the Northwest Miramichi. This inference is supported by observations of Striped Bass, some in ripe condition (males and females), in other estuaries of New Brunswick and Nova Scotia in May and June. It also includes the phenomenon of skipped spawning in which fish forego egg production until the subsequent year. Skip spawning has been reported in eastern US Striped Bass populations (Secor 2008; Gahagan et al. 2015; Secor et al. 2020) and inferred for fish from the Miramichi that had been detected off the coast of Labrador in 2017, had returned and overwintered in the Miramichi in 2017/2018 and subsequently based on behaviours from acoustic tag detections had left the Miramichi in early spring 2018 prior to spawning. These fish survived, overwintered in Miramichi in 2018/2019 and were inferred to have spawned in 2019 but not in 2018.

7.1.3. Assumptions of Fecundity at Age

The fecundity at age used in the model is a coarse approximation of fecundity values reported elsewhere. There have been efforts to collect fecundity estimates from the southern Gulf of St. Lawrence population but the analysis of these data is incomplete. Bias in the assumed fecundity at age values would bias the estimation of the Beverton-Holt stock and recruitment parameters; if fecundity was underestimated, this would result in a positive bias for the slope at the origin whereas if fecundity was overestimated, there would be the opposite effect. The direction of bias of the assumed fecundity values relative to population specific fecundities for this population is not known.

7.2. ASSUMPTIONS ON NATURAL MORTALITY AND CONSTRAINTS

Natural mortality (M) rates are difficult to estimate in most circumstances.

The acoustic tagging and tracking data provide estimates of total mortality of larger Striped Bass. It is recognized that those estimates may also include some fishing related mortality however the estimates of Z at a median value of 0.22 in recent years is strongly indicative that instantaneous natural mortality is very likely no higher than 0.2. The natural mortality value of 0.15 used in the assessment of Striped Bass on the eastern seaboard of the US is lower than what assumed in these analyses. However, there is good reason to expect natural mortality to be higher in this northern population of the southern Gulf of St. Lawrence. Douglas et al. (2006) provided information on factors that could contributed to non-fisheries related mortality.

The environment, in particular during the winter, is an important driver of the population dynamics of Striped Bass in the southern Gulf St. Lawrence. As stated in the introduction, the southern Gulf of St. Lawrence Striped Bass population is the only population where avoidance of lethal marine conditions (sub-zero water temperatures) during winter is an obligate element of its life history. The southern Gulf of St. Lawrence is a geographic region in which the coastal and estuary surface waters freeze during the winter. Rainbow Smelt, Atlantic Tomcod, and Atlantic Herring (juveniles) can produce anti-freeze proteins which lowers the freezing point of the blood thus allowing these fish to overwinter in the nearshore areas. Striped Bass do not produce these proteins and hence must overwinter in the upper estuaries near the hide of tide where the water temperatures remain above 0 ºC.

Douglas et al. (2006) identified winter thermal plumes associated with industrial infrastructure in the southern Gulf as potential contributors to winter mortality of Striped Bass. Large numbers of Striped Bass were regularly drawn to the thermal effluents of the power generating station at Trenton (NS), Dalhousie, and Belledune (NB), during late fall and winter and anglers targeted these warm water effluents because of the large concentrations of Striped Bass which seemingly continued to feed at that time of year (Douglas et al. 2006). Well over 1,000 striped bass were estimated to have died at the outflow of the Trenton (NS) station in February 2004. The cause of the fish kill was believed to be the result of an acute reduction in water temperature when the power generating station went off line and the thermal discharge was turned off (Douglas et al. 2006). Buhariwalla et al. (2016) provide details of a similar fish kill that occurred in January 2013 for the same reason; the maximum daily water temperature recorded at the discharge point before the fish kill was 12.8 °C but declined to -2.5 °C during the cold-shock event three days later. The Dalhousie NB generating station which was identified by

Douglas et al. (2006) as another source of thermal effluent utilized by Striped Bass was demolished in 2015. The thermal generating station at Belledune (NB) remains operational. There were other thermal plumes in the Miramichi, associated with pulp and paper mill discharge in the lower portion of the Northwest Miramichi; that mill closed permanently in December 2007.

Striped Bass also fast during the overwintering period and Striped Bass mortalities have been reported in some estuaries and rivers soon after ice-out. Bradford and Chaput (1998) indicated that there had been reports of Striped Bass mortalities in April and May 1997, particularly from the Richibucto River area, shortly after ice-out. The absolute number of losses in the spring of 1997 was not quantified however one mortality was examined by the DFO Fish Health Laboratory (Moncton) and no bacterial pathogens were isolated. Bradford et al. (2001b) reported that dead and moribund striped bass sampled on the Napan River (Miramichi Bay tributary) during early May 1997 were emaciated in appearance, devoid of visceral fat deposits, and exhibited atrophied digestive tracts, suggesting fish had starved.

We have no information on the natural mortality rate of young bass. Natural mortality for young age groups, 0 to 2 years, is expected to be relatively high and a general relationship relating growth parameters from von Bertalanffy relationship to M was used to provide informative priors for population modelling. High M for juvenile Striped Bass is expected because of their small body size which makes them vulnerable to a diversity of predators including Striped Bass in some circumstances (Buhariwalla et al. 2016). Small bodied fish are also more susceptible to overwinter mortality; small bodied fish may have insufficient energy reserves to survive the overwintering fast period that can extend from late October to late April. Harsh environmental conditions can also lead to mortalities, juvenile bass have been observed frozen in the ice (S. Douglas, DFO, pers. comm.).

Reductions in the intensity of a number of anthropogenic stressors likely contributed to improved survival which assisted in the rebuilding of abundance of Striped Bass. The reductions include the elimination of at least two (Dalhousie, Miramichi) thermal effluent discharges. Waste water effluents from industrial and municipal facilities are widespread throughout the southern Gulf, but their effect on striped bass or striped bass habitat is unknown (Douglas et al. 2006). Sites of particular interest in the southern Gulf were reviewed by Robichaud-LeBlanc et al. (2000). Burton et al. (1983) demonstrated significant mortality of striped bass larvae after a 72-h exposure to bleached kraft mill effluent. The number of industrial facilities discharging chemical effluents in the southern Gulf that were identified in the recovery potential assessment of 2006 (Douglas et al. 2006) has been reduced. The facilities which have closed include the paper mill at Dalhousie (NB), two mills in the Miramichi River, and more recently a mill at Pictou (NS).

7.3. ASSUMPTIONS OF STOCK STRUCTURE

There is compelling evidence that the Northwest Miramichi River is the major spawning area of for the Striped Bass population of the southern Gulf of St. Lawrence. Through the years, DFO has reported on the tagging of Striped Bass in various rivers of the southern Gulf and their subsequent recaptures on the spawning grounds of the Northwest Miramichi (DFO 2014). There have been consistent detections in the Northwest Miramichi of bass acoustically tagged from the Gaspe region and from Pictou (NS) illustrating the wide distribution range of Striped Bass in the southern Gulf and the affinity to the Miramichi River spawning area (see Table 3.8). Striped Bass tagged from the eastern boundary of the southern Gulf (Margaree River NS) to Gaspe on the western edge of the southern Gulf and locations in between have subsequently been recaptured in the Northwest Miramichi, strengthening the evidence of broad regional distribution of fish in the southern Gulf. Added to this, the evidence on the outdispersion from the Northwest

Miramich and the distribution of juveniles in other estuaries and rivers, makes the Northwest Miramichi spawning area the most important feature for production of Striped Bass.

Alternative historical spawning areas in the southern Gulf of St. Lawrence have been advocated in literature (Rulifson and Dadswell 1995; Andrews et al. 2019a) although there is no published evidence to date of annual spawning and successful recruitment from these locations. In the past two years, corresponding to a period of high Striped Bass spawner abundance, non-government organisations sampled and reported the presence of Striped Bass eggs and larvae from the Southwest Miramichi River and the Tabusintac River tidal areas, (M. Hambrook, Miramich Salmon Association, pers. comm.; Andrews et al. 2019a). Intense spawning activities were also reported from the Southwest Miramichi near the head of tide at Quarryville in spring 2020 (T. Tunney, DFO personal communication). Expansion of observations of spawning activities would be expected as the overall spawner abundance increases. Striped Bass spawning can be established in new areas, as evidenced by the colonization event of the southern Gulf by Striped Bass with the Holocene glacial retreat and the spawning and recruitment of Striped Bass in new contemporary spawning areas of the St. Lawrence River (DFO 2017). The consequence to population modelling results of the establishment of new spawning areas is that the asymptotic abundance would increase due to a higher carrying capacity although density independent survival rates from eggs to age-0 in summer would be expected to remain as estimated.

7.4. ASSUMPTIONS ON DENSITY DEPENDENT STOCK AND RECRUITMENT RELATIONSHIP

Based on the available observations, the stock and recruitment dynamic between eggs and abundance at age-3 is adequately described by a proportional function or Beverton-Holt stock and recruitment function. The near monotonic increasing trajectory of the population abundance from its low point in the late 1990s to the highest abundance in the late 2010s provides limited information to unequivocally define the asymptotic population size.

We preferentially chose a model that incorporates a limit to the carrying capacity for the southern Gulf population of Striped Bass. Based on literature, life history, and the geographic area where spawning occurs, we chose a model that set the carrying capacity limit at the early juvenile (age-0, summer) phase. The spawning / nursery habitat and food base for the larvae and post-metamorphosis juveniles are constrained to a relatively small tidal area in the Northwest Miramichi.

The model (model 6) that considered the egg to age-3 recruitment directly provides a different perspective on asymptotic abundance at age-3 and total abundance of the population. The difference in model outputs using the same observational data cannot be explained, other than by weak evidence for density dependence from available data. It is possible that the abundance of Striped Bass could continue to increase to levels indicated by model 6 of almost 1 million spawners at age 3 (Table 4.3), 4.8 million spawners and 7.8 million fish (age 3+) at B_{MSY} as the full productive potential for this population has yet to be realized.

7.5. TIME SERIES CONSIDERATIONS

As mentioned previously, the near monotonic increasing trajectory of the population abundance from its low point in the late 1990s to the highest abundance in the late 2010s provides limited information to unequivocally define the population dynamics parameters of the population model. The recruitment from the 2017 to 2019 spawner abundances have not been assessed with 3-year olds from the 2017 spawning first available for assessment in 2020, and the other year-classes in 2021 and 2022. The fork length distributions of Striped Bass in the fall of 2019 suggest a small mode at just under 35 cm FL, which would be 3-year old fish in May 2020, however, such modes at small fork lengths have been noted in previous years but the cohort tracking of these modes is not convincing (see Figure 3.4 but also Figure 3.9 which is the cohort decline analysis).

The assessments of spawners as published in DFO (2020, and previous years) are assumed to be unbiased albeit highly uncertain estimates of the true spawner abundances. The assessment model uses the commercial gaspereau fishery platform to obtain abundance indices by individual trapnet which are then raised using trapnet specific catchability indices estimated from tag and recapture experiments to estimate total abundance. There is a large contrast in catch rates of Striped Bass in these trapnets over the 1994 to 2019 time period (Figure 3.1) that are consistent with the assessed increase in abundance. In recent years, as the commercial gaspereau trapnets began fishing somewhat later to optimize the catch of gaspereau which is the target species, movements of acoustically tagged bass have been used to infer the proportion of the total spawners present in the commercial gaspereau fishing area when the fishery began.

DFO (2020) provides supplementary indices independent of the commercial gaspereau fishery catches that corroborate the trend of increased abundance of Striped Bass over this period. Specifically, index estuary trapnets installed and monitored by DFO Science in the Southwest Miramichi and in the Northwest Miramichi are used to assess the abundance of numerous anadromous species in the Miramichi River (Hayward et al. 2014). Catches of Striped Bass in the months of May and June show an important increase in abundance, however, the data for May should be interpreted with caution as the installation dates of these trapnets for sampling upstream migrating fish have varied among years. The sum of daily catches in the month of June increased over the period of sampling at both locations and with generally higher catches, particularly in the recent decade, recorded at the Northwest Miramichi trapnet located in the Striped Bass spawning area (Figure 7.1). Trapnet catches in the autumn have also greatly increased over the time period with the strongest signal for the month of October in the Northwest Miramichi and for the months of September and October in the Southwest Miramichi. Contrary to the spring, the highest catches in the fall are consistently recorded at the Southwest Miramichi index trapnet as Striped Bass return to the Miramichi River to overwinter (Figure 7.1).

7.6. FISHERIES RELATED LOSSES AND MANAGEMENT OPTIONS

There are no complete fishery catch data for Striped Bass in the southern Gulf of St. Lawrence.

Historically, fisheries statistics included only commercial harvests, exclusive of recreational and Indigenous peoples fisheries harvests.

It was noted previously that Striped Bass is particularly vulnerable to fisheries in estuaries of the southern Gulf of St. Lawrence. Although the fisheries on Striped Bass were essentially closed in 2000, DFO (2011) indicated that large numbers in the tens of thousands of Striped Bass of various life stages were intercepted in a variety of illegal fisheries, commercial fisheries, and aboriginal FSC fisheries. The activity with the greatest contribution to the total loss of Striped Bass was considered to be the illegal fishery followed by the recreational fishery (DFO 2011).

The recreational fishery for Striped Bass in the southern Gulf has a large component of catch and release, in part due to the mandatory slot size restrictions for retention but also associated with the fishing practices of individual anglers that favour a lot of angling activity without intent to retain. In the eastern US, catch and release represented 85% to 90% of the total catch (retained plus released) of Striped Bass during 2015-2017 and annual losses from catch and release . averaged 2.9 million fish during 2015 to 2017, approximately equivalent to the retained catch of 2.9 to 3.5 million for those same years (NEFSC 2019). There are differences in management measures between jurisdictions; notably the use of natural bait is prohibited in the recreational fishery in Quebec but natural bait, usually in the form of chunks of mackerel or other fish placed on hooks, is permitted in DFO Gulf Region. A catch and release mortality rate of 9% is also assumed, as used in the coastwide Striped Bass assessment of the US. Catch and release mortality rate depends upon fishing gear, water temperature, maturity state and angler practices (Millard et al. 2005; NEFSC 2019). The analysis of consequences of these fishing practices on population abundance and reference points cannot be assessed in the absence of such data.

Young of the year (YOY) Striped Bass remain susceptible to capture in the openwater autumn and winter fishing gears (boxnets and gillnets) set for Rainbow Smelt throughout the southern Gulf of St. Lawrence. Prior to the delayed opening of the fall openwater smelt fishery in the Miramichi from Oct. 15 to Nov.1, interceptions of you bass were estimated to have been in the hundreds of thousands annually, in the Miramichi river alone, most of which would be dead given the difficulty to sort and release them alive from the large quantities of fish captured in these fisheries (Bradford et al. 1997). The delayed season opening should have reduced the bycatch but no follow-up assessment has occurred.

There are additional anecdotal reports of unregulated mortality in other sectors, including Striped Bass being kept and used as bait in the lobster fishery. Striped Bass have also increased in abundance in the freshwater portions of larger rivers such as the Miramichi and Restigouche and there are numerous reports of bass being angled and killed via discarding in the woods from these inland areas.

In the absence of any monitoring of recreational catches and harvests, it is not possible to provide fisheries management advice in terms of total allowable catches nor can the status of the population relative to removal rates be assessed. In the absence of catch and harvest data from all the fisheries, the best that could be done is to track the response of the population abundances to variations in fisheries management strategies. Assessments of spawner abundances are usually provided in the fall to early winter of the spawning year and management plans are established based on the past year's abundance. This approach, used to date for management of Striped Bass of the southern Gulf of St. Lawrence results in low risk to the population if exploitation rates are relatively low. The abundance trajectory of this population indicates that to date, the exploitation rate has been less than the surplus production of the population.

8. REFERENCES CITED

Ahrens, R.N.M., Allen, M.S., Walters, C., and Arlinghaus, R. 2020. Saving large fish through harvest slots outperforms the classical minimum-length limit when the aim is to achieve multiple harvest and catch-related fisheries objectives. Fish and Fisheries 2020 DOI: 10.1111/faf.12442.

Andrews, S.N., Dadswell, M.J., Buhariwalla, C.F., Linnansaari, T., and Curry, R.A. 2019a. Looking for Striped Bass in Atlantic Canada: The reconciliation of local, scientific, and historical knowledge. Northeastern Naturalist 26: 1‑30.

Andrews, S.N., Buhariwalla, C.F., Fleet-Pardy, B., Dadswell, M.J., Linnansaari, T., and Curry, R.A. 2019b. Left out in the cold: the understudied overwintering ecology of striped bass in Canada. Environ. Biol. Fish 102: 499–518.

Atlantic States Marine Fisheries Commission (ASMFC). 2019. Addendum VI to amendment 6 to the Atlantic striped bass interstate fishery management plan. October 30, 2019.

Barneche, D.R., Robertson, D.R., White, C.R., and Marshall, D.J. 2018. Fish reproductive-energy output increases disproportionately with body size. Science 360, 642–645.

Bradford, R.G., and Chaput, G. 1996. The status of striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 1995. DFO Atl. Fish. Res. Doc. 96/62: 36 p.

Bradford, R.G., and Chaput, G. 1997. . Status of striped bass (*Morone saxatilis*) in the Gulf of St. Lawrence in 1996 and revised estimates of spawner abundance for 1994 and 1995. DFO Can. Sci. Advis. Sec. Res. Doc. 97/16: 31 p.

Bradford, R.G., and Chaput, G. 1998. Status of striped bass (*Morone saxatilis*) in the Gulf of St. Lawrence in 1997. DFO Can. Sci. Advis. Sec. Res. Doc. 98/35: 25 p.

Bradford, R.G., Chaput, G., and Tremblay, E. 1995a. Status of striped bass (*Morone saxatilis*) in the Gulf of St. Lawrence. DFO Atl. Fish. Res. Doc. 95/119: 43 p.

Bradford, R.G., Robichaud, K.A., and Courtenay, S.C. 1995b. By-catch in commercial fisheries as an indicator and regulator of striped bass (*Morone saxatilis*) abundance in the Miramichi River estuary. In Water, science, and the public: the Miramichi ecosystem. Edited by E.M.P. Chadwick. Can. Spec. Publ. Fish. Aquat. Sci. 123: 249-259.

Bradford, R.G., Chaput, G., Hurlbut, T., and Morin, R. 1997. Bycatch of striped bass, white hake, winter flounder, and Atlantic tomcod in the autumn “open water” smelt fishery of the Miramichi River estuary. Can. Tech. Rep. Fish. Aquat. Sci. 2195: vi + 37 p.

Bradford, R.G., Cairns, D., and Jessop, B. 2001a. Update on the status of striped bass (*Morone saxatilis*) in eastern Canada in 1998. DFO Can. Sci. Advis. Sec. Res. Doc. 2001/007: i + 14 p.

Bradford, R.G., Chaput, G., Douglas, S., and Hayward, J. 2001b. Status of striped bass (*Morone saxatilis*) in the Gulf of St. Lawrence in 1998. DFO Can. Sci, Advis. Sec. Res. Doc. 2001/006: iii + 30 p.

Buhariwalla, C.F., MacMillan, J.L., Gregoire, M.J., Dadswell, M.J., and Stokesbury, M.J.W. 2016. Population characteristics of Striped Bass killed by cold shock during winter shutdown of a power plant in Nova Scotia. Northeast Naturalist 23: 163–173.

Burton, D.T., Hall, L.W.Jr., Klauda, R.J., and Margrey, S.L. 1983. Effects of treated leached kraft mill effluent on eggs and prolarvae of striped bass *Morone saxatilis*. Water Resour. Bull. 19: 869-879.

Caddy, J.F. 2002. Limit reference points, traffic lights, and holistic approaches to fisheries management with minimal stock assessment input. Fish. Res. 56: 133–137.

Chaput, G. 2022. Multi-species Considerations for Defining Fisheries Reference Points for Striped Bass (*Morone saxatilis*) from the Southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2022/030. vii + 50 p.

Chaput, G.J., and Robichaud, K.A. 1995. Size and growth of striped bass, (*Morone saxatilis*), from the Miramichi River, Gulf of St. Lawrence, Canada. P. 161-176. In E.M.P. Chadwick (ed.). Water, science, and the public: the Miramichi ecosystem. Can. Spec. Publ. Fish. Aquat. Sci. 123.

Chiasson, G., Gallant, P.A., and Mallet, P. 2002. Traditional and local knowledge: estuarine fisheries by-catch in the southern Gulf of St. Lawrence; ecosystem based fisheries management considerations. Can. Man. Rep. Fish. Aqua. Sci. No. 2613: vi + 45 p.

Cook, A.M., J. Duston, and R.G. Bradford. 2006. Thermal tolerance of a northern population of Striped Bass, *Morone saxatilis*. J. Fish Biol. 69: 1482–1490.

COSEWIC 2004. COSEWIC assessment and status report on the Striped Bass *Morone saxatilis* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vii + 43 pp.

COSEWIC. 2012. COSEWIC assessment and status report on the Striped Bass *Morone saxatilis* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. iv + 82 pp.

Cowan, J.H.Jr., Rose, K.A., Rutherford, E.S., and Houde, E.D. 1993. Individual-based model of young of-the-year striped bass population dynamics. 2. Factors affecting recruitment in the Potomac River, Maryland. Trans. Am. Fish. Soc. 119: 757–778.

DFO. 2006. Recovery assessment report for the St. Lawrence estuary, southern Gulf of St. Lawrence and Bay of Fundy striped bass (*Morone saxatilis*) populations. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2006/053.

DFO. 2009. A fishery decision-making framework incorporating the Precautionary Approach

DFO. 2011. Allowable harm assessment of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2011/014

DFO. 2013. Update to 2012 on spawner abundance and biological characteristics for striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Sci. Resp. 2013/010.

DFO. 2014. Recreational fishery catches, spawner abundance, and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 2013. DFO Can. Sci. Advis. Sec. Sci. Resp. 2014/015.

DFO. 2015a. Recreational fishery catches, spawner abundance, and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 2014. DFO Can. Sci. Advis. Sec. Sci. Resp. 2015/011.

DFO. 2015b. Development of reference points for Atlantic salmon (*Salmo salar*) that conform to the Precautionary Approach. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2015/058.

DFO. 2016. Spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 2015. DFO Can. Sci. Advis. Sec. Sci. Resp. 2016/017.

DFO. 2017. Information in Support of Critical Habitat Identification for Striped Bass (*Morone saxatilis*) of the St. Lawrence River. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2017/001.

DFO. 2018. Spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 2017. DFO Can. Sci. Advis. Sec. Sci. Resp. 2018/016.

DFO. 2019. Guidance for the development of rebuilding plans under the Precautionary Approach Framework: Growing stocks out of the critical zone.

DFO. 2020. Update of spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence to 2019. DFO Can. Sci. Advis. Sec. Sci. Resp. 2020/009.

Douglas, S.G., and Chaput, G. 2011a. Assessment and status of Striped Bass (*Morone saxatilis*) in the Southern Gulf of St. Lawrence, 2006 to 2010. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/097. iv + 22 p.

Douglas, S.G., and Chaput, G. 2011b. Information on the Striped Bass (*Morone saxatilis*) population of the southern Gulf of St. Lawrence relevant to the development of a 2nd COSEWIC status report for the species. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/098. iv + 16 p.

Douglas, S.G., Bradford, R.G., and Chaput, G. 2003. . Assessment of striped bass (*Morone saxatilis*) in the Maritime Provinces in the context of species at risk. DFO Can. Sci. Advis. Sec. Res. Doc. 2003/008: iii + 49 p.

Douglas, S.G., Chaput, G., and Caissie, D. 2006. Assessment of status and recovery potential for striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2006/041: viii + 95 p.

Douglas, S.G., Chaput, G., Hayward, J., and Sheasgreen, J. 2009. Prespawning, spawning, and postspawning behavior of striped bass in the Miramichi River. Trans. Am. Fish. Soc. 138: 121-134.

Gabriel, W.L., and Mace, P.M. 1999. A Review of Biological Reference Points in the Context of the Precautionary Approach. p. 34-45. Proceedings, 5th NMFS NSAW. 1999. NOAA Tech. Memo. NMFS-F/SPO-40.

Gahagan, B.I., Fox, D.A., and Secor, D.H. 2015. Partial migration of striped bass: revisiting the contingent hypothesis. Mar. Ecol. Prog. Ser. 525: 185–197.

Gislason, H., Daan, N., Rice, J.C., and Pope, J.G. 2010. Size, growth, temperature and the natural mortality of marine fish. Fish and Fisheries 11: 149–158.

Goodyear, C.P. 1985. Toxic materials, fishing, and environmental variation: simulated effects on striped bass population trends. Trans. Am. Fish. Soc. 114: 107-113.

Goodyear, C.P. 1993. Spawning stock biomass per recruit in fisheries management: foundation and current use. pp. 67-81 In S.J. Smith, J.J. Hunt and D. Rivard [eds.]. Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Gwinn, D.C., Allen, M.S., Johnston, F.D., Brown, P., Todd, C.R., and Arlinghaus, R. 2015. Rethinking length-based fisheries regulations: The value of protecting old and large fish with harvest slot. Fish and Fisheries 16: 259–281.

Halliday, R.G. (chairman). 2001. Proceedings of the Fisheries Management Studies Working Group, 15-16 and 31 May 2001 Bedford Institute of Oceanograhy, Dartmouth, Nova Scotia. DFO Can. Sci. Advis. Sec. Proc. Ser. 2001/21.

Halliday, R.G., and Mohn, R. (chairmen). 2001. Proceedings of the Fisheries Management Studies Working Group, 8-11 January 2001 Bedford Institute of Oceanograhy, Dartmouth, Nova Scotia. DFO Can. Sci. Advis. Sec. Proc. Ser. 2001/08.

Hanson, J.M. 2020. Diet of Striped Bass in the Southern Gulf of St. Lawrence (Canada) with Emphasis on the Spring Spawning Aggregation. Northeastern Naturalist 27(3): 381-400.

Hayward, J., Sheasgreen, J., Douglas, S., and Reid, J. 2014. Diadromous fish monitoring programs in the Miramichi River system in 2011. Can. Manuscr. Rep. Fish. Aquat. Sci. No. 3042: xiii + 102 p.

Hilborn, R., and Walters, C.J. 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Chapman and Hall, New York.

Hurst, T.P. and Conover, D.O. 1998. Winter mortality of young-of-the-year Hudson River striped bass (*Morone saxatilis*): size-dependent patterns and effects on recruitment. Can. J. Fish. Aquat. Sci. 55: 1122-1130.

ICES. 1997. Report of the Study Group on the precautionary approach to fisheries management. ICES 1997/Assess:7.

ICES 2001. Report of the Working Group on Eels St. Andrews, N.B., Canada, 28 August –1 September 2000. ICES CM 2001/ACFM:03.

LeBlanc, C.H., and Chaput, G.J. 1991. Landings of estuarine fishes in the Gulf of St. Lawrence 1917 – 1988. Can. Data Rep. Fish. Aquat. Sci. 842: viii + 101 p.

LeBlanc, N.M., Gahagan, B.I., Andrews, S.N., Avery, T.S., Puncher, G.N., Reading, B.J., Buhariwalla, C.F., Curry, R.A., Whiteley, A.R., Pavey, S.A. 2020. Genomic population structure of Striped Bass (*Morone saxatilis*) from the Gulf of St. Lawrence to Cape Fear River. Evol. Appl.13: 1468–1486.

Liao, H., Sharov, A.F., Jones, C.M., and Nelson, G.A. 2013. Quantifying the Effects of Aging Bias in Atlantic Striped Bass Stock Assessment, Trans. Am. Fish. Soc. 142: 193-207.

Lunn, D., Jackson, C., Best, N., Thomas, A., and Spiegelhalter, D. 2013. The BUGS Book a Practical Introduction to Bayesian Analysis. CRC Press Taylor & Francis Group, LLC, Boca Raton, Florida. 381 pp.

Mace, P.M. 1994. Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. Can. J. Fish. Aquat. Sci. 51: 110-122.

Mace, P.M. and M.P. Sissenwine. 1993. How much spawning per recruit is enough? In S.J. Smith, J.J. Hunt and D. Rivard [eds.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120:101-118.

Millard, M.J., Mohler, J.W., Kahnle, A., and Cosman, A. 2005. Mortality associated with catch and release angling of striped bass in the Hudson River. North Amer. J. Fish. Manag. 25: 1533-1541.

Myers, R.A., Rosenberg, A.A., Mace, P.M., Barrowman, N., and Restrepo, V.R. 1994. In search of thresholds for recruitment overfishing. ICES J. Mar. Sci. 51:191-205.

Northeast Fisheries Science Center (NEFSC). 2019. 66th Northeast Regional Stock Assessment Workshop (66th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 19-08; 1170 p. Available from: http://www.nefsc.noaa.gov/ publications/

Richards, R.A., and Rago, P.J. 1999. A case history of effective fishery management: Chesapeake bay striped bass. North Amer. J. Fish. Manag. 19: 356-375.

Rideout, R.M., and Tomkiewicz, J. 2011. Skipped Spawning in Fishes: More Common than You Might Think. Mar. Coast. Fish. 3: 176-189.

Robichaud-LeBlanc, K.A., Courtenay, S.C., and Locke, A. 1996. Spawning and early life history of a northern population of striped bass (*Morone saxatilis*) in the Miramichi River estuary, Gulf of St. Lawrence. Can. J. Zool. 74: 1645-1655.

Robichaud-LeBlanc, K.A., Courtenay, S.C., and Hanson, J.M. 1997. Ontogenetic diet shifts in age-0 striped bass, *Morone saxatilis*, from the Miramichi River estuary, Gulf of St. Lawrence. Can. J. Zool. 75: 1300-1309.

Robichaud-LeBlanc, K.A., Courtenay, S.C., and Benfey, T.J. 1998. Distribution and growth of young-of-the-year striped bass in the Miramichi River estuary, Gulf of St. Lawrence. Trans. Am. Fish. Soc. 127: 56-69.

Robichaud-LeBlanc, K.A., Bradford, R.G., Flecknow, J., and Collins, H. 2000. Bibliography of Miramichi, southern Gulf of St. Lawrence striped bass and North American studies on the effects of domestic and industrial effluent on striped bass. Can. Manuscr. Rep. Fish. Aquat. Sci. 2547: iii + 31 p.

Robinson, M., Courtenay, S., Benfey, T., Maceda, L., and Wirgin, I. 2004. Origin and movements of young-of-the-year striped bass in the southern Gulf of St. Lawrence, New Brunswick. Trans. Amer. Fish. Soc. 133: 412-426.

Rulifson, R.A., and Dadswell, M.J. 1995. Life history and population characteristics of striped bass in Atlantic Canada. Trans. Am. Fish. Soc. 124: 477-507.

Rutherford, E.S., Houde, E.D., and Nyman, R.M. 1997. Relationship of larval-stage growth and mortality to recruitment of striped bass, *Morone saxatilis*, in Chesapeake Bay. Estuaries 20: 174-198.

Rutherford, E.S., Rose, K.A., and Cowan, J.H.Jr. 2003. Evaluation of the Shepherd and Cushing (1980) model of density-dependent survival: a case study using striped bass (*Morone saxatilis*) larvae in the Potomac River, Maryland, USA. ICES J. Mar. Sci. 60: 1275–1287.

Secor, D.H. 2008. Influence of skipped spawning and misspecified reproductive schedules on biological reference points in sustainable fisheries. Trans. Am. Fish. Soc. 137: 782–789.

Secor, D.H., Trice, T.M., and Hornick, H.T. 1995. Validation of otolith-based aging and a comparison of otolith and scale-based aging in mark-recaptured Chesapeake Bay striped bass, *Morone saxatilis*. Fish. Bull. 93: 186–190.

Secor, D.H., O'Brien, M.H.P., Gahagan. B.I., Watterson, J.C., and Fox, D.A. 2020. Differential migration in Chesapeake Bay striped bass. PLoS ONE 15(5): e0233103. pone.0233103.

Uphoff, J.H. 1989. Environmental effects on survival of eggs, larvae and juveniles of striped bass in the Choptank River, Maryland. Trans. Am. Fish. Soc. 118: 251–263.

Valiquette, E., Harvey, V., and Pelletier, A.-M. 2017. Information update on the Identification, Description and Spatio-Temporal Use of Striped Bass (*Morone saxatilis*) Habitat in the St. Lawrence River, Quebec. DFO Can. Sci. Advis. Sec. Res. Doc. 2017/005. ix + 55 p.

Valiquette, E., Legault, M., Mainguy, J., Bujold, V., et Pelletier, A.-M. 2018. Répartition du bar rayé au Québec – mise à jour des connaissances, Ministère des Forêts, de la Faune et des Parcs, Québec, v + 17 p.

Walters, C.J., and Martell, S.J.D. 2004. Fisheries Ecology and Management. Princeton University Press, Princeton, New Jersey.

Walters, C.J., Hilborn, R., and Christensen, V. 2008. Surplus production dynamics in declining and recovering fish populations. Can. J. Fish. Aquat. Sci. 65: 2536–2551.

Wirgin, I.I., Ong, T., Maceda, L., Waldman, J.R., Moore, D., and Courtenay, S. 1993. Mitochondrial DNA variation in striped bass (*Morone saxatilis*) from Canadian rivers. Can. J. Fish. Aquat. Sci. 50: 80-87.

Wirgin, I., Maceda, L., Tozer, M., Stabile, J., and Waldman, J. 2020. Atlantic coastwide population structure of striped bass *Morone saxatilis* using microsatellite DNA analysis. Fish. Res. 226 (2020) 105506.

TABLES

Table 2.1. Management milestones for Striped Bass fisheries from the southern Gulf of St. Lawrence, 1992 to 2013.

Year	Commercial	Recreational	Indigenous FSC
Prior to 1992	Gillnet licences (mesh restriction 127 mm) Bow net fishery open Incidental catches retained and marketed	No retention of bass < 38 cm Total Length (TL), unless in Kent Co. waters	> 68 cm TL
1992	DFO conservation strategy written: - closure of all directed fisheries - incidental catches of bass > 38 cm TL to be released - bycatch tolerance for bass < 38 cm in gaspereau and smelt fisheries - bow net fishery designated as recreational, with recreational bag limit and size restrictions	July 1 to Oct. 31 One bass per day > 68 cm TL	> 68 cm TL
1993	DFO conservation strategy implemented	July 1 to Oct. 31 One bass per day > 68 cm TL	July 1 to Oct. 31 > 68 cm TL
1994	DFO conservation strategy implemented	July 1 to Oct. 31 One bass per day > 68 cm TL	July 1 to Oct. 31 > 68 cm TL
1995	Release of bass > 38 cm not imposed Some voluntary release of spawning fish in Miramichi (May – June) 17 t recorded harvest	July 1 to Oct. 31 One bass per day > 68 cm TL	> 68 cm TL
1996	Commercial fisheries closed Sale of wild caught Striped Bass prohibited Tolerance limit for retention but not sale of bass < 35 cm TL in gaspereau and smelt fisheries 15 t recorded harvest	Hook and release only May 1 to Oct. 31	July 1 to Oct. 31 Size restrictions lifted (impractical because of gillnets)
1997	Commercial fisheries closed Sale of wild caught Striped Bass prohibited Tolerance limit for retention but not sale of bass < 35 cm TL in gaspereau and smelt fisheries	Hook and release only May 1 to Oct. 31	July 1 to Oct. 31
1998	Commercial fisheries closed Sale of wild caught Striped Bass prohibited Tolerance conditions revoked, no retention of bycatch of any size	Hook and release only April 15 to Oct. 31 (opening corresponds to opening date of black salmon and trout fishereries)	July 1 to Oct. 31
1999	Commercial fisheries closed Sale of wild caught Striped Bass prohibited Tolerance conditions revoked, no retention of bycatch of any size	Hook and release only April 15 to Oct. 31	July 1 to Oct. 31

Year	Commercial	Recreational	Indigenous FSC
	Delayed opening to Nov. 1 (from Oct. 15) of fall openwater smelt boxnet fishery in Miramichi		
2000	Continued from 1999	Inland and coastal waters closed to directed fishing for Striped Bass	FSC allocations suspended
2012	Continued from 1999	Inland and coastal waters remain closed to directed fishing for Striped Bass	Re-instatement of Indigenous FSC allocations
2013	Continued from 1999	Re-opening of retention fishery	FSC allocations maintained

Table 2.2. Recreational fisheries management measures for Striped Bass since the re-opening of the fishery in 2013. (see DFO 2016 for management breakdown in 2013 to 2015).

Year	Region	Season for tidal waters	Retention days	Daily bag limit	Retention size limit	Notes
2012	DFO Gulf	Closed	na	na	na	na
2013	DFO Gulf	May 1 to Sept. 30	25	1	55 – 65 cm TL	na
	Prov. of Quebec (Chaleur Bay; Zone 21)	June 15 - Sept. 30	0	0	na	Catch and release only Single hook
2014	DFO Gulf	May 1 to Sept. 30	53	1 (May 1-21) 2 (May 22-25)* 1 (Aug. 1-21) 1 (Sept. 24-30)	50 – 65 cm TL	*Due to cold weather and poor angler success, the retention period in May 2014 was extended for four days to May 25. During this extension, anglers were permitted to retain two Striped Bass per day and possess no more than two at any given time.
	Prov. of Quebec (Chaleur Bay; Zone 21)	June 15 - Sept. 30	30 (July 26 – Aug. 24)	1	< 65 cm TL	Single hook maximum 3 per line Artificial lures only, bait prohibited
2015	DFO Gulf	May 1 to Oct. 31	56 May 11 – 31 Aug. 1 – 23 Sept. 4 – 7 Oct. 24 - 31	1	50 – 65 cm TL	na
	Prov. of Quebec (Chaleur Bay; Zone 21)	June 15 - Sept. 30	56 (July 1 – Aug. 25)	1	50 – 65 cm TL	Single hook maximum 3 per line Artificial lures only, bait prohibited
2016	DFO Gulf	May 1 to Oct. 31	104	1	50 – 65 cm TL	na
	Prov. of Quebec (Chaleur Bay; Zone 21)	June 15 - Oct. 31	109 ((July 1 – Aug. 26; Sept. 9 – Oct. 31)	1	50 – 65 cm TL	Single hook maximum 3 per line Artificial lures only, bait prohibited
2017	DFO Gulf	April 15 to Oct. 31	200 (April 15 to Oct. 31)	1(April 15 – June 14) 2 (June 15 – Aug. 31) 1 (Sept. 1 – Oct. 31)	50 – 65 cm TL	na

Year	Region	Season for tidal waters	Retention days	Daily bag limit	Retention size limit	Notes
	Prov. of Quebec (Chaleur Bay; Zone 21)	June 15 - Oct. 31	139 (June 15 - Oct. 31)	2	50 – 65 cm TL	Single hook maximum 3 per line Artificial lures only, bait prohibited
2018	DFO Gulf	April 15 to Oct. 31	200 (April 15 to Oct. 31)	3	50 – 65 cm TL	na
	Prov. of Quebec (extended Zone 21)	June 15 - Oct. 31	139 (June 15 - Oct. 31)	3	50 – 65 cm TL	Single hook maximum 3 per line Artificial lures only, bait prohibited Extension of Zone 21 upstream in St. Lawrence River to a line approximately joining Rimouski and Forestville and extending to the north shore of the St. Lawrence, including Magdalene Islands
2019	DFO Gulf	April 15 to Oct. 31	200 (April 15 to Oct. 31)	3	50 – 65 cm TL	ns
	Prov. of Quebec (extended Zone 21)	June 15 - Oct. 31	139 (June 15 - Oct. 31)	3	50 – 65 cm TL	Same as 2018 Including in most rivers that flow into Zone 21

Table 2.3. Summary of spawning area closures to all recreational fisheries activities on the spawning grounds of the Northwest Miramichi, 2017 to 2020.

Year	DFO Gulf Region Variation Order	Start Date	End Date	Total days of closure	Length of spawning area closed
2017	GVO-2017-038	1 June	9 June	9	9.8 km
2018	GVO-2018-032	4 June	8 June	5	6.5 km
2019	GVO-2019-035	5 June	9 June	5	6.5 km
2020	GVO-2020-044	28 May	1 June	5	6.5 km

Table 2.4. Recorded landings (t) of Striped Bass from the fisheries statistical districts that are located in the vicinity of the Miramichi River, and overall in the southern Gulf of St. Lawrence. Data for the period 1917 to 1988 are from LeBlanc and Chaput (1991). Data for 1989 to 1994 are from Bradford et al. (1995a). There were no recorded landings for the years 1935 to 1967. Detailed harvests by statistical districts in DFO Gulf NB as well as by season and regions are provided in Bradford et al. (1995a) and Douglas et al. (2003). "ns" means no information specified.

	Miramichi area districts						Southern Gulf
Year	68	70	71	72	73	Total	
1917	8.2	ns	4	0.4	1.5	14.1	61.4
1918	7.2	ns	1.1	4.5	1.5	14.3	54.4
1919	4.1	0.5	1.2	2.3	3.6	11.7	33.7
1920	17.3	ns	2.2	0.5	4.2	24.2	28.3
1921	1.1	ns	1.5	ns	2.7	5.3	15.9
1922	1.4	ns	1.2	ns	ns	2.6	19.1
1923	0.9	ns	0.2	ns	5.4	6.5	25.5
1924	ns	ns	0.9	7.2	ns	8.1	39.8
1925	0.9	ns	0.7	0.4	4.1	6.1	22.1
1926	ns	ns	1.9	0.4	ns	2.3	20.0
1927	ns	ns	ns	ns	6.5	6.5	22.8
1928	ns	ns	0.2	ns	3.7	3.9	10.3
1929	ns	ns	ns	ns	1.7	1.7	5.8
1930	ns	ns	0.5	0.5	0.9	1.9	4.0
1931	ns	ns	ns	0.5	0.9	1.4	3.2
1932	ns	0.8	ns	0.5	1.1	2.4	3.9
1933	ns	0.2	ns	0.1	ns	0.3	0.7
1934	ns	ns	ns	0.3	ns	0.3	0.4
1935	ns	ns	ns	ns	ns	ns	ns
1967	ns	ns	ns	ns	ns	ns	ns
1968	ns	0.4	1.8	1.1	0.1	3.4	8.2
1969	ns	0.1	0.4	1.6	0.1	2.2	9.4
1970	0.1	2.6	0.9	3.4	0.4	7.4	10.6
1971	ns	0.7	1.4	8.5	0.4	11	13.3
1972	ns	0.1	1.8	3.4	0.5	5.8	8.8
1973	ns	0.2	0.1	3.8	ns	4.1	6.1
1974	0.1	ns	0.3	3.6	ns	4	5.4
1975	0.7	3.2	1	ns	ns	4.9	7.2
1976	0.1	1.9	1.6	3.1	ns	6.7	8.6
1977	ns	0.9	1.2	ns	ns	2.1	5.1
1978	ns	1.5	ns	ns	ns	1.5	5.1
1979	0.1	2.2	1.2	ns	ns	3.5	6.8
1980	0.1	9.7	2.9	ns	ns	12.7	15.3
1981	0.9	5.5	4.7	ns	ns	11.1	47.8
1982	1	3.8	2.4	ns	ns	7.2	32.4
1983	2	3	6.9	ns	0.1	12	23.4
1984	0.1	9.9	2.2	ns	ns	12.2	17.3
1985	0.8	2.3	8	ns	ns	11.1	22.0
1986	2.2	3.5	ns	ns	ns	5.7	12.5
1987	ns	0.6	ns	ns	0.1	0.7	2.3
1988	0.1	2	0.9	ns	ns	3	4.1
1989	ns	ns	0.1	ns	ns	0.1	4.0
1990	ns	ns	0.1	ns	ns	0.1	1.0
1991	ns	ns	0.1	ns	ns	0.1	1.3
1992	ns	ns	0.5	ns	ns	0.5	8.9
1993	ns	ns	ns	ns	ns	ns	0.6
1994	ns	ns	ns	ns	ns	ns	1.0
1995	ns	ns	ns	ns	ns	ns	17.3
1996	ns	ns	ns	ns	ns	ns	15.25

Table 2.5. Summary of available estimated recreational fisheries catches since the re-opening of the Striped Bass recreational fisheries in the Gulf of St. Lawrence in 2013 to 2015. Data for 2013 are from DFO (2014) and data for 2014 are summarized in DFO (2015a).

Year	Management authority / daily limit	Survey period	Estimated fish retained Point estimate (95% confidence interval)	Estimated fish released [1]	Estimated total losses due to fishing (assumed 10% mortality from catch and release)
2013	DFO Gulf Region 1 fish per day 55 – 65 cm TL for retention	May 1 – 15 Miramichi River	2,400	29,224	5,322
		Aug. 2 -11 Eight locations in the southern Gulf	244	2,911	535
2014	DFO Gulf Region 1 fish per day 50 – 65 cm TL for retention	17 of 25 days during May 1 to 25 Miramichi River	400	9,637	1,364
		August and September retention periods	na	na	na
	Province of Quebec 1 fish per day < 65 cm TL for retention	July / August	554 (299 to 809)	8,456 (4,865 to 12,047)	1,400 (1,146 to 2,013)
2015	DFO Gulf Region 1 fish per day 50 – 65 cm TL for retention	na	na	na	na
	Province of Quebec 1 fish per day 50 – 65 cm TL for retention	July / August	1,172 (790 to 1,554)	20,797 (14,225 to 27,368)	3,252

[1] for the province of Quebec survey, the value for total catch and release is the value for total catch (retained plus released)

Table 2.6. *Characteristics of the recreational fishery in Chaleur Bay, 2014 to 2019. Data for 2014 are presented in DFO (2015a). Data for 2016 to 2019 were provided by Quebec MFFP (unpubl. data).*

Year	Management regulation	Catch category	Size group	Percentage of catch category
2014	1 fish per day < 65 cm Total Length for retention	Retained	< 50 cm	27%
			50 – 65 cm	73%
		Released	< 50 cm	33%
			>65 cm	13%
2016	1 fish per day 50 - 65 cm Total Length for retention	Prob. of retaining 1 or more fish		14.2%
		Catch (retained and released)	< 50 cm	69%
			50 – 65 cm	22%
			>65 cm	9%
2017	2 fish per day < 65 cm Total Length for retention	Prob. of retaining 1 or more fish		6.5%
		Catch (retained and released)	< 50 cm	58%
			50 – 65 cm	37%
			>65 cm	5%
2018	3 fish per day 50 - 65 cm Total Length for retention	Prob. of retaining 1 or more fish		22.5%
		Catch (retained and released)	< 50 cm	49%
			50 – 65 cm	44%
			>65 cm	7%
2019	3 fish per day 50 - 65 cm Total Length for retention	Prob. of retaining 1 or more fish		6.5%
		Catch (retained and released)	< 50 cm	55%
			50 – 65 cm	43%
			>65 cm	3%

Table 3.1. Summary of tagging locations, tagging years, as well as overwintering and spawning histories of acoustically tagged Striped Bass with tag identification codes detected at the acoustic receiver line at Port Hope Simpson (Labrador) in 2017. The detections data at the Port Hope line were provided by M. Robertson (DFO Newfoundland and Labrador Region). Striped Bass were tagged in the St. Lawrence and in Gaspe by personnel from the MFFP Quebec.

Location of tagging	Year tagged	Overwinter history	Spawning history	Detected in Labrador	Acoustic detections (n)
St. Lawrence	2015	Never seen in Miramichi	Never seen in Miramichi	28-Sep-17	5
Gaspe	2016	Never seen in Miramichi	Never seen in Miramichi	4-Sep-17	1
Gaspe	2014	Miramichi – 2014/15, 2015/16, 2016/17	Miramichi - 2015, 2016, 2017	5-Sep-2017, 22-Sep-2017	3
Gaspe	2014	Miramichi - 2014/15, 2015/16, 2016/17	Miramichi - 2015, 2016, 2017	30-Aug-2017, 28-Sep-2017	13
Gaspe	2014	Miramichi - 2014/15, 2015/16, 2016/17	Miramichi - 2015, 2016, 2017	29-Aug-17	1
Gaspe	2014	Miramichi - 2014/15, 2015/16, 2016/17	Miramichi - 2015, 2016, 2017	5-Sep-2017, 27-Sep-2017	5
Gaspe	2014	Miramichi - 2014/15, 2015/16, 2016/17, 2017/18, 2018/19	Miramichi - 2015, 2016, 2017, 2019	3-Sep-2017, 22-Sep-2017	6
St. Lawrence	2014	Miramichi - 2016/17, 2017/18, 2018/19	Miramichi - 2017, 2019	29-Aug-17	3
Miramichi	2013	Miramichi – 2013/14, 2014/15, 2015/16, 2016/17, 2017/18, 2018/19	Miramichi - 2015, 2016, 2017, 2019	22-Sep-17	2

Table 3.2. Summary statistics of selected samples of fork length (cm) at scale-interpreted ages of Striped Bass from the Miramichi River used in the von Bertalanffy growth model analysis.

					Posterior summaries of von Bertalanffy predictions		
Age	N retained (available)	Mean (cm)	Std. dev.	CV	Mean (cm)	Std. dev.	Growth increment (cm)
1	71 (71)	17.8	1.5	0.083	17.5	1.5	-
2	200 (562)	28.0	3.2	0.116	29.0	2.6	11.4
3	200 (2606)	40.4	3.6	0.088	38.5	3.4	9.6
4	200 (2542)	46.8	3.9	0.082	46.7	4.2	8.2
5	200 (1485)	52.6	3.9	0.073	53.6	4.8	6.9
6	200 (769)	58.1	4.5	0.077	59.4	5.3	5.8
7	124 (124)	63.6	5.5	0.086	64.4	5.7	5.0
8	94 (94)	69.1	5.3	0.076	68.6	6.1	4.2
9	62 (62)	72.7	5.5	0.076	71.9	6.3	3.3
10	20 (20)	77.1	6.3	0.082	75.0	6.6	3.1
11	21 (21)	78.2	6.3	0.081	77.6	6.8	2.6
12	10 (10)	83.5	5.2	0.062	79.4	6.9	1.9
13	2 (2)	75.5	7.6	0.101	81.4	7.2	2.0
14	5 (5)	78.2	7.2	0.093	82.8	7.4	1.3
15	3 (3)	86.9	16.4	0.189	84.2	7.5	1.4

Table 3.3. Posterior parameter estimates of the von Bertalanffy growth function to fork length (cm) at age (years) data for Striped Bass from the Miramichi River.

			Correlations	
Parameter	Median	5th to 95th percentile	L∞ to	K to
L_∞ (cm)	90.8	88.5 to 93.3	na	na
K	0.1685	0.1598 to 0.1771	-0.974	-
$a_0(year)$	-0.2680	-0.3176 to -0.2218	-0.748	0.857
σ $(\log scale)$	0.088	0.085 to 0.091	na	na
Pred. length at age 3 (cm)	38.4	33.2 to 44.3	na	na

*Table 3.4. Fork length (cm) to whole weight (kg) relationship for Striped Bass sampled during May and June 2013 to 2105 from the Miramichi River. The equation is: log(WWkg) = intercept + slope * log(FLcm) + ε with $\varepsilon \sim N(0, sigma^2)$.*

		Maximum Likelihood	
Sex	Parameter	Mean	Standard error
Combined	Slope	3.0027	0.0094
	Intercept	-11.3428	0.0363
	sigma	0.087	
	N	1,839	
By sex			
Female	Slope	3.0742	0.0156
	Intercept	-11.6014	0.0613
	N	643	
Male	Slope	2.9327	0.0196
	Intercept	-11.0879	0.0760
	N	1,196	
sigma		0.085	

Table 3.5. Number of female and male Striped Bass by age from opportunistic samples collected in May and June in the southern Gulf of St. Lawrence, 1970 to 2018. These could be biased to males because in many cases, the sex was identified by external characteristics (ripe and running) which is more easily detected in males than females.

Age	N - Females	N - Males	Proportion female
2	5	53	0.086
3	32	2053	0.015
4	120	1524	0.073
5	201	487	0.292
6	124	160	0.437
7	41	40	0.506
8	32	18	0.640
9	19	16	0.543
10	8	4	0.667
11	7	7	0.500
12	7	0	1.000
13	0	1	0.000
14	1	2	0.333
15	2	1	0.667

Table 3.6. Summary of assumptions on proportion mature at age and the proportion female at age of spawners for Striped Bass of the southern Gulf of St. Lawrence.

	Age (years)			
Characteristic	3	4	5	6 and older
Proportion mature at age (assumed)				
Male	0.5	0.9	1	1
Female	0.1	0.5	0.9	1
Proportion female at age on spawning grounds assuming similar proportions at age of male and female mature recruits are spawners on the spawning grounds				
Proportion female	0.17	0.36	0.47	0.50

Table 3.7. Predicted M at age of Striped Bass based on the fitted von Bertalanffy growth characteristics and the empirical relationship of M to growth characteristics of Gislason et al. (2010). Mean sizes at age are shown in Table 3.2.

Age	Mid-season mean size (mm) ($L_{a,t}$ to $L_{a+1,\ t+1}$)	Predicted M	Predicted S
0	135 (110 to 160)[1]	1.97	0.14
1	232 (175 to 290)	0.82	0.44
2	337 (290 to 385)	0.45	0.64
3	426 (385 to 467	0.31	0.73
4	501 (467 to 536)	0.24	0.79

[1] Modal length range of young of the year going into their first winter

Table 3.8. Data used in the estimation of survival probabilities from Striped Bass tagged with acoustic tags and detected in the Miramichi River. The data for 2003 to 2009 are from Douglas and Chaput (2011a). N tags is the number of tags from the tagging group detected in the Miramichi that represents the initial number of animals tracked in subsequent years. The size group categories represent the fork length (cm) retention size limits for the recreational fishery, in place since 2014. Fish are assigned to a size group based on their fork length at time of tagging.

Location tagged	Year tagged	Season tagged	Tag type	Size group	N tags	Tags detected in year of inferred survival								
						2003	2004	2008	2009	2014	2015	2016	2017	2018
Miramichi	2003	spring	V16	Total	19	13	na	na	na	na	na	na	na	na
Miramichi	2004	spring	V16	Total	21	na	13	na	na	na	na	na	na	na
Miramichi	2008	spring	V16	Total	20	na	na	14	10	na	na	na	na	na
Miramichi	2009	spring	V16	Total	21	na	na	na	14	na	na	na	na	na
Gaspe	2013	summer	V13	< 46	1	na	na	na	na	1	na	na	na	na
				46 - 61	23	na	na	na	na	22	na	na	na	na
				> 61	15	na	na	na	na	13	na	na	na	na
				Total	39	na	na	na	na	36	na	na	na	na
Miramichi	2013	fall	V16	46 - 61	15	na	na	na	na	12	8	5	5	5
				> 61	21	na	na	na	na	17	15	13	9	9
				Total	36	na	na	na	na	29	23	18	14	14
Gaspe	2014	summer	V13	< 46	3	na	na	na	na	na	3	1	na	na
				46 - 61	12	na	na	na	na	na	10	8	na	na
				Total	15	na	na	na	na	na	13	9	na	na
Gaspe	2014	summer	V16	46 - 61	25	na	na	na	na	na	18	14	6	5
				> 61	18	na	na	na	na	na	16	12	10	9
				Total	43	na	na	na	na	na	34	26	16	14
Pictou	2015	winter	V16	Total	5	na	na	na	na	na	na	5	3	2
Gaspe	2015	summer	V13	Total	1	na	na	na	na	na	na	1	na	na
Gaspe	2016	late fall	V13	Total	8	na	na	na	na	na	na	na	8	3
Gaspe	2016	late fall	V16	Total	4	na	na	na	na	na	na	na	2	1
Miramichi	2016	fall	V16	< 46	4	na	na	na	na	na	na	na	4	4
				46 - 61	14	na	na	na	na	na	na	na	12	11
				> 61	6	na	na	na	na	na	na	na	6	6
				Total	24	na	na	na	na	na	na	na	22	21
Miramichi	2017	fall	V16	< 46	3	na	na	na	na	na	na	na	na	3
				46 - 61	19	na	na	na	na	na	na	na	na	14
				> 61	1	na	na	na	na	na	na	na	na	1
				Total	23	na	na	na	na	na	na	na	na	18

Table 4.1. Model specific parameters and prior assumptions for the life cycle age structured model. In OpenBUGS, the normal distribution is parameterized by the mean and the precision (1/variance) and C(#,) indicates the distribution is constrained to values greater than the first element. The gamma distribution is parameterized on the inverse gamma scale.

Model variant	Parameters with associated priors	Parameter translations
Model 1	δ ~ N(1,0.001)C(0,)	Beverton-Holt; α = exp(-δ)
	K ~ N(1,0.001)C(1,)	Z(0) = -log(S.0)
	S.0 ~ Beta(139,861)	Z[1] = -log(S[1])
	S[1] ~ Beta(440,560)	Z[2] = -log(S[2]
	S[2] ~ Beta(638,362)	Z[3+] = -log(S[3+])
	S[3+] ~ Beta(720,280)	p.rec.to.sp[7:15+] = p.rec.to.sp[6]
	p.rec.to.sp[3] ~ Beta(270,730)	
	p.rec.to.sp[4] ~ Beta(630,370)	
	p.rec.to.sp[5] ~ Beta(855,145)	
	p.rec.to.sp[6] ~ Beta(900,100)	
	log(σ) [3:8, Total] ~ U(0,3)	
Model 2	δ ~ N(1,0.001)C(0,)	Beverton-Holt; α = exp(-δ)
	K ~ N(1,0.001)C(1,)	Z(0) = -log(S.0)
	S.0 ~ Beta(139,861)	Z[1] = -log(S[1])
	S[1] ~ Beta(440,560)	Z[2] = -log(S[2]
	S[2] ~ Beta(638,362)	Z[3:8] = -log(S[3:8])
	S[3] ~ Beta(72,28)	Z[9:15+] = Z[8]
	S[4] ~ Beta(75,25)	p.rec.to.sp[7:15+] = p.rec.to.sp[6]
	S[5] ~ Beta(80,20)	
	S[6] ~ Beta(85,15)	
	S[7] ~ Beta(90,10)	
	S[8] ~ Beta(95,5)	
	p.rec.to.sp[3] ~ Beta(270,730)	
	p.rec.to.sp[4] ~ Beta(630,370)	
	p.rec.to.sp[5] ~ Beta(855,145)	
	p.rec.to.sp[6] ~ Beta(900,100)	
	log(σ) [3:8, Total] ~ U(0,3)	
Model 3	δ ~ N(1,0.001)C(0,)	Beverton-Holt; α = exp(-δ)
	K ~ N(1,0.001)C(1,)	Z(0) = -log(S.0)
	S.0 ~ Beta(139,861)	Z[1] = -log(S[1])
	S[1] ~ Beta(440,560)	Z[2] = -log(S[2]
	S[2] ~ Beta(638,362)	Z[3:8] = -log(S[3:8])
	S[3:8] ~ Beta(6,4)	Z[9:15+] = Z[8]
	p.rec.to.sp[3] ~ Beta(270,730)	p.rec.to.sp[7:15+] = p.rec.to.sp[6]
	p.rec.to.sp[4] ~ Beta(630,370)	
	p.rec.to.sp[5] ~ Beta(855,145)	
	p.rec.to.sp[6] ~ Beta(900,100)	
	log(σ) [3:8, Total] ~ U(0,3)	
Model 4	δ ~ N(1,0.001)C(0,)	Beverton-Holt; α = exp(-δ)
	K ~ N(1,0.001)C(1,)	Z(0) = -log(S.0)
	S.0 ~ Beta(14,86)	Z[1] = -log(S[1])
	S[1] ~ Beta(44,56)	Z[2] = -log(S[2]
	S[2] ~ Beta(64,36)	Z[3:8] = -log(S[3:8])
	S[3:8] ~ Beta(6,4)	Z[9:15+] = Z[8]
	p.rec.to.sp[3] ~ Beta(4,12)	p.rec.to.sp[7:15+] = p.rec.to.sp[6]
	p.rec.to.sp[4] ~ Beta(3,3)	
	p.rec.to.sp[5] ~ Beta(5,2)	
	p.rec.to.sp[6] ~ Beta(4,1)	
	log(σ) [3:8, Total] ~ U(0,3)	
Model 5	δ ~ N(1,0.001)C(0,)	Beverton-Holt; α = exp(-δ)
	K ~ N(1,0.001)C(1,)	Z(0to3) = -log(S[0to3])
	S[0to3] ~ Beta(5,45)	Z[3:8] = -log(S[3:8])
	S[3:8] ~ Beta(6,4)	Z[9:15+] = Z[8]
	p.rec.to.sp[3] ~ Beta(4,12)	p.rec.to.sp[7:15+] = p.rec.to.sp[6]

Model variant	Parameters with associated priors	Parameter translations
	p.rec.to.sp[4] ~ Beta(3,3) p.rec.to.sp[5] ~ Beta(5,2) p.rec.to.sp[6] ~ Beta(4,1) log(σ) [3:8, Total] ~ U(0,3)	
Model 6	δ ~ N(1,0.001)C(0,) K ~ N(1,0.001)C(1,) S[3:8] ~ Beta(6,4) p.rec.to.sp[3] ~ Beta(4,12) p.rec.to.sp[4] ~ Beta(3,3) p.rec.to.sp[5] ~ Beta(5,2) p.rec.to.sp[6] ~ Beta(4,1) log(σ) [3:8, Total] ~ U(0,3)	Beverton-Holt; α = exp(-δ) Z[3:8] = -log(S[3:8]) Z[9:15+] = Z[8] p.rec.to.sp[7:15+] = p.rec.to.sp[6]
Model 7 Power stock and recruitment function	α ~ Beta(1,1) β ~ Gamma(6,4) S[3:8] ~ Beta(6,4) p.rec.to.sp[3] ~ Beta(4,12) p.rec.to.sp[4] ~ Beta(3,3) p.rec.to.sp[5] ~ Beta(5,2) p.rec.to.sp[6] ~ Beta(4,1) log(σ) [3:8, Total] ~ U(0,3)	Z[3:8] = -log(S[3:8]) Z[9:15+] = Z[8] p.rec.to.sp[7:15+] = p.rec.to.sp[6]

Table 4.2. Description of the models examined for estimating the life history and population dynamics parameters of Striped Bass from the southern Gulf of St. Lawrence. A summary of model fits (deviance, approximate Aikike Information Criterion (AIC'), and the DIC from the OpenBUGS) are also shown. In all models, the weight at age, fecundity, and proportion female at age on the spawning grounds are known or assumed with no uncertainty (Appendix 3).

Model variant	Fit statistics	Comments
Model 1	Deviance: 2440 Parameters: 17 AIC' = Dev+2*p = 2474 DIC = 2448 (pD = 8)	Poor fit to total spawners (residuals are positive generally) Very poor fit to observed abundances at age, dominant residual patterns
Model 2	Deviance: 2442 Parameters: 22 AIC' = Dev+2*p = 2484 DIC = 2450 (pD = 7.6)	Poor fit to total spawners (residuals are positive) Residuals mostly positive for age-3, negative for ages 7 and 8 Temporal trend in residuals for ages 7 and 8
Model 3	Deviance: 2403 Parameters: 22 AIC' = Dev+2*p = 2447 DIC = 2412 (pD = 9.1)	Good fit to spawners at age Temporal trend in residuals for ages 7 and 8 Mostly positive residuals for total spawners No autocorrelation for residual Survival age 3 higher than S for ages 4 to 7 which is not consistent with expectations
Model 4	Deviance: 2396 Parameters: 22 AIC' = Dev+2*p = 2440 DIC = 2401 (pD = 5.0)	Good fit to spawners at ages 3 to 6 A few more positive residuals for total spawners Temporal trend in residuals for ages 7 and 8 No autocorrelation for residuals Survival age 3 higher than for ages 4 to 7 which is not consistent with expectations Negative correlation between α and K, α and S[0]
Model 5	Deviance: 2395 Parameters: 20 AIC' = Dev+2*p = 2435 DIC = 2394 (pD = -1.4)	Good fit to spawners at ages 3 to 6 Almost balanced residual pattern for total spawners Temporal trend in residuals for ages 7 and 8 No autocorrelation for residuals. Survival age 3 higher than for ages 4 to 7 which is not consistent with expectations Negative correlation between α and K, α and S.0to3
Model 6	Deviance: 2391 Parameters: 19 AIC' = Dev+2*p = 2429 DIC = 2392 (pD = 0.3)	Good fit to spawners at age Temporal trend in residuals for ages 7 and 8 No autocorrelation for residuals. Survival age 3 higher than S for ages 4 to 7 which is not consistent with expectations Positive correlation between Bev-Holt alpha and S[3]
Model 7	Deviance: 2385 Parameters: 19 AIC' = Dev+2*p = 2423 DIC = 1330 (pD = -1055)	Equally good fit to spawners at age and total spawners as model with Beverton-holt assumption Beta (power term) is centered on 1, no density dependence (abundance increasing without limit Strong positive correlation between beta and gamma of the power function

Table 4.3. Summary (median; 5th to 95th percentiles range) of posterior estimates of the stock and recruitment parameters and predicted abundances for three models with a Beverton-Holt stock and recruitment function. The asymptotic abundance estimates are based on runs of the equilibrium model with life history parameters from the specific model fits and no fishing.

Feature	Model 4 (BH-eggs to age-0)	Model 5 (BH-eggs to age-0)	Model 6 (BH-eggs to age-3)
Survival eggs to age-0			
α	5.34 E-4 (3.53 E-4 to 8.27 E-4)	2.28 E-4 (1.32 E-4 to 4.02 E-4)	na
Survival age-0 to 3			
assumptions	S[0]*S[1]*S[2]	S[0to3]	na
S	0.0631 (0.0449 to 0.0869)	0.163 (0.103 to 0.249)	na
Survival eggs to age-3 in absence of density dependence			
S	3.34 E-5 (2.45 E-5 to 4.76 E-5)	3.65 E-5 (2.51 E-5 to 5.65 E-5)	4.20 E-5 (2.74 E-5 to 6.92 E-5)
Lifetime reproductive rate (number of recruits at age-3 per lifetime contribution of a spawner in absence of density-dependent compensatory survival)			
Age-3 (number)	5.5 (4.9 to 7.1)	5.0 (3.7 to 7.6)	4.9 (3.7 to 7.4)
Asymptotic abundance (K; Beverton-Holt model)			
Age-0 (millions)	9.10 (6.25 to 12.46)	6.80 (4.06 to 10.27)	na
Age-3 recruitment (thousands)	566 (383 to 834)	1,074 (640 to 1,799)	3,705 (1,622 to 7,373)
Equilibrium modelling abundance			
Age-0 (millions)	7.37 (4.94 to 10.22)	5.23 (2.87 to 8.38)	na
Age-3 recruitment (thousands)	456 (314 to 685)	824 (444 to 1,466)	2,848 (1,251 to 5,686)
Age 3 spawners (thousands)	170 (109 to 265)	288 (159 to 508)	819 (351 to 1,812)
Eggs (millions) [1]	66,175 (37,433 to 182,588)	105,676 (35,939 to 381,738)	286,682 (106,334 to 908,776)

[1] Egg abundances corresponding to the asymptotic abundances of age-0 or age-3 from equilibrium modelling are very high with large uncertainty because the stock and recruitment curve at that point (replacement point) is very flat hence similar levels of recruitment are realized for a very large range of spawners.

Table 6.1. Example management strategies based on size limits that could be considered to define fishery reference points for Striped Bass.

Retention regulations	Minimum size (fork length, cm)	Maximum size (fork length, cm)	Comment
No size limits	na (30)	na (150)	Although no size limits are given, for purposes of modelling, a minimum size of 30 cm was assumed as the smallest fish that would be retained. Although no maximum size limit is given, a maximum size (150 cm) that exceeds the expected size of any fish is assumed
Slot size	47	61	As per recreational fisheries plan of 2016 to 2020
Maximum size only	na (30)	65	Although no minimum size limit is given, for purposes of modelling, a minimum size of 30 cm was assumed as the smallest fish that would be retained.

Table 6.2a. Model 5 - reference levels (median; 5th to 95th percentile range) derived from the equilibrium modelling based on life history parameters and population dynamics parameters for the two scenarios of values of M specific to the management strategy without any size limit for retention and no accounting for catch and release mortality.

References for Model 5	M = Z from modelling	M informed from observations
Equilibrium abundance (ages 3 to 15+) at F = 0		
Total abundance (biomass, t)	4,140 (2,120 to 11,450)	13,980 (8,040 to 24,710)
Total abundance (number, thousands)	2,320 (1,380 to 4,340)	4,700 (2,800 to 8,060)
Spawner abundance (biomass, t)	2,810 (1,430 to 8,100)	10,340 (5,400 to 19,410)
Spawner abundance (number, thousands)	1,360 (800 to 2,620)	3,110 (1,760 to 5,610)
Spawner abundance (eggs, millions)	104,300 (51,300 to 317,300)	413,900 (214,100 to 783,600)
MSY references (ages 3 to 15+)		
Total abundance (biomass; t)	1,620 (890 to 3,600)	4,610 (2,680 to 8,000)
Total abundance (number, thousands)	1,230 (740 to 2,230)	2,430 (1,460 to 4,130)
Spawner abundance (biomass, t)	1,010 (550 to 2,350)	3,200 (1,770 to 5,830)
Spawner abundance (number, thousands)	660 (390 to 1,240)	1,450 (850 to 2,550)
Spawner abundance (eggs, millions)	34,560 (18,190 to 85,230)	121,680 (65,990 to 224,330)
Fishing rate and yield at MSY		
F_{MSY} (fully recruited F)	0.18 (0.12 to 0.23)	0.17 (0.15 to 0.19)
Fcrash (fully recruited F)	0.69 (0.6 to 0.78)	0.87 (0.73 to 1)
Catch at MSY (biomass, t)	210 (130 to 380)	650 (370 to 1140)
Catch at MSY (number, thousands)	160 (100 to 270)	340 (190 to 590)
Equilibrium abundance (age-3)		
Total abundance (number, thousands)	840 (500 to 1420)	1000 (590 to 1690)
Spawner abundance (number, thousands)	290 (170 to 520)	350 (210 to 610)
Lifetime reproductive rate (number of recruits at age-3 per lifetime contribution of a spawner in absence of density-dependent compensatory survival)		
Age-3 (number)	5.01 (3.73 to 7.59)	15.55 (11.01 to 23.29)
Spawner potential per recruit references (ages 3 to 15+) (fully-recruited F)		
F at 50%SPR	0.19 (0.14 to 0.27)	0.12 (0.11 to 0.13)
F at 30%SPR	0.39 (0.28 to 0.53)	0.24 (0.22 to 0.27)

Table 6.2b. Model 4 - reference levels (median; 5th to 95th percentile range) derived from the equilibrium modelling based on life history parameters and population dynamics parameters for the two scenarios of values of M specific to the management strategy without any size limit for retention and no accounting for catch and release mortality.

References for Model 4	M = Z from modelling	M informed from observations
Equilibrium abundance (ages 3 to 15+) at F = 0		
Total abundance (biomass, t)	2,540 (1,470 to 6,620)	8,050 (5,210 to 12,600)
Total abundance (number, thousands)	1,380 (920 to 2,340)	2,670 (1,780 to 3,990)
Spawner abundance (biomass, t)	1,790 (1,040 to 4,730)	6,100 (3,600 to 10,080)
Spawner abundance (number, thousands)	860 (570 to 1,480)	1,850 (1,180 to 2,870)
Spawner abundance (eggs, millions)	66,700 (37,600 to 188,100)	244,000 (142,700 to 405,600)
MSY references (ages 3 to 15+)		
Total abundance (biomass; t)	970 (600 to 1,990)	2,620 (1,730 to 4,000)
Total abundance (number, thousands)	720 (490 to 1,180)	1,360 (920 to 2,020)
Spawner abundance (biomass, t)	650 (400 to 1,340)	1,900 (1,190 to 3,010)
Spawner abundance (number, thousands)	420 (280 to 700)	860 (570 to 1,300)
Spawner abundance (eggs, millions)	22,200 (13,400 to 48,700)	72,100 (44,400 to 115,700)
Fishing rate and yield at MSY		
F_{MSY} (fully recruited F)	0.19 (0.12 to 0.24)	0.17 (0.15 to 0.19)
Fcrash (fully recruited F)	0.73 (0.65 to 0.82)	0.88 (0.76 to 1)
Catch at MSY (biomass, t)	140 (100 to 220)	370 (240 to 580)
Catch at MSY (number, thousands)	100 (70 to 150)	190 (130 to 290)
Equilibrium abundance (age-3)		
Total abundance (number, thousands)	450 (300 to 670)	530 (350 to 780)
Spawner abundance (number, thousands)	170 (110 to 260)	200 (130 to 300)
Lifetime reproductive rate (number of recruits at age-3 per lifetime contribution of a spawner in absence of density-dependent compensatory survival)		
Age-3 (number)	5.45 (4.08 to 7.95)	15.72 (11.63 to 22.05)
Spawner potential per recruit references (ages 3 to 15+) (fully-recruited F)		
F at 50%SPR	0.19 (0.14 to 0.26)	0.12 (0.12 to 0.13)
F at 30%SPR	0.39 (0.28 to 0.52)	0.24 (0.22 to 0.27)

Table 6.3a. Model 5 - reference point summaries (median; 5th to 95th percentile range) from the equilibrium modelling based on life history and population dynamics parameters for the two scenarios of values of M and for the default management strategy of no size limit for retention and no accounting for catch and release mortality.

Reference	Units	M = Z from modelling	M informed from observations
Upper Stock Reference (spawners ages 3 to 15+)			
80%B_{MSY}	Eggs (millions)	25,780 (13,460 to 63,420)	91,320 (49,990 to 168,040)
	Biomass (t)	780 (420 to 1,780)	2,450 (1,360 to 4,450)
	Number (thousands)	530 (310 to 1,020)	1,210 (710 to 2,110)
	Eggs per fish	48,210 (4,0200 to 66,970)	75,670 (64,820 to 86,000)
	Eggs per kg	33,370 (31,370 to 36,070)	37,290 (35,950 to 38,280)
	Mean age of spawners	4.33 (4.05 to 5.04)	5.28 (4.91 to 5.64)
	Mean weight (kg) of spawners	1.45 (1.28 to 1.86)	2.03 (1.8 to 2.25)
50%SPR	Eggs (millions)	32,440 (16,110 to 72,360)	165,250 (82,190 to 315,970)
	Biomass (t)	950 (500 to 2,020)	4,280 (2,190 to 8,120)
	Number (thousands)	620 (360 to 1,150)	1,760 (990 to 3,130)
Limit Reference Point (spawners ages 3 to 15+)			
Brecover	Eggs (millions)	200	
	Biomass (t)	6.5	
	Number (thousands)	4.5	
40%B_{MSY}	Eggs (millions)	11,600 (5,970 to 28,660)	40,580 (22,430 to 74,480)
	Biomass (t)	370 (200 to 850)	1,160 (650 to 2,090)
	Number (thousands)	280 (160 to 580)	700 (410 to 1,220)
	Eggs per fish	41,440 (34,840 to 53,150)	58,030 (50,080 to 65,850)
	Eggs per kg	31,750 (29,660 to 34,220)	35,150 (33,640 to 36,380)
	Mean age of spawners	4.1 (3.88 to 4.52)	4.66 (4.39 to 4.92)
	Mean weight (kg) of spawners	1.31 (1.17 to 1.55)	1.65 (1.49 to 1.81)
30%SPR	Eggs (millions)	7,020 (1,520 to 20,350)	82,420 (38,020 to 161,800)
	Biomass (t)	230 (50 to 620)	2,220 (1,070 to 4,300)
	Number (thousands)	180 (50 to 430)	1,120 (610 to 2,040)
Half K – Bev Holt	Eggs (millions)	29,950 (17,450 to 54,180)	
	Biomass (t)	890 (530 to 1,590)	870 (520 to 1,560)
	Number (thousands)	590 (360 to 1030)	560 (350 to 980)
	Eggs per fish	50,430 (43,480 to 59,350)	53,250 (46,240 to 60,640)

Reference	Units	M = Z from modelling	M informed from observations
	Eggs per kg	33,810 (32,240 to 35,240)	34,360 (32,870 to 35,610)
	Mean age of spawners	4.41 (4.16 to 4.74)	4.49 (4.25 to 4.74)
	Mean weight (kg) of spawners	1.49 (1.35 to 1.69)	1.55 (1.40 to 1.70)
Half equilibrium	Eggs (millions)	19,300 (10,820 to 37,630)	26,160 (15,420 to 47,040)
	Biomass (t)	590 (340 to 1,110)	770 (460 to 1,360)
	Number (thousands)	420 (260 to 750)	510 (310 to 880)
	Eggs per fish	45,350 (38,360 to 55,650)	51,470 (44,590 to 58,450)
	Eggs per kg	32,720 (30,840 to 34,630)	34,020 (32,470 to 35,270)
	Mean age of spawners	4.23 (3.99 to 4.61)	4.43 (4.20 to 4.67)
	Mean weight (kg) of spawners	1.39 (1.24 to 1.61)	1.51 (1.37 to 1.66)
Removal rate reference point (fully recruited F)			
MSY	MSY	0.18 (0.12 to 0.23)	0.17 (0.15 to 0.19)
50%SPR	50%SPR	0.19 (0.14 to 0.27)	0.12 (0.11 to 0.13)
30%SPR	30%SPR	0.39 (0.28 to 0.53)	0.24 (0.22 to 0.27)

Table 6.3b. Model 4 - reference point summaries (median; 5th to 95th percentile range) from the equilibrium modelling based on life history and population dynamics parameters derived for the two scenarios of values of M and for the default management strategy of no size limit for retention and no accounting for catch and release mortality.

Reference	Units	M = Z from modelling	M informed from observations
Upper Stock Reference (spawners ages 3 to 15+)			
80%B_{MSY}	Eggs (millions)	16,700 (10,000 to 36,500)	54,300 (33,700 to 86,400)
	Biomass (t)	500 (310 to 1,030)	1,460 (920 to 2,290)
	Number (thousands)	340 (220 to 580)	720 (480 to 1,090)
	Eggs per fish	49,200 (41,700 to 66,100)	75,400 (65,600 to 85,000)
	Eggs per kg	33,600 (31,800 to 36,000)	37,300 (36,100 to 38,200)
	Mean age of spawners	4.36 (4.11 to 5)	5.28 (4.94 to 5.61)
	Mean weight (kg) of spawners	1.46 (1.31 to 1.83)	2.02 (1.82 to 2.23)
50%SPR	Eggs (millions)	22,500 (12,900 to 44,900)	98,200 (56,000 to 163,600)
	Biomass (t)	650 (390 to 1,240)	2,540 (1,480 to 4,190)
	Number (thousands)	420 (280 to 680)	1,050 (660 to 1,620)
Limit Reference Point (spawners ages 3 to 15+)			
Brecover	Eggs (millions)	200	
	Biomass (t)	6.5	
	Number (thousands)	4.5	
40%B_{MSY}	Eggs (millions)	7,600 (4,500 to 16,700)	24,500 (15,400 to 38,500)
	Biomass (t)	240 (150 to 500)	700 (450 to 1,080)
	Number (thousands)	180 (120 to 330)	420 (280 to 630)
	Eggs per fish	41,900 (35,900 to 52,700)	58,000 (50,700 to 65,200)
	Eggs per kg	31,900 (30,100 to 34,200)	35,200 (33,800 to 36,300)
	Mean age of spawners	4.11 (3.91 to 4.5)	4.65 (4.41 to 4.90)
	Mean weight (kg) of spawners	1.31 (1.2 to 1.54)	1.65 (1.50 to 1.80)
30%SPR	Eggs (millions)	6,400 (2,400 to 14,900)	49,000 (26,400 to 84,200)
	Biomass (t)	200 (80 to 440)	1,320 (730 to 2,230)
	Number (thousands)	160 (70 to 300)	670 (410 to 1,050)
Half K – Bev Holt	Eggs (millions)	17,300 (11,300 to 26,500)	
	Biomass (t)	510 (340 to 770)	510 (340 to 760)
	Number (thousands)	350 (240 to 510)	330 (220 to 490)
	Eggs per fish	49,600 (43,600 to 57,100)	52,400 (46,400 to 59,100)

Reference	Units	M = Z from modelling	M informed from observations
	Eggs per kg	33,700 (32,300 to 35,000)	34,200 (32,900 to 35,400)
	Mean age of spawners	4.38 (4.17 to 4.65)	4.46 (4.26 to 4.69)
	Mean weight (kg) of spawners	1.47 (1.35 to 1.63)	1.53 (1.41 to 1.67)
Half equilibrium	Eggs (millions)	11,600 (7,300 to 19,300)	15,200 (10,000 to 23,000)
	Biomass (t)	350 (230 to 570)	450 (300 to 670)
	Number (thousands)	260 (170 to 390)	300 (210 to 440)
	Eggs per fish	45,300 (39,100 to 53,600)	50,800 (44,900 to 57,200)
	Eggs per kg	32,700 (31,100 to 34,400)	33,900 (32,600 to 35,100)
	Mean age of spawners	4.23 (4.02 to 4.53)	4.41 (4.21 to 4.62)
	Mean weight (kg) of spawners	1.38 (1.26 to 1.56)	1.50 (1.38 to 1.63)
Removal rate reference point (fully recruited F)			
MSY	MSY	0.19 (0.12 to 0.24)	0.17 (0.15 to 0.19)
50%SPR	50%SPR	0.19 (0.14 to 0.26)	0.12 (0.12 to 0.13)
30%SPR	30%SPR	0.39 (0.28 to 0.52)	0.24 (0.22 to 0.27)

Table 6.4a. Model 5 - comparison of calculated reference points for different fishing strategies conditioned by size limits. The equilibrium simulations were run based on life history characteristics from model 5 and assuming M for ages 4 to 15+ based on acoustic tagging observations. There is no accounting for catch and release mortality in these scenarios. Summary statistics shown are the median with the 5th to 95th percentile range.

Reference	Unit	No size restrictions (slot = 30 to 150)	Slot size (47 to 61 cm FL)	Maximum size limit (30 to 65 cm FL)
MSY references (ages 3 to 15+)				
B_{MSY}	Total abundance (biomass, t)	4,610 (2,680 to 8,000)	3,720 (2,210 to 6,450)	3,800 (2,250 to 6,630)
	Total abundance (number, thousands)	2,430 (1,460 to 4,130)	2,060 (1,250 to 3,520)	1,990 (1,200 to 3,390)
	Spawners (biomass, t)	3,200 (1,770 to 5,830)	2,550 (1,460 to 4,540)	2,610 (1,480 to 4,700)
	Spawners (number, thousands)	1,450 (850 to 2,550)	1,180 (720 to 2,040)	1,140 (690 to 1,970)
	Spawners (eggs, millions)	121,680 (65,990 to 224,330)	94,930 (53,650 to 169,950)	98,600 (55,150 to 179,550)
	Catch at MSY (weight, t)	650 (370 to 1,140)	530 (300 to 940)	490 (280 to 850)
	Catch at MSY (number, thousands)	340 (190 to 590)	360 (210 to 640)	400 (230 to 700)
Upper Stock Reference (spawners 3 to 15+)				
80%B_{MSY}	Eggs (millions)	91,320 (49,990 to 168,040)	71,270 (40,530 to 127,900)	74,590 (41,800 to 135,270)
	Biomass (t)	2,450 (1,360 to 4,450)	1,960 (1,130 to 3,490)	2,010 (1,150 to 3,630)
	Number (thousands)	1,210 (710 to 2,110)	990 (600 to 1,710)	940 (570 to 1,630)
	Eggs per spawner	75,670 (64,820 to 86,000)	71,890 (61,140 to 82,490)	79,000 (66,840 to 90,880)
	Eggs per kg of spawner	37,290 (35,950 to 38,280)	36,390 (35,060 to 37,410)	37,080 (35,740 to 38,070)
	Mean age of spawners	5.28 (4.91 to 5.64)	5.26 (4.87 to 5.66)	5.52 (5.06 to 5.96)
	Mean weight (kg) of spawners	2.03 (1.8 to 2.25)	1.97 (1.74 to 2.21)	2.13 (1.87 to 2.39)
50%SPR	Eggs (millions)	165,250 (82,190 to 315,970)	189,420 (94,270 to 362,900)	189,550 (94,260 to 363,200)
	Biomass (t)	4,280 (2,190 to 8,120)	4,880 (2,480 to 9,300)	4,850 (2,480 to 9,240)
	Number (thousands)	1,760 (990 to 3,130)	1,820 (1,030 to 3,250)	1,770 (1,000 to 3,170)
Limit Reference Point (spawners 3 to 15+)				
Brecover	Eggs (millions)		200	
	Biomass (t)		6.5	
	Number (thousands)		4.5	
40%B_{MSY}	Eggs (millions)	40,580 (22,430 to 74,480)	30,960 (17,970 to 55,620)	32,880 (18,660 to 59,550)
	Biomass (t)	1,160 (650 to 2,090)	920 (540 to 1,620)	940 (540 to 1,680)
	Number (thousands)	700 (410 to 1,220)	580 (360 to 1,000)	540 (330 to 930)
	Eggs per spawner	58,000 (50,080 to 65,850)	53,100 (45,640 to 61,190)	61,200 (51,850 to 70,680)
	Eggs per kg of spawner	35,150 (33,640 to 36,380)	33,830 (32,310 to 35,110)	35,040 (33,470 to 36,270)
	Mean age of spawners	4.66 (4.39 to 4.92)	4.57 (4.3 to 4.87)	4.86 (4.52 to 5.21)

Reference	Unit	No size restrictions (slot = 30 to 150)	Slot size (47 to 61 cm FL)	Maximum size limit (30 to 65 cm FL)
	Mean weight (kg) of spawners	1.65 (1.49 to 1.81)	1.57 (1.41 to 1.74)	1.75 (1.55 to 1.95)
30%SPR	Eggs (millions)	82,420 (38,020 to 161,800)	97,590 (44,980 to 192,980)	98,420 (45,770 to 192,480)
	Biomass (t)	2,220 (1,070 to 4,300)	2,620 (1,260 to 5,080)	2,610 (1,260 to 5,030)
	Number (thousands)	1120 (610 to 2,040)	1200 (660 to 2,150)	1,130 (620 to 2,040)
Half K – Bev Holt	Eggs (millions)	29,950 (17,450 to 54,180)	29,840 (17,310 to 53,970)	29,920 (17,400 to 54,370)
	Biomass (t)	870 (520 to 1,560)	890 (520 to 1,580)	860 (510 to 1,540)
	Number (thousands)	560 (350 to 980)	570 (350 to 990)	500 (310 to 870)
	Eggs per spawner	53,250 (46,240 to 60,640)	52,380 (44,720 to 60,950)	59,430 (50,690 to 68,430)
	Eggs per kg of spawner	34,360 (32,870 to 35,610)	33,700 (32,100 to 35,090)	34,780 (33,260 to 36,000)
	Mean age of spawners	4.49 (4.25 to 4.74)	4.54 (4.26 to 4.86)	4.79 (4.47 to 5.13)
	Mean weight (kg) of spawners	1.55 (1.4 to 1.7)	1.55 (1.39 to 1.74)	1.71 (1.52 to 1.9)
Half equilibrium	Eggs (millions)	26,160 (15,420 to 47,040)	25,980 (15,350 to 46,800)	26,160 (15,450 to 47,010)
	Biomass (t)	770 (460 to 1,360)	780 (470 to 1,380)	760 (460 to 1,350)
	Number (thousands)	510 (310 to 880)	520 (320 to 890)	460 (280 to 790)
	Eggs per spawner	51,470 (44,590 to 58,450)	50,240 (42,910 to 58,420)	57,320 (48,860 to 66,070)
	Eggs per kg of spawner	34,020 (32,470 to 35,270)	33,300 (31,680 to 34,710)	34,450 (32,890 to 35,700)
	Mean age of spawners	4.43 (4.2 to 4.67)	4.47 (4.2 to 4.77)	4.72 (4.41 to 5.04)
	Mean weight (kg) of spawners	1.51 (1.37 to 1.66)	1.51 (1.35 to 1.68)	1.66 (1.49 to 1.85)
Fishing rate (fully recruited F)	MSY	0.17 (0.15 to 0.19)	0.66 (0.58 to 0.74)	0.33 (0.29 to 0.38)
	50%SPR	0.12 (0.11 to 0.13)	0.36 (0.34 to 0.38)	0.19 (0.18 to 0.20)
	30%SPR	0.24 (0.22 to 0.27)	0.64 (0.60 to 0.72)	0.34 (0.32 to 0.36)
Exploitation rate (catch number divided by total abundance number at F)	MSY	0.14 (0.12 to 0.16)	0.17 (0.15 to 0.20)	0.20 (0.17 to 0.23)
	50%SPR	0.10 (0.09 to 0.11)	0.09 (0.08 to 0.11)	0.11 (0.10 to 0.12)
	30%SPR	0.19 (0.17 to 0.21)	0.17 (0.15 to 0.2)	0.20 (0.19 to 0.22)

Table 6.4b. Model 4 - comparison of calculated reference points for different fishing strategies conditioned by size limits. The equilibrium simulations were run based on life history characteristics from model 4 and assuming M for ages 4 to 15+ based on acoustic tagging observations. There is no accounting for catch and release mortality in these scenarios. Summary statistics shown are the median with the 5th to 95th percentile range.

Reference	Unit	No size restrictions (slot = 30 to 150)	Slot size (47 to 61 cm FL)	Maximum size limit (30 to 65 cm FL)
MSY references (ages 3 to 15+)				
B_{MSY}	Total abundance (biomass, t)	2,620 (1,730 to 4,000)	2,110 (1,410 to 3,170)	2,180 (1,440 to 3,280)
	Total abundance (number, thousands)	1,360 (920 to 2,020)	1,150 (780 to 1,690)	1,110 (760 to 1,640)
	Spawners (biomass, t)	1,900 (1,190 to 3,010)	1,510 (970 to 2,330)	1,560 (990 to 2,430)
	Spawners (number, thousands)	860 (570 to 1,300)	700 (480 to 1,050)	680 (460 to 1,010)
	Spawners (eggs, millions)	72,100 (44,400 to 115,700)	56,000 (35,500 to 87,500)	58,500 (36,700 to 92,800)
	Catch at MSY (weight, t)	370 (240 to 580)	310 (200 to 470)	280 (180 to 430)
	Catch at MSY (number, thousands)	190 (130 to 290)	210 (140 to 320)	230 (150 to 340)
Upper Stock Reference (spawners 3 to 15+)				
80%B_{MSY}	Eggs (millions)	54,300 (33,700 to 86,400)	42,300 (26,900 to 65,600)	44,500 (27,900 to 70,200)
	Biomass (t)	1,460 (920 to 2,290)	1,170 (750 to 1,790)	1,200 (770 to 1,870)
	Number (thousands)	720 (480 to 1,090)	590 (410 to 880)	570 (380 to 840)
	Eggs per spawner	75,400 (65,600 to 85,000)	71,100 (61,500 to 80,700)	78,800 (67,800 to 89,900)
	Eggs per kg of spawner	37,300 (36,100 to 38,200)	36,300 (35,100 to 37,300)	37,100 (35,900 to 38,000)
	Mean age of spawners	5.28 (4.94 to 5.61)	5.23 (4.87 to 5.6)	5.51 (5.1 to 5.93)
	Mean weight (kg) of spawners	2.02 (1.82 to 2.23)	1.96 (1.75 to 2.17)	2.12 (1.89 to 2.37)
50%SPR	Eggs (millions)	98,200 (56,000 to 163,600)	111,800 (63,300 to 189,000)	111,800 (63,200 to 188,200)
	Biomass (t)	2,540 (1,480 to 4,190)	2,880 (1,660 to 4,790)	2,860 (1,650 to 4,770)
	Number (thousands)	1,050 (660 to 1,620)	1,080 (690 to 1,680)	1,050 (670 to 1,630)
Limit Reference Point (spawners 3 to 15+)				
Brecover	Eggs (millions)	200		
	Biomass (t)	6.5		
	Number (thousands)	4.5		
40%B_{MSY}	Eggs (millions)	24,500 (15,400 to 38,500)	18,600 (12,100 to 28,600)	19,800 (12,700 to 31,100)
	Biomass (t)	700 (450 to 1,080)	550 (370 to 840)	570 (370 to 870)
	Number (thousands)	420 (280 to 630)	350 (240 to 520)	330 (220 to 480)
	Eggs per spawner	58,000 (50,700 to 65,200)	52,600 (46,000 to 60,000)	61,100 (52,700 to 69,900)
	Eggs per kg of spawner	35,200 (33,800 to 36,300)	33,800 (32,400 to 35,000)	35,100 (33,700 to 36,200)
	Mean age of spawners	4.65 (4.41 to 4.90)	4.55 (4.31 to 4.82)	4.86 (4.55 to 5.18)

Reference	Unit	No size restrictions (slot = 30 to 150)	Slot size (47 to 61 cm FL)	Maximum size limit (30 to 65 cm FL)
	Mean weight (kg) of spawners	1.65 (1.50 to 1.80)	1.56 (1.42 to 1.72)	1.74 (1.57 to 1.93)
30%SPR	Eggs (millions)	49,000 (26,400 to 84,200)	57,700 (30,700 to 100,300)	58,100 (31,100 to 100,700)
	Biomass (t)	1,320 (730 to 2,230)	1,550 (850 to 2,630)	1,540 (850 to 2,620)
	Number (thousands)	670 (410 to 1,050)	720 (450 to 1,120)	670 (420 to 1,050)
Half K – Bev Holt	Eggs (millions)	17,300 (11,400 to 26,500)	17,200 (11,200 to 26,400)	17,300 (11,300 to 26,600)
	Biomass (t)	510 (340 to 760)	510 (340 to 770)	500 (330 to 760)
	Number (thousands)	330 (220 to 490)	340 (230 to 490)	290 (200 to 440)
	Eggs per spawner	52,400 (46,400 to 59,100)	51,100 (44,500 to 58,800)	58,600 (51,000 to 66,900)
	Eggs per kg of spawner	34,200 (32,900 to 35,400)	33,500 (32,100 to 34,800)	34,700 (33,400 to 35,800)
	Mean age of spawners	4.46 (4.26 to 4.69)	4.5 (4.26 to 4.78)	4.76 (4.49 to 5.06)
	Mean weight (kg) of spawners	1.53 (1.41 to 1.67)	1.53 (1.39 to 1.69)	1.69 (1.53 to 1.86)
Half equilibrium	Eggs (millions)	15,200 (10,000 to 23,000)	15,100 (10,000 to 22,800)	15,200 (10,000 to 22,900)
	Biomass (t)	450 (300 to 670)	460 310 to 680)	440 (300 to 660)
	Number (thousands)	300 (210 to 440)	310 210 to 450)	270 (180 to 390)
	Eggs per spawner	50,800 (44,900 to 57,200)	49,100 (42,800 to 56,400)	56,600 (49,200 to 64,600)
	Eggs per kg of spawner	33,900 (32,600 to 35,100)	33,100 (31,700 to 34,400)	34,400 (33,000 to 35,600)
	Mean age of spawners	4.41 (4.21 to 4.62)	4.42 (4.19 to 4.69)	4.69 (4.42 to 4.98)
	Mean weight (kg) of spawners	1.50 (1.38 to 1.63)	1.48 (1.35 to 1.64)	1.65 (1.49 to 1.82)
Fishing rate (fully recruited F)	MSY	0.17 (0.15 to 0.19)	0.68 (0.6 to 0.74)	0.34 (0.3 to 0.37)
	50%SPR	0.12 (0.12 to 0.13)	0.36 (0.34 to 0.38)	0.19 (0.18 to 0.2)
	30%SPR	0.24 (0.22 to 0.27)	0.66 (0.62 to 0.72)	0.34 (0.32 to 0.36)
Exploitation rate (catch number divided by total abundance number at F)	MSY	0.14 (0.13 to 0.16)	0.18 (0.16 to 0.21)	0.20 (0.18 to 0.23)
	50%SPR	0.10 (0.10 to 0.11)	0.10 (0.09 to 0.11)	0.11 (0.10 to 0.12)
	30%SPR	0.19 (0.18 to 0.21)	0.18 (0.16 to 0.20)	0.20 (0.19 to 0.22)

Table 6.5. Model 5 - comparison of calculated reference points for different fishing strategies conditioned by size limits and considering whether catch and release mortality is included (A.CR = 0) or excluded (A.CR = 9%) in the equilibrium modelling. The equilibrium simulations were run based on life history characteristics from model 5 and assuming M for ages 4 to 15+ based on informed observations.

Reference values for Model 5	Slot size (47 to 61 cm FL)		Maximum size limit (30 to 65 cm FL)	
	A.CR = 0	A.CR = 9%	A.CR = 0	A.CR = 9%
MSY references (ages 3 to 15+)				
Total abundance (thousands)	2,060 (1,250 to 3,520)	2,020 (1,240 to 3,420)	1,990 (1,200 to 3,390)	1,960 (1,200 to 3,330)
Spawners (thousands)	1,180 (720 to 2,040)	1,160 (700 to 2,020)	1,140 (690 to 1,970)	1,120 (680 to 1,960)
Catch at MSY (number, thousands)	360 (210 to 640)	310 (180 to 550)	400 (230 to 700)	380 (220 to 670)
Losses at MSY (number, thousands)		360 (210 to 640)		390 (230 to 690)
Upper Stock Reference (spawners 3 to 15+)				
80%B_{MSY} (number, thousands)	990 (600 to 1,710)	970 (590 to 1,690)	940 (570 to 1,630)	930 (560 to 1,630)
50%SPR (number, thousands)	1820 (,1030 to 3,250)	,1800 (1,020 to 3,220)	1,770 (1,000 to 3,170)	1,760 (1,000 to 3,150)
Limit Reference Point (spawners 3 to 15+)				
Brecover (number, thousands)	4.5			
40%B_{MSY} (number, thousands)	580 (360 to 1,000)	560 (340 to 980)	540 (330 to 930)	530 (320 to 930)
30%SPR (number, thousands)	1,200 (660 to 2,150)	1,160 (640 to 2,090)	1,130 (620 to 2,040)	1,120 (610 to 2,010)
Half K – Bev Holt (number, thousands)	570 (350 to 990)	580 (350 to 1,000)	500 (310 to 870)	520 (320 to 900)
Half equilibrium (number, thousands)	520 (320 to 890)	520 (320 to 900)	460 (280 to 790)	470 (290 to 810)
Fishing rate (fully recruited F for retained catch)				
MSY	0.66 (0.58 to 0.74)	0.56 (0.48 to 0.62)	0.33 (0.29 to 0.38)	0.32 (0.28 to 0.36)
50%SPR	0.36 (0.34 to 0.38)	0.30 (0.28 to 0.32)	0.19 (0.18 to 0.20)	0.18 (0.17 to 0.19)
30%SPR	0.64 (0.60 to 0.72)	0.56 (0.52 to 0.62)	0.34 (0.32 to 0.36)	0.32 (0.31 to 0.35)

FIGURES

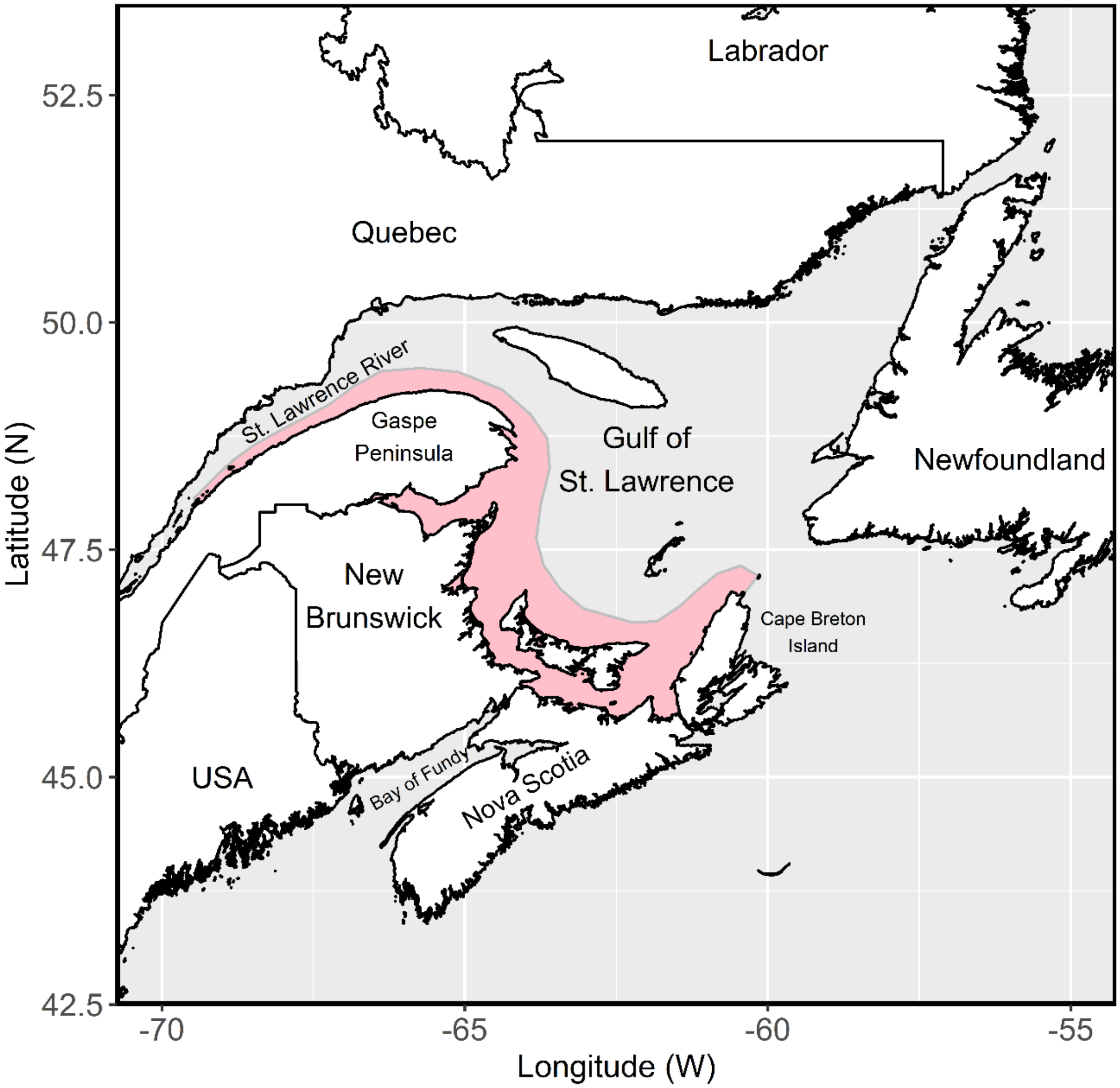


Figure 1.1. Geographic distribution (red shaded area) of the southern Gulf of St. Lawrence Striped Bass population in eastern Canada.

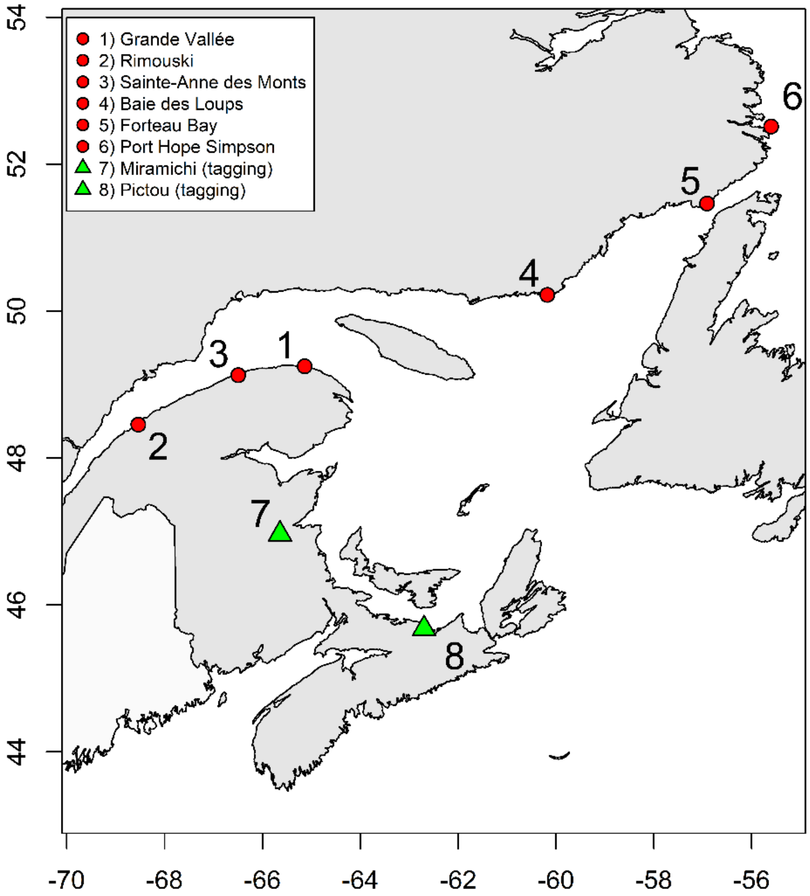


Figure 1.2. Recoveries (circle symbols) in 2017 of Striped Bass tagged in the southern Gulf of St. Lawrence in exceptional areas outside the historic range of the population, including in the estuary of the St. Lawrence River, the north shore of the St. Lawrence, and at the southern Labrador Port Hope Simpson acoustic array deployed by Fisheries and Oceans Canada. Details on external tag recoveries are provided in DFO (2018). The figure is amended from DFO (2018) to show the acoustic array location (Port Hope Simpson) in southern Labrador. Acoustic tag identification codes detected in Labrador were from fish tagged in Miramichi, in the Gaspe region, and in the St. Lawrence River (see Table 3.1 for details). No acoustic tag identification codes for Striped Bass were recorded at the southern Labrador acoustic array in 2018 (M. Robertson, DFO Newfoundland and Labrador Region, pers. comm.).

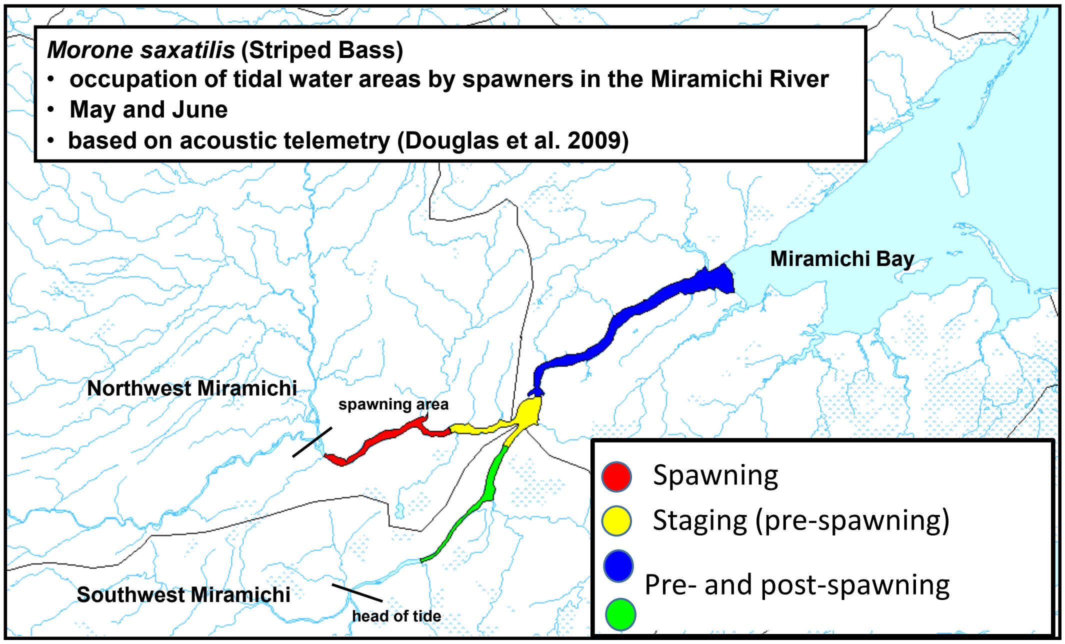


Figure 1.3. Location of the spawning area of the Northwest Miramichi as well as pre- and post-spawning areas of the Miramichi River occupied by Striped Bass, based on the acoustic telemetry study of Douglas et al. (2009).

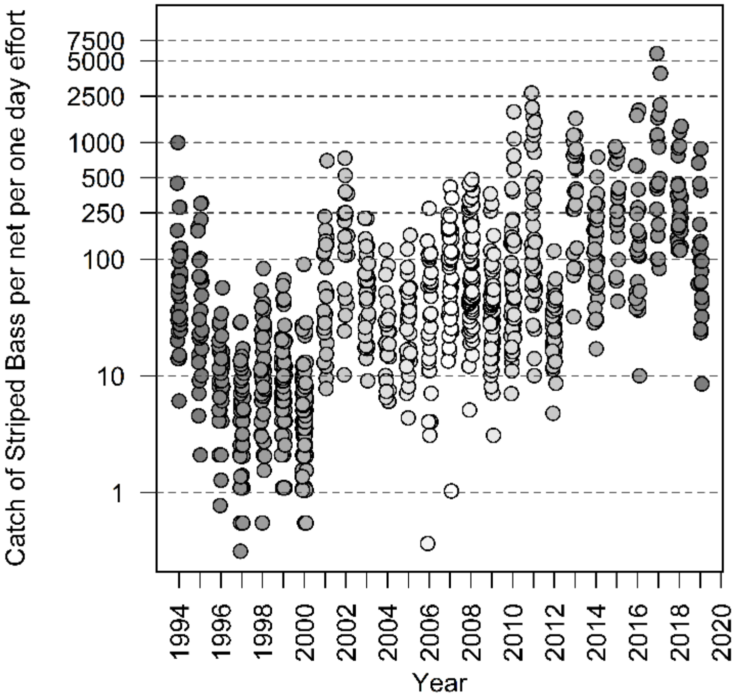


Figure 3.1. The number of Striped Bass captured per net per day of effort from monitoring of the commercial gaspereau fishery in the Northwest Miramichi, 1994 to 2019. The catch rates are not adjusted for the proportion of the spawners available for capture in the fishery. In 2012, the spawning was very early and the majority of the fish was considered to have left the area and were not available to the fishery, hence no estimate was provided for that year. The points within a year are jittered slightly for clarity. The figure is taken from DFO (2020).

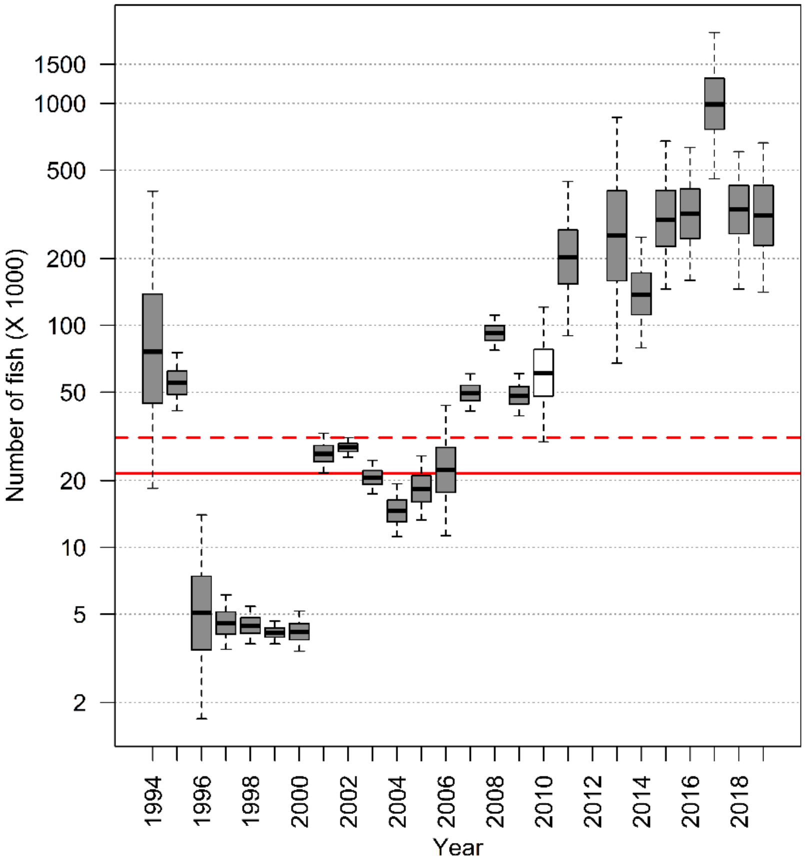


Figure 3.2. Estimated abundance of adult Striped Bass spawners in the Northwest Miramichi estuary between 1994 and 2019. The estimates are shown on a logarithmic scale for visibility of the full range of abundance values over the time series. The estimate for 2010 (unshaded interquartile box) is considered to be an underestimate due to the earlier timing of the spawning events (Douglas and Chaput 2011a). There is no estimate for 2012 because spawning was very early and Striped Bass left the sampling area prior to monitoring activities (DFO 2013). Box plots are interpreted as follows: dash is the median, boxes are the interquartile range, and the vertical dashes are the 5th to 95th percentile ranges. The solid and dashed horizontal lines show the recovery objectives defined in the Recovery Potential Assessment in support of the Species at Risk Act listing decision process (DFO 2006). The figure is taken from DFO (2020).

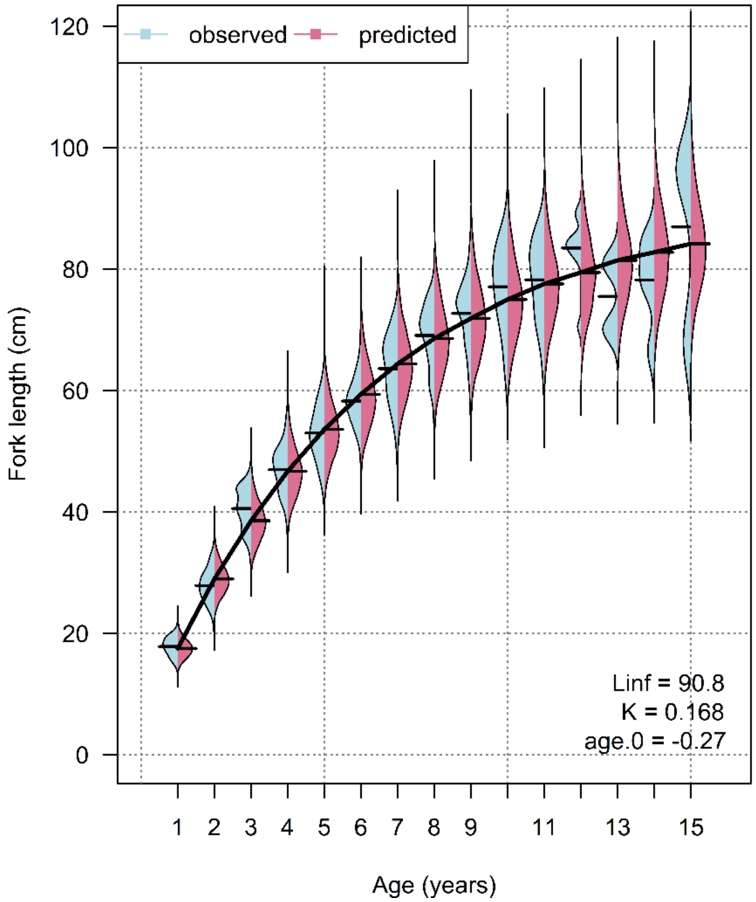


Figure 3.3. Bean plot summaries of the fork length distributions (cm) at age for the observed data used in the von Bertalanffy model fits (light blue) and the posterior distribution of the predicted fork length at age (light red) of Striped Bass from the Miramichi River. The solid black line is the mean predicted fork length at age from the posterior distributions

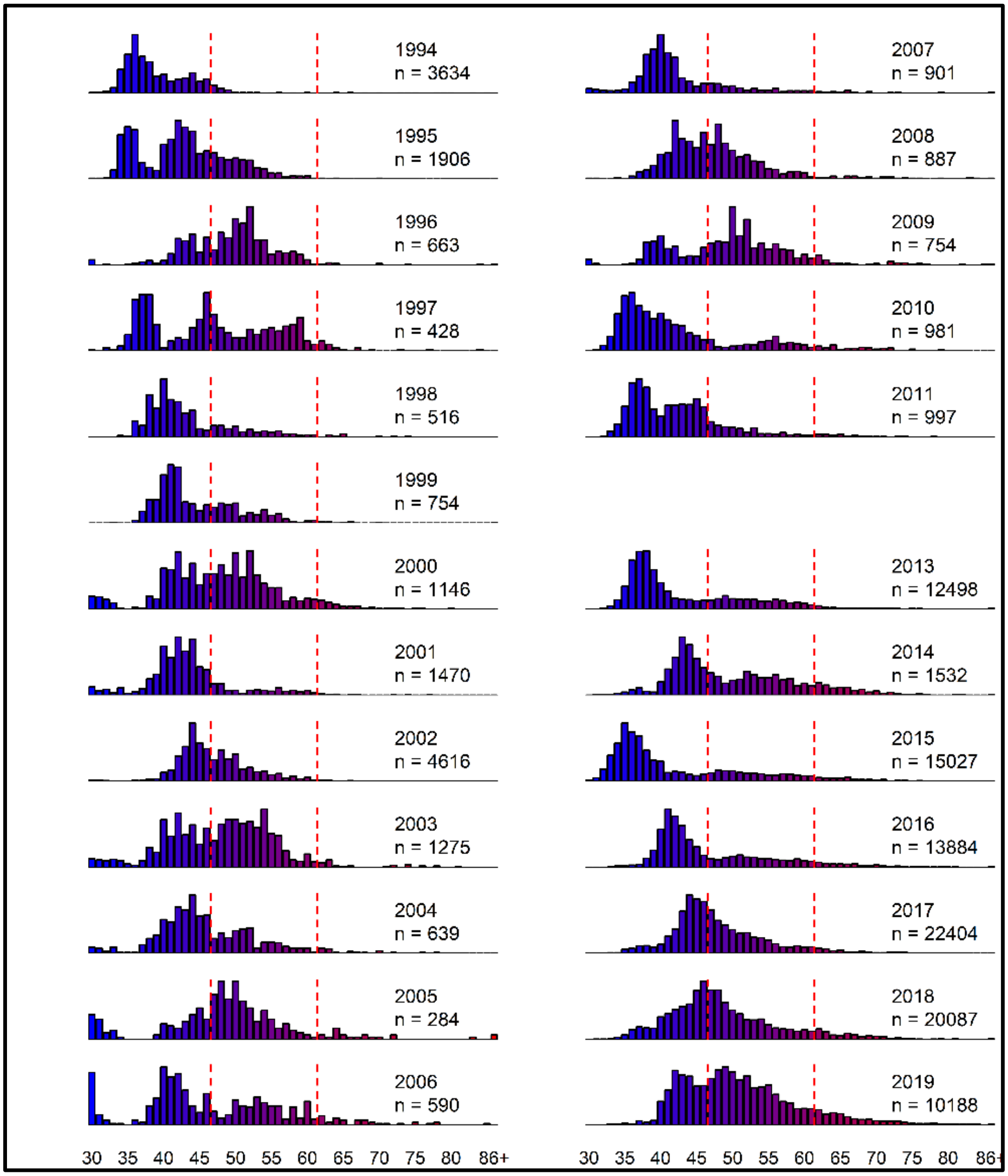


Figure 3.4. Sampled fork length distributions (cm bins) of Striped Bass on the spawning grounds, 1994 to 2019. The dashed vertical lines in each panel correspond to the minimum and maximum length range (47 to 61 cm) of the recreational fishery retention slot limit in effect since 2014.

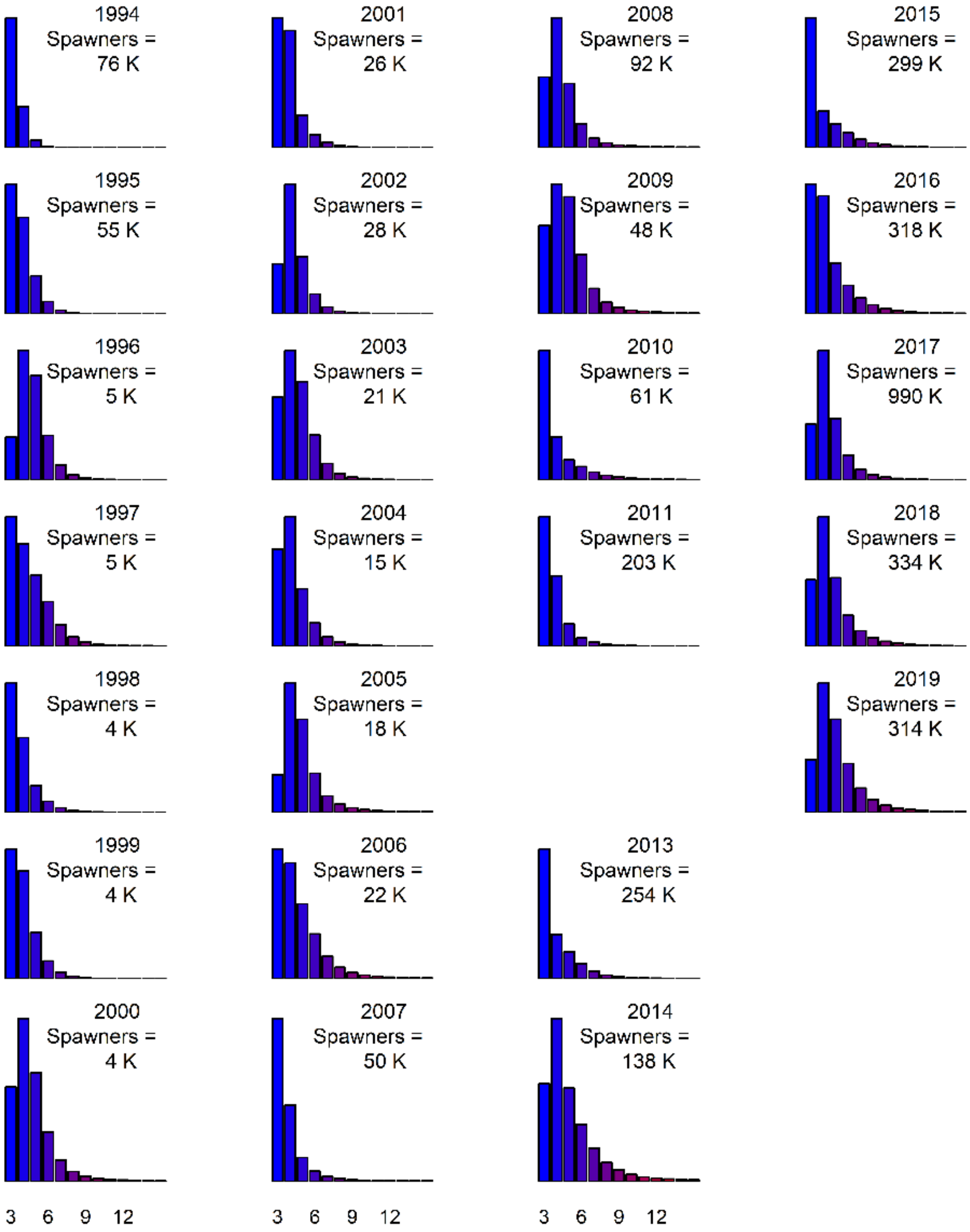


Figure 3.5. The estimated abundances (number, on a relative scale) at age of Striped Bass on the spawning grounds, 1994 to 2019, excluding 2012. The last age group includes fish age 15 and older. In each panel, the median of the estimated spawner abundance (thousands, K) is shown. The estimate for 2010 is considered to be an underestimate due to the earlier timing of the spawning events (Douglas and Chaput 2011a). There is no estimate for 2012 because spawning was very early and Striped Bass left the sampling area prior to monitoring activities (DFO 2013). Estimated spawner abundances at age are provided in Appendix 3.

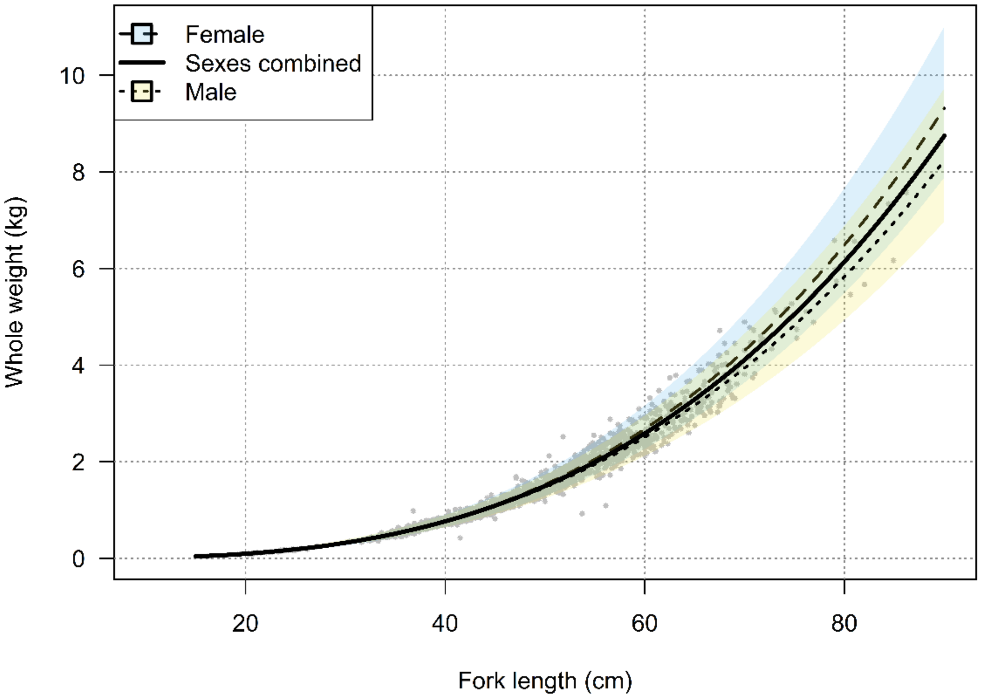


Figure 3.6. Fork length (cm) to whole weight (kg) relationship for Striped Bass from the Miramichi River, obtained from samples collected in May and June 2013 to 2015. The solid line is the mean regression line for sexes combined, the dashed line is for female bass, and the dashed line is for male bass. The coloured polygons represent the approximate 95% confidence interval for the mean line for females (light blue) and males (light yellow), respectively.

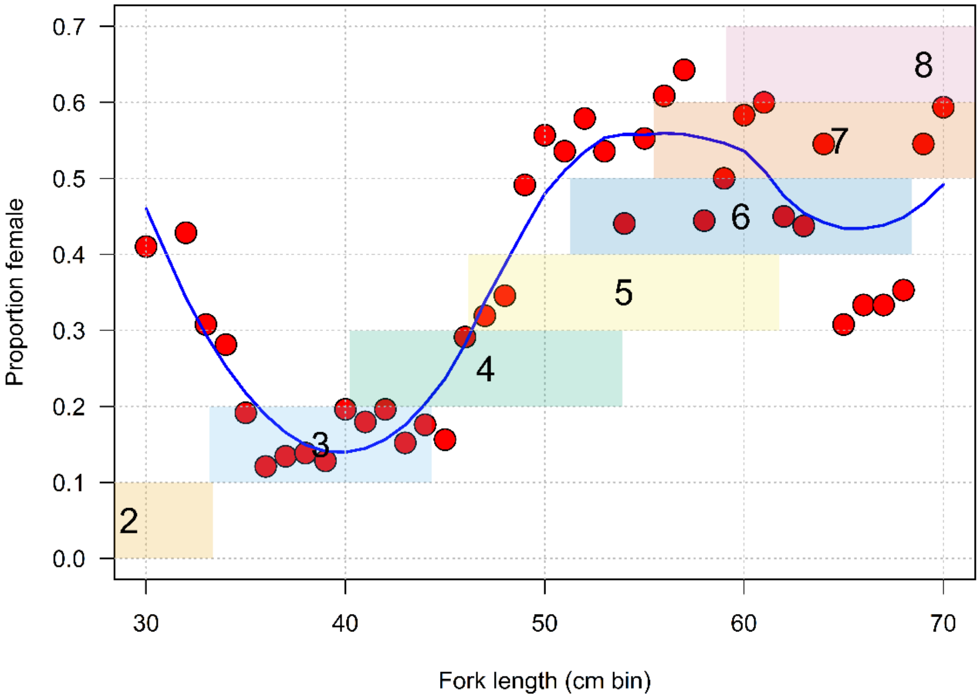


Figure 3.7. Proportion female by fork length (cm bin) of Striped Bass from the Miramichi River, obtained from sacrificed samples collected in May and June, 2013 to 2015. The blue line is a LOESS smoother of the proportion female at age (span = 0.5). The shaded rectangles illustrate the 95% confidence interval range of the predicted fork length for ages 2 to 8; for ages 2, 7 and 8, the confidence range extends beyond the fork length axis range. The size range of samples collected was 19.2 to 86.2 cm fork length. The symbol for the 30 cm fork length bin includes all bass less than or equal to 30 cm (n = 39) and the symbol for the 70 cm fork length bin includes all bass greater than or equal to 70 cm (n = 32). Sample sizes in fork length bins for other symbols range from n = 7 at 32 cm to n = 102 at 42 cm.

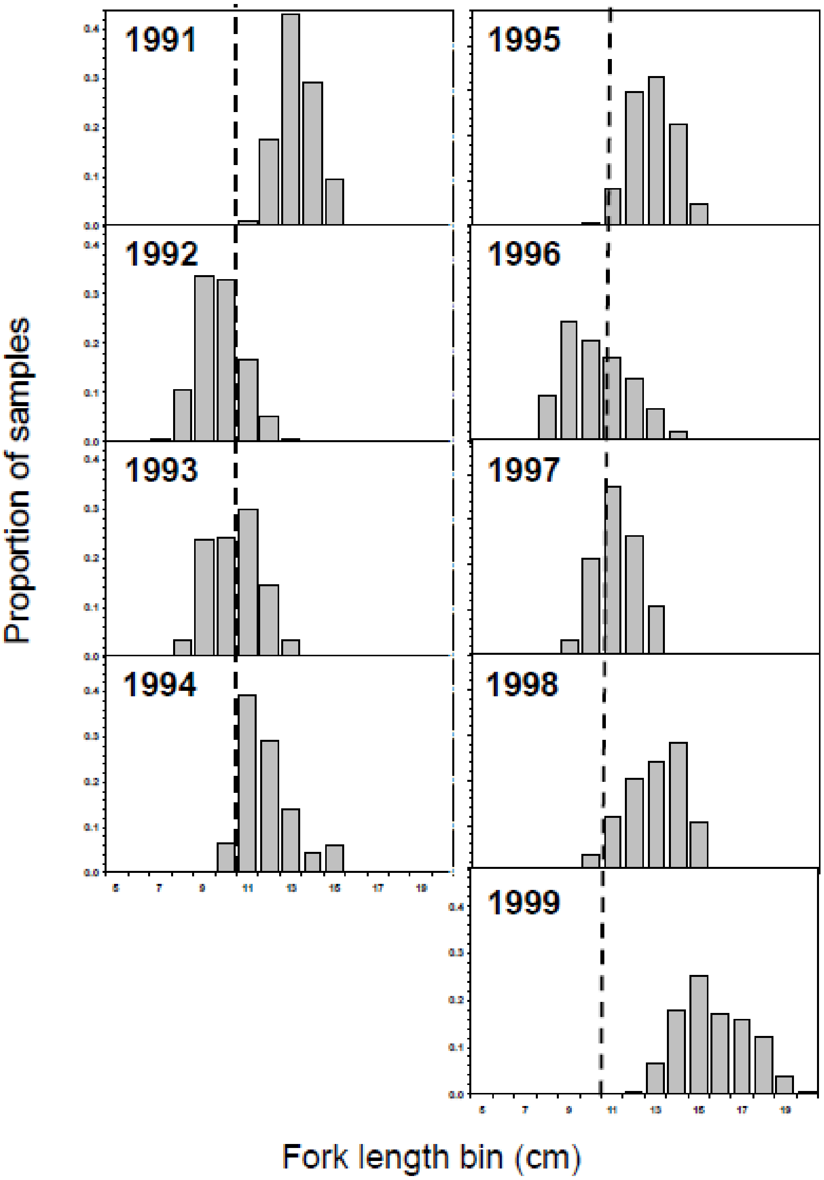


Figure 3.8. Length frequencies of young-of-the-year Striped Bass sampled in the fall open-water smelt fishery of Miramichi Bay (1991 to 1998) and the Tabusintac estuary (1999). The vertical hatched line at the interval between the 10 and 11 cm bins is included to illustrate the size variability among years. The figure is taken from Douglas et al. (2006).

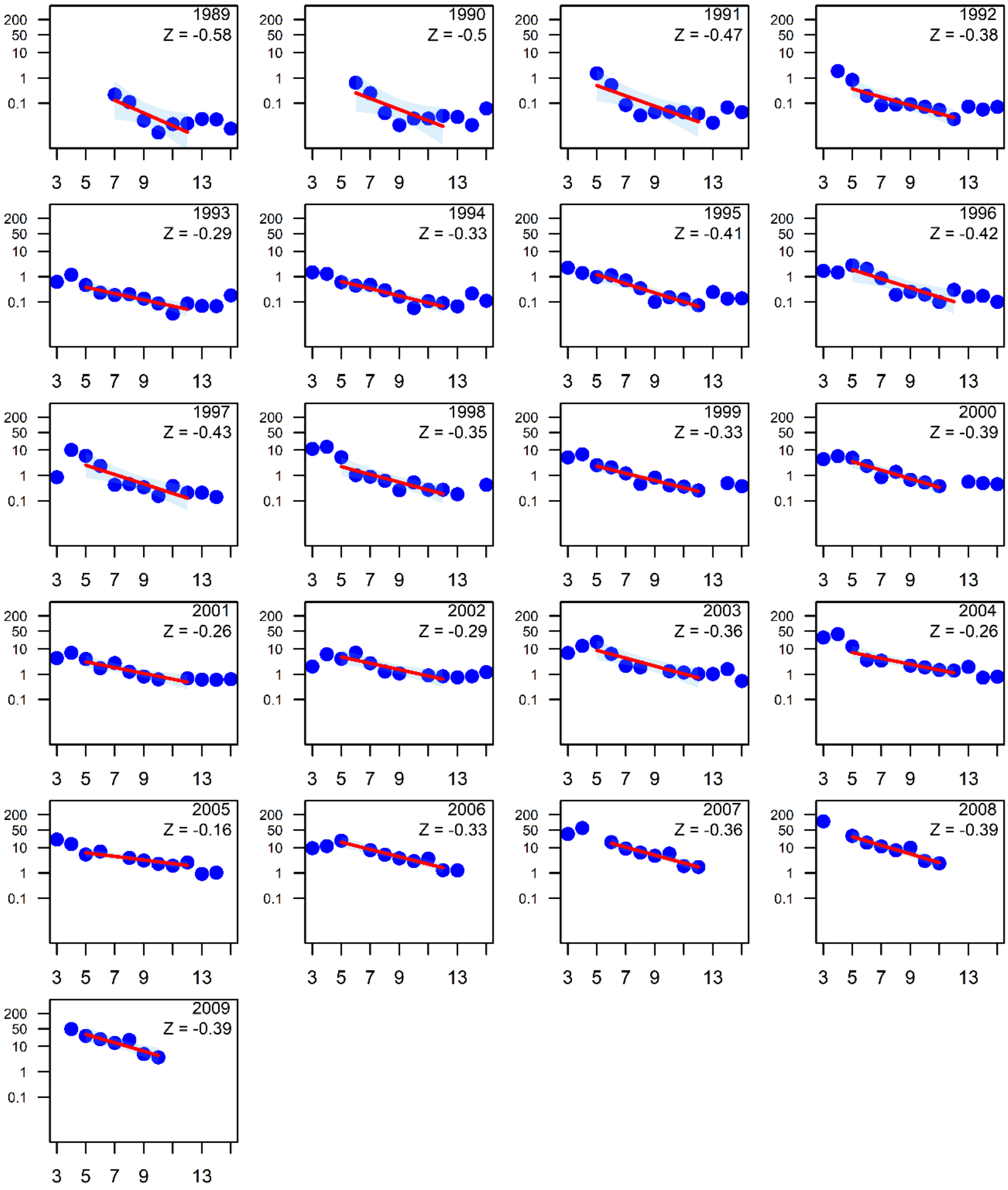


Figure 3.9. Cohort decline analysis based on changes in the natural log of assessed abundances at age by cohort for Striped Bass from the southern Gulf of St. Lawrence. The vertical axis in each plot is on the scale of the natural log of assessed abundance at age (axis labels are in thousands of fish) and the horizontal axis is the age. The red line is the predicted log of abundance over the range of ages 5 to 12 for cohorts with at least six observations within the age range 5 to 12.

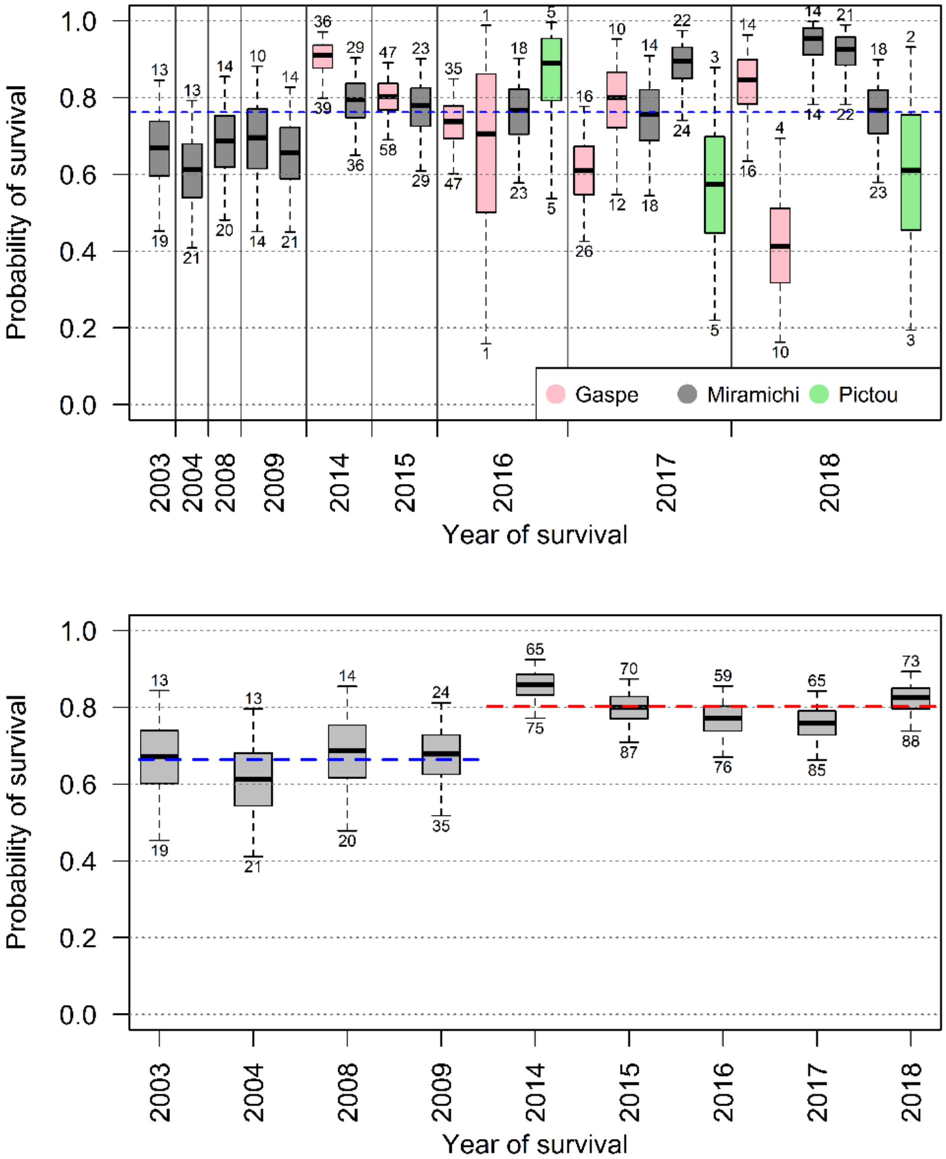


Figure 3.10. Posterior distributions of estimated probabilities of survival of acoustically tagged Striped Bass based on sequential detections in the Miramichi by inferred year of survival for Striped Bass tagged (V13 and V16 tag groups and size groups combined) and released in three locations. The upper panel shows the posterior distributions by year of inferred survival with the horizontal dashed line representing the median across all years and tagging locations. The bottom panel shows the posterior distributions of inferred survival by year, pooled over size groups, tag types and release locations. The horizontal dashed lines represent the median annual survival probabilities for the 2003 to 2009 period and the 2014 to 2018 time period. The inferred year of survival represents the calendar year (eg. 2017 is the survival over the period winter 2016/17 to winter 2017/18). Boxplots show the 2.5 to 97.5 percentile ranges as whiskers, the interquartile range as the rectangle, and the median as the internal dash. The numbers shown in each panel for each boxplot are the numbers of fish detected (above) and the number of tags available (below) used in the estimation of the survival rates.

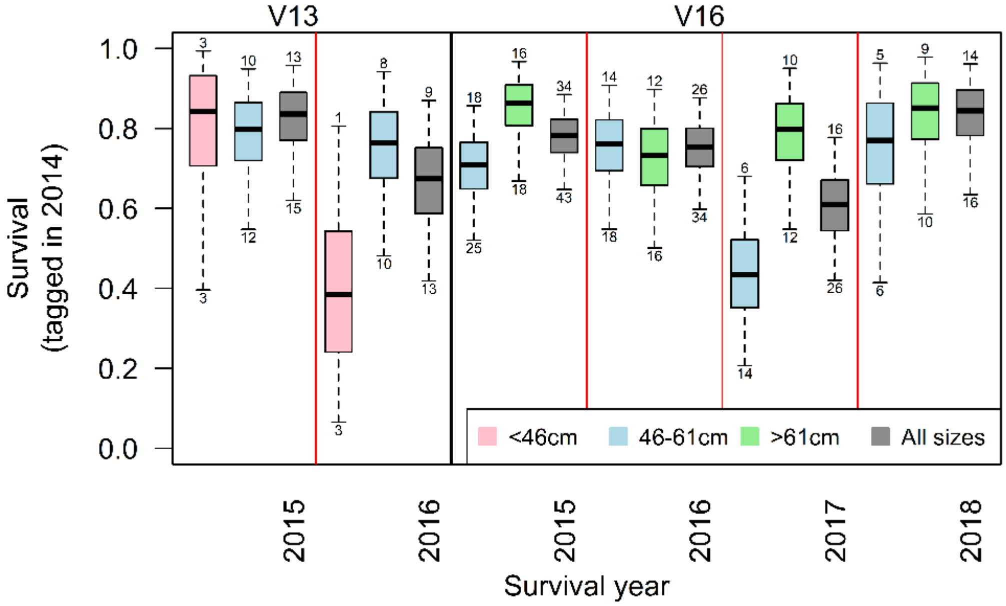


Figure 3.11. Posterior distributions of the sequential survival rate estimates of Striped Bass by size group tagged with V13 or V16 acoustic tags from the Gaspe release location in 2014. Size groups correspond to the fork length size group of the fish at tagging. Box plots are interpreted as in Figure 3.10. The numbers shown in each panel for each boxplot are the numbers of fish detected (above) and the number of tags available (below) used in the estimation of the survival rates.

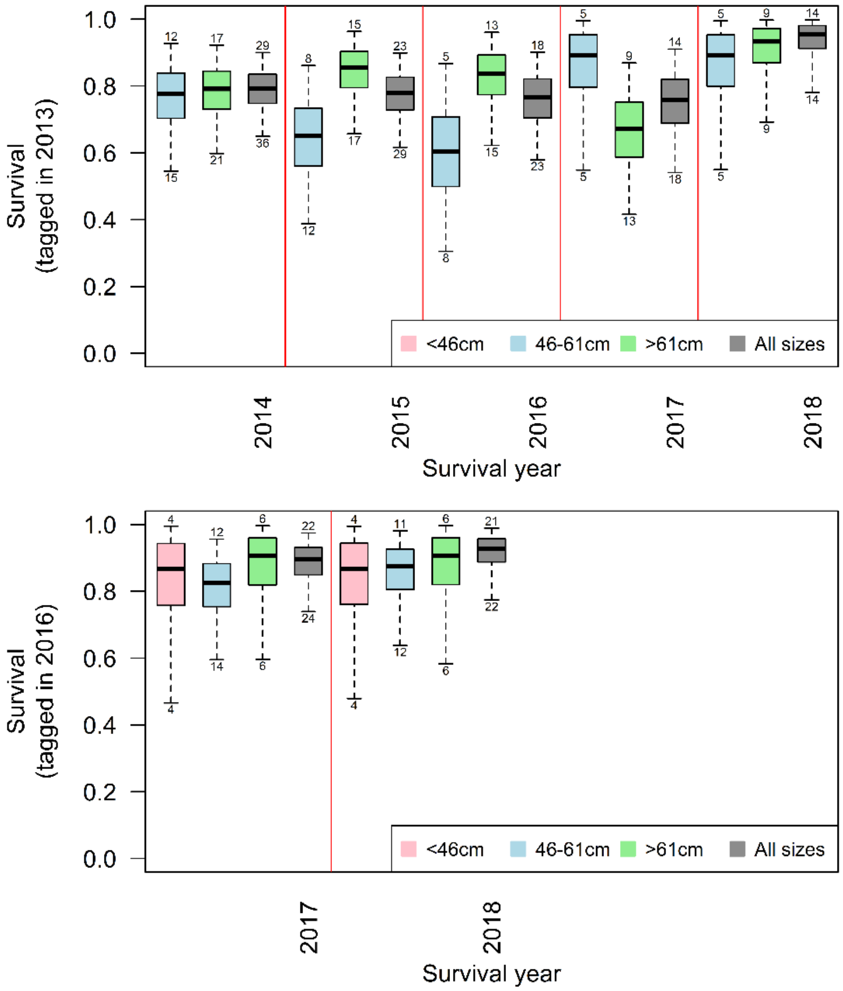


Figure 3.12. Posterior distributions of the sequential survival rate estimates of Striped Bass by size group tagged with V16 acoustic tags from the Miramichi release location in 2013 (upper panel) and 2016 (lower panel). Size groups correspond to the fork length size group of the fish at tagging. Box plot are interpreted as in Figure 3.10. The numbers shown in each panel for each boxplot are the numbers of fish detected (above) and the number of tags available (below) used in the estimation of the survival rates.

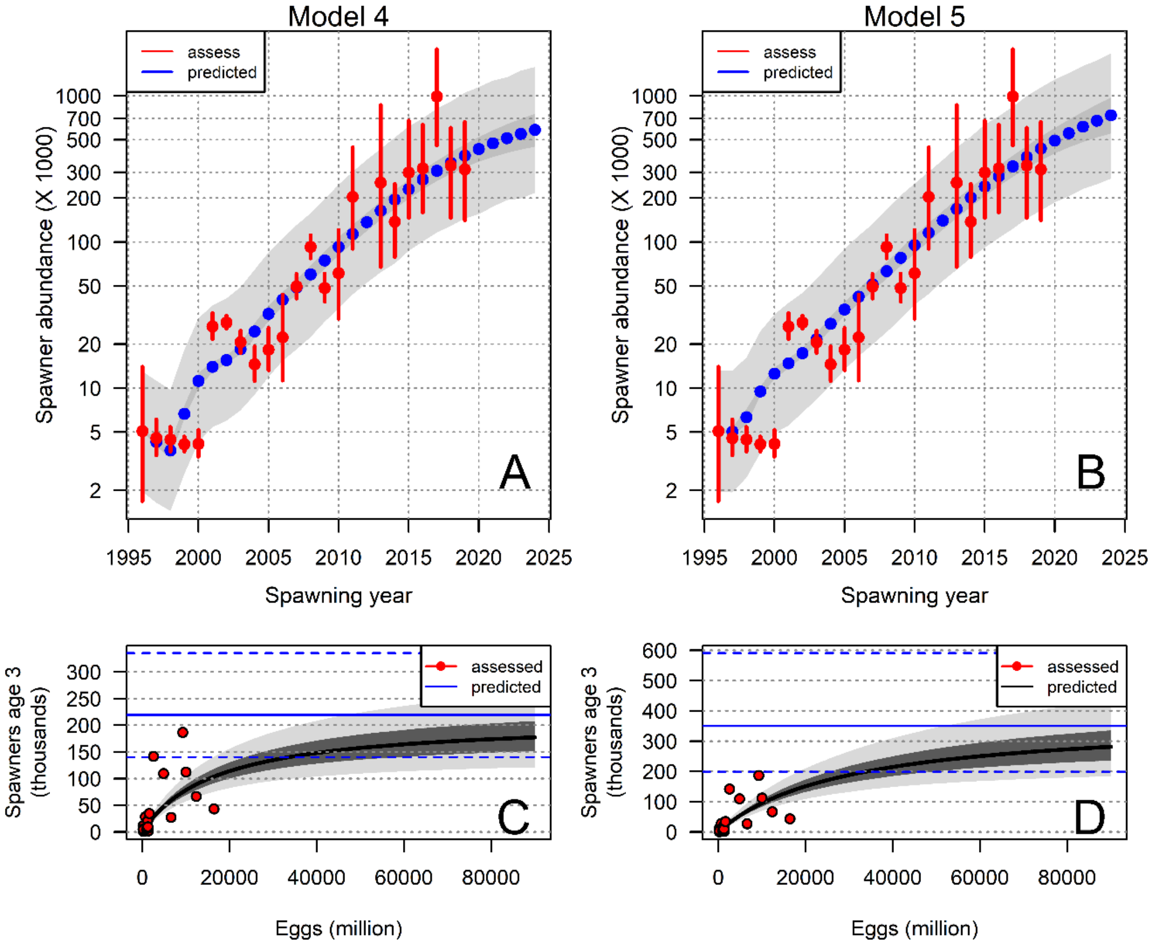


Figure 4.1. Observed and predicted total spawners of Striped Bass from the southern Gulf of St. Lawrence (upper row; A, B) and the stock and recruitment predicted abundance of spawners at age 3 years old (lower row; C, D) based on Model 4 (left panels A and C) and Model 5 (right panels B and D). In the upper row of panels, the assessed abundances are shown as red symbols for the median with 5th to 95th percentiles ranges as red vertical lines. The blue symbols are the predicted abundances, the darker grey shading is the 5th to 95th percentile range of mean predicted abundance and the light grey shading represents the 5th to 95th percentile range of the predicted spawner abundance accounting for the full process uncertainty. Note the y-axis abundance is shown on the log scale. In the lower panel, the assessed abundance of 3-year old spawners is shown as red symbols and the predicted median line with 25th to 75th and 5th to 95th percentile intervals are dark and light grey shading, respectively. The upper (blue) solid horizontal line (median) and the dashed horizontal lines (5th to 9th percentile range) are the Beverton-Holt asymptotic abundance (K) whereas the lower (red) solid horizontal line (median) and the dashed horizontal lines (5th to 9th percentile range) are half saturation values (50 % K) from the Beverton-Holt model.

Figure 4.

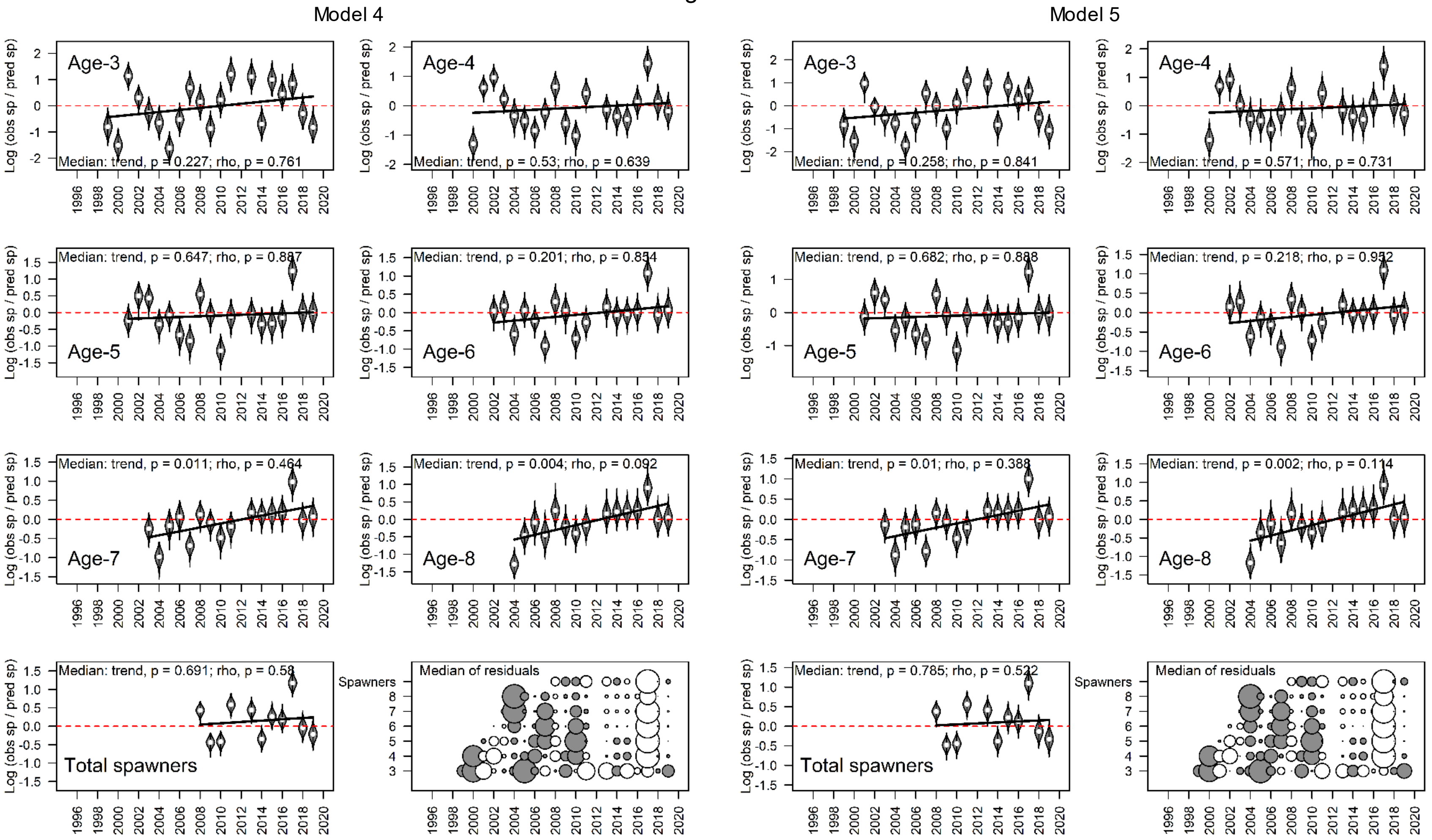
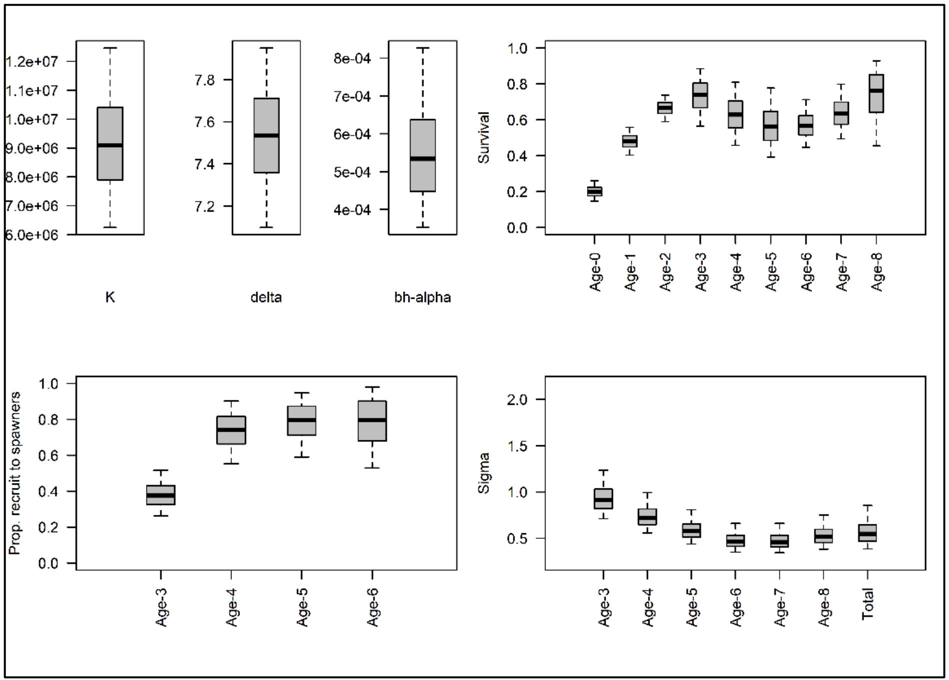


Figure 4.2. Residual plots expressed as log(assessed abundance / predicted abundance) at ages 3 to 8 and total spawners, and relative (by age group) bubble plot of logged residual patterns. Also shown in each panel of residuals are the p-value for the temporal linear trend in residuals and the p-value for the first order autocorrelation of the residuals (from package EnvStats in R).

A)
Model 4



B)
Model 5

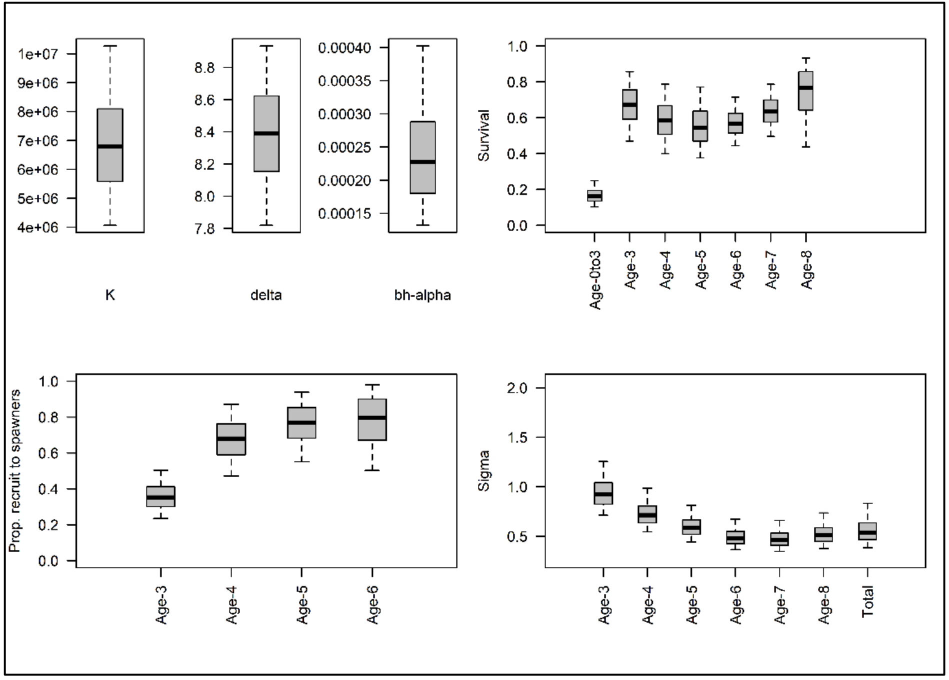


Figure 4.3. Posterior distributions of parameter estimates of the population dynamics for the Striped Bass population of the southern Gulf, from model 4 (A, upper panel) and model 5 (B, lower panel). The boxplot summaries show (from top left to bottom right): K (carrying capacity), delta (-log(bh.alpha)), bh.alpha (survival rate at the origin), survivals at age, proportion recruits that become spawners, and sigma (log of the standard deviation of the likelihood of observed spawner abundance at ages 3 to 8 and for total spawners). Boxplots summarize the following statistics of the posterior distributions: vertical dashed lines are the 5th to 95th percentile range, the box encompasses the interquartile range and the horizontal dash is the median.

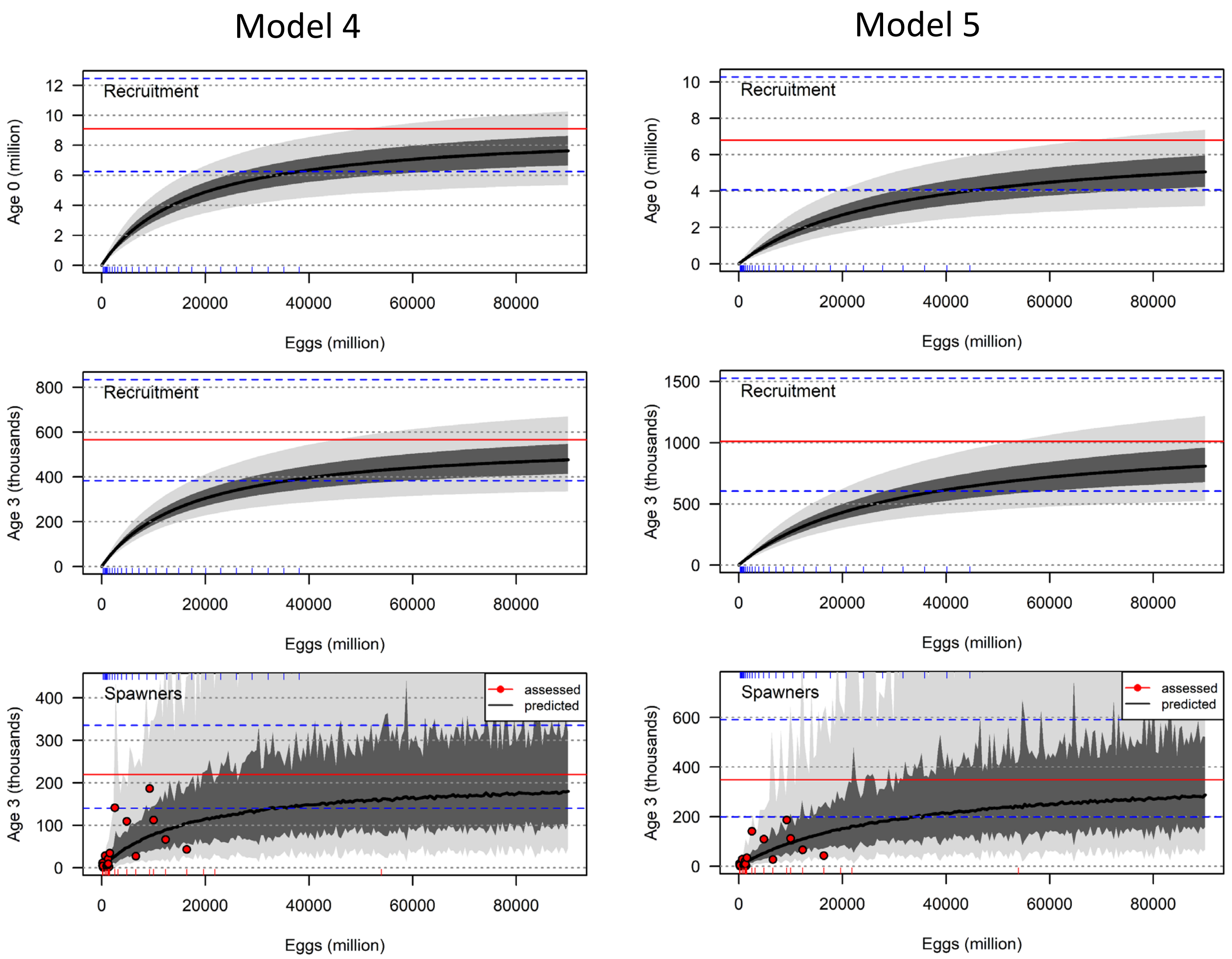


Figure 4.4. Predicted Beverton-Holt stock and recruitment function for abundance of Striped Bass at age-0, (upper panel), adjusted to abundance at age-3 (middle panel) and for predicted spawners at age-3 (bottom panel) based on Model 4 (left column) and Model 5 (right column). The light and dark shading in the upper and middle panels are the interquartile range and the 5th to 95th percentiles range, respectively, of the mean abundance. In the lower panel, the dark and light shading are the interquartile range and the 5th to 95th percentile range accounting for the uncertainty (log sigma-3) in the predicted abundance of spawners at age-3. The assessed abundance of spawners at age-3 corresponding to assessed egg production by cohort is overlain as red symbols in the bottom panel. In all three panels, the solid horizontal line is the median of the predicted carrying capacity scaled to recruitment and spawners at age-3 and the lower and upper dashed lines are the 5th to 95th percentile range.

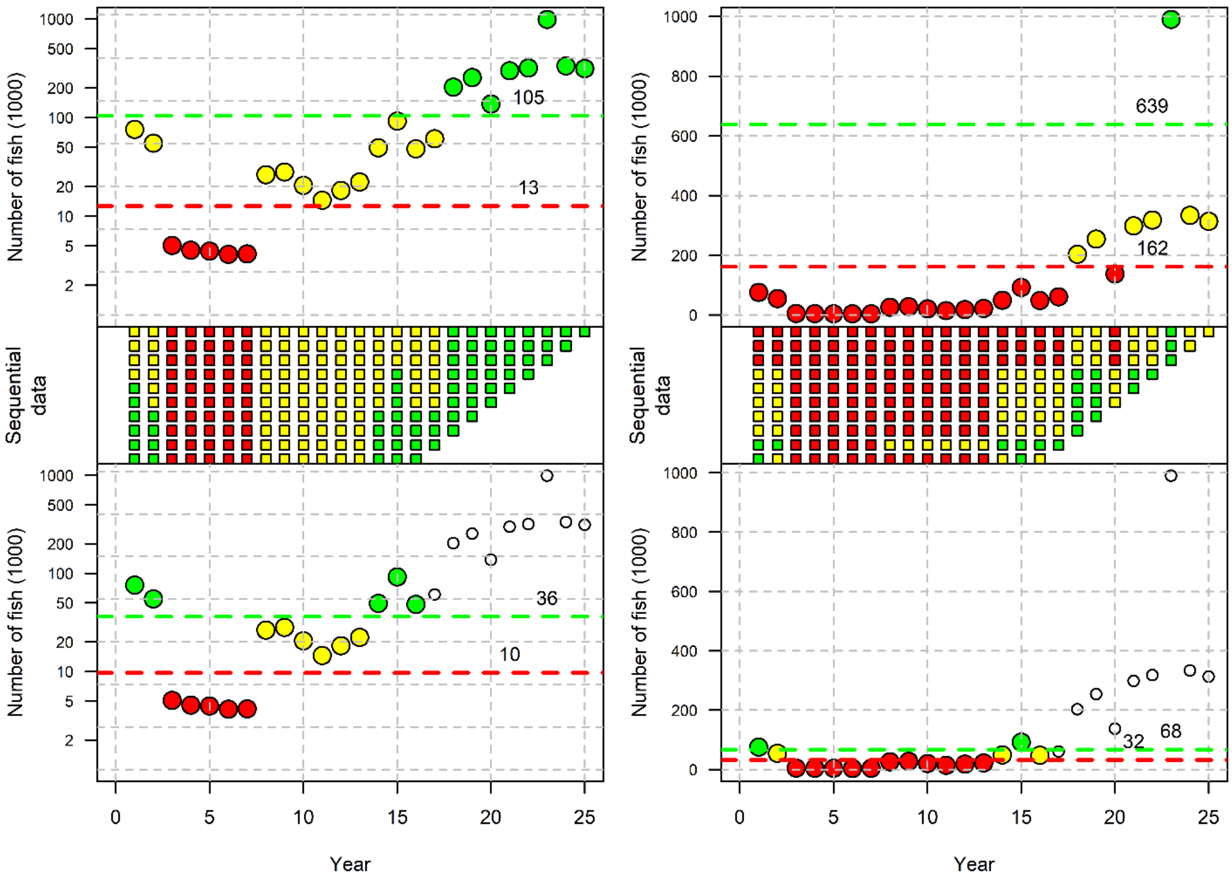


Figure 6.1. The effect of scale of the observations and the effect of the included time series used to assign annual abundance estimates of Striped Bass to one of three zones. The left column presents the results when the log(abundance) is used and the right column presents the results for the natural scale. The upper row of plots shows the status (red equivalent to critical, yellow equivalent to cautious, green equivalent to healthy) of the annual estimated abundances based on groups defined using the entire time series, 1994 to 2019 (excluding 2012). The bottom row shows the status of the annual estimated abundances for 1994 to 2009 based on groups defined using only the 1994 to 2009 time series (symbols in white are not assigned). In both the upper and bottom rows, the lower horizontal dashed line is the proxy LRP based on the average of the lower and middle group centroids and the upper horizontal dashed line is the proxy for the USR based on the average of the middle and upper group centroids; values for each are printed above each corresponding line. The middle row shows the assigned status colour for the corresponding year (on the vertical) relative to the time series included (by increments of one year for each horizontal line) beginning with the 1996 to 2009 time series in the bottom line of symbols to the 1996 to 2019 time series for the upper line of symbols. In this figure the year sequence represents: 1 = 1994, 18 = 2011, 19 = 2013, to 25 = 2019.

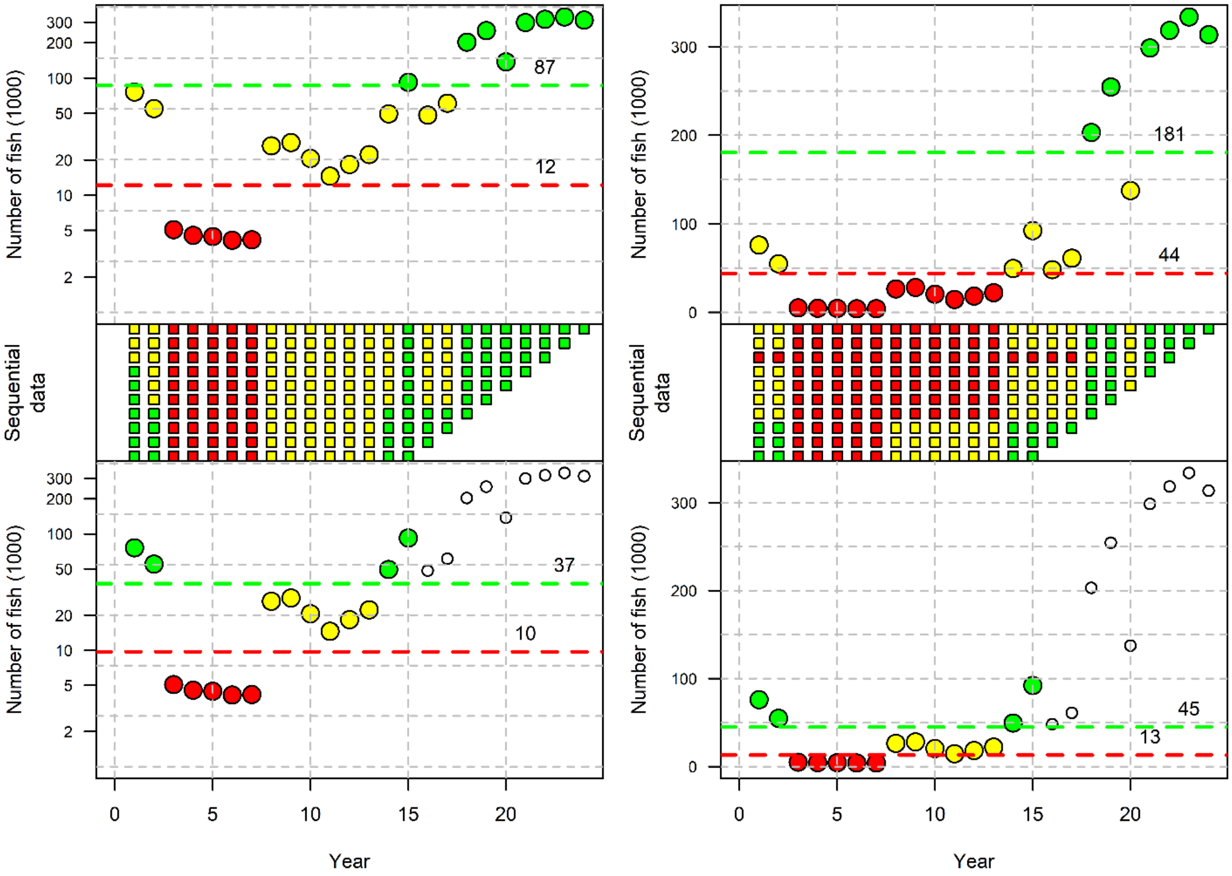


Figure 6.2. The effect of scale of the observations and the effect of the included time series used to assign annual abundance estimates of Striped Bass to one of three zone using the time series of observations that excludes 2017. The plots are interpreted as in Figure 6.1. In this figure the year sequence represents: 1 = 1994, 18 = 2011, 19 = 2013, 22 = 2016, 23 = 2018, 24 = 2019.

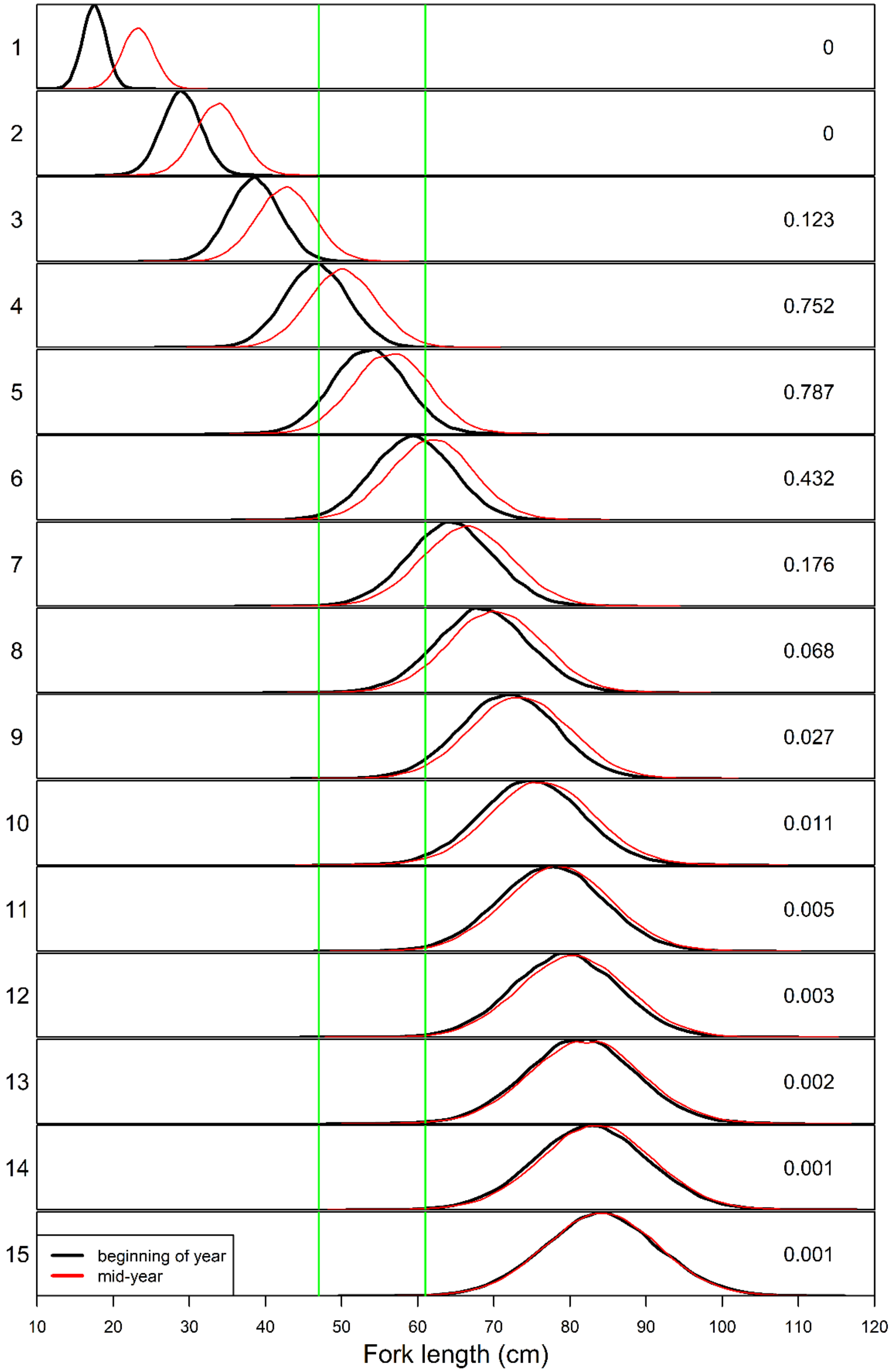


Figure 6.3. Predicted fork length at age distributions (black line start of year, red line mid-year) and the estimate of selectivity at age (s_a) to fully-recruited F (proportion of the age group that is within the fisheries size limits) is shown in the upper right of each age panel. The fisheries management strategy shown (vertical green lines) is a slot size for retention of 47 to 61 cm fork length,.

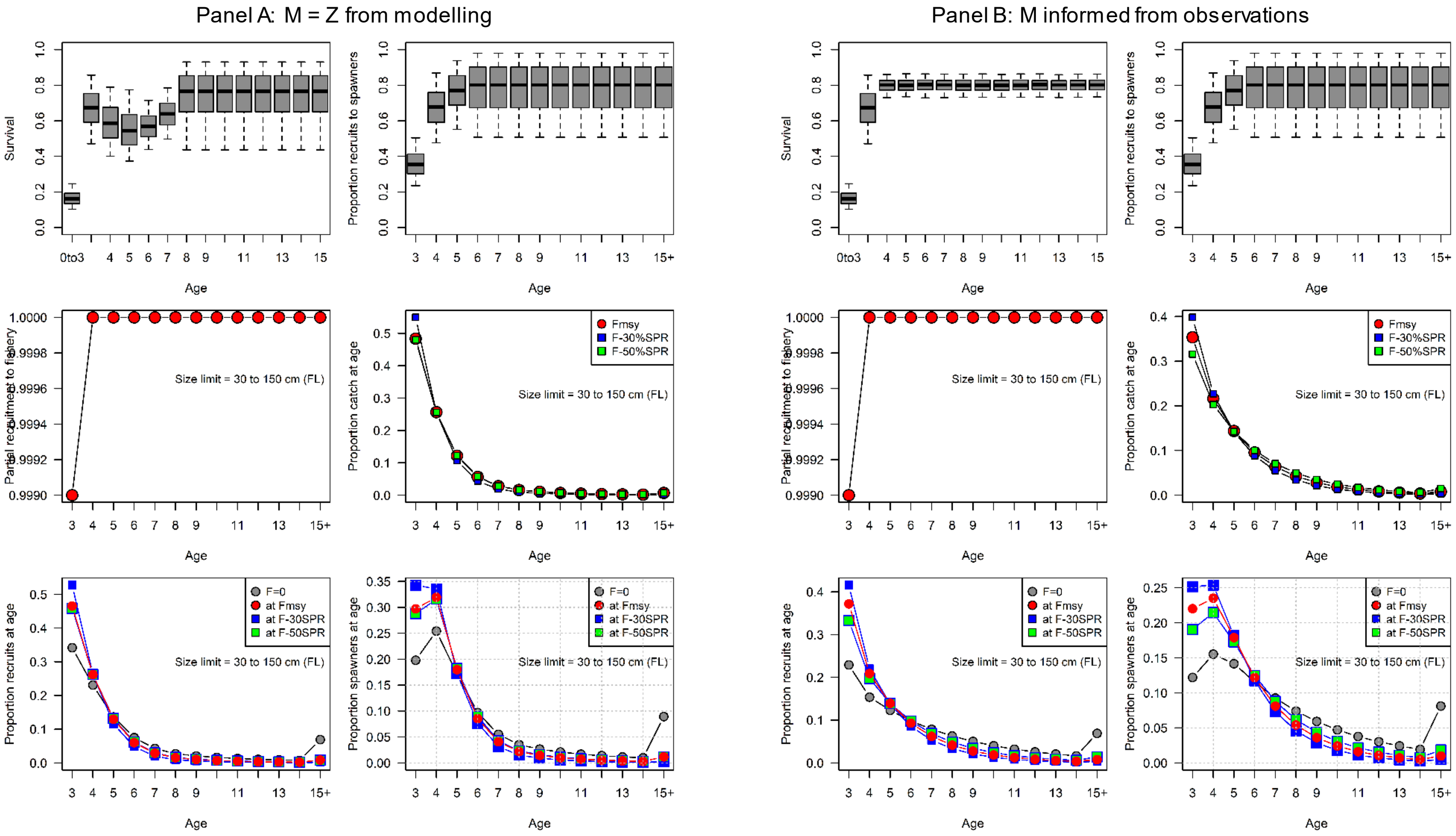


Figure 6.4a. Model 5 - distributions of life history parameters and age structure from equilibrium modelling of Striped Bass, based on parameter estimates and (panel A) modelled estimates of survival rates at ages 4 to 15+ or (panel B) with assumed values for M at ages 4 to 15+ informed from acoustic tag observations. The plots in each panel in reading order from top left to bottom right refer to: survival at age, proportion of recruits that are spawners at age, partial selectivity to the fishery, average proportion catch at age for different fishing rate reference values, average proportion of total abundance at age for different fishing rates, and average proportion spawners at age for fishing rate reference values.

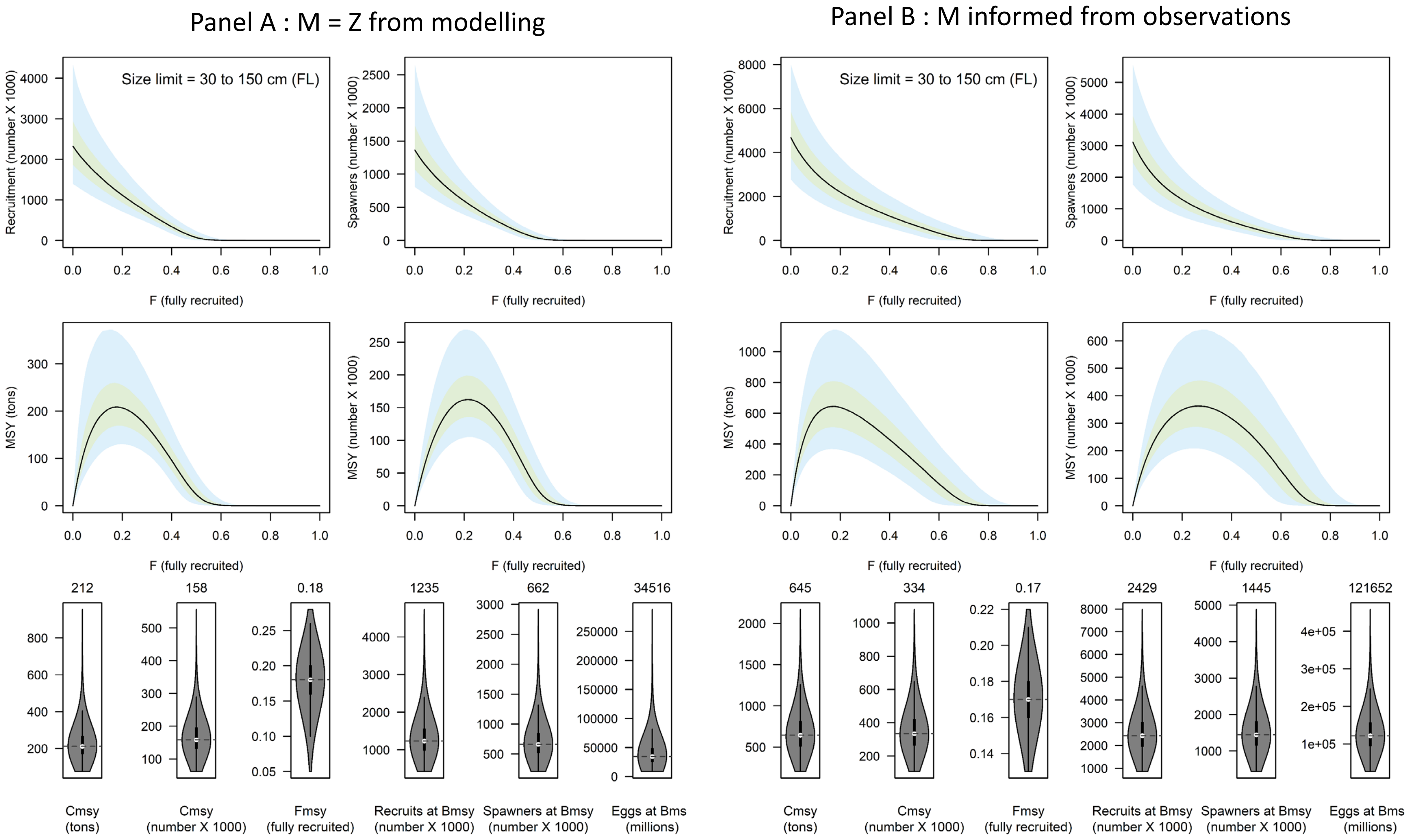


Figure 6.4b. Model 5 - equilibrium modelling abundance, and catch trajectories for increasing levels of F and summary boxplots of reference values associated with MSY, based on parameter estimates from modelling including M=Z (panel A) or with assumed values for M at ages 4 to 15+ (panel B). The plots in each panel refer, in reading order from top left to bottom right to: total abundance for ages 3 to 15+, total spawners aged 3 to 15+, yield in tons, yield in number of fish, and MSY reference values for C_{MSY} (tons), C_{MSY} (number), F_{MSY}, total abundance (number) at B_{MSY}, spawners (number) at B_{MSY}, and total eggs at B_{MSY}.

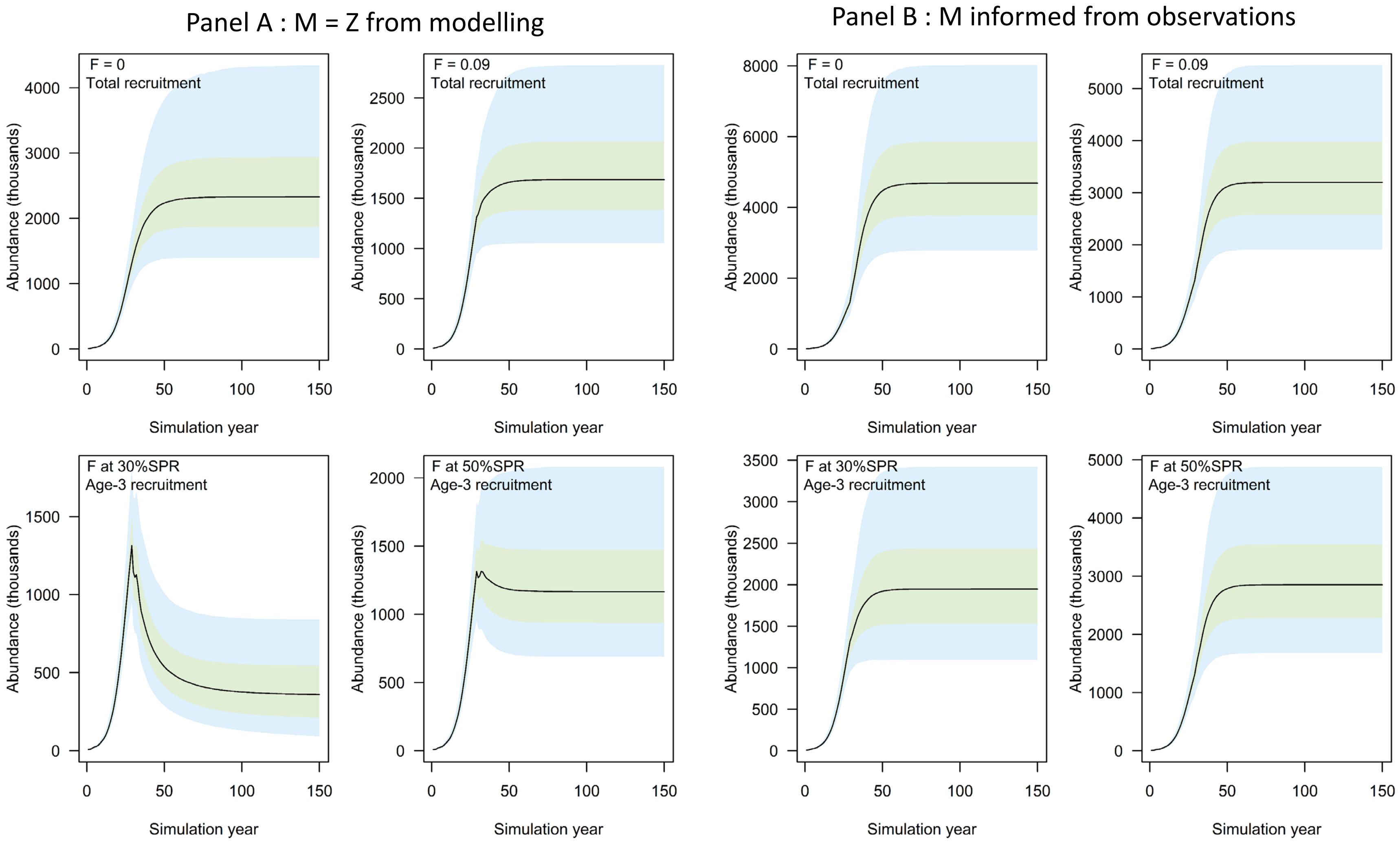


Figure 6.4c. Model 5 - equilibrium modelling of abundance by year of simulation to confirm attainment of equilibrium conditions based on life history parameter estimates from modelling including M=Z (panel A) or with assumed values for M at ages 4 to 15+ (panel B). Estimated abundances for years 1 to 29 are from population modelling, abundances for years 30 to 150 are projected forward.

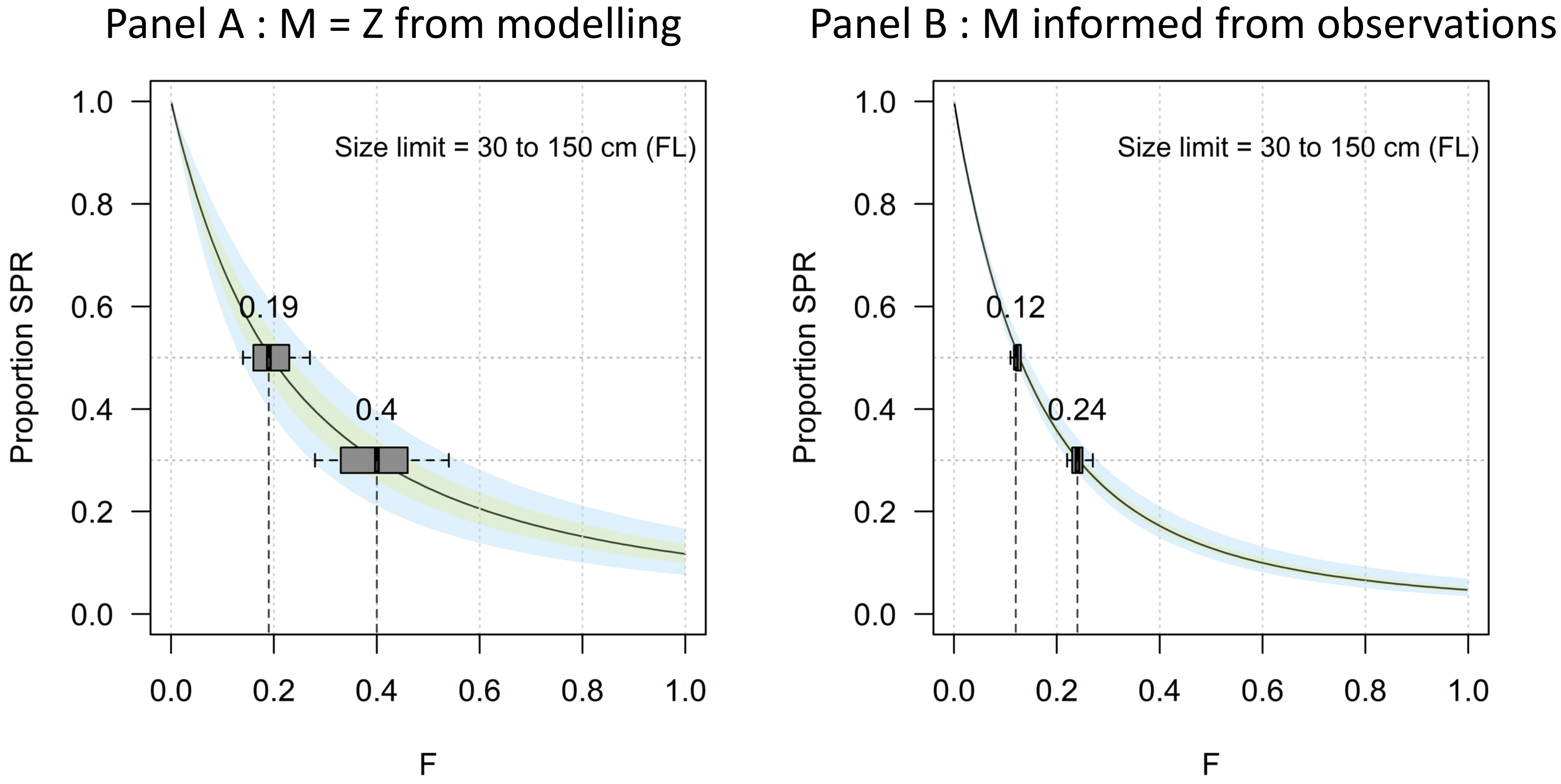


Figure 6.4d. Model 5 - equilibrium modelling of Spawner per Recruit trajectories based on life history parameter estimates from modelling including M=Z (panel A) or with assumed values for M at ages 4 to 15+ (panel B).

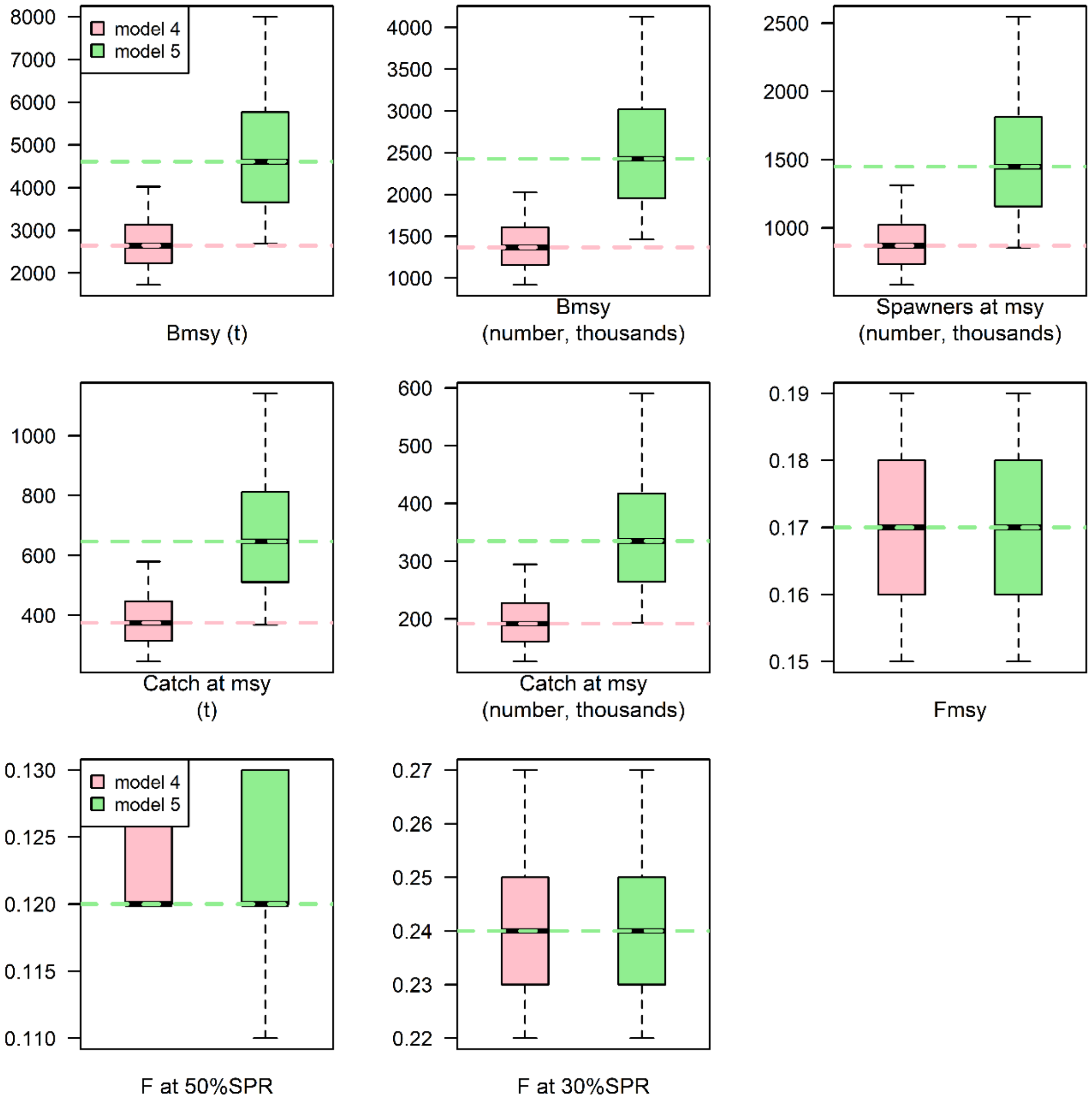


Figure 6.5. Comparison of MSY and SPR reference levels from Model 4 and Model 5 for scenarios with M informed by observations and for the default fishing strategy with no size limit and excluding catch and release mortality. The boxplot summaries are interpreted as follows: vertical dashed lines encompass the 5th to 95th percentile range, the boxes encompass the interquartile range, and the internal dash and dashed horizontal lines are the medians.

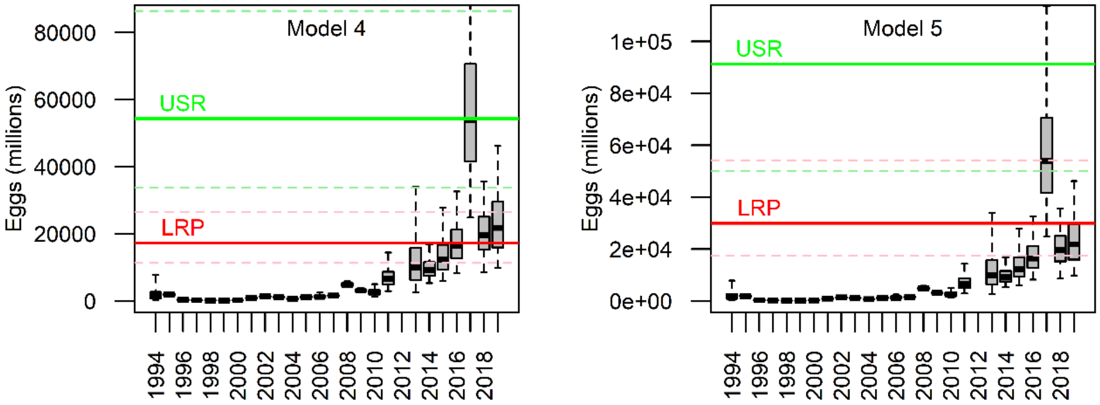


Figure 6.6. Assessed abundance of eggs in spawners (boxplots; eggs in millions) and status relative to the USR (upper green horizontal line) and the LRP (lower red horizontal line) candidate references from Model 4 (left panel) and Model 5 (right panel) for Striped Bass from the southern Gulf of St. Lawrence, 1994 to 2019. For Model 4 and Model 5, the USR corresponds to the median estimate of eggs at 80%B_{MSY} and the LRP corresponds to the median estimate of eggs that result in 50% of Beverton-Holt K (half saturation). The dashed red lines and green lines are the 5th to 95th percentile ranges of the LRP and USR respectively. Note the 95th percentile line of the USR and the 95th percentile point of eggs in 2017 are off scale in both panels.

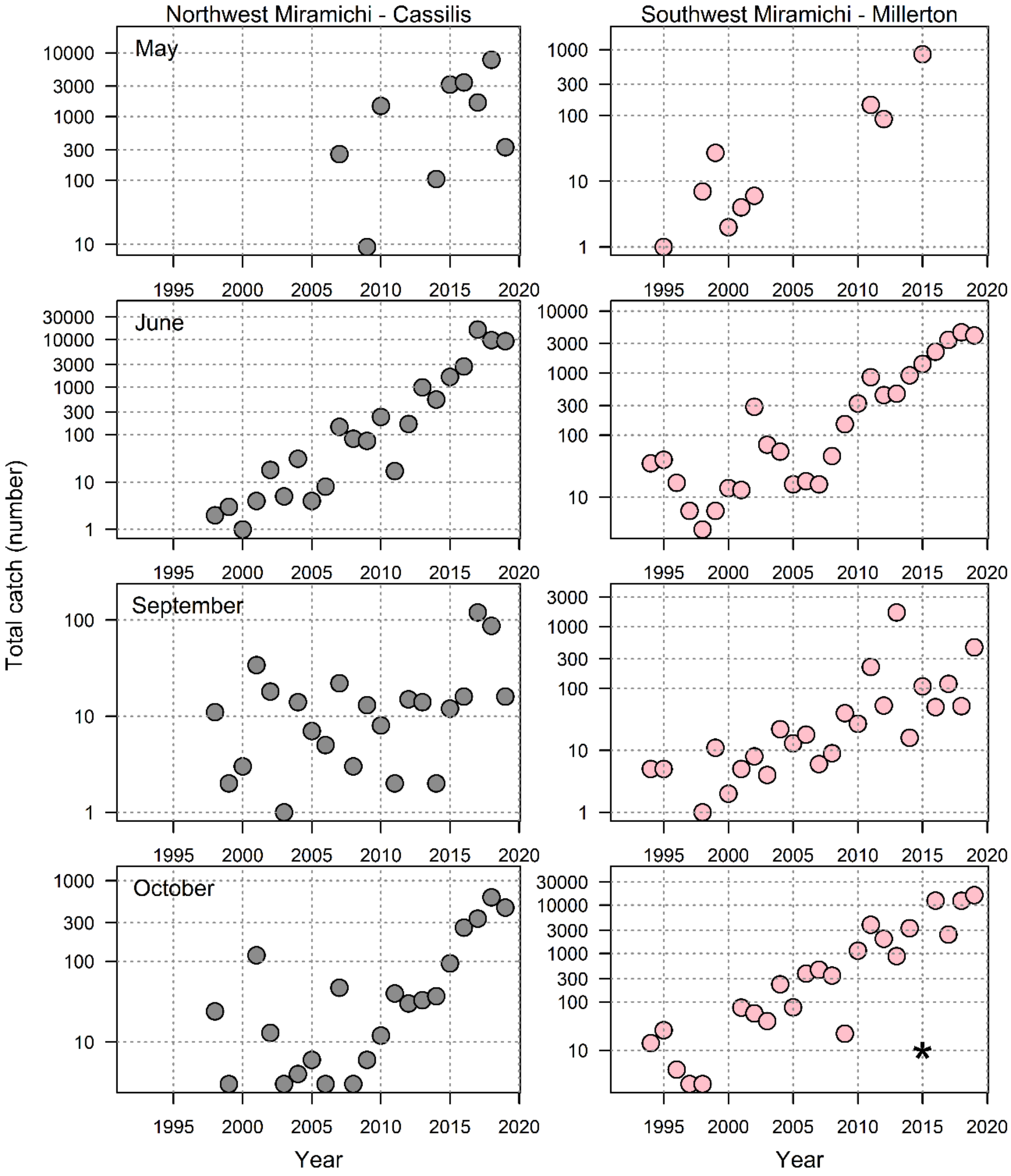


Figure 7.1. The monthly catches of Striped Bass at the DFO index trapnets of Cassilis on the Northwest Miramichi River (left column) and for Millerton on the Southwest Miramichi River (right column) during the months of May (top row), June (second row), September (third row) and October (bottom row), 1998 to 2019 for Cassilis and 1994 to 2019 for Millerton. In the lower right panel (Millerton, October), the asterisk indicates that the trapnet was not operational due to flood conditions which ended the monitoring program on Sept. 30 for the Southwest Miramichi trapnet; the Northwest Miramichi trapnet was not operating for five days during the first week of October in 2015.

APPENDICES

APPENDIX 1. REVIEW OF FISHERIES MANAGEMENT OF STRIPED BASS

Striped Bass have been exploited in numerous fisheries of the southern Gulf of St. Lawrence for over a century of records. The first mention of Striped Bass found in the Canada Gazette is from 1895 referring to fisheries regulations for Bass in New Brunswick; of note it mentions that no bass less than two pounds weight could be retained in any fishery.

In Canada Gazette 1927, the minimum size limit for retention is changed to 12 inches in length, measured from "the tip of the nose to the end of the tail".

There is an absence of reported landings from the southern Gulf of St. Lawrence during the period 1933 to 1968. This is not interpreted to be a period without fisheries. In Canada Gazette Part II (1949; SOR/49-223), an amendment is made to the Special Fishery Regulations for the province of New Brunswick (Council P.C. 5357 of 31st December 1947) that changed the retention conditions for Striped Bass, effectively closing the commercial fishery by authorizing the retention of Striped Bass from angling only and detailing precise restrictions to a number of rivers:

> "By deleting therefrom subsection one of section one and substituting the following new subsection one:
>
> 1. (a) No one shall fish for or kill any striped bass, otherwise than by angling, from the first day of April to the thirtieth day of November, both dates inclusive; nor otherwise than by angling at any time in the waters of the Miramichi, Kouchigouguac, Tabusintac, Tracadie, or Richibucto Rivers or any of the tributaries of these rivers.
>
>   (b) No one shall buy, sell, or have in his possession any striped bass unless, the proof whereof shall be on him, the same has been legally caught or killed."

The Nova Scotia fisheries regulations for Striped Bass combined them with other bass species (smallmouth bass, largemouth bass or occasionally referred to as black bass). The 1954 Canada Gazette Part II Amendment to the Nova Scotia Fishery Regulations (SOR/54-728) included the following:

- Definition of sport fish that includes salmon, trout and bass;
- No fishing for bass except by angling;
- No fishing for smallmouth and largemouth bass during Nov. 1 to 30 June;
- Total daily limit of bass (all species) defined as: "No person shall fish for, catch or kill, in any one day by angling or shall carry away a greater number of bass than, in the aggregate, shall weigh more than twenty pounds plus one such fish and no greater number than thirty, although the said number weigh less than twenty pounds."; and
- Size limit for striped bass as: "No person shall retain any striped bass less than twelve inches in length nor any black bass less than nine inches in length, measured from the tip of the nose to the end of the tail; any one who catches any such bass shall immediately return it to the water."

In the province of Quebec (14 July 1965, volume 99 number 13; Canada Gazette Part II):

- No person shall fish for, catch or kill any striped bass from the first day of December in any year to the thirty-first day of May next following, both days inclusive;

- Every person that catches a striped bass that is less than sixteen inches in length, measured from the tip of the snout to the posterior edge of the tail, shall return it to the water immediately.

In January 1960 (Canada Gazette Part II Volume 94 Number 1, 13 January 1960), the New Brunswick Fishery Regulations were amended and authorized the sale of incidentally captured Striped Bass captured in nets , traps, or weirs set for catching fish other than striped bass:

> "1. Paragraph(t) of section 2 of the New Brunswick Fishery Regulations is revoked and the following substituted therefor:
>
> (t) "sport fish" includes salmon, trout, and black bass
>
> 2. (1) Subsection (2) of section 3 of the said Regulations is revoked and the following substituted therefor:
>
> (2) Striped bass may be retained and marketed when unintentionally caught in nets, traps or weirs set for the catching of fish other than striped bass.

In 1965, a complete revision to the New Brunswick Fishery Regulations was published in Canada Gazette Part II (SOR/65-111, April 14, 1965) that included the following definitions and regulations related to striped bass:

- Sport fish includes salmon, trout and black bass
- No person shall fish for, catch or kill striped bass except with gill nets or by angling ("directed fishing")
- No person shall fish for striped bass with a net except under a licence
- No person shall fish for, catch or kill striped bass by means of a gill net, the mesh of which is less than five inches, extension measure, when in use
- Striped bass that are unintentionally caught in nets, traps or weirs set for other fish may be retained and marketed.
- No person shall make a hole in the ice for the purpose of fishing for striped bass unless he marks the hole with four evergreen trees, each tree being six feet in height (was in regs since1895).

Bow net specific, SOR/80-434: defined what a bow net, season for Kent County (NB) and size limit minimum of 38 cm.

In 1993, the Nova Scotia Fishery Regulations C.R.C. c848, the New Brunswick Fishery Regulations C.R.C. c844, and the Prince Edward Island Fishery Regulations C.R.C., c850 were revoked and replaced with the Maritime Provinces Fishery Regulations (SOR/95-55, 4 Feb. 1993) that specified regulations specific to fishing in the three Maritime provinces and in adjacent tidal waters. For Gulf Region waters:

- For striped bass, length referred to a straight line from the tip of the nose to the tip of the tail;
- Sport fish were defined as smallmouth bass, landlocked salmon, salmon and trout;
- Retention of striped bass incidentally caught with any fishing gear operated under the authority of a licence;
- No person shall use as bait or possess for use as bait in a province any: live or dead, bass, bullhead, sunfish, white perch, yellow perch, or other spiny fin-rayed fish;
- No person shall fish for striped bass except by angling or with a bow net;

- Bow net fishing is only allowed in tidal waters of Kent County (NB) during the period 1 December to 31 March, with a bow net with a mesh not less than 127 mm;
- Angling is open year round (excluding Dec. 30 and 31) in tidal waters but closed in inland waters during April 15 to Sept. 30 in NS, May 1 to Sept. 15 in NB, and April 15 to Sept. 30 in PEI;
- Daily quota and size restrictions of:
  - Nova Scotia: angling inland and tidal waters, 1 per day, 68 cm minimum length
  - New Brunswick: angling inland and tidal waters, 1 per day, 68 cm minimum length
  - New Brunswick: bow net fishing in tidal waters, no quota ,minimum length 38 cm
  - Prince Edward Island: angling inland and tidal waters, 10 fish per day, minimum length 30 cm.

A major amendment to Maritime Fisheries Regulations was introduced in 1996 to prohibit the retention and sale of bycatch of Striped Bass (Canada Gazette 1996 SOR/96-125):

- In 1996, Paragraph 4(2)b of the Maritime Provinces Fisheries Regulations which permitted the retention of unlimited bycatch of Striped Bass in commercial fishing gears for gaspereau, Rainbow Smelt, American Shad, and American Eel was repealed (Canada Gazette Part II, Vol. 130, No. 5; SOR/96-125). The regulatory impact analysis statement stated:

  > “Striped bass are currently being caught in large numbers as a by-catch in other fisheries, notably those for gaspereau, shad, smelt and eel. Although there is no fishery specifically directed at striped bass, the species is being taken in sufficient quantities through by-catches to threaten its survival. Because paragraph 4(2)b of the Maritime Provinces Fishery Regulations allows an unlimited by-catch of striped bass, it is necessary to remove this provision from the regulations. If striped bass stocks return to healthy numbers, by-catches in the commercial fishery can be regulated through licence conditions. This amendment applies to fishers in Nova Scotia, New Brunswick and Prince Edward Island.

- Alternatives Considered:

  > The Department asked fishers to release striped bass voluntarily, but few compiled with this request during the trial period. Fisheries manages found that a large number of striped bass were still being retained in other fisheries and sold commercially. The only acceptable solution is to prohibit by-catches of this species outright.

- Benefits and Costs:

  > The primary benefit of this amendment is the conservation and protection of striped bass. Recreational fishers will benefit in the short-term and commercial fishers could benefit in the long-term.
  >
  > Commercial fishers will lose the opportunity to catch and sell striped bass unless they are allowed to do so through licence conditions. However, since no directed fishery of this species currently occurs, the impact of this measure on them should be minor. The amendment is necessary to ensure the conservation of striped bass.”

Subsequent modifications to the Striped Bass fisheries management of the southern Gulf were made via licence conditions (for commercial fisheries) and variation orders for recreational fisheries. Additional restrictions to various fisheries interacting with Striped Bass were introduced from 1996 to 2000 which culminated in the closure of all legal Striped Bass fisheries (Table 2.1).

The commercial fisheries for Striped Bass were closed in March 1996 and commercial fishers were required to release all Striped Bass that are incidentally caught in commercial gear while fishing for other species. An exception to this in 1996 was made through condition of licence for gaspereau and smelt fisheries where a bycatch tolerance for fish <35 cm total length was in effect recognizing the difficulty of sorting bass less than 35 cm TL from large quantities of similar-sized fish, however these fish could not be sold. Bradford and Chaput (1998) provide a breakdown of the reported harvests of Striped Bass from 1996: harvests were reported from three statistical districts in DFO Gulf Region New Brunswick including:

- 14.5 t during January and February 1996 from district 76 (Richibucto district)
- 0.25 t in June 1996 from district 66 (Acadian peninsula, Miscou area)
- 0.25 t in October to December 1996 from district 77 (Bouctouche area).

Subsequent modifications to the Striped Bass fisheries management of the southern Gulf were made via licence conditions (for commercial fisheries) and variation orders for recreational fisheries. Additional restrictions to various fisheries interacting with Striped Bass were introduced from 1996 to 2000 which culminated in the closure of all legal Striped Bass fisheries (Table 2.1).

Although the fisheries on Striped Bass were essentially closed in 2000, mortality of Striped Bass from fishing activities continued (Chiasson et al. 2002; Douglas et al. 2006; DFO 2011). DFO (2011) indicated that Striped Bass of various life stages continued to be intercepted in a variety of illegal fisheries, commercial fisheries, and aboriginal FSC fisheries, with a total estimated loss of medium and large sized Striped Bass in all southern Gulf of St. Lawrence fisheries in the range of 60,000 fish per year. The total number of bass handled in the fisheries was estimated to be 152,000 fish, of which 41% were estimated to have died or been killed (DFO 2011). The activity with the greatest contribution to the total loss of Striped Bass is considered to be the illegal fishery, accounting for over 50% of the estimated adult losses, followed by the recreational fishery (illegal retention and bycatch) at about 15% (DFO 2011).

Following indications of sustained increases in abundance, re-initiation of Indigenous FSC allocations began in 2012 and the recreational fisheries were re-opened in 2013, followed by a pilot commercial fishery licence to an Indigenous community in 2018 and 2019 (Table 2.1).

Striped Bass originating from the southern Gulf are also exploited in fisheries along the coast of Chaleur Bay in Quebec. Fisheries management measures for the recreational Striped Bass fishery in Quebec that paralleled the fisheries management measures in DFO Gulf Region were introduced in 2013. Based on elemental composition analyses of otoliths and different characterizations of these signatures in Striped Bass originating from the Miramichi River and from the St. Lawrence River spawning areas, Valiquette et al. (2018) indicated that the southern Gulf of St. Lawrence Striped Bass distribution extended around Chaleur Bay and upstream along the Gaspe peninsula to Rivière du Loup. Occasionally, as noted in the samples of Striped Bass from 2017, southern Gulf bass were also distributed along the lower north shore of the St. Lawrence River (Valiquette et al. 2018). Tag returns of bass marked in the southern Gulf and reports of the presence of Striped Bass in southern Labrador in late summer and into the winter (DFO 2018) as well as detections of acoustically tagged Striped Bass on the receiver line at Port Hope (Labrador) confirmed the broader excursion of southern Gulf Striped Bass outside its historic range in 2017 and its exploitation in various fisheries in the Gulf of St. Lawrence.

Indigenous Peoples have allocations for Striped Bass within Food, Social, and Ceremonial fisheries agreements. In 1997, FSC agreements included 290 Striped Bass for three groups in the Miramichi River area, 500 bass from the Richibucto River and 172 bass from the Buctouche River (Bradford and Chaput 1998).

There are no complete fishery catch data for Striped Bass. Historically, fisheries statistics reported commercial harvests exclusive of recreational and Indigenous peoples fisheries harvests (LeBlanc and Chaput 1991; Bradford et al. 1995a; Douglas et al. 2003). Since the re-opening of the recreational fisheries in 2013, partial catch data for some sectors of the recreational fishery have been collated but they are incomplete.

In addition to the directed fishery management measures, short-term closures to directed recreational fisheries in the spawning area of the Northwest Miramichi to preclude harm to spawning fish were instituted since 2017. The temporary closure to all recreational fisheries of the spawning area in the Northwest Miramichi during the peak spawning period was previously identified as one of several management measures that would enhance the protection of Striped Bass and promote its recovery and justify the decision not to list the add the population to the schedule under the *Species at Risk Act* (List of Wildlife Species at Risk (Decisions Not to Add Certain Species) Order; Canada Gazette Part II Vol. 147 No. 7 (2013), SI/2013-27 March 27, 2013).

APPENDIX 2. DERIVATION OF AGE LENGTH KEY AND SPAWNERS AT AGE

A2.1. Interpretation of Ages

Ages of Striped Bass are interpreted from scales. Size-at-age has been reported previously by Chaput and Robichaud (1995) and in Douglas et al. (2006). Sampling and age determination has occurred opportunistically. There has not been any age validation nor is a reference scale set available for doing reader tests.

Striped Bass grow during the open-water season in the southern Gulf (May to October). No growth occurs through the winter when bass are overwintering under the ice in the upper areas of estuaries and they do not feed; this is evident from an examination of size distributions of bass sampled in the fall in the Miramichi at DFO index trapnets which are identical to those of bass sampled the following spring in the Miramichi (for example, see DFO 2020).

A total of 8,497 age and length data combinations are available from sampling in the southern Gulf of St. Lawrence; from the samples available, maximum age interpreted is 15 years and maximum fork length recorded is 97.0 cm (Table 3.2). There is a broad size distribution at age (Table A2.1; Figure A2.1). Samples were restricted to those collected in May and June (n = 8,376), corresponding to the spawning period.

Length distributions at age from sampling show annual variations, although there is no statistically significant trend over time (Figure A2.2).

A2.2. Von Bertalanffy Growth Model

A von Bertalanffy growth function was adjusted to the selected age and length data over all years:

$$L_a = L_\infty \left(1 - e^{-K(a - a_0)}\right) e^{\varepsilon}$$

with L_a = fork length (cm) at age a, L_∞ = predicted asymptotic fork length (cm), K = predicted metabolic parameter, a_0 = predicted apparent age at time of hatching, and $\varepsilon \sim N(0, \sigma^2)$.

The von Bertalanffy model parameters were estimated with OpenBugs using non-informative priors for the parameters $(L_\infty, K, a_0, \sigma)$ to be estimated (Section 3.1.1). The posterior distributions of the parameters are summarized in Table 3.3 and a visualization of the data, model fits and predicted length distributions at age are presented in Figure 3.3.

A2.3. Spawner Abundance at Age

Sampling for and age interpretations are not available for all assessment years, nor are there sufficient samples of older and larger fish in any year to adequately estimate their relative abundances. Consequently, the von Bertalanffy model predicted length at age distributions were used to derive an age length key which was then used to estimate the annual abundance at age of spawners based on the assessed annual length distributions of the spawners.

The posterior predicted fork length at age distributions show a large size overlap at age, particularly for ages 6 and older (Figure A2.3). Fork length distributions at age for the purpose of developing the age–length key were derived assuming a normal distribution defined by the posterior median and standard deviation of the predicted fork length at age (Table 3.2; Figure A2.3).

The age-length key proportions at age by cm fork length bins (Figure A2.4) were estimated from the length bin standardized proportions at age as:

$p.Age_{a,fl} = {DN_{a,fl}}/{\sum_a^A DN_{a,fl}}$

with $p.Age_{a,fl}$ = proportion of fish age *a* within the fork length bin *fl* (cm) and

$DN_{a,fl}$ = density at fork length *fl* for age *a* assuming a normal distribution of fork length at age (Table A2.2; Figure A2.5; $N(\mu_a, \sigma_a)\ in\ cm$).

$DN_{a,fl}$ is calculated as the difference in the left-tailed cumulative distributions between two fork length bins (fl-0.5 to fl+0.5) for each value of fl (cm, 10 to 100) over ages a = 1 to 15+:

$DN_{a,fl} = pNorm(fl + 0.5, \mu_a, \sigma_a) - pNorm(fl - 0.5, \mu_a, \sigma_a)\ \ (R\ code)$.

This age-length key (Figure A2.4) was applied to the sampled fork length distributions of spawners (Figure 3.4) to derive the number of sampled spawners at age by year, as:

$n_{y,a,fl} = p.Age_{a,fl} * n.fl_{y,fl}$

with $n_{y,a,fl}$ = number of fish in year y, of age a, in fork length bin fl from the sampled length distribution in year y,

$p.Age_{a,fl}$ as above, age-length key proportion of fish of age a in fork length bin fl, and

$n.fl_{y,fl}$ = number of fish in fork length bin *fl* from the sampled length distribution in year y.

The number of sampled spawners at age by year is:

$n.Sp_{y,a} = \sum_{fl} n_{y,a,fl}$ for fl = 10 to 100 cm.

and the proportion of sampled spawners at age, assuming spawners are age 3 and older is:

$p.Sp_{y,a} = {n.Sp_{y,a}}/{\sum_a^A n.Sp_{y,a}}$ for a = 3 to A = 15.

Finally the number of spawners at age by year y is calculated using:

$Sp.at.age_{y,a} = p.Sp_{y,a} * med.Sp_y$ for a = 3 to 15

with $med.Sp_y$ = median of the estimated spawner abundance in year y (from the assessment).

A2.4. OpenBugs Code for Von Bertalanffy Modelling

```
# bugs model for von Bertalanffy length age data for striped bass
# data are fork length in cm (FLcm[y]), age in years (Age[y]), total observations Y
model {
  # priors for von B parameters
Linf ~ dlnorm(0,0.001)
K ~ dlnorm(0, 0.001)
age.0 ~ dunif(-5, 0)
sig.eps ~ dunif(0,5)
prec.eps <- pow(sig.eps, -2)

for (y in 1:Y){
    Flcm[y] ~ dlnorm(u.logfl[y], prec.eps)
    u.logfl[y] <- log(Linf * (1 - exp(-K * (Age[y] - age.0))))
    }   # end likelihood loop
# predicting length distributions at ages 1 to 15
for (a in 1:15){
    pred.FL[a] ~ dlnorm(u.pred.fl[a], prec.eps)
    u.pred.fl[a] <-log(Linf * (1 - exp(-K * (a - age.0))))
  } # end predicted length at age loop
} # end model
```

Table A2.1. Summary statistics of fork length (mm) at scale-interpreted ages of Striped Bass from the Miramichi River.

Age (years)	mean	Std. dev.	n	min	max	median	p0.025	p0.975	CV
1	178	15	71	146	210	179	152	200	0.083
2	282	31	562	140	414	278	232	345	0.109
3	406	35	2606	244	512	410	338	463	0.085
4	473	38	2542	290	592	475	399	542	0.080
5	528	41	1485	348	658	527	455	606	0.077
6	580	40	769	445	726	580	495	659	0.069
7	636	55	124	480	740	645	524	724	0.086
8	691	53	94	515	780	702	588	764	0.076
9	727	55	62	572	848	738	609	822	0.076
10	771	63	20	644	858	781	644	851	0.082
11	782	63	21	640	861	801	671	861	0.081
12	835	52	10	705	897	841	730	894	0.062
13	755	76	2	701	809	755	704	806	0.101
14	782	72	5	665	847	784	676	846	0.093
15	869	164	3	680	970	958	694	969	0.189

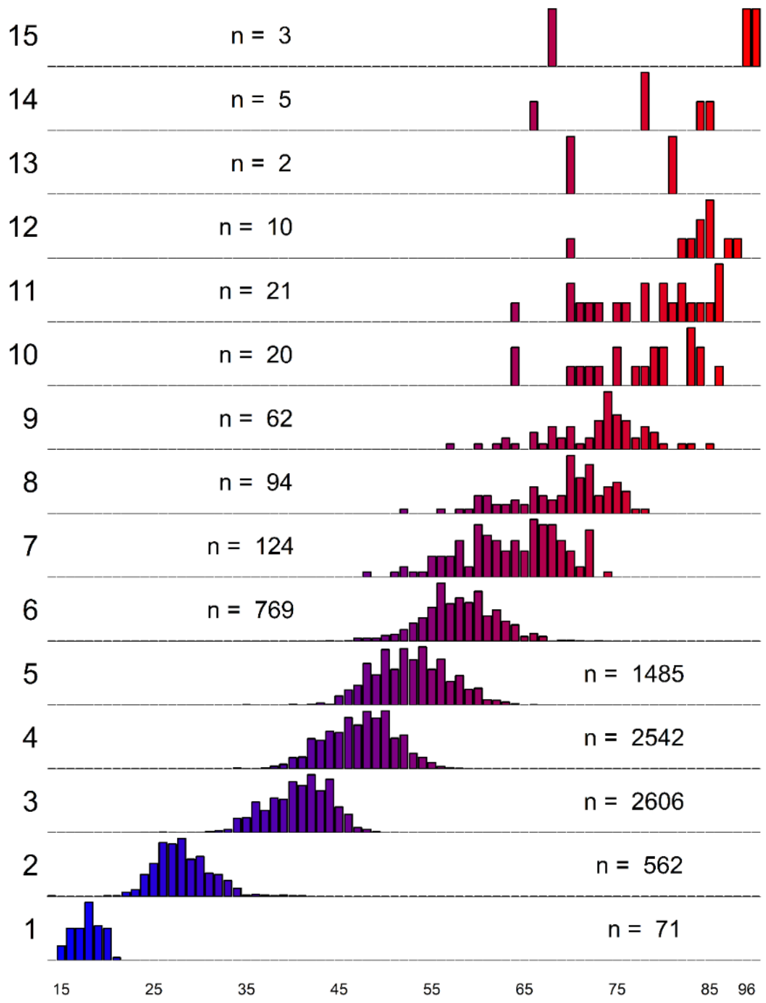


Figure A2.1. Length (cm) frequency distributions at age (year; rows) of Striped Bass from the southern Gulf of St. Lawrence, based on available samples with scale age interpretations.

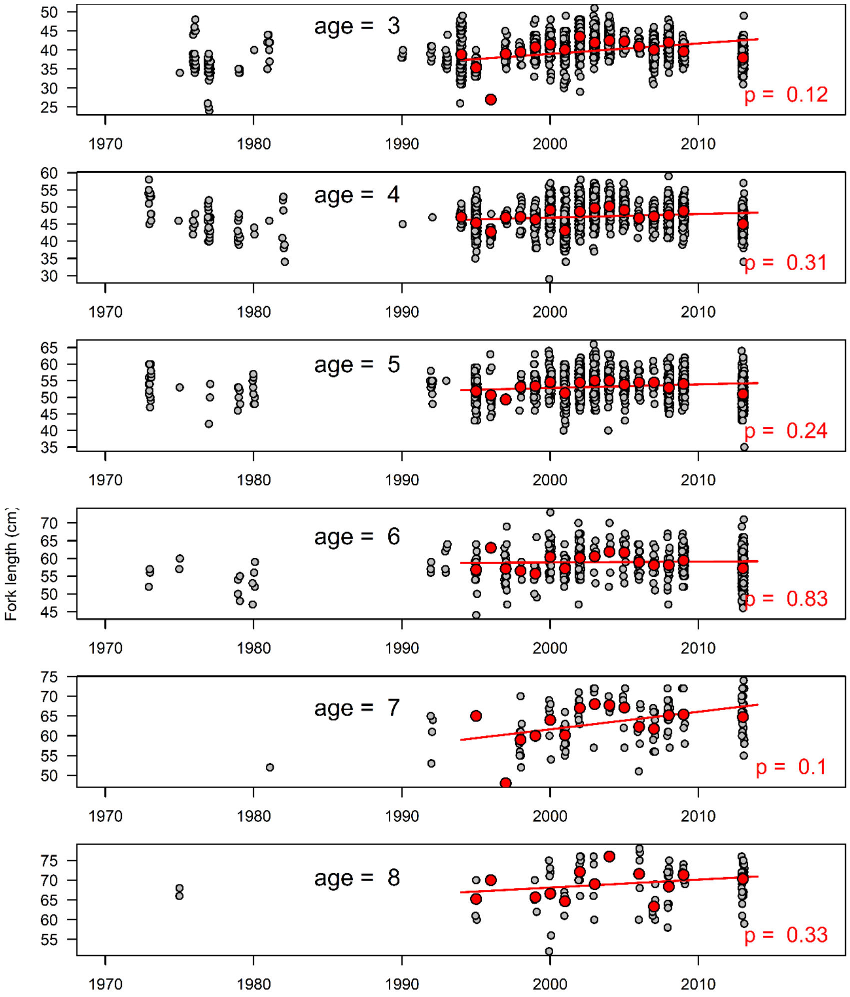


Figure A2.2. Fork length (cm) at interpreted age of Striped Bass, by year of sampling from the southern Gulf of St. Lawrence, based on scale interpretations. Shown are individual (jittered slightly for clarity) lengths at ages 3 to 8 (successive rows) by year of sampling. The linear regression of median annual size at age (red symbols) versus year is shown in each age plot with the corresponding p-value of the slope of the regression of median values over years.

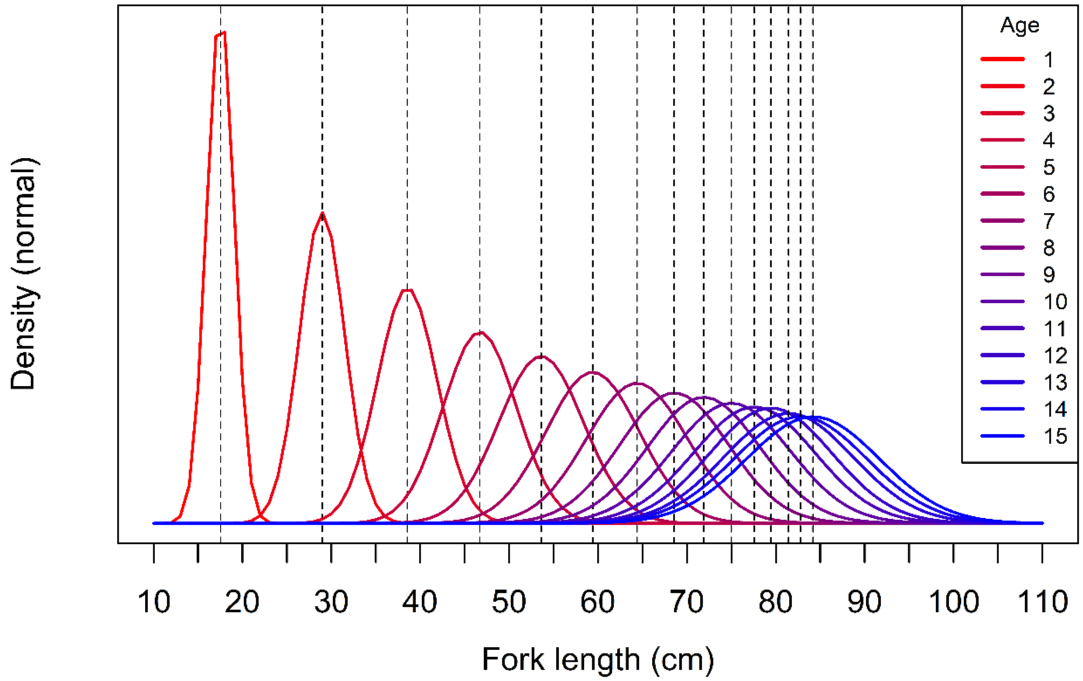


Figure A2.3. Fork length distributions (cm) at age (1 to 15) assuming a normal distribution with parameters mean and standard deviation of the posterior predicted distributions of the von Bertalanffy growth model (Table 3.2).

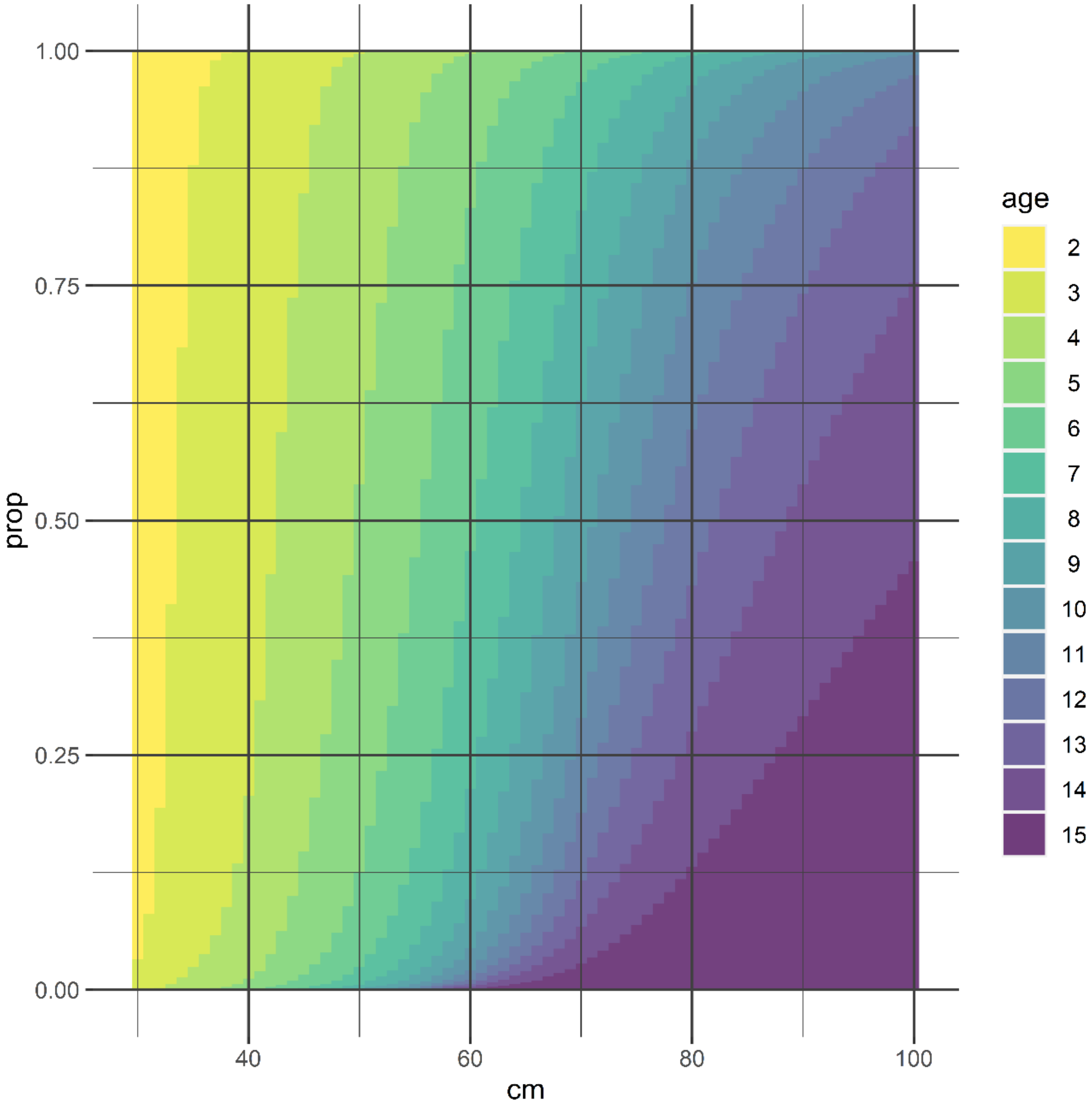


Figure A2.4. The proportions at age by cm fork length bin (30 to 100) ($p.Age_{a,fl}$) *used to estimate the abundance at age of spawners on the spawning grounds in the Miramichi. For illustration, the fork length bin range shown is 30 to 100 cm and the age range shown is 2 to 15+ years.*

APPENDIX 3. INPUT DATA USED IN THE POPULATION MODELLING

Table A3.1. Posterior summaries of the estimated abundance (number of fish) of Striped Bass spawners in the Miramichi River, 1994 to 2019. There are no estimates for 2012.

Year	mean	sd	2.5pc	5.0pc	25.0pc	median	75.0pc	95.0pc	val97.5pc
1994	130600	217300	12560	18440	44450	76020	138400	402000	594500
1995	56220	10560	39070	41230	48740	55020	62320	75420	80160
1996	6104	4390	1219	1687	3449	5072	7408	13980	17600
1997	4634	818	3294	3465	4057	4545	5112	6108	6489
1998	4484	534	3546	3677	4107	4447	4820	5421	5636
1999	4141	300	3591	3672	3933	4128	4335	4655	4766
2000	4208	539	3280	3403	3827	4163	4540	5164	5385
2001	26670	3378	20900	21660	24280	26400	28750	32640	34030
2002	28210	1738	24990	25470	27010	28150	29340	31180	31800
2003	20760	2219	16850	17400	19200	20600	22150	24630	25540
2004	14820	2510	10640	11170	13040	14570	16330	19330	20460
2005	18770	3912	12520	13270	16010	18310	21000	25820	27720
2006	24210	10870	9161	11300	17670	22260	28190	43620	51580
2007	49980	5933	39700	41080	45800	49530	53670	60440	62860
2008	93000	10380	74690	77180	85660	92320	99610	111200	115300
2009	48930	6677	37610	39120	44230	48320	52970	60730	63670
2010	66450	30560	23860	29840	48000	61090	77830	120800	143500
2011	227100	121700	69860	89790	153800	203200	268700	444400	537200
2012	na	na	na	na	na	na	na	na	na
2013	333900	304500	46420	67460	158800	254500	403400	864600	1131000
2014	147400	55990	66260	79090	111600	137600	172100	249500	288400
2015	339800	179200	118600	145800	226400	298600	405300	675300	807200
2016	346900	155700	132800	159600	245700	318400	411700	633000	750700
2017	1093000	553400	275800	456500	763400	990200	1296000	2083000	2507000
2018	348900	142300	55170	145800	258500	333800	425800	605100	674500
2019	346100	168800	116500	140900	229100	313600	426000	663200	773600

Table A3.2. Estimated abundance (number of fish) at age (3 to 15+ years) of Striped Bass spawners in the Miramichi River, 1994 to 2019. Abundance at age is derived using an age-length key applied to length distribution of spawners and raised to total abundance based on the median of the assessed total abundance of spawners (Table A3.1). There are no estimates for 2012.

	Age (years)												
Year	3	4	5	6	7	8	9	10	11	12	13	14	15+
1994	54669	17302	3135	490	124	67	48	41	36	32	28	25	22
1995	25182	18710	7387	2398	738	267	114	62	43	35	31	27	25
1996	627	1888	1526	649	221	81	33	16	9	7	6	6	5
1997	1512	1194	832	522	250	111	52	26	15	11	8	7	5
1998	2279	1317	474	199	86	41	21	12	7	5	3	2	2
1999	1711	1421	613	235	83	33	14	7	4	3	2	2	2
2000	858	1480	989	448	190	88	45	25	15	10	7	5	4
2001	11311	10209	2837	1149	483	205	93	46	25	16	11	8	6
2002	5211	13539	5979	2063	718	292	137	73	45	32	24	20	16
2003	4409	6906	5244	2397	880	350	162	88	55	39	29	23	18
2004	4321	5769	2558	1039	425	196	102	58	35	24	17	14	10
2005	2026	6976	5019	2121	901	454	255	156	108	87	75	69	62
2006	6923	6180	3999	2395	1212	617	340	200	129	92	70	57	45
2007	28335	13253	4097	1732	851	459	263	155	100	75	68	69	72
2008	20928	38426	19008	7087	2780	1388	822	539	387	305	249	217	184
2009	9504	13918	12576	6364	2725	1257	674	408	274	208	163	137	112
2010	34618	11537	5411	3615	2127	1272	813	532	366	274	210	175	140
2011	109574	59488	18928	7102	3451	1880	1078	627	386	259	181	143	102
2012	na	na	na	na	na	na	na	na	na	na	na	na	na
2013	141492	48472	29316	16641	8006	3954	2189	1333	905	697	568	498	428
2014	27270	45493	26051	15897	9164	5248	3116	1867	1184	828	609	494	378
2015	171327	48442	31348	19425	11187	6422	3834	2312	1470	1029	747	602	453
2016	112323	102255	43883	24683	13819	7899	4774	2952	1932	1389	1023	833	636
2017	186751	433056	205540	83570	36494	18002	9879	5815	3710	2637	1946	1582	1218
2018	66676	129860	68724	31012	15572	8592	5026	2987	1871	1285	919	733	543
2019	43325	106070	76332	40063	19945	10657	6182	3742	2422	1736	1283	1042	801

Table A3.3. Predicted mean fork length (cm) at age, predicted mean weight (kg) at age, assumed proportion mature at age, and derived proportion female at age of spawners of Striped Bass from the Miramichi River.

Age (years)	Fork length (predicted mean; cm)	Whole weight (predicted mean; kg) [1]	Proportion mature		Prop. female spawners
			male	female	
3	38.5	0.677	0.5	0.1	0.167
4	46.7	1.204	0.9	0.5	0.357
5	53.6	1.818	1.0	0.9	0.474
6	59.4	2.474	1.0	1.0	0.50
7	64.4	3.166	1.0	1.0	0.50
8	68.6	3.814	1.0	1.0	0.50
9	71.9	4.400	1.0	1.0	0.50
10	75.0	4.999	1.0	1.0	0.50
11	77.6	5.544	1.0	1.0	0.50
12	79.4	5.946	1.0	1.0	0.50
13	81.4	6.399	1.0	1.0	0.50
14	82.8	6.746	1.0	1.0	0.50
15+	84.2	7.058	1.0	1.0	0.50

[1] Predicted mean whole weight is based on a weight to length relationship parametrized as
$\ln(WW_{kg}) = -11.3428 + 3.0027 * \ln(FL_{cm})$

APPENDIX 4. CODES FOR MODELS

Appendix 4a. Model 4-Bayesian life cycle model code in OpenBugs.

```
model {
# Y is total years of matrix, 1996 to 2019
# priors for Bev Holt parameters
bh.alpha <- exp(-delta)  # survival rate as e(-Z)
delta ~ dnorm(1,0.001)C(0,)
K.prime ~ dnorm(1,0.001)C(1,)
K <- K.prime*100000
# priors for mortality rates
z.0 <- -log(S.0)
S.0 ~ dbeta(s.0.a, s.0.b)
for (a in 1:8){
  S[a] ~ dbeta(s.age.a[a], s.age.b[a])
  z.at.age[a] <- -log(S[a])
  }
for (a in 9:15){
  z.at.age[a] <- z.at.age[8]
  }
# priors for proportion recruits to spawners at age assumed similar for male and female
# spawners are for ages 3 to 15 so index runs from 1 to 13
for (a in 1:4){  # spawner ages 3 to 6, strongly informative prior
  p.rec.sp.at.age[a] ~ dbeta(p.rec.sp.a[a], p.rec.sp.b[a])
  }
for (a in 5:13){ # ages 7 to 15
  p.rec.sp.at.age[a] <- p.rec.sp.at.age[4]
  }
# initial year 1996
for (y in 1:1){
   # for ages 1 and 2, use age 3 spawners to estimate recruits
   pred.R[y,1] <- obs.sp.at.age[y,1] / p.rec.sp.at.age[1] / exp(-(z.at.age[1] + z.at.age[2]))
   pred.R[y,2] <- obs.sp.at.age[y,1] / p.rec.sp.at.age[1] / exp(-z.at.age[2])
   for (a in 3:15){
     pred.R[y,a] <- obs.sp.at.age[y,(a-2)] / p.rec.sp.at.age[a-2]
     pred.S[y,a-2] <- pred.R[y,a]*p.rec.sp.at.age[a-2]
     u.log.S.a[y,a-2] <- log(pred.S[y,a-2])
     eggs[y,a-2] <-pred.S[y,a-2]*wt.at.age[a-2]*p.fem.sp.at.age[a-2]*eggs.kg
     }
  tot.eggs[y] <- sum(eggs[y,])
  pred.R0[y] <- bh.alpha*tot.eggs[y]/(1 + tot.eggs[y] * bh.alpha/K)
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds, fill in first year
  }  # end first year
for (y in 2:Y){  # year loop 1997 to 2019
  pred.R[y,1] <- pred.R0[y-1] * exp(-z.0)
  pred.R[y,2] <- pred.R[y-1,1]*exp(-z.at.age[1])
  for (a in 3:14){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])
    pred.S[y,a-2] <- pred.R[y,a]*p.rec.sp.at.age[a-2]
    u.log.S.a[y,a-2] <- log(pred.S[y,a-2])
    eggs[y,a-2] <-pred.S[y,a-2]*wt.at.age[a-2]*p.fem.sp.at.age[a-2]*eggs.kg
    }  # end age 3 to 14 loop
  for (a in 15:15){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1]) + pred.R[y-1,a]*exp(-z.at.age[a])
    pred.S[y,a-2] <- pred.R[y,a]*p.rec.sp.at.age[a-2]
    u.log.S.a[y,a-2] <- log(pred.S[y,a-2])
    eggs[y,a-2] <-pred.S[y,a-2]*wt.at.age[a-2]*p.fem.sp.at.age[a-2]*eggs.kg
    }  # end age 12+ loop
  tot.eggs[y] <- sum(eggs[y,])
  pred.R0[y] <- bh.alpha*tot.eggs[y]/(1 + tot.eggs[y] * bh.alpha/K)
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds
```

```
  } # end year loop
# likelihoods
# age 3 likelihood
for (y in 4:Y){
   obs.sp.at.age[y,1] ~ dlnorm(u.log.S.a[y,1], tau.sp[1])   # likelihood of spawner abundance
   res.S.3[y] <- log(obs.sp.at.age[y,1]/pred.S[y,1]) # residual for spawners age 3
   }
# age 4 likelihood
for (y in 5:Y){
   obs.sp.at.age[y,2] ~ dlnorm(u.log.S.a[y,2], tau.sp[2])   # likelihood of spawner abundance
   res.S.4[y] <- log(obs.sp.at.age[y,2]/pred.S[y,2]) # residual for spawners age 4
   }
# age 5 likelihood
for (y in 6:Y){
   obs.sp.at.age[y,3] ~ dlnorm(u.log.S.a[y,3], tau.sp[3])   # likelihood of spawner abundance
   res.S.5[y] <- log(obs.sp.at.age[y,3]/pred.S[y,3]) # residual for spawners age 5
   }
# age 6 likelihood
for (y in 7:Y){
   obs.sp.at.age[y,4] ~ dlnorm(u.log.S.a[y,4], tau.sp[4])   # likelihood of spawner abundance
   res.S.6[y] <- log(obs.sp.at.age[y,4]/pred.S[y,4]) # residual for spawners age 6
   }
# age 7 likelihood
for (y in 8:Y){
   obs.sp.at.age[y,5] ~ dlnorm(u.log.S.a[y,5], tau.sp[5])   # likelihood of spawner abundance
   res.S.7[y] <- log(obs.sp.at.age[y,5]/pred.S[y,5]) # residual for spawners age 7
   }
# age 8 likelihood
for (y in 9:Y){
   obs.sp.at.age[y,6] ~ dlnorm(u.log.S.a[y,6], tau.sp[6])   # likelihood of spawner abundance
   res.S.8[y] <- log(obs.sp.at.age[y,6]/pred.S[y,6]) # residual for spawners age 8
   }
# total spawner likelihood beginning in year 2008
for (y in 13:Y){
   u.log.S[y] <- log(sum.S[y])
   obs.med.sp[y] ~ dlnorm(u.log.S[y], tau.sp[7])   # likelihood of spawner abundance
   res.S[y] <- log(obs.med.sp[y]/sum.S[y])  # residual for total spawners
   }
for (s in 1:7){
 logsigmaS[s] ~ dunif(0,3)
 tau.sp[s] <- pow(logsigmaS[s],-2)}
# predictions
for (y in Y2:Y3){  # predictions Y+1 to Y+more
  pred.R[y,1] <- pred.R0[y-1] * exp(-z.0)
  pred.R[y,2] <- pred.R[y-1,1]*exp(-z.at.age[1])
 for (a in 3:14){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])
    pred.S[y,a-2] <- pred.R[y,a]*p.rec.sp.at.age[a-2]
    eggs[y,a-2] <- pred.S[y,a-2]*wt.at.age[a-2]*p.fem.sp.at.age[a-2]*eggs.kg
    }  # end age 3 to 11 loop
  for (a in 15:15){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1]) + pred.R[y-1,a]*exp(-z.at.age[a])
    pred.S[y,a-2] <- pred.R[y,a]*p.rec.sp.at.age[a-2]
    u.log.S.a[y,a-2] <- log(pred.S[y,a-2])
    eggs[y,a-2] <-pred.S[y,a-2]*wt.at.age[a-2]*p.fem.sp.at.age[a-2]*eggs.kg
    }  # end age 15+ loop
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds
  u.log.S[y] <- log(sum.S[y])  # log mean of total spawners, for likelihood
  tot.eggs[y] <- sum(eggs[y,])
  pred.R0[y] <- bh.alpha*tot.eggs[y]/(1 + tot.eggs[y] * bh.alpha/K)
   } # end year loop
} # end model
```

Appendix 4b. Model 5-Bayesian life cycle model code in OpenBugs.

```
model {
# Y is total years of matrix, 1996 to 2019
# priors for Bev Holt parameters
bh.alpha <- exp(-delta)  # survival rate as e(-Z)
delta ~ dnorm(1,0.001)C(0,)
K.prime ~ dnorm(1,0.001)C(1,)
K <- K.prime*100000
# priors for mortality rates
# cumulative mortality age 0 to 3
S.0to3 ~ dbeta(s.0to3.a,s.0to3.b)  # survival from summer age 0 to age 3
z.0to3 <- -log(S.0to3)/3  # annual instantaneous rate for age 0, age 1, age 2 non-cohort
for (a in 1:6){  # ages 3 to 8
  S[a] ~ dbeta(s.age.a[a], s.age.b[a])
  z.at.age[a] <- -log(S[a])
  }
for (a in 7:13){
  z.at.age[a] <- z.at.age[6]
  }
# priors for proportion recruits to spawners at age assumed similar for male and female
# spawners are for ages 3 to 15 so index runs from 1 to 13
for (a in 1:4){  # spawner ages 3 to 6, strongly informative prior
  p.rec.sp.at.age[a] ~ dbeta(p.rec.sp.a[a], p.rec.sp.b[a])
  }
for (a in 5:13){ # ages 7 to 15
  p.rec.sp.at.age[a] <- p.rec.sp.at.age[4]
  }
# initial year 1996
for (y in 1:1){
   for (a in 1:13){
     pred.R[y,a] <- obs.sp.at.age[y,a] / p.rec.sp.at.age[a]
     pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
     u.log.S.a[y,a] <- log(pred.S[y,a])
     eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
     }
  tot.eggs[y] <- sum(eggs[y,])
  pred.R0[y] <- bh.alpha*tot.eggs[y]/(1 + tot.eggs[y] * bh.alpha/K)
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds, fill in first year
  }  # end first year
for (y in 2:3){  # year loop 1997 and 1998
  for (a in 1:1){  # age 3
    pred.R[y,a] <- obs.sp.at.age[y,a] / p.rec.sp.at.age[a]
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age 3 loop
  for (a in 2:12){ }  # ages 4 to 14 loop
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end ages 4 to 14 loop
  for (a in 13:13){  # 15 plus group
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1]) + pred.R[y-1,a]*exp(-z.at.age[a])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end 15 plus group
  tot.eggs[y] <- sum(eggs[y,])
  pred.R0[y] <- bh.alpha*tot.eggs[y]/(1 + tot.eggs[y] * bh.alpha/K)
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds
```

```
  } # end year loop
for (y in 4:Y){  # year loop 1999 to 2019
   for (a in 1:1){
    pred.R[y,a] <- pred.R0[y-3]* S.0to3
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age 3 loop
  for (a in 2:12){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end ages 4 to 14 loop
  for (a in 13:13){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1]) + pred.R[y-1,a]*exp(-z.at.age[a])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end 15 plus group

  tot.eggs[y] <- sum(eggs[y,])
  pred.R0[y] <- bh.alpha*tot.eggs[y]/(1 + tot.eggs[y] * bh.alpha/K)
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds
  } # end year loop
# likelihood
# age 3 likelihood
for (y in 4:Y){
   obs.sp.at.age[y,1] ~ dlnorm(u.log.S.a[y,1], tau.sp[1])   # likelihood of spawner abundance
   res.S.3[y] <- log(obs.sp.at.age[y,1]/pred.S[y,1]) # residual for spawners age 3
   }
# age 4 likelihood
for (y in 5:Y){
   obs.sp.at.age[y,2] ~ dlnorm(u.log.S.a[y,2], tau.sp[2])   # likelihood of spawner abundance
   res.S.4[y] <- log(obs.sp.at.age[y,2]/pred.S[y,2]) # residual for spawners age 4
   }
# age 5 likelihood
for (y in 6:Y){
   obs.sp.at.age[y,3] ~ dlnorm(u.log.S.a[y,3], tau.sp[3])   # likelihood of spawner abundance
   res.S.5[y] <- log(obs.sp.at.age[y,3]/pred.S[y,3]) # residual for spawners age 5
   }
# age 6 likelihood
for (y in 7:Y){
   obs.sp.at.age[y,4] ~ dlnorm(u.log.S.a[y,4], tau.sp[4])   # likelihood of spawner abundance
   res.S.6[y] <- log(obs.sp.at.age[y,4]/pred.S[y,4]) # residual for spawners age 6
   }
# age 7 likelihood
for (y in 8:Y){
   obs.sp.at.age[y,5] ~ dlnorm(u.log.S.a[y,5], tau.sp[5])   # likelihood of spawner abundance
   res.S.7[y] <- log(obs.sp.at.age[y,5]/pred.S[y,5]) # residual for spawners age 7
   }
# age 8 likelihood
for (y in 9:Y){
   obs.sp.at.age[y,6] ~ dlnorm(u.log.S.a[y,6], tau.sp[6])   # likelihood of spawner abundance
   res.S.8[y] <- log(obs.sp.at.age[y,6]/pred.S[y,6]) # residual for spawners age 8
   }
# total spawner likelihood beginning in year 2008
for (y in 13:Y){
   u.log.S[y] <- log(sum.S[y])
   obs.med.sp[y] ~ dlnorm(u.log.S[y], tau.sp[7])   # likelihood of spawner abundance
   res.S[y] <- log(obs.med.sp[y]/sum.S[y])  # residual for total spawners
   }
```

```
for (s in 1:7){
 logsigmaS[s] ~ dunif(0,3)
 tau.sp[s] <- pow(logsigmaS[s],-2)
  }
# predictions
for (y in Y2:Y3){  # predictions Y+1 to Y+more
  for (a in 1:1){
    pred.R[y,a] <- pred.R0[y-3]* S.0to3
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age 3 loop
  for (a in 2:12){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    eggs[y,a] <- pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age loop
  for (a in 13:13){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])+ pred.R[y-1,a]*exp(-z.at.age[a])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    eggs[y,a] <- pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age loop
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds
  u.log.S[y] <- log(sum.S[y])  # log mean of total spawners, for likelihood
  tot.eggs[y] <- sum(eggs[y,])
  pred.R0[y] <- bh.alpha*tot.eggs[y]/(1 + tot.eggs[y] * bh.alpha/K)
   } # end year loop
} # end model
```

Appendix 4c. Model 6-Bayesian life cycle model code in OpenBugs.

```
model {
# Y is total years of matrix, 1996 to 2019
# priors for Bev Holt parameters
bh.alpha <- exp(-delta)  # survival rate as e(-Z)
delta ~ dnorm(1,0.001)C(0,)
K.prime ~ dnorm(1,0.001)C(1,)
K <- K.prime*100000
# priors for mortality rates
for (a in 1:6){  # ages 3 to 8
  S[a] ~ dbeta(s.age.a[a], s.age.b[a])
  z.at.age[a] <- -log(S[a])
  }
for (a in 7:13){
  z.at.age[a] <- z.at.age[6]
  }
# priors for proportion recruits to spawners at age assumed similar for male and female
# spawners are for ages 3 to 15 so index runs from 1 to 13
for (a in 1:4){  # spawner ages 3 to 6, weakly informative prior
  p.rec.sp.at.age[a] ~ dbeta(p.rec.sp.a[a], p.rec.sp.b[a])
  }
for (a in 5:13){ # ages 7 to 15
  p.rec.sp.at.age[a] <- p.rec.sp.at.age[4]
  }
# initial year 1996
for (y in 1:1){
   for (a in 1:13){  # spawners at age 3 to 15
     pred.R[y,a] <- obs.sp.at.age[y,a] / p.rec.sp.at.age[a]
     pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
     u.log.S.a[y,a] <- log(pred.S[y,a])
     eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
     }
  tot.eggs[y] <- sum(eggs[y,])
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds, fill in first year
  }  # end first year
for (y in 2:3){  # year loop 1997 and 1998
  for (a in 1:1){
    pred.R[y,a] <- obs.sp.at.age[y,a] / p.rec.sp.at.age[a]
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age 3 loop
  for (a in 2:12){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end ages 4 to 14 loop
  for (a in 13:13){  # age 15 loop
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1]) + pred.R[y-1,a]*exp(-z.at.age[a])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end 15 plus group
  tot.eggs[y] <- sum(eggs[y,])
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds
  } # end year loop
for (y in 4:Y){  # year loop 1999 to 2019
  for (a in 1:1){
    pred.R[y,a] <- bh.alpha*tot.eggs[y-3]/(1 + tot.eggs[y-3] * bh.alpha/K)
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
```

```
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age 3 loop
  for (a in 2:12){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end ages 4 to 14 loop
  for (a in 13:13){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1]) + pred.R[y-1,a]*exp(-z.at.age[a])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end 15 plus group
  tot.eggs[y] <- sum(eggs[y,])
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds
  } # end year loop
# likelihoods
# age 3 likelihood
for (y in 4:Y){
   obs.sp.at.age[y,1] ~ dlnorm(u.log.S.a[y,1], tau.sp[1])   # likelihood of spawner abundance
   res.S.3[y] <- log(obs.sp.at.age[y,1]/pred.S[y,1]) # residual for spawners age 3
   }
# age 4 likelihood
for (y in 5:Y){
   obs.sp.at.age[y,2] ~ dlnorm(u.log.S.a[y,2], tau.sp[2])   # likelihood of spawner abundance
   res.S.4[y] <- log(obs.sp.at.age[y,2]/pred.S[y,2]) # residual for spawners age 4
   }
# age 5 likelihood
for (y in 6:Y){
   obs.sp.at.age[y,3] ~ dlnorm(u.log.S.a[y,3], tau.sp[3])   # likelihood of spawner abundance
   res.S.5[y] <- log(obs.sp.at.age[y,3]/pred.S[y,3]) # residual for spawners age 5
   }
# age 6 likelihood
for (y in 7:Y){
   obs.sp.at.age[y,4] ~ dlnorm(u.log.S.a[y,4], tau.sp[4])   # likelihood of spawner abundance
   res.S.6[y] <- log(obs.sp.at.age[y,4]/pred.S[y,4]) # residual for spawners age 6
   }
# age 7 likelihood
for (y in 8:Y){
   obs.sp.at.age[y,5] ~ dlnorm(u.log.S.a[y,5], tau.sp[5])   # likelihood of spawner abundance
   res.S.7[y] <- log(obs.sp.at.age[y,5]/pred.S[y,5]) # residual for spawners age 7
   }
# age 8 likelihood
for (y in 9:Y){
   obs.sp.at.age[y,6] ~ dlnorm(u.log.S.a[y,6], tau.sp[6])   # likelihood of spawner abundance
   res.S.8[y] <- log(obs.sp.at.age[y,6]/pred.S[y,6]) # residual for spawners age 8
   }
# total spawner likelihood beginning in year 2008
for (y in 13:Y){
   u.log.S[y] <- log(sum.S[y])
   obs.med.sp[y] ~ dlnorm(u.log.S[y], tau.sp[7])   # likelihood of spawner abundance
   res.S[y] <- log(obs.med.sp[y]/sum.S[y])  # residual for total spawners
   }
for (s in 1:7){
 logsigmaS[s] ~ dunif(0,3)
 tau.sp[s] <- pow(logsigmaS[s],-2)
  }
# predictions
for (y in Y2:Y3){  # predictions Y+1 to Y+more
  for (a in 1:1){
```

```
    pred.R[y,a] <- bh.alpha*tot.eggs[y-3]/(1 + tot.eggs[y-3] * bh.alpha/K)
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    u.log.S.a[y,a] <- log(pred.S[y,a])
    eggs[y,a] <-pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age 3 loop
  for (a in 2:12){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    eggs[y,a] <- pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age loop
  for (a in 13:13){
    pred.R[y,a] <- pred.R[y-1,a-1]*exp(-z.at.age[a-1])+ pred.R[y-1,a]*exp(-z.at.age[a])
    pred.S[y,a] <- pred.R[y,a]*p.rec.sp.at.age[a]
    eggs[y,a] <- pred.S[y,a]*wt.at.age[a]*p.fem.sp.at.age[a]*eggs.kg
    }  # end age loop
  sum.S[y] <- sum(pred.S[y,]) # total spawners on the spawning grounds
  u.log.S[y] <- log(sum.S[y])  # log mean of total spawners, for likelihood
  tot.eggs[y] <- sum(eggs[y,])
   } # end year loop
} # end model
```

APPENDIX 5. DIAGNOSTICS OF MODEL FITS

The following outputs from models 4, 5, and 6 are provided.

- Table summarizing the model structure, parameters, priors, fitting diagnostic (deviance, AIC’) and comments on fits;
- Density plots of prior versus posteriors for model parameters;
- Boxplots of posterior distributions of parameters;
- Correlation plots of parameters;
- Residuals plot;
- Observed vs predicted total spawners;
- Observed versus predicted proportions at age of spawners.

Table A5.1. Parameters, priors and diagnostics of model 4.

Feature	Specifics
Parameters and prior assumptions	Non-informative: Bev-Holt (α, K) σ (3:8, Total) Weakly Informative: S[0:2] S[3:8] p.rec.to.spawner[3:6]
Parameter	Beverton-Holt $\alpha = \exp(-\delta)$ Z(0) = -log(S.0) Z[1] = -log(S[1]) Z[2] = -log(S[2] Z[3:8] = -log(S[3:8]) Z[9:15+] = Z[8] p.rec.to.sp[7:15+] = p.rec.to.sp[6]
Prior	δ ~ N(1,0.001)C(0,) K ~ N(1,0.001)C(1,) S.0 ~ Beta(14,86) S[1] ~ Beta(44,56) S[2] ~ Beta(64,36) S[3:8] ~ Beta(6,4) p.rec.to.sp[3] ~ Beta(4,12) p.rec.to.sp[4] ~ Beta(3,3) p.rec.to.sp[5] ~ Beta(5,2) p.rec.to.sp[6] ~ Beta(4,1) log(σ) [3:8, Total] ~ U(0,3)
Fit statistics	Deviance: 2396 Parameters: 22 AIC' = Dev+2*p = 2440 DIC = 2401 (pD = 5.0)
Comments	Good fit to spawners at ages 3 to 6 Mostly positive residuals for total spawners Temporal trend in residuals for ages 7 and 8 No autocorrelation for residuals Survival age 3 higher than for ages 4 to 7 which is not consistent with expectations Negative correlation between α and K, α and S[0]

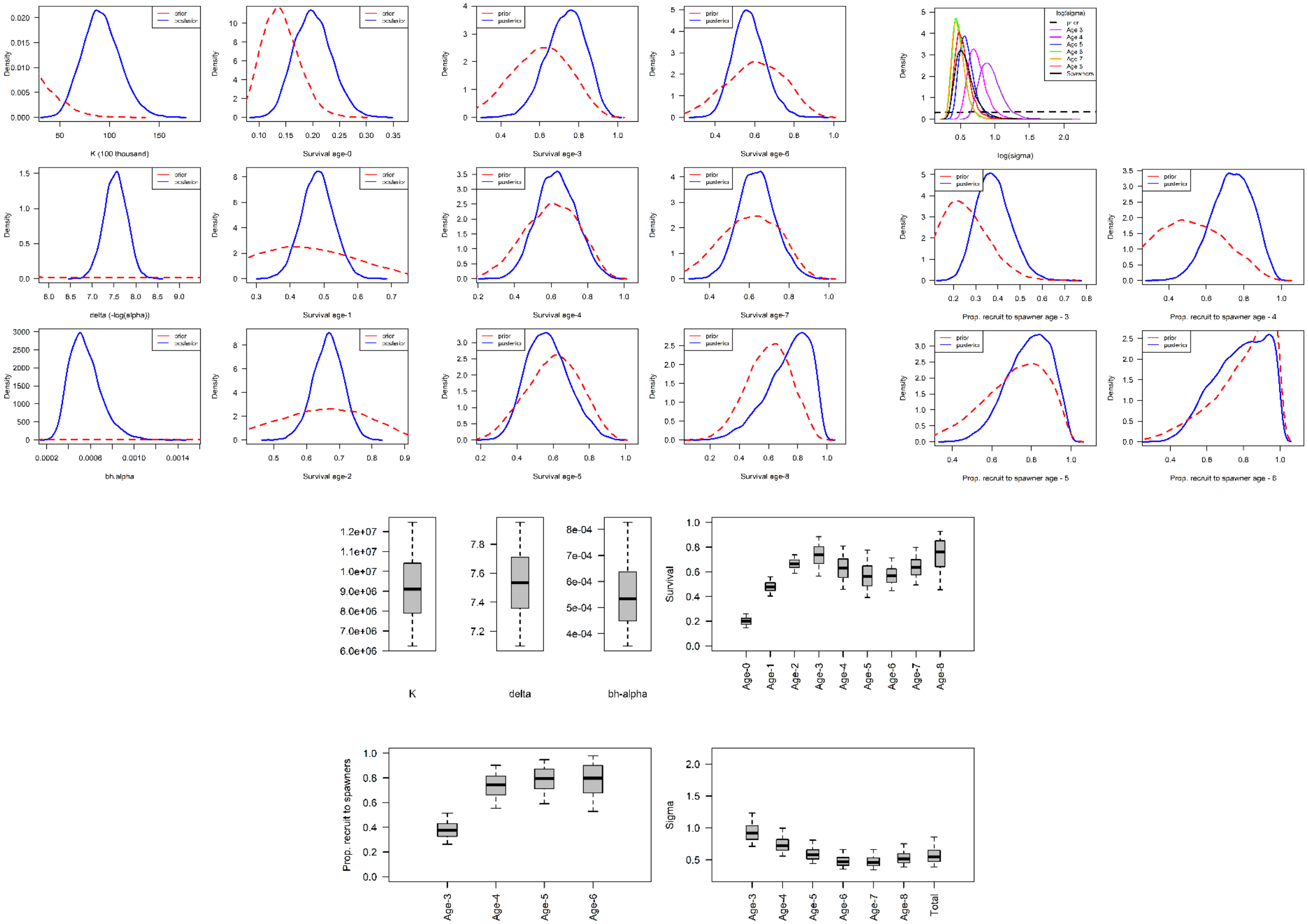


Figure A5.1. Parameter posterior distributions of model 4.

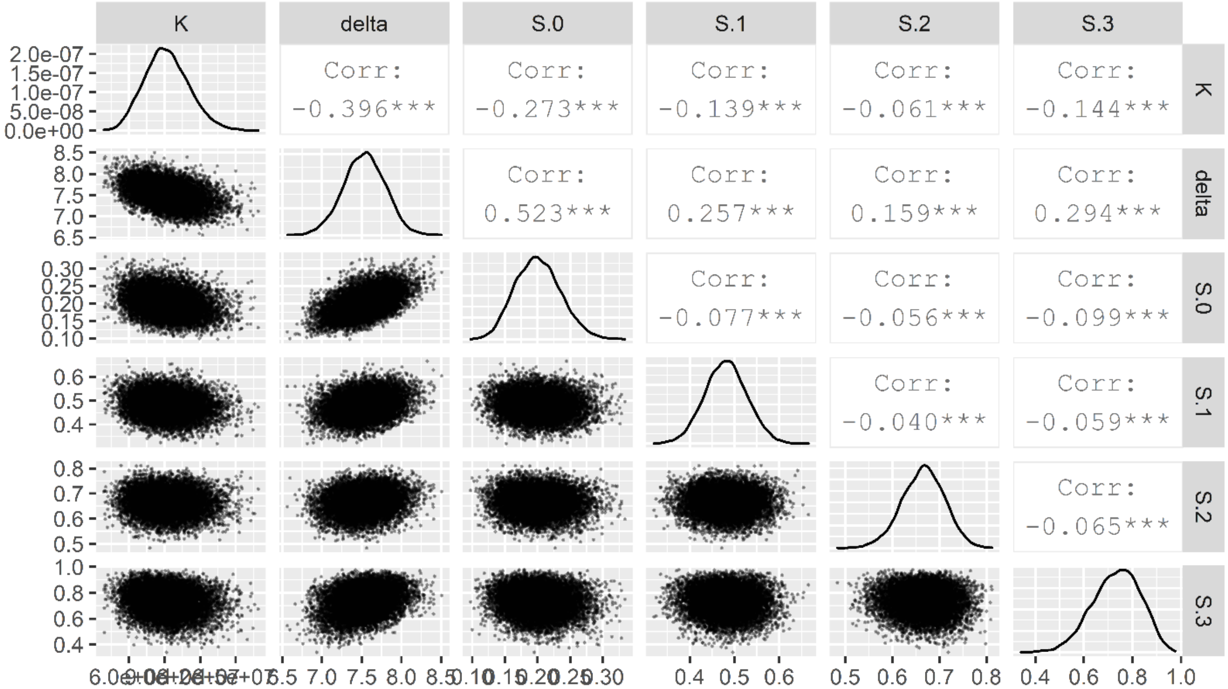


Figure A5.1 (continued). Parameter scatter plots and Pearson correlations of key parameters from the model fits of model 4.

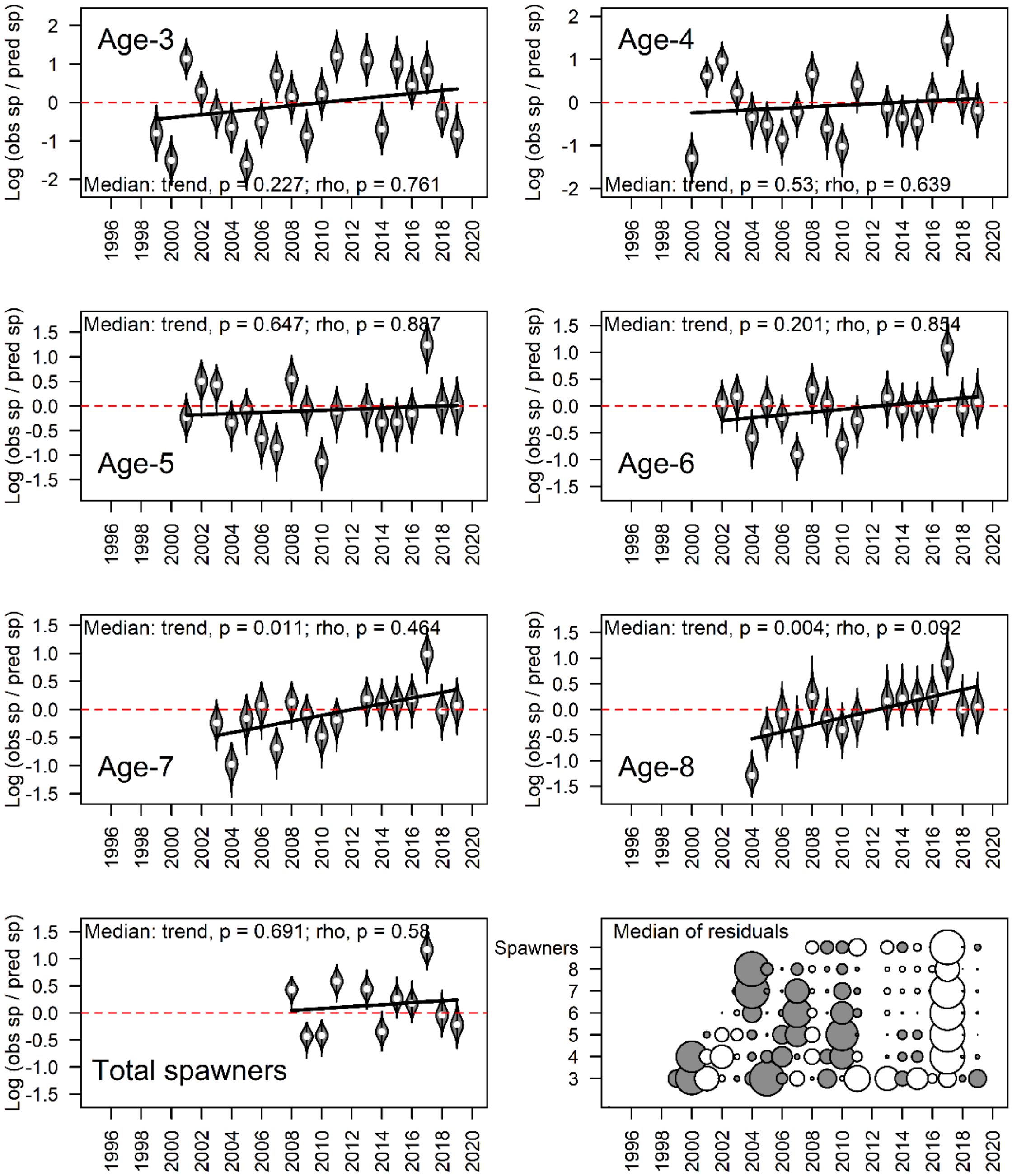


Figure A5.1 (continued). Residual plots expressed as log(assessed abundance / predicted abundance) at ages 3 to 8+, for total spawners, and relative (by age group) bubble plot of logged residual patterns of model 4. Also shown in each panel of residuals are the p-value for the temporal linear trend in residuals and the p-value for the first order autocorrelation of the residuals (from package EnvStats in R).

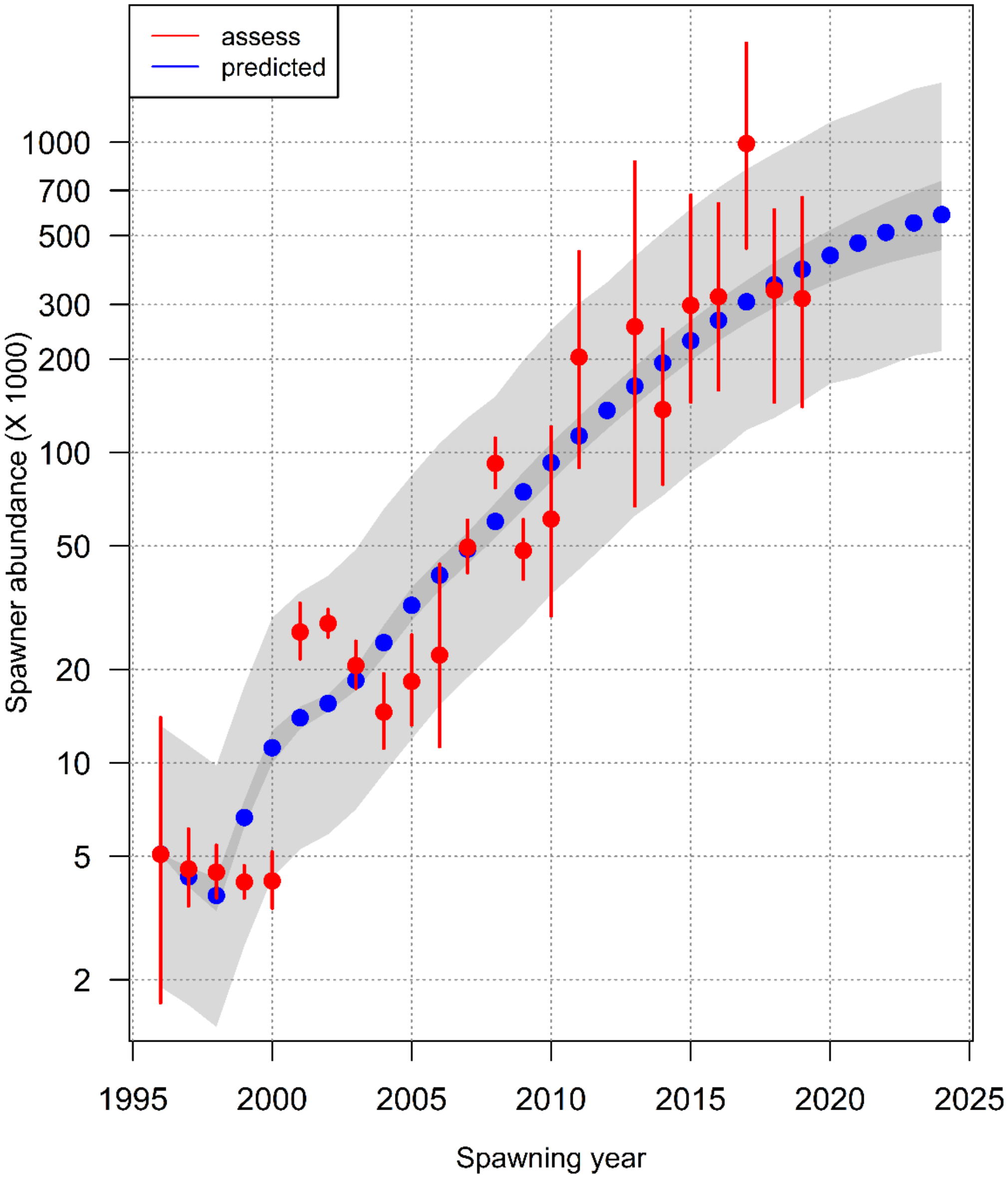


Figure A5.1 (continued). Observed vs predicted total spawners of model 4. The assessed abundances are shown as red symbols for the median with 5th to 95th percentiles ranges as red vertical lines. The blue symbols are the predicted abundances, the darker grey shading is the 5th to 95th percentile range of mean predicted abundance and the light grey shading represents the 5th to 95th percentile range of the predicted spawner abundance accounting for the full process uncertainty (logσ). Note the y-axis abundance is shown on the log scale.

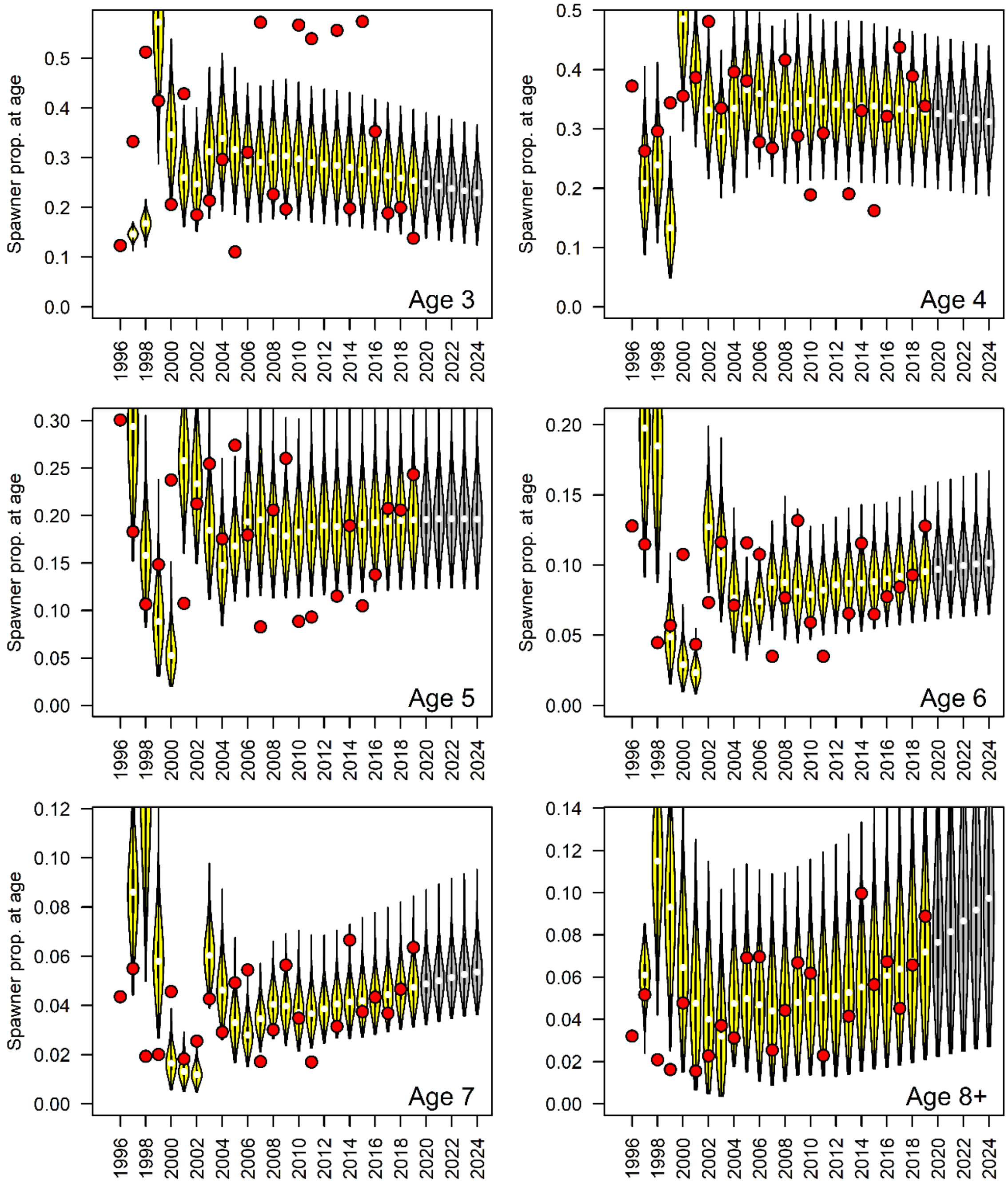


Figure A5.1 (continued). Observed (red symbols) vs predicted (yellow violin plots) proportions at age of spawners, for ages 3 to 7, and for ages 8+ of model 4.

Table A5.2. Parameters, priors and diagnostics of model 5.

Feature	Specifics
Parameters and prior assumptions	Non-informative: Bev-Holt (α, K) σ (3:8, Total) Weakly Informative: S[3:8] p.rec.to.spawner[3:6] S[0to3]
Parameter	Beverton-Holt α = exp(-δ) Z(0to3) = -log(S[0to3]) Z[3:8] = -log(S[3:8]) Z[9:15+] = Z[8] p.rec.to.sp[7:15+] = p.rec.to.sp[6]
Prior	δ ~ N(1,0.001)C(0,) K ~ N(1,0.001)C(1,) S[0to3] ~ Beta(5,45) S[3:8] ~ Beta(6,4) p.rec.to.sp[3] ~ Beta(4,12) p.rec.to.sp[4] ~ Beta(3,3) p.rec.to.sp[5] ~ Beta(5,2) p.rec.to.sp[6] ~ Beta(4,1) log(σ) [3:8, Total] ~ U(0,3)
Fit statistics	Deviance: 2395 Parameters: 20 AIC' = Dev+2*p = 2435 DIC = 2394 (pD = -1.4)
Comments	Good fit to spawners at ages 3 to 6 Better fit to total spawners, balanced residuals Temporal trend in residuals for ages 7 and 8 No autocorrelation for residuals. Survival age 3 higher than for ages 4 to 7 which is not consistent with expectations Negative correlation between α and K, α and S.0to3

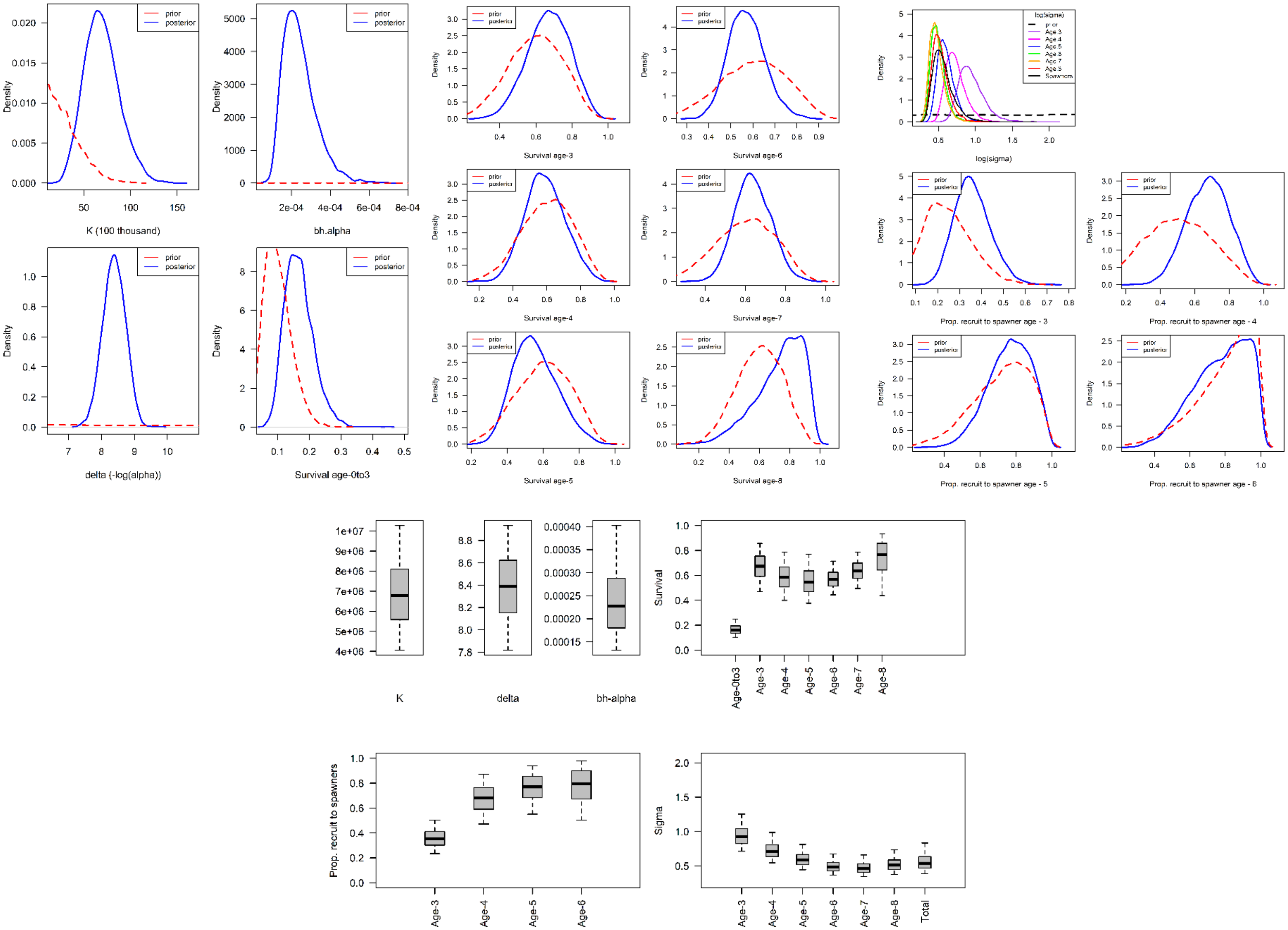


Figure A5.2. Parameter posterior distributions of model 5.

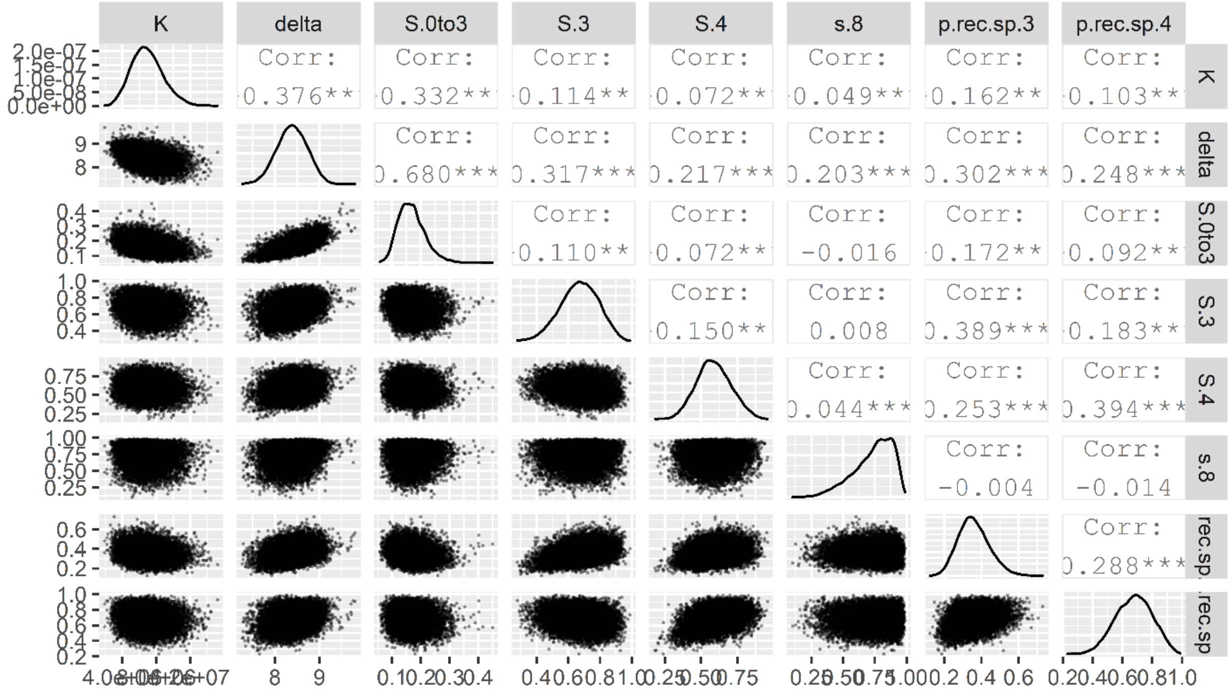


Figure A5.2 (continued). Parameter scatter plots and Pearson correlations of key parameters from the model fits of model 5.

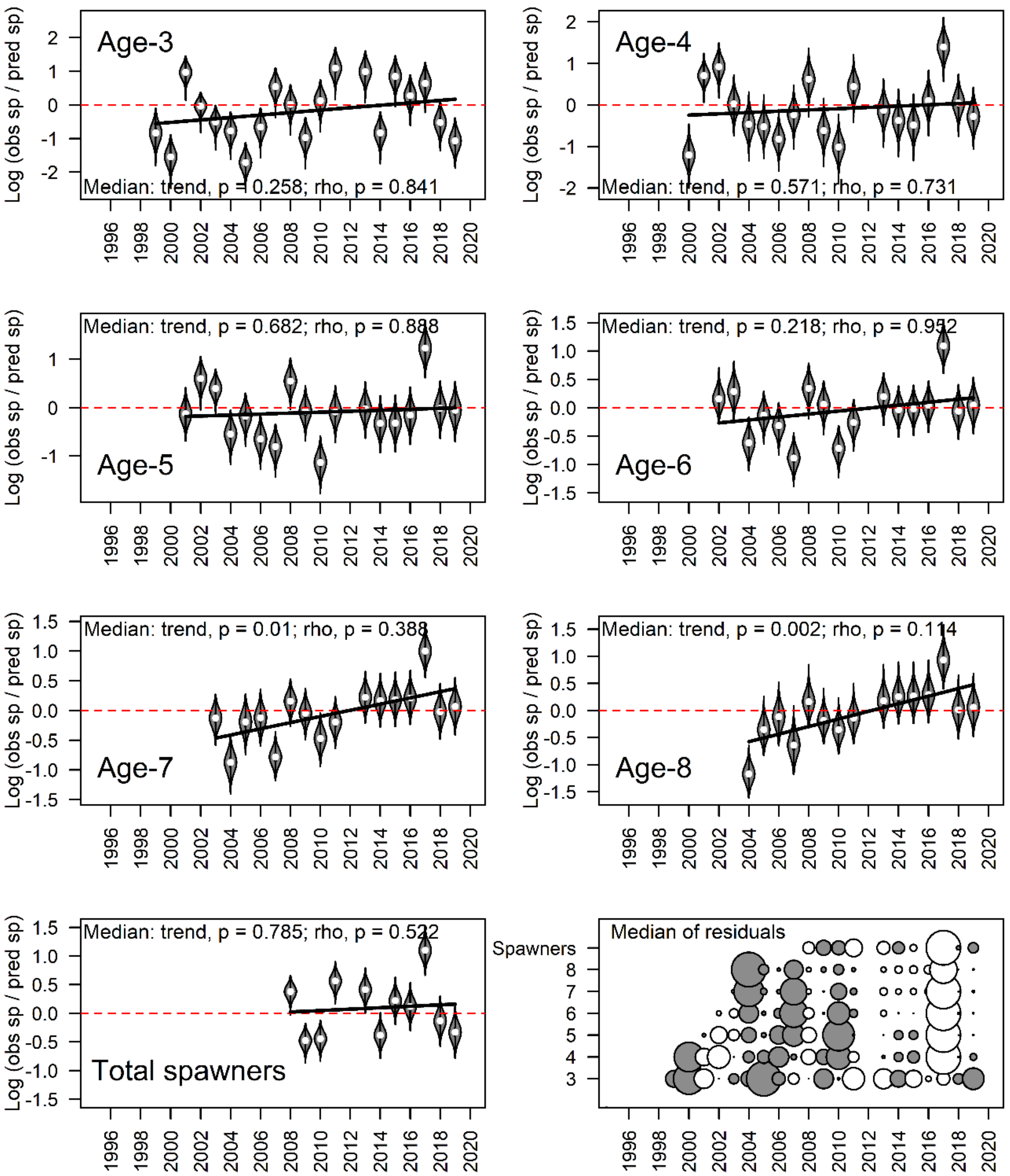


Figure A5.2 (continued). Residual plots expressed as log(assessed abundance / predicted abundance) at ages 3 to 8+, for total spawners, and relative (by age group) bubble plot of logged residual patterns of model 5. Also shown in each panel of residuals are the p-value for the temporal linear trend in residuals and the p-value for the first order autocorrelation of the residuals (from package EnvStats in R).

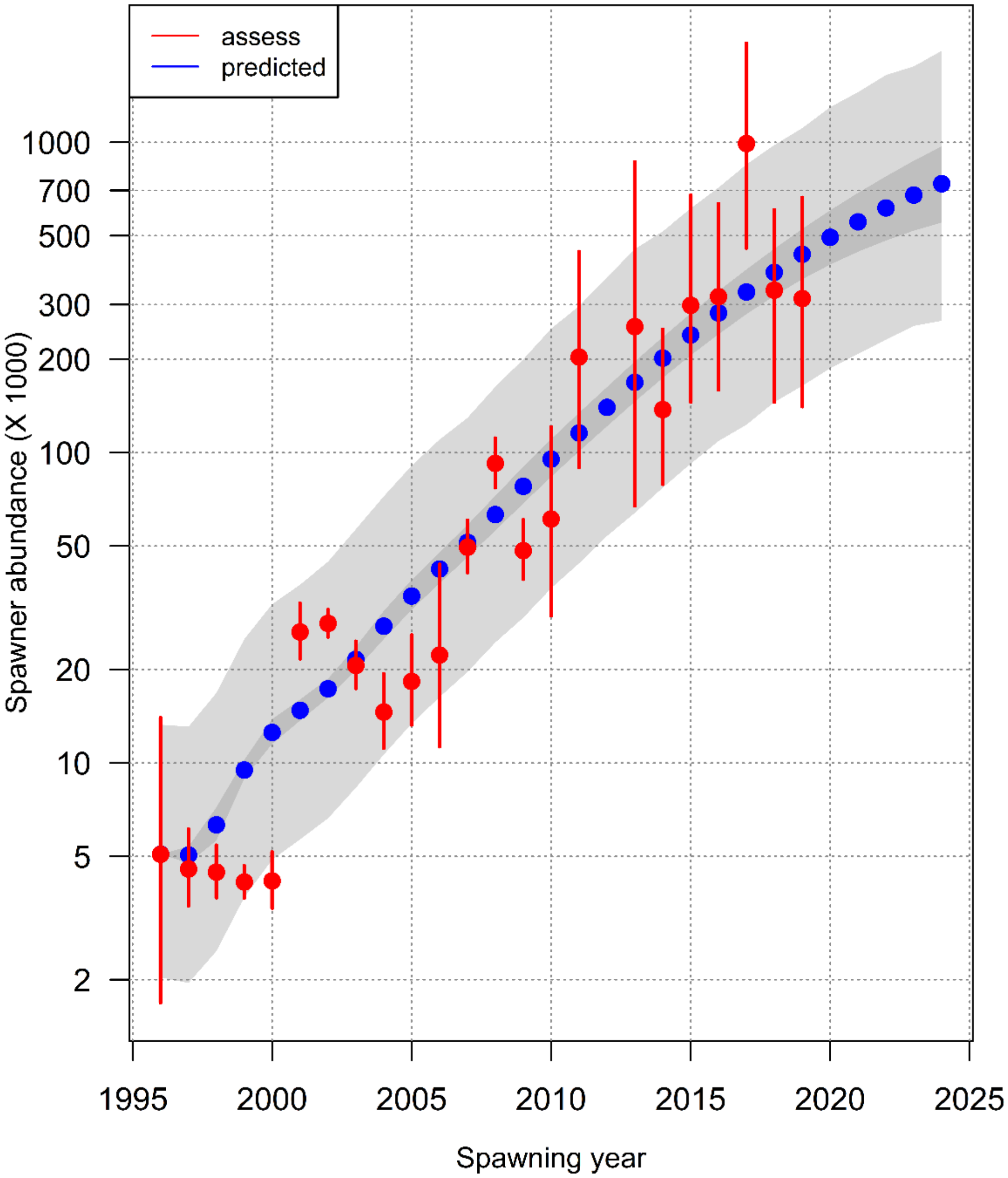


Figure A5.2 (continued). Observed vs predicted total spawners of model 5. The assessed abundances are shown as red symbols for the median with 5th to 95th percentiles ranges as red vertical lines. The blue symbols are the predicted abundances, the darker grey shading is the 5th to 95th percentile range of mean predicted abundance and the light grey shading represents the 5th to 95th percentile range of the predicted spawner abundance accounting for the full process uncertainty (logσ). Note the y-axis abundance is shown on the log scale.

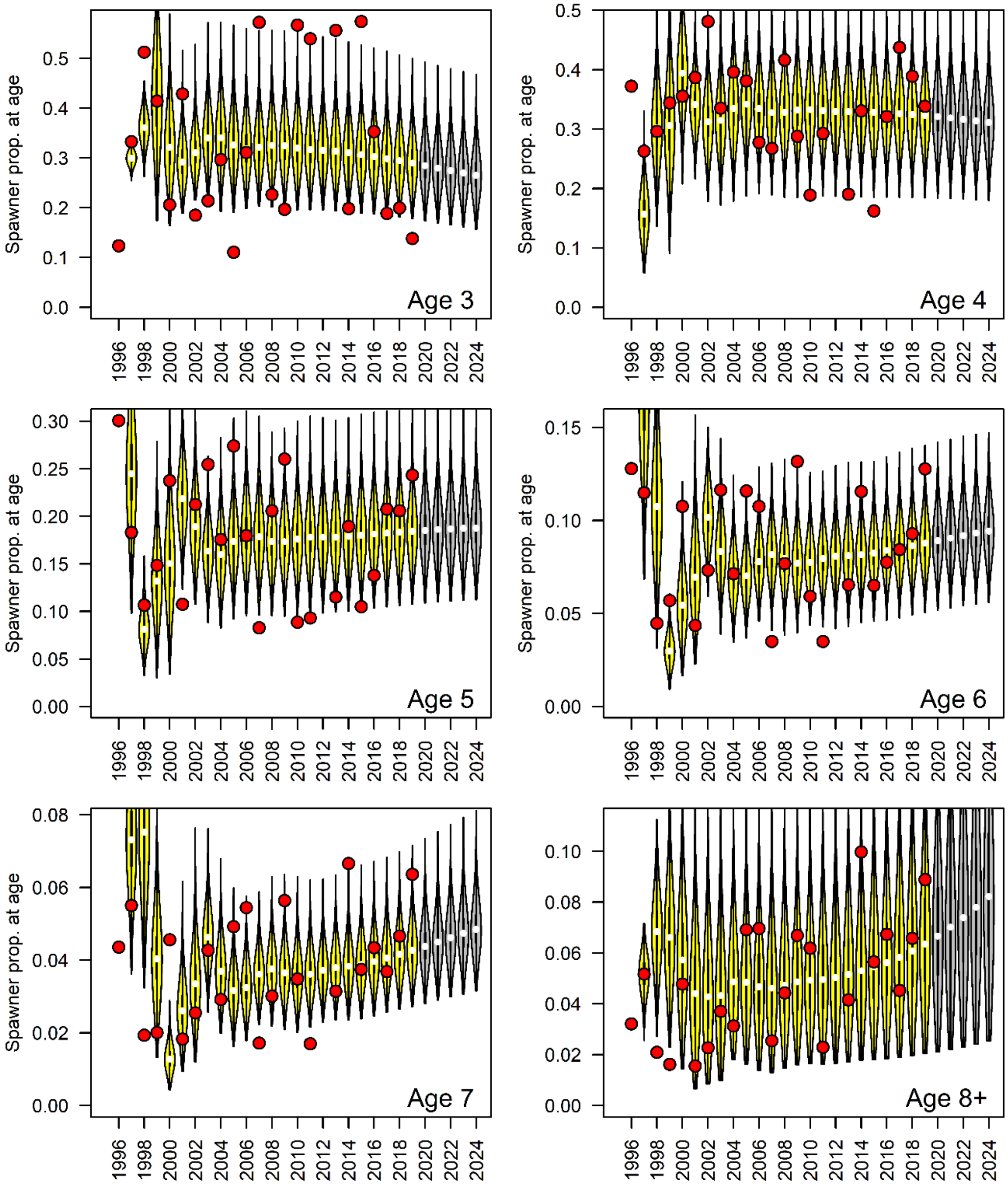


Figure A5.2 (continued). Observed (red symbols) vs predicted (yellow violin plots) proportions at age of spawners, for ages 3 to 7, and for ages 8+ of model 5.

Table A5.3. Parameters, priors and diagnostics of model 6.

Feature	Specifics
Parameters and prior assumptions	Non-informative: Bev-Holt (α, K) for age 3 σ (3:8, Total) Weakly Informative: S[3:8] p.rec.to.spawner[3:6]
Parameter	Beverton-Holt $\alpha = \exp(-\delta)$ Z[3:8] = -log(S[3:8]) Z[9:15+] = Z[8] p.rec.to.sp[7:15+] = p.rec.to.sp[6]
Prior	δ ~ N(1,0.001)C(0,) K ~ N(1,0.001)C(1,) S[3:8] ~ Beta(6,4) p.rec.to.sp[3] ~ Beta(4,12) p.rec.to.sp[4] ~ Beta(3,3) p.rec.to.sp[5] ~ Beta(5,2) p.rec.to.sp[6] ~ Beta(4,1) log(σ) [3:8, Total] ~ U(0,3)
Fit statistics	Deviance: 2391 Parameters: 19 AIC' = Dev+2*p = 2429 DIC = 2392 (pD = 0.3)
Comments	Good fit to spawners at age Temporal trend in residuals for ages 7 and 8 No autocorrelation for residuals. Survival age 3 higher than S for ages 4 to 7 which is not consistent with expectations Positive correlation between Bev-Holt alpha and S[3]

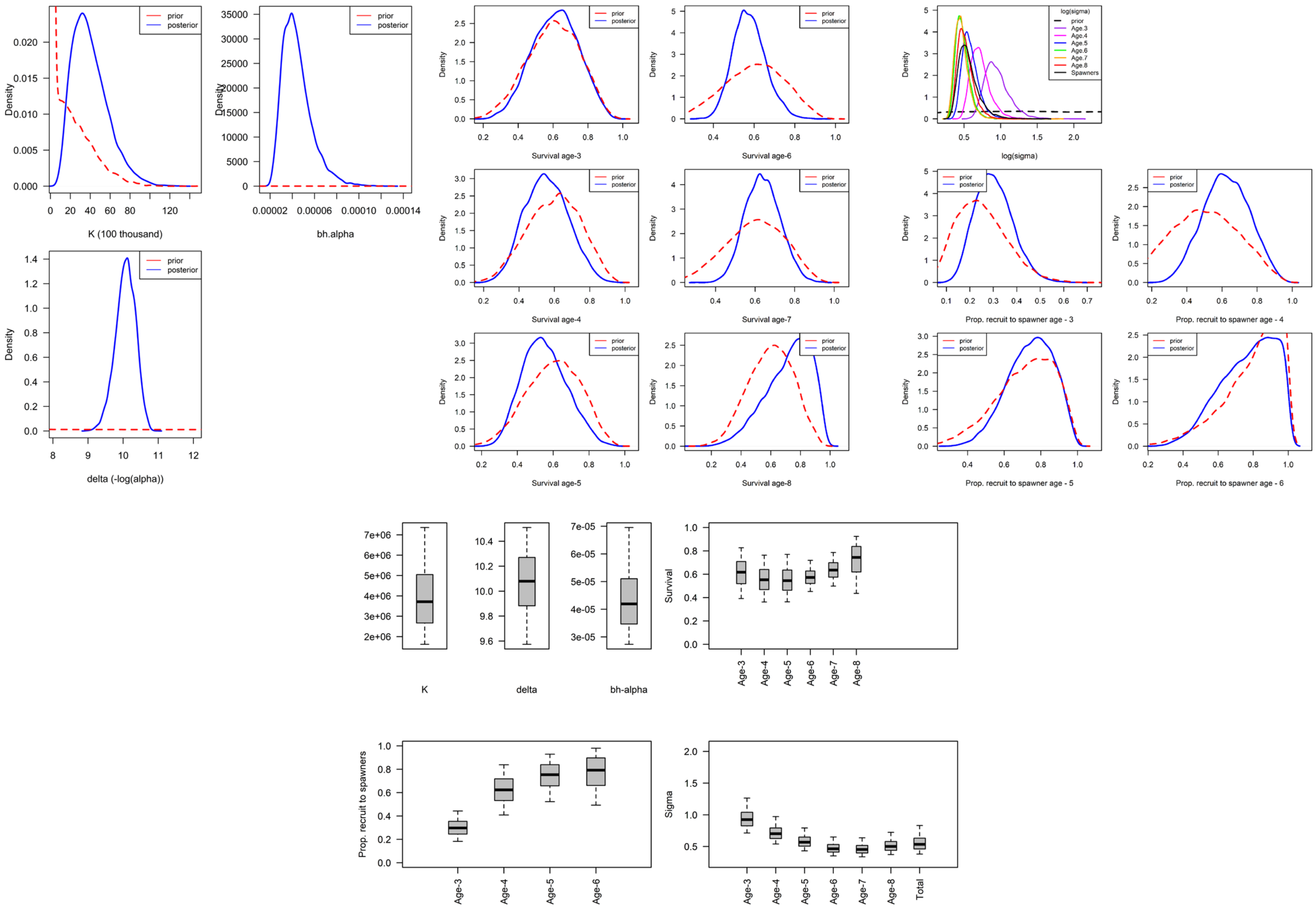


Figure A5.3. Parameter posterior distributions of model 6.

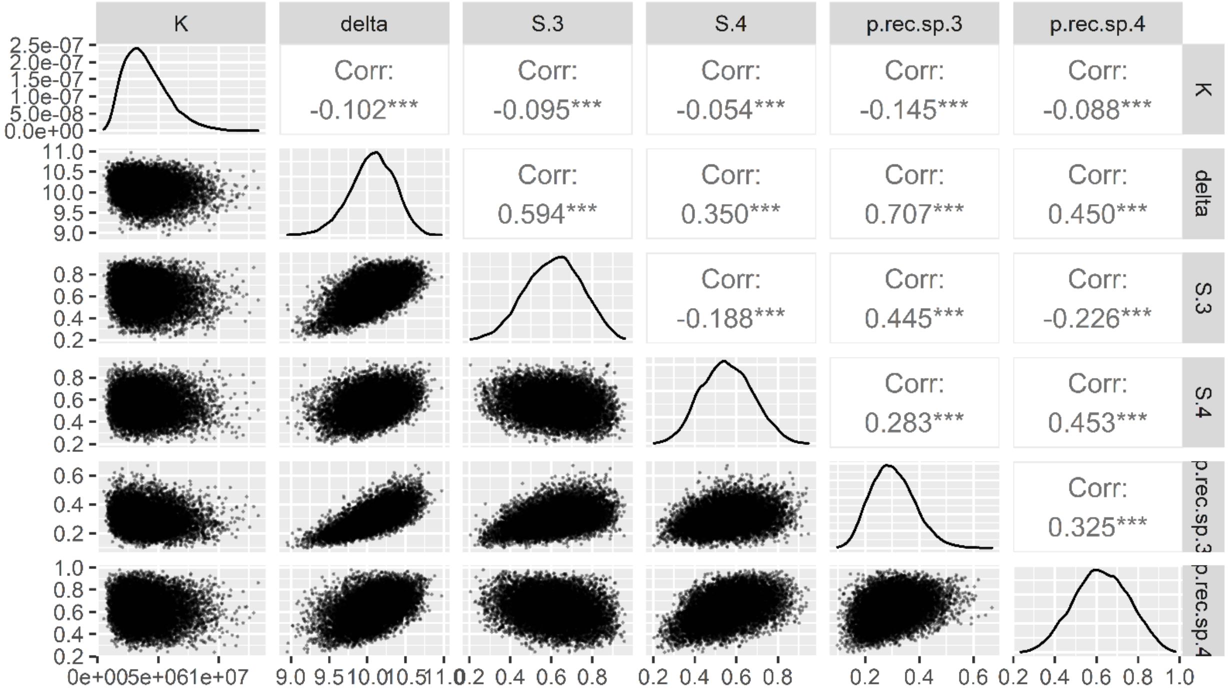


Figure A5.3 (continued). Parameter scatter plots and Pearson correlations of key parameters from the model fits of model 6.

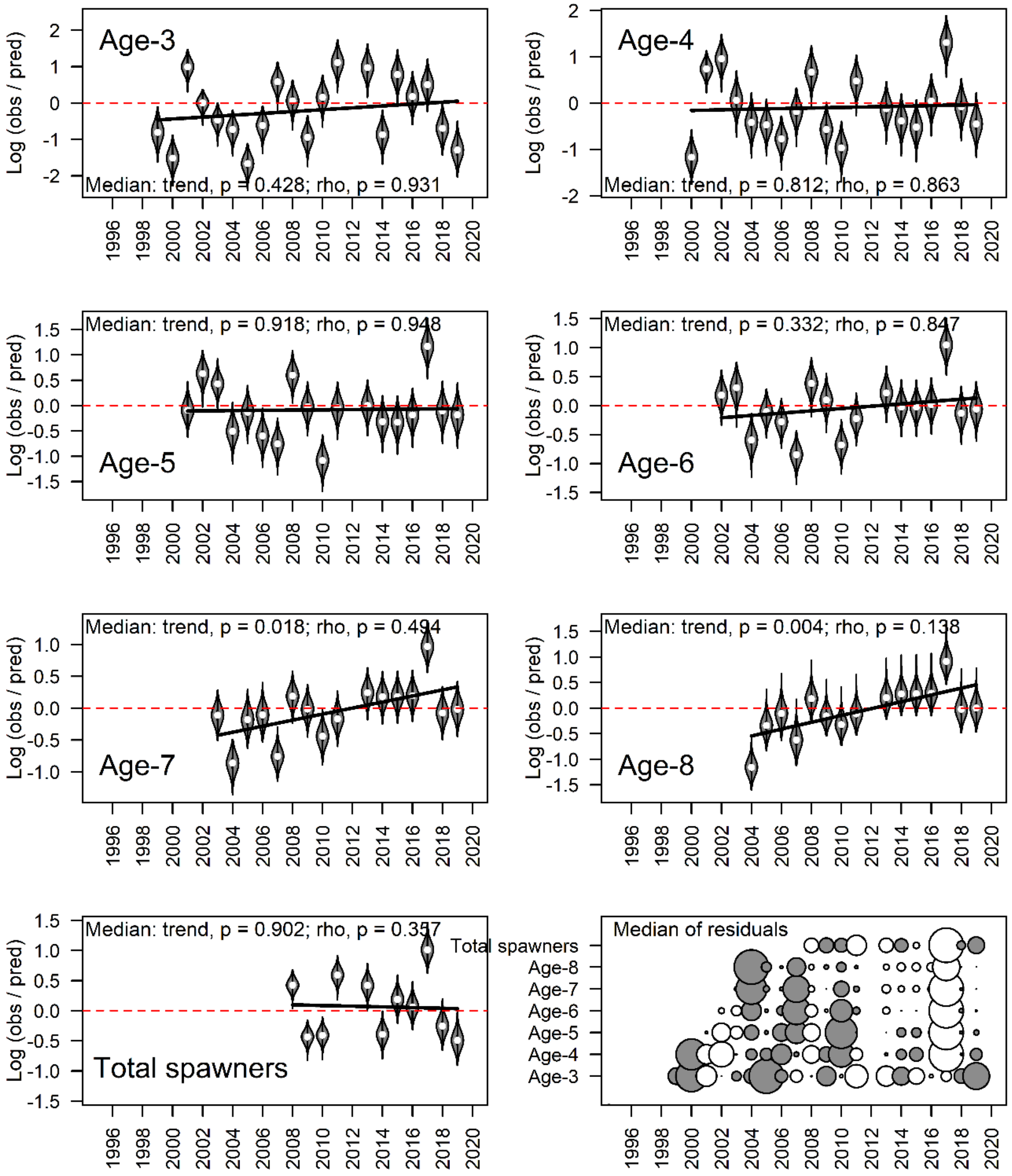


Figure A5.3 (continued). Residual plots expressed as log(assessed abundance / predicted abundance) at ages 3 to 8+, for total spawners, and relative (by age group) bubble plot of logged residual patterns of model 6. Also shown in each panel of residuals are the p-value for the temporal linear trend in residuals and the p-value for the first order autocorrelation of the residuals (from package EnvStats in R).

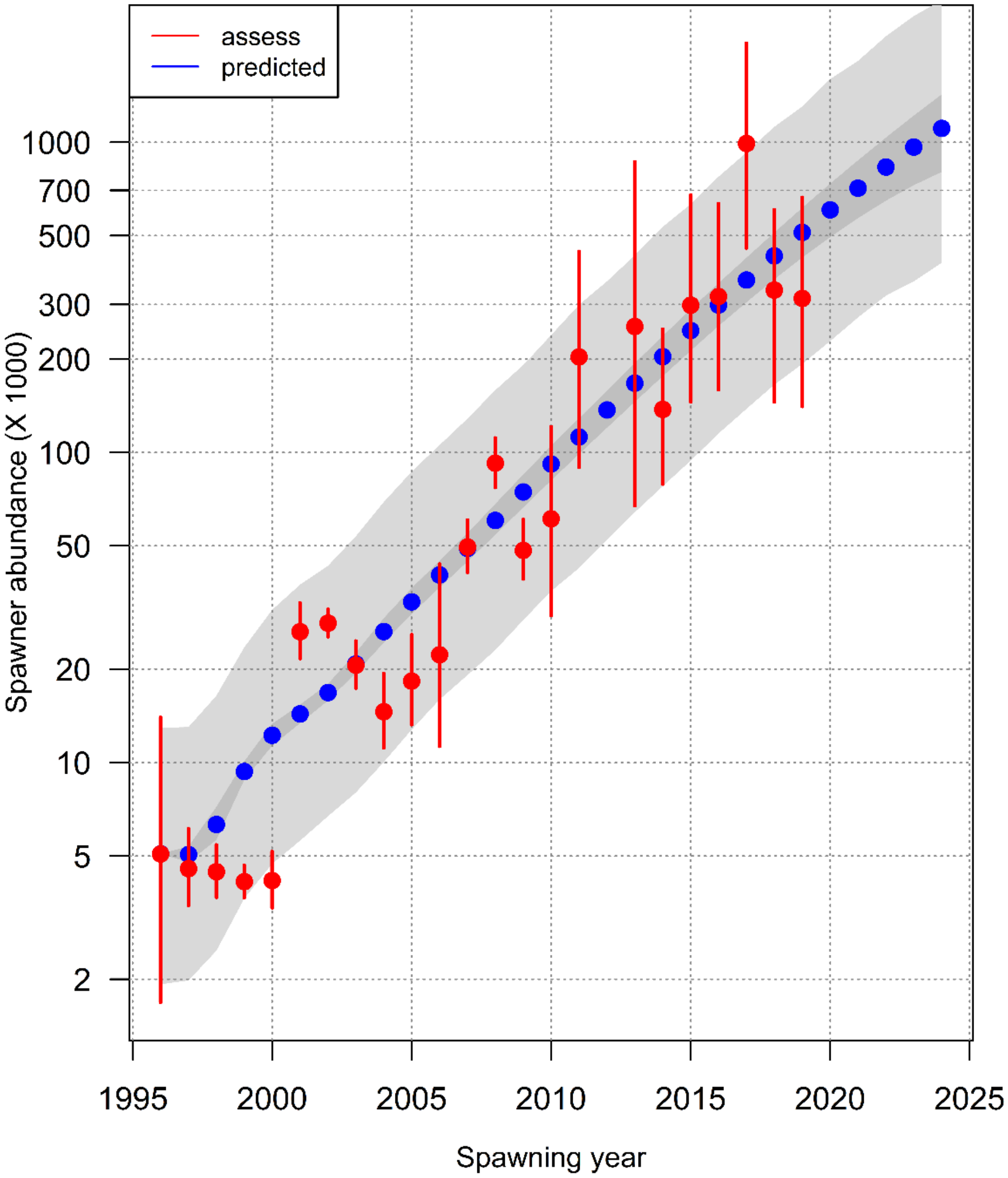


Figure A5.3 (continued). Observed vs predicted total spawners of model 6. The assessed abundances are shown as red symbols for the median with 5th to 95th percentiles ranges as red vertical lines. The blue symbols are the predicted abundances, the darker grey shading is the 5th to 95th percentile range of mean predicted abundance and the light grey shading represents the 5th to 95th percentile range of the predicted spawner abundance accounting for the full process uncertainty (logσ). Note the y-axis abundance is shown on the log scale.

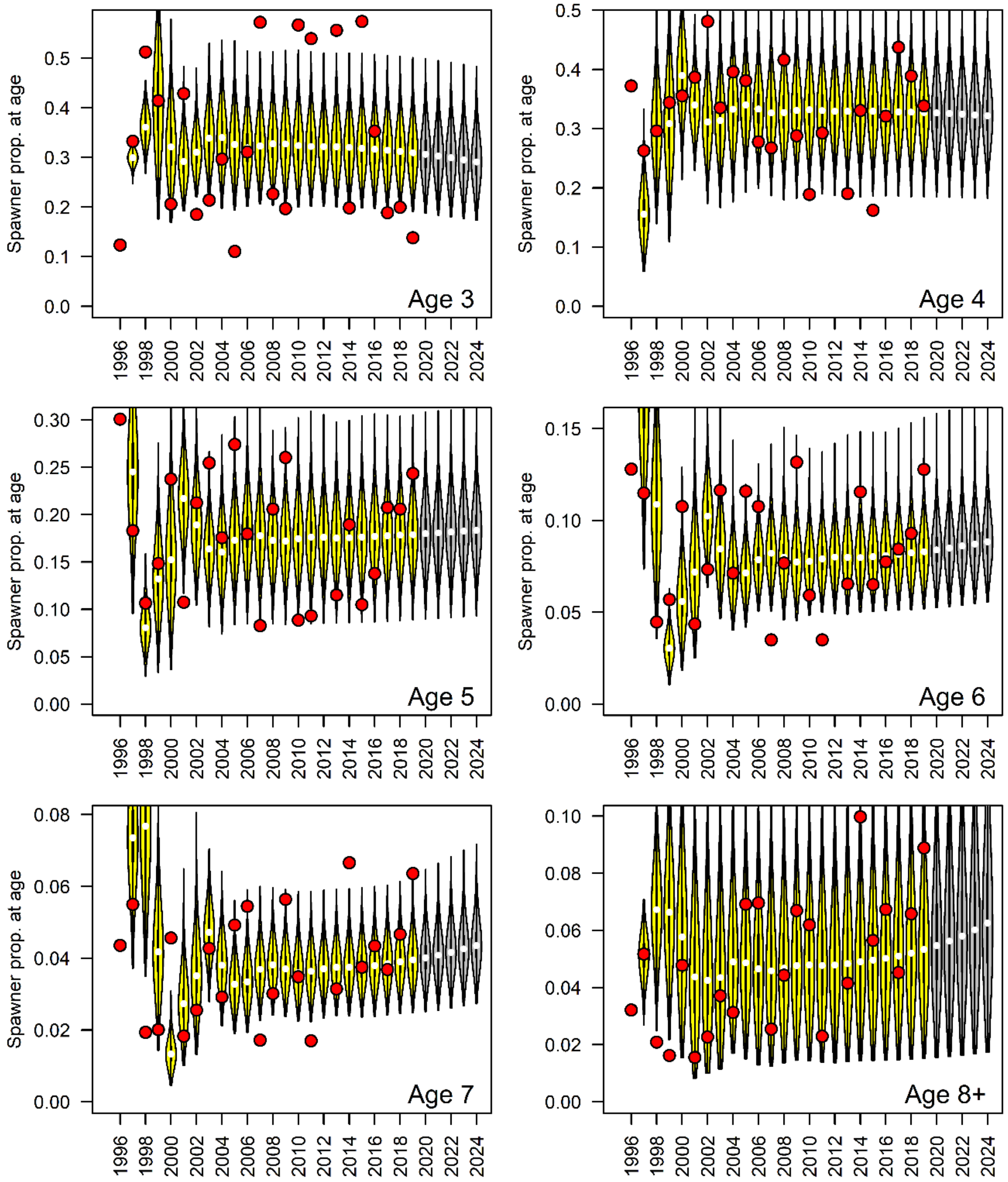


Figure A5.3 (continued). Observed (red symbols) vs predicted (yellow violin plots) proportions at age of spawners, for ages 3 to 7, and for ages 8+ of model 6.

Fisheries and Oceans Canada Pêches et Océans Canada
Science Sciences
Gulf Region
Canadian Science Advisory Secretariat
Science Advisory Report 2011/014

ALLOWABLE HARM ASSESSMENT OF STRIPED BASS (*MORONE SAXATILIS*) IN THE SOUTHERN GULF OF ST. LAWRENCE

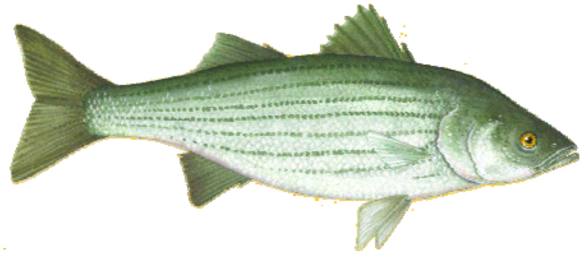


Illustration by Jeffrey C. Domm

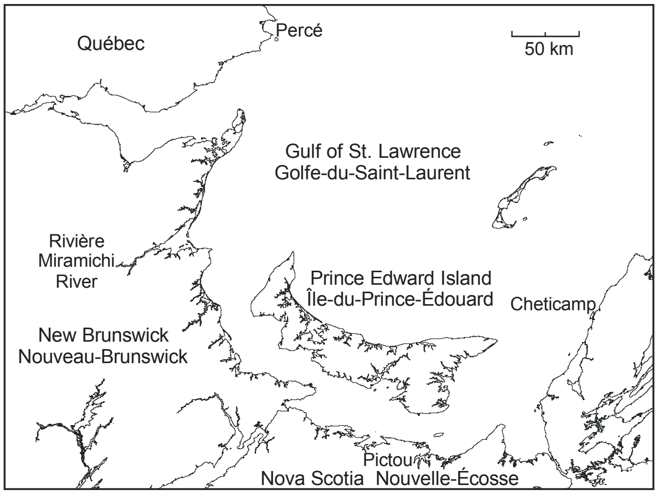


Figure 1: The southern Gulf of St. Lawrence.

Context :

*The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) identified three designatable units (DUs) of Striped Bass (*Morone saxatilis*) in eastern Canada with the southern Gulf of St. Lawrence DU assessed as 'Threatened' (COSEWIC 2004). The rationale for COSEWIC's listing recommendation of 'Threatened' for the southern Gulf of St. Lawrence DU was largely based on the population's single spawning location in the Northwest Miramichi estuary. The Recovery Potential Assessment (RPA) for Striped Bass evaluated whether the survival or recovery of the species was compromised by impacts of existing activities on the species (DFO 2006, Douglas et al. 2006). Due to a lack of quantitative information, the RPA for the southern Gulf of St. Lawrence DU did not provide any assessment of the mortality resulting from any specific fishery or other potential threat, but rather provided a qualitative description of the expected effects from many possible sources of mortality. The RPA concluded that illegal fishing and incidental mortalities in some fisheries constituted the most important constraints to the recovery of Striped Bass in the southern Gulf of St. Lawrence.*

Supplementary advice to the 2006 RPA was requested by the Species at Risk Secretariat (Gulf Region) and by the Ecosystems and Fisheries Management Branch of Fisheries and Oceans Canada (DFO EFM) to identify those activities which could be authorized by permit as defined by Section 73 or those activities to which the prohibitions of the Species at Risk Act (SARA) could be excluded as defined by Section 83(4). This advice will be considered as complementary advice to that provided in the 2006 RPA (DFO 2006).

A DFO Regional Advisory Process meeting was held February 4, 2011 in Moncton (NB) to respond to the request for advice on allowable harm. Participants at the science peer review meeting included DFO scientists and fishery managers, DFO Aboriginal Fishery coordinators, DFO SARA staff, non-government external experts, Aboriginal peoples, a representative from the commercial fishing industry, and academia.

SUMMARY

- Abundance of Striped Bass returning to the Northwest Miramichi River to spawn increased during 2007 to 2010, the result of three good recruiting year classes which followed more than 10 years of weak year classes.
- The constricted length and age distributions of Striped Bass in the southern Gulf of St. Lawrence (sGSL) is consistent with the high levels of annual mortality (28% to 47%) estimated for this population.
- Results from a questionnaire circulated to all DFO Conservation and Protection detachments and sub-detachments in Gulf Region indicated that nearly 70% of all adult Striped Bass fishery-related losses are the result of illegal (55%) and recreational (14%) fisheries. Losses from striped bass bycatch in commercial fisheries for gaspereau, rainbow smelt, Atlantic herring, American shad, American eel and Atlantic silverside were less important.
- Responses from First Nations in Gulf Region with Food, Social, and Ceremonial (FSC) gillnet and trapnet fisheries for Atlantic salmon indicated that the losses of striped bass were less than 5% of the total fisheries related losses.
- The uncertainty around the estimates of striped bass mortality for each fishery is not quantifiable with the available information but is considered to be large.
- In the absence of any fishery related losses, there is a high probability (>75%) that striped bass in the southern Gulf of St. Lawrence will meet the recovery limit of 21,600 spawners by 2015 and remain above it into 2020.
- Striped bass losses associated with illegal fishing by themselves result in a low probability (<25%) of meeting the recovery limit.
- An allocation of 2,000 adult Striped Bass to First Nations in the sGSL was not considered additional mortality on the population as this level of bycatch mortality is currently being experienced in FSC fisheries for Atlantic salmon. By itself, this fishery allocation would result in a medium chance (25% to 75%) of meeting the recovery limit for the population.
- Under any commercial fisheries bycatch scenario, and assuming no illegal or recreational fishery losses, there is a medium probability (25% to 75%) of meeting the recovery limit by 2015 and being compliant into 2020.
- With the cumulative mortality from all sources remaining at current levels, there is no measurable chance that Striped Bass from the sGSL will meet and remain above the recovery objectives into 2020.

INTRODUCTION

The Recovery Potential Assessment (RPA) for sGSL Striped Bass evaluated whether the survival or recovery of the species was compromised by impacts of existing activities on the population (DFO 2006). The RPA also assessed whether the recovery potential could be improved by reducing mortality on Striped Bass, primarily for those mortalities associated with fishing. Due to a lack of quantitative information, the RPA did not provide any assessment of the mortality resulting from any specific fishery or other potential threat but rather provided a qualitative description of the expected effects from several possible sources of mortality. The RPA concluded that illegal fishing and incidental mortalities in some fisheries constituted the most important constraints to the recovery of Striped Bass in the sGSL.

The RPA indicated that, under existing conditions, the cumulative mortality on Striped Bass from all activities resulted in a small chance (<25%) of achieving the recovery objective within ten years. The chance of recovery within ten years would be much higher if the mortality on incidentally captured adults and young-of-the-year (YOY) Striped Bass was reduced. The RPA provided some examples of measures to be considered to reduce these mortalities (DFO 2006).

The RPA did not provide any assessment of the activities which could be authorized if it was determined that they did not jeopardize the survival or recovery of the species. Complementary advice to that provided in 2006 is required for the following:

- To the extent possible, assess the impacts of activities described in Table 4 of the RPA (DFO 2006) on the survival and recovery of Striped Bass from the sGSL.
- To the extent possible and relative to objectives defined by management, assess the impacts of the activities on the survival and recovery objective trajectories.
- For the assessed activities above, determine if the probability of survival and the trajectory of the population can be improved by the application of mitigation measures and assess any residual effects.
- Based on the analyses above and relative to criteria defined by management (recovery objective, timeline), define those activities which would not jeopardize the survival or recovery of Striped Bass from the sGSL and therefore whose incidental effects could be authorized or exempted under the SARA. Also identify those activities whose effects are considered negligible.

The increased abundance of Striped Bass in the sGSL in recent years has renewed the interest of First Nation communities and the public for access to the resource. The interest is related to aboriginal FSC fisheries, to increased angling opportunities, and to the perceived fear that the increased abundance of Striped Bass will have (or is having) a negative impact on other species, in particular out-migrating Atlantic salmon smolts in the Miramichi River. Consequently, additional advice was requested for the following:

- Assess specifically the consequences to the population trajectory of annual FSC allocations to aboriginal fisheries of 2,000 Striped Bass.
- What additional mortality would be permitted on Striped Bass other than current bycatch mortality in FSC aboriginal and other fisheries?
- What size or sizes of Striped Bass could be retained with minimal impact on recovery?
- What is the mortality of Striped Bass, by size, from hook and release angling with a fly or with a fly with a barbless hook?

ASSESSMENT

Biology and Status of Southern Gulf of St. Lawrence Striped Bass

Striped Bass is widely distributed throughout the estuaries and coastal waters of the sGSL and exists at the northern limit of the species distribution. Based on tagging studies and catches from various fisheries, the coastal areas from Percé Quebec in the northwest, to Chéticamp, Cape Breton, in the east, and to Prince Edward Island in the north, represent the extent of occurrence of Striped Bass in the sGSL (Fig. 1). This population is geographically isolated within the sGSL and distinct from any other Striped Bass population, including those in the U.S.A. and the Bay of Fundy. Anadromous Striped Bass leave wintering areas in spring and return to estuaries where spawning occurs at the upper extent of the salt wedge. The Northwest Miramichi estuary remains the only known spawning location for Striped Bass in the sGSL and the spring spawning migration to this river is annually predictable in time and space. Female Striped Bass are highly fecund averaging 50,000 eggs per kg of body weight (over half a million eggs for larger females) and can reach lengths exceeding 1 m and ages older than 15 years in the sGSL.

Striped Bass spawning occurs in late May and early June in the upper portions of the Northwest Miramichi estuary when water temperatures exceed 10^oC. Fertilized eggs remain in suspension until hatching occurs in 2 to 3 days depending on the temperature of the water. Larvae move to the near-shore habitats of the estuary where they grow rapidly and metamorphose into the adult body form by early July. YOY Striped Bass progress downstream and into salt water over the course of the summer months and spread along the coast both northwest and southeast of the Miramichi system, reaching as far south and east as Pictou, NS, by early fall. Striped Bass of all ages return to estuaries to overwinter and cease feeding when water temperatures fall below 10^oC.

Management and Recovery Objectives

Striped Bass in the sGSL is managed as a single unit. Efforts to rebuild the low spawner abundance during the mid 1990s included the introduction of restrictive management measures, most notably the closure of directed commercial fishing in 1996, and the closure of recreational and aboriginal FSC fisheries in 2000. There has been no change in the management regime for the species since these closures took effect.

The RPA for Striped Bass in the sGSL proposed a recovery limit and compliance rule of 21,600 spawners in 5 of 6 years (DFO 2006). Douglas et al. (2006) further proposed that once the recovery limit was met, achieving an increased level of 31,200 spawners in 3 of 6 years could be a recovery target to consider for managing access to the resource.

Current Status

Levels of spawner abundance in the last 6 years (2005-2010) have not been sufficient to meet the recovery limit and respect the compliance rule (Fig. 2). A complete estimate of spawners was not possible in 2010 but spawner abundance was considered to have been sufficient to satisfy the recovery limit level of 21,600 spawners. If the lower confidence limit of the number of spawners returning to the Northwest Miramichi in 2011 exceeds 31,200 fish, the recovery limit and recovery target will have both been met.

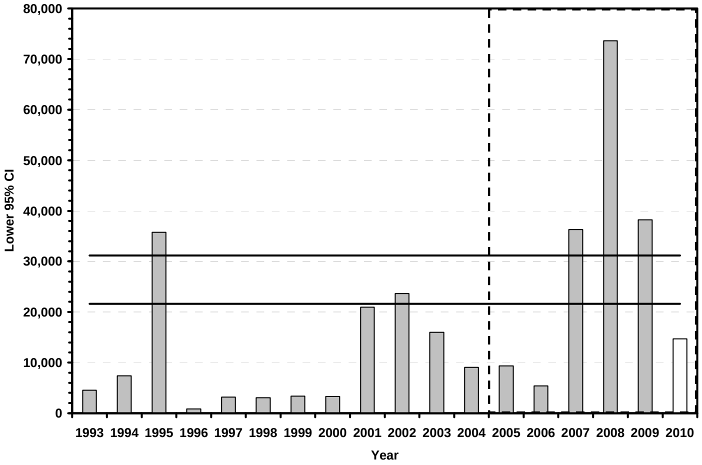


***Figure 2**. The lower confidence limit (95% CL) of the spawner abundance estimates for Striped Bass from the Northwest Miramichi relative to the recovery limit of 21,600 spawners (bottom horizontal line) and recovery target of 31,200 spawners (top horizontal line). The hatched box represents the 6 year sliding window for evaluation of compliance.*

There has been some relatively strong recruitment of Striped Bass in the last decade, specifically the year classes of 2003, 2004, 2005, and 2007 have been well represented. The 2004 year class was the first strong year class since the 1991 year class; all the year classes between 1991 and 2003 were weak (Fig. 3). Contributions to the spawning stock of Striped Bass aged 5 to 7 have improved since 2003 but remain low relative to younger year classes.

The increased abundance of Striped Bass spawners in 2007 to 2010 is the result of three good recruiting year classes, which follow more than ten years of weak year classes. Such variation in yearclass strength has also been reported in Striped Bass populations in the eastern US (Goodyear 1985).

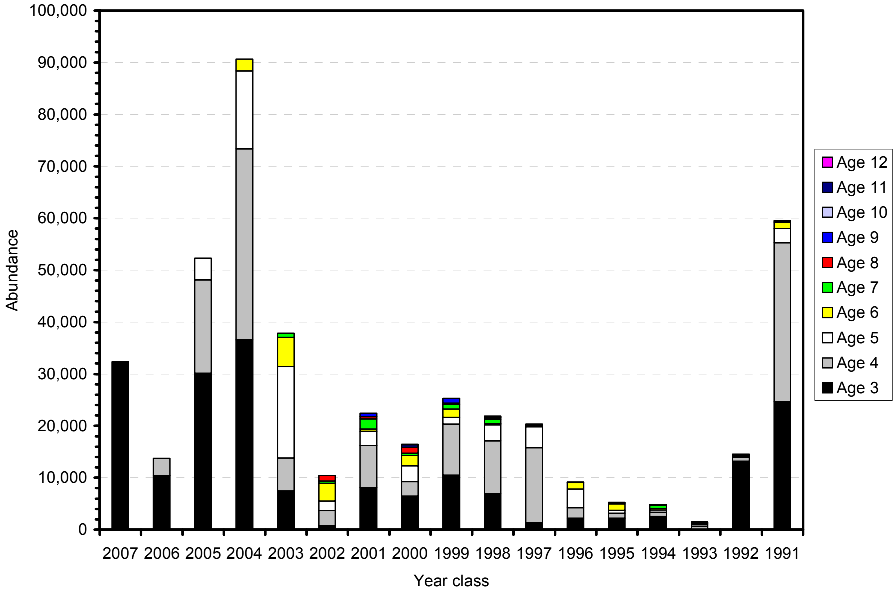


***Figure 3:** Striped Bass abundance at age by year class and total contribution to the spawning stock for year classes 1991 to 2007.*

Sources of Mortality on Southern Gulf of St. Lawrence Striped Bass

Based on the average abundance at ages 3 to 9 years over the period 1997 to 2010, the annual mortality rate of adult Striped Bass is about 47% (Z = 0.63), values marginally lower than the estimates of 50% to 60% (Z = 0.8 - 0.9) previously calculated for this population (Douglas et al. 2006). Estimates of annual mortality of Striped Bass based on acoustic tagging programs are lower than estimates derived from the catch curve analysis. In five replicates, estimates of Z consistently ranged between 0.34 and 0.48, corresponding to annual total mortality rates of 28% to 38%. It is not clear why the estimates of Striped Bass mortality differ between the two methods.

Potential threats and their associated mortality to Striped Bass in the sGSL were described previously and qualitatively ranked as low, medium, or high (DFO 2006, Douglas et al. 2006). The threats discussed below include only activities known to have a direct negative impact on Striped Bass in the sGSL. No further information has emerged on the ‘Other potential sources of Striped Bass mortality/harm’ that were reported in the RPA and these will not be discussed further (specifically Fisheries impacts on habitat, Direct mortality under permit, Habitat alterations under permit, Ecotourism and recreation, Shipping, transport and noise, Fisheries on food supplies, Aquaculture, Military activities, and Non-domestic) (Douglas et al. 2006).

Levels of Striped Bass bycatch in many of the region’s estuarial fisheries have previously been ranked qualitatively by DFO Conservation and Protection officers (Chiasson et al. 2002). In 2010-2011, DFO’s Conservation and Protection Branch (22 responses from all 19 detachments and sub-detachments in coastal areas) were asked to respond to a questionnaire in order to estimate the level of Striped Bass bycatch in various fisheries, including illegal fisheries, throughout the sGSL. In addition, First Nation communities with access to Atlantic salmon in the Aboriginal Fisheries agreements (12 respondents from 5 communities) were asked to respond to the same questionnaire but only specifically to striped bass bycatch in their FSC Atlantic salmon gillnet and trapnet fisheries.

The responses were collected with a questionnaire which was modified after initial testing internally. The questionnaire asked respondents to semi-quantitatively rank (in broad catch categories of 0, <100, 101-1000, 1001-5000, 5001-10,000, >10,000), the level of bycatch (number of fish) released dead and released alive in various fisheries throughout the sGSL. Midpoints of the bycatch categories (except 0 was used for 0, and 10,000 was used for >10,000) were used to derive total estimates of Striped Bass handled, released, and killed in the various fisheries. Individuals were also asked to differentiate between the sizes of captured Striped Bass as small (< 12 inches), medium (12-24 inches) or large (>24 inches).

Striped Bass mortality as a proportion of the total number of bass handled was used to estimate the impact of the bycatch activity in a fishery. The estimated mortality of Striped Bass in a fishery divided by the total mortality from all fisheries was used to describe the proportion of the total fishery induced mortality attributed to a specific fishery.

Estimated Losses Among Fisheries

Due to the type of data collected and the calculation method, the point estimates for Striped Bass mortality in the various fisheries reported below provide an estimated scale of impact but should not be considered absolute values. The uncertainty around the estimates is not quantifiable with the available information but is considered to be large.

Striped Bass of various life stages continue to be intercepted in a variety of illegal fisheries, commercial fisheries, and aboriginal FSC fisheries. The total estimated loss of medium and large sized Striped Bass in all southern Gulf of St. Lawrence fisheries was estimated to be in the range of 60,000 fish per year. It was estimated that 90,000 fish were handled and released alive. The total number of bass handled was estimated to be 152,000 fish, of which 41% were estimated to have died or been killed (retained) (Table 1). Loss estimates are not considered to include post-release mortality of fish that were angled and released. These values are high considering the estimates of spawner abundance between 2007 and 2010. The numbers reported in the questionnaires most likely apply to recent years when Striped Bass abundance has increased. Removal levels estimated from the questionnaire would not have been realistic in the mid 1990s and early 2000s when spawner abundance was less than 20,000 fish annually (Fig. 2).

The fisheries that occur with gillnets have the highest mortality rates on Striped Bass. All (100%) of the Striped Bass captured in the illegal fisheries are considered to be dead. Over 80% and 65% of Striped Bass handled in the gillnet fisheries for American shad and Atlantic herring, respectively, were estimated to have died (Table 1). About 60% of the Striped Bass captured in FSC fisheries for Atlantic salmon are considered to be dead.

The activity with the greatest contribution to the total loss of Striped Bass is considered to be the illegal fishery, accounting for over 50% of the estimated adult losses, followed by the recreational fishery (illegal retention and bycatch) at about 15% (Table 1). The Atlantic silverside and American eel fisheries were estimated to contribute the least to the total losses of adult Striped Bass in the sGSL.

***Table 1**. Summary of estimated losses of medium and large sized striped bass in fisheries of the southern Gulf of St. Lawrence. All values have been rounded to the nearest 100.*

	Total			Percentage	Percentage of
Fishery	**Released**	**Dead**	**Handled**	**killed**	**total killed**
Atlantic silverside	400	0	400	0.0%	0.0%
American eel	15,500	1,300	16,800	7.7%	2.1%
FSC salmon	1,200	1,900	3,100	61.3%	3.1%
American shad	500	2,500	3,000	83.3%	4.1%
Rainbow smelt	12,900	3,900	16,800	23.2%	6.3%
Atlantic herring	2,300	4,500	6,800	66.2%	7.3%
Gaspereau	37,900	4,800	42,700	11.2%	7.8%
Recreational	19,600	8,900	28,500	31.2%	14.4%
Illegal	0	33,900	33,900	100.0%	54.9%
Total	90,300	61,700	152,000	40.6%	

Projections and Probabilities of Stock Recovery

The life history model used in the RPA for Striped Bass in the sGSL was used to assess the probability of achieving the recovery limit (21,600 spawners) by 2015 and being compliant into 2020 for various fishing scenarios. The inputs were the same as those used previously and included assumed values for fishing and natural mortality: F=0.1 for YOY bycatch in the autumn rainbow smelt fishery, F=0.2 on adult bass (ages 2 and older) and M = 0.6 (ages 2 and older) (Douglas et al. 2006). Given the level of uncertainty in many of the life history model parameters, the outcomes of the projections are evaluated against the recovery objectives and the results are presented as low probability (<25%), medium probability (25% to 75%) and high probability (> 75%) of meeting the recovery objective.

In the absence of any fishing related mortality (M = 0.6 on adults), the probability of meeting the recovery limit of 21,600 fish by 2015 and respecting the compliance rule into 2020 is high.

Table 2. *Assessment of the probability (low <25%; medium 25% to 75%; high >75%) of meeting the recovery limit of 21,600 Striped Bass spawners by 2015 and being compliant into 2020, for various fishery scenarios in the southern Gulf of St. Lawrence.*

Activity		**Prob. of recovery**
No fishing related losses		High
Only illegal fishing losses		Low
FSC only (2,000 adult bass)		Medium
FSC (as above) and	Atlantic silverside fishery	Medium
	American eel fishery	Medium
	American shad fishery	Medium
	Rainbow smelt fishery	Medium
	Atlantic herring fishery	Medium
	Gaspereau fishery	Medium
	All coastal and estuarine commercial fisheries.	Medium
	Recreational fishery	Medium
All fisheries - FSC, commercial, recreational, excluding illegal		Low
All fisheries - FSC, commercial, recreational, and illegal (status quo)		Low (~0%)

Fishery specific effects

Over 50% of adult Striped Bass losses were estimated to occur in **illegal fisheries** throughout the sGSL. In the absence of any other fishery related losses, this activity results in a low probability of meeting the recovery limit by 2015.

Food, social, and ceremonial fisheries presently contribute to Striped Bass mortality on adult sized fish. The losses occur as a result of bycatch in fisheries targeting Atlantic salmon. These fisheries are limited geographically but can intercept several thousand Striped Bass with corresponding high mortality on individual fish. First Nation estimates of current annual Striped Bass losses in FSC fisheries for Atlantic salmon were greater than 2,000 fish (Table 1). An FSC fishery of 2,000 adults, in the absence of any other fishing related mortality on any life stage (and assuming M = 0.6 for age 2 and older), would result in a medium chance of meeting the recovery limit by 2015. FSC allocations greater than 2,000 fish will further reduce the chance of meeting and respecting the compliance rule into 2020. The removal of 2,000 Striped Bass for an FSC allocation in the sGSL is not considered to be an additional mortality on Striped Bass as this level of mortality already occurs as bycatch in gillnets set for Atlantic salmon. Alternative gear to gillnets, such as trapnets or angling, are options for conducting these fisheries which would allow for selective harvesting of salmon or Striped Bass.

There are two fisheries which are expected to have a low impact on Striped Bass abundance. The **Atlantic silverside** fishery, as it is presently carried out, mostly in the eastern counties of Prince Edward Island, likely has a low to negligible impact in terms of absolute number of Striped Bass lost. The consequences of this fishery are that there is a medium probability of recovery.

Commercial fisheries for **American eel** exist throughout the region and are carried out with fyke nets and spears. Little quantitative information is known about the bycatch in eel fisheries but all sizes of Striped Bass have the potential of being intercepted albeit in low numbers (Bradford et al. 1995). The eel fishery, practiced with fyke nets and with appropriate culling practices can return bycatch to the water alive, and should have a low impact on Striped Bass. The eel fishery was estimated to contribute about 2% of the total losses of adult Striped Bass and there is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2). The spear fishery for American eel is not expected to have any impact on Striped Bass.

The **gaspereau** fishery has a high potential of intercepting Striped Bass (Douglas et al. 2006). This fishery occurs in many estuaries of the Gulf Region and can intercept large numbers of Striped Bass, mostly aged 2 and older. Results from the questionnaire indicated that losses of Striped Bass in the gaspereau fishery represented 8% of the losses from all fisheries (Table 1) and there is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2). The type of gear used (trapnet), the time of year of the fishery (late spring, early summer), and the possibility of adopting effective culling practices to return bycatch with minimal harm to the water should result in a low impact on Striped Bass. Fishers in the NW Miramichi have already adopted efficient culling practices (single dipnet scoops, immediate culling) for Striped Bass which produces minimal handling mortality. Similar practices could be implemented in other areas of the sGSL.

The Atlantic herring and American shad gillnet fisheries are carried out along the coast and have the potential of intercepting Striped Bass. Gillnet mesh sizes ($2^{1/4}$ to $2^{7/8}$ inches; 57 to 73 mm) used in the **Atlantic herring** fishery should not be very effective at capturing adult-sized Striped Bass. Losses of Striped Bass in the herring fishery were estimated to be 7% of the total losses (Table 1) and there is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2). The **American shad** fishery deploys set gillnets with a minimum mesh size of 127 mm in an area where there is a high probability of intercepting Striped Bass. According to reports from shad fishers and C&P, the Striped Bass bycatch in this fishery can be in the hundreds of fish. The estimated loss of Striped Bass from this fishery was 2,500 fish annually and represented 4% of the total annual losses and there is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).

The open-water **rainbow smelt** fishery with boxnets, bagnets, and gillnets is the fishery with the largest impact on YOY Striped Bass. This fishery is geographically widespread in the sGSL and YOY are captured in most estuaries where the fishery occurs. The number of YOY killed in this fishery would be proportional to the abundance, and in some cases, can result in losses equivalent to thousands of adults. There is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2). The only quantitative assessment of Striped Bass bycatch in any commercial smelt fishery is from the Miramichi River during the open-water season. During a two year study, 100,000 to 400,000 YOY, and depending on the year, >1000 each of Age 1 and Age ≥2 fish were captured in Miramichi box nets (Bradford et al. 1997). Even though the catch in this fishery is loaded alive, the mortality on YOY Striped Bass can be high (almost 100%) and is largely attributed to the difficulty of culling large numbers of small bass from smelt catches. The magnitude of the Striped Bass bycatch in smelt boxnets and other fisheries elsewhere in the sGSL is not well documented but known to occur and qualified as substantial at times (Chiasson et al. 2002). The loss of 1,000 YOY equates to the loss of about 100 adult bass over the lifespan of the animal (life history model assumptions). Some of the YOY captured in the smelt fishery in the fall may not survive the winter due to their small body size but the smelt fishery is not selective for size; small and large YOY are captured in the nets. A two-week delay to Nov. 1 in the opening date of the open-water smelt fishery in Miramichi Bay was introduced in 1999 to reduce the high bycatch of YOY Striped Bass. The number of Striped Bass captured in the winter component of the smelt fishery is low and less than the open-water component because YOY avoid the relatively cold and saline waters of the lower estuary where the smelt fishery is concentrated (Hanson and Courtenay 1995). YOY Striped Bass that are captured in the winter fishery generally die when they are removed from the water and placed on the ice.

Tidal and marine recreational fisheries occur along the entire coast of the southern Gulf of St. Lawrence. Striped Bass are targeted, released or retained, but under the guise of angling for a different species with a legal season and bag limit (trout in estuaries, mackerel from wharves and coasts, etc.). There are large numbers of people angling and Striped Bass are extremely vulnerable to these fisheries. DFO C&P estimated that 9,000 Striped Bass were killed and 20,000 were released alive throughout the sGSL, representing over 14% of the total mortality from all fisheries bycatch combined with the result that there is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).

Striped Bass fishing with barbed or barbless artificial flies under the guise of fishing for other species is not a common practice in the sGSL. Striped Bass are intercepted in angling fisheries that occur in estuaries and coastal waters, with spin casting and surf casting equipment. Terminal tackle for spin and surf casting equipment tends to be jigs and lures in estuarine environments while natural bait is the norm for coastal fisheries. There is no information on mortality rates for Striped Bass hooked and then released in the sGSL. On the eastern seaboard of the U.S.A., hooking mortality estimates range from a low of 3% to a high of 74% and vary considerably depending on the type of tackle used and the water conditions (Millard et al. 2005).

Cumulative effects of bycatch losses in FSC fisheries and in commercial estuarine and coastal fisheries results in a medium probability of meeting the recovery limit by 2015 (Table 2). When recreational fishery losses are added, the probability of meeting the recovery limit is reduced to low (Table 2). Under what are considered status quo conditions (F = 0.1 on YOY and age 1, F = 0.2 for Striped Bass age 2 and older in all other fisheries including illegal fisheries, and M = 0.6 on age 2+), there is no measurable chance (0%) of meeting the recovery limit and complying with the rule into 2020.

Uncertainties and Knowledge Gaps

Losses of Striped Bass in recent years from incidental bycatch and handling mortality, and from illegal retention fisheries, have not been directly measured in the sGSL. Questionnaires to C&P personnel indicated that the losses from fishing are in the tens of thousands of Striped Bass per year. The values reported in the 2010 questionnaire may be more representative of the losses in the recent years (past three years) when the overall spawning stock of Striped Bass was estimated to be between 50,000 and 100,000 animals annually. Even so, the losses and fish handled would represent a very high proportion of the estimated spawner abundance, which may in part explain the high mortality rate estimated from catch curve analyses (Z = 0.63 in the recent assessment).

The design, delivery, and results of the questionnaire are presently under review by an external expert. A number of issues were raised regarding the design and how the data were analysed, including:

- Generally, the total number of respondents was low and the representativeness of the results are unknown. For the DFO C&P respondents however, the questionnaires were completed by detachments throughout Gulf Region, were based on the experience and opinions of several officers, and the information was considered to be reliable and representative of the situation.

- Only the DFO C&P information for fisheries other than FSC and the aboriginal community information specific to the FSC fishery were used to estimate the bycatch of Striped Bass. The few responses from commercial fishermen (n = 11) were not used in this analysis, but it may be informative to compare their responses with those from DFO C&P to help evaluate the reliability of the results.
- There are responses in the questionnaire which contrast to findings from other studies. For example, there are low numbers of small bass reported captured in the smelt fisheries in contrast to previous studies indicating that YOY bass are frequent bycatch in the open-water smelt fisheries. It is possible that respondents did not consider YOY bass within the small fish category, due to the lack of a clear definition of what was considered small Striped Bass.
- Definitions of recreational fisheries or illegal fisheries were not clear.
- The use of midpoints in the categories and the use of a single number for the high and low categories is not standard practice in the analysis of ordinal data.

The population model used to determine reference levels for sGSL striped bass assumed or borrowed features from other populations (Douglas et al. 2006). The same model was used to assess the probability of meeting the recovery objectives under various scenarios of Striped Bass bycatch mortality in FSC, commercial, recreational tidal and illegal fisheries.

In order to adequately assess consequences of fishing activities on recovery potential, a partitioning of total mortality (Z) into natural (M) and fishing related (F) mortality is required. The constricted age distribution of adult Striped Bass observed in the Northwest Miramichi is indicative of some or all of the following: high natural mortality on adult size animals (in the range of M = 0.6), lower natural mortality but high fishing mortality, or lower natural mortality but higher size-dependent fishing mortality on older ages. The most recent catch curve estimate of Z = 0.63 is lower than the value previously reported by Douglas et al. (2006). If Z is 0.63, then M should be less than the value of 0.6 assumed in the life history model for the RPA (Douglas et al. 2006). At the level of reported losses of Striped Bass estimated over all fisheries, M should be much lower than the assumed value of 0.6. It is not simply a matter of changing the value of M in the life history model and running it to evaluate the probability of meeting the defined objectives in the RPA (DFO 2006). If M is in fact lower, then the recovery limits and targets would be higher than those identified in the RPA and which were based on M = 0.6.

Recent acoustic tracking data indicates that the total loss (Z) for Striped Bass implanted with a sonic transmitter in one year that returned to the Miramichi to spawn in the following year is in the range of 0.3 to 0.4. Knowing that some of these losses of tagged fish occurred in fisheries, M is therefore less than 0.4 and a value of 0.2 as assumed in other parts of the species range (eastern USA) may not be unreasonable. If M is in fact this low, then mortality from other factors, primarily fishing, must be high to account for the absence of old fish and the constricted age structure of the spawning population.

CONCLUSIONS AND ADVICE

To the extent possible, assess the impacts of activities described in Table 4 of the 2006 RPA on the survival and recovery of Striped Bass from the southern Gulf.

- No further information was presented on the ‘Other potential sources of Striped Bass mortality/harm’ that were reported in the RPA (specifically Fisheries impacts on habitat, Direct mortality under permit, Habitat alterations under permit, Ecotourism and recreation, Shipping, transport and noise, Fisheries on food supplies, Aquaculture, Military activities, and Non-domestic) (Douglas et al. 2006). The threats examined in this review included only activities known to have a direct negative impact on Striped Bass in the sGSL. Results from the questionnaire circulated to all DFO Conservation and Protection detachments and sub-detachments in Gulf Region and to First Nations in Gulf Region indicated that nearly 70% of adult Striped Bass losses from fishing activities occur in illegal (55%) and recreational (14%) fisheries. In order of importance, the remainder of the losses are in the commercial fisheries for gaspereau, rainbow smelt, Atlantic herring, American shad, American eel and Atlantic silverside fisheries of the sGSL. Responses from First Nations in Gulf Region with Food, Social, and Ceremonial (FSC) gillnet and trapnet fisheries for Atlantic salmon indicated that the losses of striped bass were less than 5% of the total fisheries related losses. Mortality from all other potential sources is considered minimal relative to those caused by fisheries.

To the extent possible and relative to objectives defined by management, assess the impacts of the activities on the survival and recovery objective trajectories.

- Based on the assumptions of the life history model used to define recovery objectives for Striped Bass in the sGSL (DFO 2006), there is no measurable chance of achieving the recovery objectives at the current levels of total mortality experienced by the population. Strong year classes have occurred periodically (1991 followed by the 2004 year class) and these may produce sufficient abundance of spawners to occasionally achieve the defined recovery objectives. This scenario is not expected to occur frequently given the biology of sGSL Striped Bass and the high number of adult removals which continue to occur on an annual basis.

For the assessed activities above, determine if the probability of survival and the trajectory of the population can be improved by the application of mitigation measures and assess any residual effects.

- In the absence of any fishing related mortality and assuming natural mortality of adults at $M = 0.6$, the probability of meeting the recovery limit of 21,600 fish by 2015 and respecting the compliance rule into 2020 is considered to be high. The probability of meeting the recovery limit and the recovery target by 2015 and respecting the compliance rule into 2020 is medium.

Based on the analyses above and relative to criteria defined by management (recovery objective, timeline), define those activities which would not jeopardize the survival or recovery of Striped Bass from the southern Gulf and therefore whose incidental effects could be authorized or exempted under the SARA. Also identify those activities whose effects are considered negligible.

- The Atlantic silverside fishery and the American eel fisheries, as presently carried out, likely have a low to negligible impact on Striped Bass. There is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).
- The gaspereau trapnet fishery has a high potential of intercepting Striped Bass (the fishery occurs in many estuaries of the southern Gulf) and can intercept large numbers of Striped Bass, but the type of gear used (trapnet), the time of year of the fishery (late spring, early summer), and the possibility of adopting effective culling practices to return bycatch with minimal harm to the water should result in a low impact on Striped Bass. There is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).
- Aboriginal FSC fisheries presently contribute to Striped Bass mortality on adult sized fish as a result of bycatch in gillnet fisheries targeting Atlantic salmon. These fisheries are limited geographically but can intercept several thousand Striped Bass with corresponding high mortality on individual fish. Alternative gear to gillnets, such as trapnets or angling, are options for conducting these fisheries which would allow for selective harvesting of salmon or Striped Bass. There is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).
- Atlantic herring gillnet fisheries were reported to capture important numbers of Striped Bass and to have a high mortality rate. Gillnet mesh sizes ($2^{1/4}$ to $2^{7/8}$ inches; 57 to 73 mm) used in the Atlantic herring fishery should not be effective at capturing the larger adult-sized Striped Bass. The extent of the losses associated with this bait fishery in both the spring and fall is unknown. There is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).
- The limited American shad fishery deploys set gillnets with a minimum mesh size of 127 mm in an area where there is a high probability of intercepting Striped Bass. There are presently 10 licences fishing in the Pointe Sapin area in the spring. Considering the limited amount of gear, the single location and short season, this fishery has a relatively large impact on Striped Bass. According to reports from shad fishers, the Striped Bass bycatch in this fishery can be in the hundreds of fish with estimated losses of 2,500 fish annually. There is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).
- The open-water rainbow smelt fishery with boxnets, bagnets, and gillnets has the largest impact on YOY Striped Bass. This fishery is geographically widespread in the sGSL and YOY are captured in most estuaries where the fishery occurs. The mortality on YOY Striped Bass can be high (almost 100%) and is largely attributed to the difficulty of rapidly culling large numbers of small bass from smelt catches. The number of YOY killed in this fishery would be proportional to the abundance, and in some cases, can result in losses equivalent to thousands of adults annually. There are no proposed mitigation measures that could reduce the bycatch of YOY Striped Bass without negatively affecting the catch of the targeted species. The abundance of Striped Bass captured in the winter component of the smelt fishery is low because YOY avoid the relatively cold and saline waters of the lower estuary where the smelt fishery is concentrated in the winter. There is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).
- Tidal and marine recreational fisheries occur along the entire coast of the sGSL and are practiced by large numbers of people fishing for Striped Bass under the guise of fishing for other fish species. Striped Bass are intercepted in these angling fisheries with spin casting and surf casting equipment. There is no information on mortality rates for Striped Bass hooked and then released in the sGSL with any angling gears. On the eastern seaboard of the U.S.A., hooking mortality estimates range from a low of 3% to a high of 74% and vary considerably depending on the type of tackle used and the water conditions. It was estimated that 14% of the total mortality of Striped Bass from all fisheries was the result of recreational fisheries. There is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).

Assess specifically the consequences to the population trajectory of a potential FSC allocation of 2,000 Striped Bass to First Nations in the sGSL.

- More than 2,000 adult Striped Bass are currently being intercepted and killed in aboriginal gillnet fisheries that target Atlantic salmon for FSC purposes. An allocation of Striped Bass at this level (2,000) is not considered to represent additional mortality on the population and there is a medium probability of meeting the recovery objective at this level of fishery related loss (Table 2).

What additional mortality would be permitted on Striped Bass other than current bycatch mortality in food, social, and ceremonial aboriginal and other fisheries?

- Nearly 70% of adult Striped Bass losses were estimated to occur in illegal (55%) and recreational (14%) fisheries throughout the sGSL. Illegal fisheries are those where Striped Bass are targeted and retained. In recreational fisheries, Striped Bass are targeted, released or retained, but under the guise of angling for a different species with a legal season and bag limit (trout in estuaries, mackerel from wharves and coasts, etc.). Any additional authorized mortality would further reduce the chance of achieving and remaining above the recovery objectives. A legalized hook and release angling fishery on Striped Bass for example would represent an additional source of mortality on the population as there would be increased participation by anglers currently abstaining from the illegal recreational fishery on Striped Bass.

What size or sizes of striped bass could be retained with minimal impact on recovery?

- The number of eggs per fish increases with size so larger females contribute substantially more eggs than younger and smaller females. Striped Bass recruitment is annually variable due mostly to the variations in survival of the early life stages (egg to summer YOY). As a result, there is benefit in maintaining a broad age distribution in the spawning population so that the spawning potential is maintained and distributed over a number of spawning years to take advantage of favourable environmental conditions that can produce good year classes of recruitment. Allowing a potentially high harvest rate on fish larger than a minimum length could result in further constricting the age structure in the spawning population with low numbers of older animals. Managing on a slot size or a maximum size limit, would result in high exploitation rates on a specific age component, reducing the abundance of older animals once they grow through the slot or above the limit. Neither of these measures is considered good practice for a species whose life history (high fecundity, multiple years of spawning, old age) suggests that persistence depends upon the presence of a sufficient spawning stock to take advantage of infrequent favourable environmental conditions that produce strong year classes. The use of a minimum size limit, a maximum size limit, or a slot limit may not be effective measures to manage the exploitation and sustain the spawning population of adult Striped Bass in the sGSL.

What is the mortality of Striped Bass, by size, from hooked and released by angling with a fly or with a fly with a barbless hook?

- Angling for Striped Bass with a barbed or barbless artificial fly is not a common practice in the sGSL, and hooking mortality by this means should be very low. Artificial baits such as jigs, lures, and single hooks with natural bait are the most common terminal tackle used to angle Striped Bass in the sGSL. There is no evidence that hooking mortality is related to the size of Striped Bass, however water temperature, salinity, and types of terminal tackle have all been identified as important factors influencing hooking mortality, which can vary from a low of 3% to a high of 74%.

OTHER CONSIDERATIONS

Nearly 70% of adult Striped Bass losses were estimated to occur in illegal (55%) and recreational (14%) fisheries throughout the sGSL. Illegal fisheries are those where Striped Bass are targeted and retained while in recreational fisheries, Striped Bass are targeted, released or retained, but under the guise of angling for a different species with a legal season and bag limit (trout in estuaries, mackerel from wharves and coasts, etc.). There is no easy solution to this problem.

Striped Bass in the sGSL spawn at a single location in the upper portion of the Northwest Miramichi estuary. Staging occurs first at the confluence of the Northwest and Southwest Miramichi rivers (locally known as Strawberry Marsh) during the month of May. The spawning period is brief and the majority of spent Striped Bass have returned to the coastal environment by mid June.

Striped bass in the sGSL are at the northern limit of the species distribution where environmental conditions may play an important role in the success or failure of a year class. Striped Bass eggs are highly sensitive to environmental conditions in the spring and young-of-the-year must attain a critical size by the end of the first growing season to survive the winter. Striped bass abundance in the sGSL is expected to be variable even in the absence of fishing related mortality.

Habitat quality on the spawning area should be improving as the two large pulp and paper mill operations in the Miramichi have recently closed down and are being dismantled, eliminating the treated effluent discharge in the area immediately below the spawning grounds in the Northwest Miramichi.

SOURCES OF INFORMATION

This Science Advisory Report has resulted from a Fisheries and Oceans Canada, Canadian Science Advisory Secretariat Gulf Regional Advisory Process of February 4, 2011 on Allowable Harm assessment for the striped bass population from the southern Gulf of St. Lawrence. Additional publications resulting from this process will be posted as they become available on the DFO Science Advisory Schedule at http://www.dfo-mpo.gc.ca/csas-sccs/index-eng.htm.

Bradford, R.G., Robichaud, K.A., and Courtenay, S.C. 1995. By-catch in commercial fisheries as an indicator and regulator of striped bass (*Morone saxatilis*) abundance in the Miramichi River estuary. *In* Water, science, and the public: the Miramichi ecosystem. *Edited by* E.M.P. Chadwick. Can. Spec. Publ. Fish. Aquat. Sci. 123: 249-259.

Bradford, R.G., Chaput, G., Hurlbut, T., and Morin, R. 1997. Bycatch of striped bass, white hake, winter flounder, and Atlantic tomcod in the autumn “open water” smelt fishery of the Miramichi River estuary. Can. Tech. Rep. Fish. Aquat. Sci. 2195: vi + 37 p.

Chiasson, G., Gallant, P.A., and Mallet, P. 2002. Traditional and local knowledge: estuarine fisheries by-catch in the southern Gulf of St. Lawrence; ecosystem based fisheries management considerations. Can. Man. Rep. Fish. Aqua. Sci. No. 2613: vi + 45 p.

COSEWIC 2004. COSEWIC assessment and status report on the Striped Bass *Morone saxatilis* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vii + 43 p.

DFO. 2006. Recovery assessment report for the St. Lawrence estuary, southern Gulf of St. Lawrence and Bay of Fundy striped bass (*Morone saxatilis*) populations. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2006/053: 21 p.

Douglas, S.G., Chaput, G., and Caissie, D. 2006. Assessment of status and recovery potential for striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence. DFO CSAS Res. Doc. 2006/041: viii + 95 p.

Goodyear, C.P. 1985. Toxic materials, fishing, and environmental variation: simulated effects on striped bass population trends. Trans. Am. Fish. Soc. 114: 107-113.

Hanson, J.M. and Courtenay, S.C. 1995. Seasonal abundance and distribution of fishes in the Miramichi estuary. *In* Water, science, and the public: the Miramichi ecosystem. *Edited by* E.M.P. Chadwick. Can. Spec. Publ. Fish. Aquat. Sci. 123: 141-160.

Millard, M.J., Mohler, J.W., Kahnle, A., and Cosman, A. 2005. Mortality associated with catch and release angling of striped bass in the Hudson River. North Amer. J. Fish. Manag. 25: 1533-1541.

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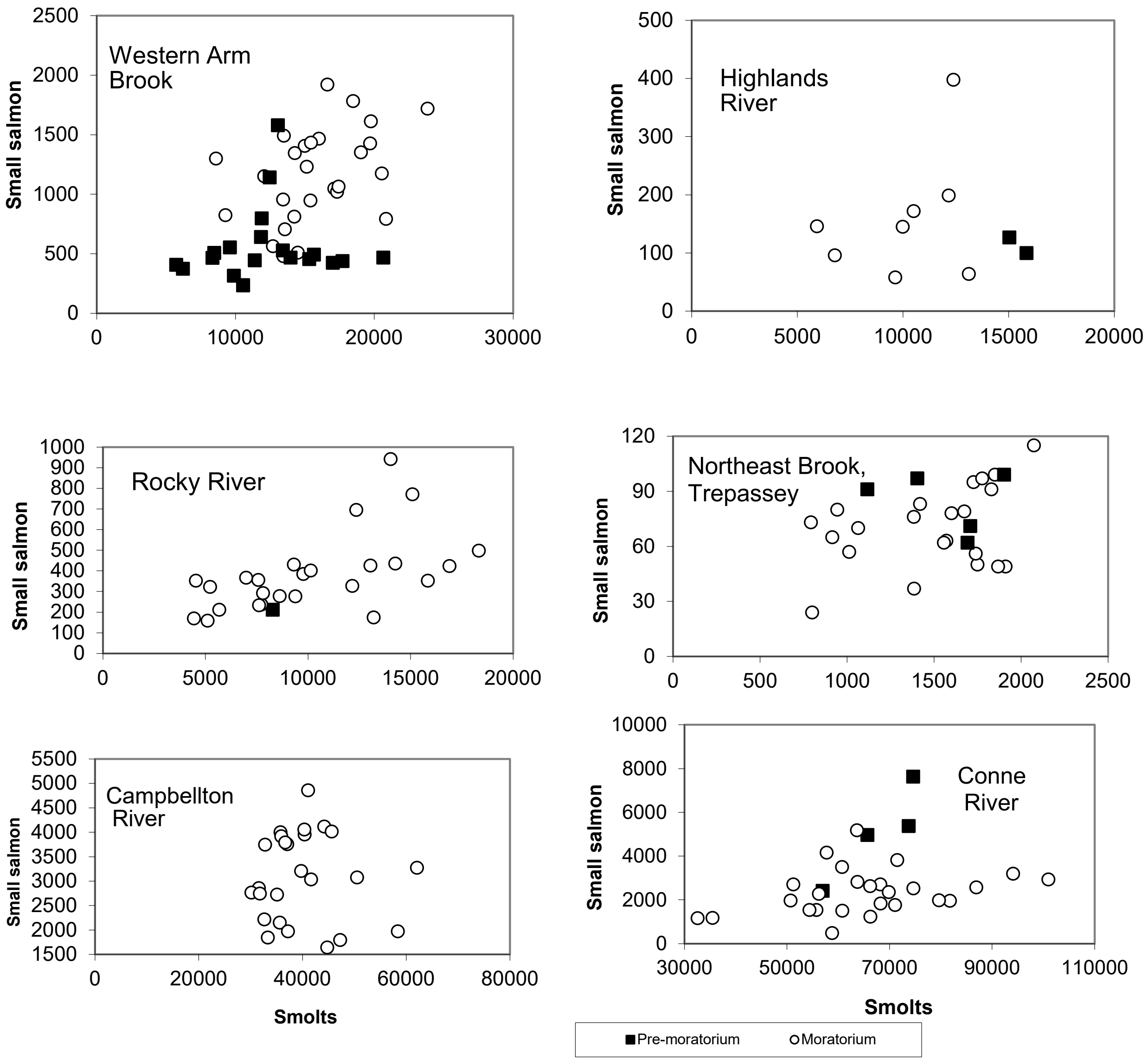
E-Mail: csas-sccs@dfo-mpo.gc.ca
Internet address: www.dfo-mpo.gc.ca/csas

ISSN 1919-5079 (Print)
ISSN 1919-5087 (Online)

La version française est disponible à l’adresse ci-dessus.

CORRECT CITATION FOR THIS PUBLICATION

DFO. 2011. Allowable harm assessment of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2011/014.


Western Arm Brook
Small salmon
0
500
1000
1500
2000
2500
0
10000
20000
30000
Highlands River
Small salmon
0
100
200
300
400
500
0
5000
10000
15000
20000
Rocky River
Small salmon
0
100
200
300
400
500
600
700
800
900
1000
0
5000
10000
15000
20000
Northeast Brook, Trepassey
Small salmon
0
30
60
90
120
0
500
1000
1500
2000
2500
Campbellton River
Small salmon
1500
2000
2500
3000
3500
4000
4500
5000
5500
0
20000
40000
60000
80000
Smolts
Conne River
Small salmon
0
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Smolts
■ Pre-moratorium
○ Moratorium

Fisheries and Oceans Canada
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Canadian Science Advisory Secretariat (CSAS)

Research Document 2016/032

Maritimes Region

Recovery Potential Assessment for Outer Bay of Fundy Atlantic Salmon (*Salmo salar*): Population Dynamics and Viability

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June 2016

Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

Research documents are produced in the official language in which they are provided to the Secretariat.

Published by:

Fisheries and Oceans Canada
Canadian Science Advisory Secretariat
200 Kent Street
Ottawa ON K1A 0E6

http://www.dfo-mpo.gc.ca/csas-sccs/
csas-sccs@dfo-mpo.gc.ca

ISSN 1919-5044

Correct citation for this publication:

Gibson, A.J.F., Jones, R.A. and MacAskill, G.J. 2016. Recovery Potential Assessment for Outer Bay of Fundy Atlantic Salmon (*Salmo Salar*): Population Dynamics and Viability. DFO Can. Sci. Advis. Sec. Res. Doc. 2016/032. v + 87 p.

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ABSTRACT

The purpose of this research document is to provide information about the population dynamics and viability of Outer Bay of Fundy Atlantic Salmon in support of recovery planning for this designatable unit. It covers the topics in the Terms of Reference for the Recovery Potential Assessment for Outer Bay of Fundy Atlantic Salmon relating to estimation of age- and stage-specific life history parameters (mortality rates and stage transition probabilities), the past and present population dynamics and viability of these populations, and scenario analyses to help identify and prioritize among recovery actions.

Analyses are presented for the Nashwaak River and the Tobique River salmon populations. For the Nashwaak River population, life history parameter estimates were obtained by fitting a life history model to population-specific data including: annual estimates of juvenile densities; egg depositions; the number and age composition of emigrating smolts; and the numbers of returning adults. The results indicate that at-sea survival for salmon maturing after two winters at sea has decreased by a factor of about two to three, whereas recent increases in the return rates for salmon maturing after one winter at sea are nearer the historical values. Maximum lifetime reproductive rates decreased from an average of 2.49 in the 1970’s to 1.13 in the 2000’s for the Nashwaak River population, and was estimated to be 0.18 for the Tobique River population using data from 1989 to 2005. Based on these values, in the absence of human intervention or a change in these rates, the Tobique River population is expected to extirpate, whereas, although the Nashwaak River population has a equilibrium population size greater than zero, it has very little capacity to rebuild and is at risk of extirpation from random variability and stochastic events.

Population viability analyses indicate that relatively small increases in either freshwater productivity or at-sea survival are expected to markedly decrease extinction probabilities for the Nashwaak River population, although larger changes in at-sea survival will be required to restore the population to levels above their conservation requirements. Larger changes in survival are expected to be necessary to prevent the extirpation of the Tobique River population due to the reduced rate of survival for emigrating smolts.

Évaluation du potentiel de rétablissement (EPR) du saumon de l'Atlantique (*Salmo salar*) de l’extérieur de la baie de Fundy : Analyses de la viabilité de la population

RÉSUMÉ

Le présent document de recherche a pour objet de fournir des renseignements à propos de la dynamique et de la viabilité des populations du saumon de l'Atlantique de l'extérieur de la baie de Fundy à l'appui de la planification du rétablissement de cette unité désignable. Il traite des points du cadre de référence pour l'évaluation du potentiel de rétablissement du saumon de l'Atlantique de l'extérieur de la baie de Fundy relatifs à l'estimation des paramètres du cycle biologique propres à l'âge et au stade (les taux de mortalité et les probabilités de transition de stades), à la dynamique des populations passées et présentes et à la viabilité de celles-ci, et aux analyses de scénarios afin d'aider à définir des mesures de rétablissement et à établir des priorités entre elles.

Les analyses sont présentées pour les populations de saumons des rivières Nashwaak et Tobique. Les estimations des paramètres du cycle biologique de la population de la rivière Nashwaak ont été obtenues en intégrant un modèle de cycle biologique aux données d'une population précise, notamment les estimations annuelles des densités de juvéniles, les pontes, le nombre et la composition selon l'âge des saumoneaux qui émigrent et le nombre d'adultes qui reviennent. Les résultats montrent que le taux de survie en mer des saumons qui atteignent la maturité après avoir passé deux hivers en mer a diminué d'un facteur d'environ deux à trois, tandis que les récentes augmentations dans les taux de montaison de saumons qui atteignent la maturité après avoir passé un hiver en mer sont plus proches des valeurs historiques. Les taux de reproduction maximaux ont diminué, passant d'une moyenne de 2,49 dans les années 1970 à 1,13 dans les années 2000 pour la population de la rivière Nashwaak, et ils ont été estimés à 0,18 pour la population de la rivière Tobique à l'aide des données de 1989 à 2005. En fonction de ces valeurs, en l'absence d'une intervention humaine ou d'un changement dans ces taux, on s'attend à ce que la population de la rivière Tobique disparaisse. De son côté, bien que sa taille à l'équilibre soit supérieure à zéro, la population de la rivière Nashwaak a une faible capacité de rétablissement et elle est à risque de disparition en raison de la variabilité aléatoire et d'événements stochastiques.

Selon des analyses de viabilité démographiques, des augmentations relativement faibles de la productivité en eau douce ou de la survie en mer devraient permettre de grandement réduire les probabilités de disparition de la population de la rivière Nashwaak, même si des changements plus importants sur le plan de survie en mer seront nécessaires pour rétablir la population à des niveaux supérieurs à ceux qui sont requis pour sa conservation. En revanche, des changements plus importants concernant la survie devraient être nécessaires pour empêcher la disparition de la population de la rivière Tobique en raison du faible taux de survie des saumoneaux qui émigrent.

1.0 INTRODUCTION

The Outer Bay of Fundy Designatable Unit of Atlantic Salmon (*Salmo salar*) occupies rivers in New Brunswick between the Canada-USA border and the city of Saint John, including the Saint John River (Figure 1). This designatable unit was designated as “endangered” by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2011). To aid in consultative processes following the designation, and to serve as a basis for recovery planning, information about Outer Bay of Fundy Atlantic Salmon populations has been compiled in support of the Recovery Potential Assessment (RPA) for this designatable unit, addressing the 27 Terms of Reference (TORs) developed to guide the process.

This research document contains information about the population dynamics of Outer Bay of Fundy Atlantic Salmon, including:

1. life history parameters such as stage-specific mortality rates and stage transition probabilities;
2. the past and present population dynamics;
3. population viability; and
4. scenario analyses to identify and prioritize among recovery actions.

This information is based on new analyses of data from the Nashwaak River salmon population and a summary of previously published information about the dynamics of the Tobique River salmon population.

Specifically, this document addresses the following TORs:

1. Estimate, to the extent that information allows, the current or recent life-history parameters (total mortality, natural mortality, fecundity, maturity, recruitment, etc.) or reasonable surrogates; and associated uncertainties for all parameters.
2. Project expected population trajectories over three generations (or other biologically reasonable time), and trajectories over time to the recovery target (if possible to achieve), given current parameters for population dynamics and associated uncertainties using Fisheries and Oceans Canada (DFO) guidelines on long-term projections (Shelton et al. 2007).
3. Assess the probability that the recovery targets can be achieved under current rates of parameters for population dynamics, and how that probability would vary with different mortality (especially lower) and productivity (especially higher) parameters.
4. Project expected population trajectory (and uncertainties) over three generations (or other biologically reasonable time), and to the time of reaching recovery targets when recovery is feasible; given mortality rates and productivities associated with specific scenarios identified for exploration (as above). Include scenarios which provide as high a probability of survivorship and recovery as possible for biologically realistic parameter values.
5. Recommend parameter values for population productivity and starting mortality rates, and where necessary, specialized features of population models that would be required to allow exploration of additional scenarios as part of the assessment of economic, social, and cultural impacts of listing the species.
6. Evaluate maximum human-induced mortality which the species can sustain and not jeopardize survival or recovery of the species.

2.0 INFORMATION ABOUT LIFE HISTORY

COSEWIC (2011) identified 17 rivers considered to contain, or historically to have contained Outer Bay of Fundy Atlantic Salmon, although it is likely salmon may have used the smaller coastal or un-assessed rivers in southwest New Brunswick as well (Marshall et al. 2014). Although there is some variability in life history characteristics, the life cycle of Atlantic Salmon is relatively similar among populations throughout the region. In this section of the research document, the life cycle is described, estimates of life history parameters considered representative of populations in the region are provided, and these estimates are used to characterize the dynamics of the populations both in the past and at present. Specifically, information is provided about the life cycle of Atlantic Salmon, which served as the basis for the population models (Section 2.1) and the life history parameter values for the Nashwaak River and Tobique River populations derived using a statistical, life history-based population model (Section 2.2).

2.1 LIFE CYCLE OF OUTER BAY OF FUNDY ATLANTIC SALMON

Outer Bay of Fundy Atlantic Salmon are anadromous fish, meaning that while they are obligated to reproduce in fresh water, most spend part of their lives in the ocean to feed and grow. They are iterparous, meaning that they can spawn several times before they die. After spawning for the first time, some individuals may spawn again in consecutive years, while others may spawn in alternate years and others may switch between alternate and consecutive repeat spawning. Spawning typically occurs in late October and early November. After spawning, adults (known as “kelts”) may return to the sea or may remain in fresh water until the following spring. Eggs are deposited in nests (referred to as “redds”) excavated in the gravel substrate. Hatching begins in April and the yolk-sac larvae (known as “alevins”), remain in the gravel until May or June. After emergence from the gravel, the young (now called “fry”) begin feeding. As they grow, their behaviour changes and they tend to be found in different places in the river. By autumn, they are referred to as “parr”. Wild-origin parr in Outer Bay of Fundy rivers typically remain in fresh water for 2 to 4 years, although as described in Section 2.3, most leave the rivers at age-2 or age-3. Prior to leaving the river, parr undergo physical changes that allow them to survive in the ocean. These juvenile salmon are now referred to as “smolt” and will migrate to the sea during late April, May and early June. Timing of the smolt run varies somewhat with environmental conditions. Some male parr become sexually mature at a small size while still in the river (these are called “precocious parr” and are not included in the population model).

Within Outer Bay of Fundy populations, salmon mature after either one or two winters at sea (called “one sea-winter salmon” or 1SW, “two sea-winter salmon” or 2SW, respectively), although historically a small proportion also matured after three winters at sea (called “three sea-winter salmon” or 3SW). The proportion of salmon maturing after a given number of winters at sea can be highly variable among salmon populations, although populations in the Outer Bay of Fundy typically have a higher incidence of maturity as 2SW salmon than do salmon of the Inner Bay of Fundy designatable unit, which have a higher incidence of maturity as 1SW salmon. Three sea-winter salmon are now very rare or absent from most populations in the Outer Bay of Fundy. Adult run timing is variable. The terms “small salmon” and “large salmon” are used at times. Small salmon are <63 cm fork length and are virtually all 1SW salmon. Large salmon are >63 cm fork length, and include 2SW salmon, 3SW salmon as well as repeat spawning salmon (“multi-sea-winter” or MSW). A very small component of 1SW salmon may be greater than 63 cm fork length, but these are rare in the Outer Bay of Fundy. Similarly, a small component of the 1SW repeat spawning salmon may be less than 63 cm fork length, but these are also rare.

2.2 ESTIMATION OF LIFE HISTORY PARAMETERS

Life history parameter estimates were derived using a statistical, life history-based population dynamics model developed by Gibson et al. (2008b, 2009). The model, described in Appendix 1, follows the general theory developed by Fournier and Archibald (1982) and Deriso et al. (1985) for statistical catch-at-age models for stock assessment that allows auxiliary data to be incorporated for model fitting. This approach was adapted to use with several abundance life-stage specific indices (auxiliary data) to derive estimates of the age- and stage-specific abundances and survival rates required to analyze the dynamics of these populations. This information is then used to determine how recovery actions may be expected to change population size and viability.

The population dynamics model consists of two parts: a freshwater production model that provides estimates of the expected smolt production as a function of egg deposition and an egg-per-smolt (EPS) model that provides estimates of the rate at which smolts produce eggs throughout their lives. These components are combined via an equilibrium analysis that provides estimates of the abundance at which the population would stabilize if the input parameters remained unchanged. This combined model is then used to evaluate how equilibrium population size has changed through time, as well as how the population would be expected to change in response to changes in carrying capacity, survival, or life stage transition probabilities, as described in Section 2.4.

There are two Atlantic Salmon populations in the Outer Bay of Fundy Designatable Unit with sufficient data for estimating values for life history parameters (i.e. with enough data that the above modeling approach can be used). These are the Nashwaak River and the Tobique River populations. The dynamics of the Tobique River population were analyzed by Gibson et al. 2009), the results of which are summarized here. The model equations, symbolism and statistical considerations for the model used for the Nashwaak River population are described in Appendix 1, and the population specific details of the modeling, model diagnostics and sensitivity analyses are provided in Appendix 2. The results of the analyses are summarized in the text below. For the Nashwaak River population, the model is set up using data from 1970 to 2011, whereas the analysis for the Tobique River population was fit to data for the years 1989 to 2005 (Gibson et al. 2009). Although this analysis was not updated here, preliminary analyses including more recent data provided estimates not dissimilar to those of Gibson et al. (2009).

Nashwaak River Population

The model results summarized in this section are for the base model run described in Appendix 1. Sensitivity analyses are provided in Appendix 2 and a comparison of the various model runs is provided in Appendix 2: Table A2.8.

Parameter estimates obtained for the Nashwaak River population are biologically plausible (e.g. mortality estimates between 0 and 1) and the standard errors of the estimates were not large relative to the means indicating reasonable model fits (Table 2.1, Appendix 2: Table A2.8). Mortality from the egg to the fry stages (at the time of the electrofishing surveys in mid summer) was estimated to be 0.96, meaning 96% of individuals die from the time of egg deposition to the time of the survey. The α parameter is the maximum survival from the fry stage to the age-1 parr stage, which occurs at low abundance in the absence of density dependence. The value of 0.54 means that, at most, an average of 54% of fry survive to age-1 (during the time period between the electrofishing surveys in one year and the surveys in the next). The carrying capacity of age-1 parr, R_{asy}, was estimated to be 28.0 parr per 100 m^2. This is the maximum density of age-1 parr that would occur if there were a very large (infinite) number of spawners in the river. The parameter value for j_2 of 0.61 is interpreted to mean that on average, 61% of the age-2 parr undergo smoltification in the spring at that age and emigrate from the river. The value of j_3 is interpreted to mean that on average, 99% of the parr that remain alive in the cohort, undergo smoltification at age-3. The estimate of the annual mortality rate for parr older than age-1, M_{parr}, is estimated to be 0.53. This parameter is the proportion of parr older than age-1 that die annually (i.e. the mortality rate between age-1 and age-2, or between age-2 and age-3) and is assumed to be the same for all age classes in this category.

As described in Appendix 1, these parameter values can be combined to summarize the dynamics of juvenile salmon in freshwater (i.e. to describe survival from egg deposition to smoltification). The analyses indicate that the maximum number of smolts produced per egg is 0.007 (Table 2.1). This value is low relative to that seen in many other rivers (see discussion). The carrying capacity for smolt was estimated to be slightly more than 104,000 smolt. Again, these values can loosely be interpreted as averages for the time. Similar to the interpretation for parr above, the maximum number of smolts produced per egg occurs at very low abundance in the absence of density dependence and as abundance increases, survival decreases, resulting in a lower number of smolt per egg. The carrying capacity for smolt is the maximum number of smolt that would occur if there were a very large (infinite) number of spawners in the river.

The model is set up to provide estimates of the abundance of salmon at several life stages. In the case of the egg depositions, there are empirical estimates of egg depositions (calculated from adult escapement) and the model is fit to these data. In the case of the Nashwaak population, the model fits the data well (Figure 2.1). The time series do not show an increase in the egg depositions in the mid-1980's when the commercial salmon fisheries in the region were closed. Smolt abundance estimates from monitoring data are available from 1998 to 2011 (Figure 2.1) and the model is fit to these data as well. A nice feature of the model is that it can be used to estimate what the smolt abundances would have been over the entire time period. Based on these estimates, smolt abundance in the 1970's and 1980's would have been roughly two to five times higher than at present. The Nashwaak River contains about 53,505 habitat units (1 unit = 100 m^2 of habitat) upstream of the counting fence and smolt enumeration site (Jones et al. 2014). If these abundance estimates are scaled by amount of habitat in the river, they indicate that smolt production per unit area has decreased from a maximum of nearly 0.94 smolts per 100 m^2 to an average of 0.25 smolts per 100 m^2 during the last 10 years. These values are low relative to those seen in other rivers, but alone are not necessarily indicative of poor habitat quality. Rather, adult abundance in the Nashwaak River was never high enough during this time period to fully realize the production potential in the river (See Section 2.4). However, the very low maximum survival rate from egg to smolt (0.007) could be considered indicative of poor habitat quality.

Because the model is set up to produce estimates of smolt abundance in the past, and because adult counts are also available, return rates (both to the mouth of the river and to spawning escapement) can be estimated. This comparison of past and present return rates would not be possible without the model. The observed and estimated return rates of 1SW and 2SW salmon to the river mouth are shown in Figure 2.2. Both the 1SW and 2SW return rates show decreasing trends during the 1970's to the mid-1980's and increasing trends during the 2000's.

A summary (mean, minimum and maximum) of the return rate estimates for the 1973-82 and 2000-09 time periods is provided in Table 2.2. In the early period, return rates varied between 3.02% and 12.14% for 1SW salmon and between 1.30% and 9.07% for 2SW salmon. In the 2000's, return rates varied between 1.38% and 11.48% for 1SW salmon and between 0.41% and 1.96% for 2SW salmon. Return rates through to spawning escapement are also provided in Table 2.2. The difference in the return rates to the river mouth and to spawning escapement is the effect of the recreational fishery. In the earlier period, the fishery reduced the 1SW and

MSW returns to 63% and 68% of those to the river. In the 2000's, there is no effect because the fishery has been closed.

Bayesian posterior probability densities (Figure 2.3) indicate that the mean return rates for the two time periods are reasonably well estimated by the model, and 95% Bayesian credible intervals (an indicator of the probability that the rates are different) for 2SW return rates do not overlap. The difference in the return rates between the two time periods for 1SW salmon is less than for 2SW salmon.

Tobique River Population

The results presented here are a summary of the analyses by Gibson et al. (2009) for the Tobique River population using data from 1989 to 2005. The freshwater model parameters have the same interpretation as for the Nashwaak River population. Comparing the freshwater dynamics between these two populations: the proportion of age-2 smolts in a cohort, the carrying capacity for parr, and the survival of older parr are estimated to be lower in the Tobique than in the Nashwaak; but the maximum survival rate from age-0 to age-1 is estimated to be higher (Table 2.1).

Based on the analyses of Gibson et al. (2009), under recent conditions the maximum number of smolts produced per egg ($\breve{\alpha}$) was estimated to be 0.0046 with an asymptotic population size ($\breve{R}_{asy}$) of 27,009 smolts. Both these estimates are quite low, the latter equating to 0.34 smolts per 100 m^2. For comparison, 3.8 smolts per 100 m^2, calculated for Atlantic Salmon in the 1970's (Symons 1979) is sometimes used as a general reference value.

The model for the Tobique River population was not set up to estimate return rates because the effects of the Mactaquac program (broodstock removals, trapping and trucking) make it impossible (with the current) to determine which salmon were destined for which tributary upstream of Mactaquac Dam. For this reason, Gibson et al. (2009) used the return rates to the Nashwaak River as a proxy for those to the Tobique River when analysing the dynamics of the Tobique River salmon population. An important difference between these populations is that mortality of salmon migrating downstream through head ponds and past turbines is thought to be 45.3%. All other things being equal, this would reduce return rates to the Tobique River to just over one half those to the Nashwaak River.

2.3 MORTALITY OF ADULT SALMON BETWEEN SPAWNING EVENTS

Estimates of the proportion of salmon in the Nashwaak River that return to spawn for a second or a third time are provided in Appendix 2 (Table A2.4) based on cohorts spawning for the first time during the 1993 to 2009 time period. Within the 1SW component, alternate-year repeat spawning is more common, whereas the rates of repeat spawning using an alternate-year or a consecutive-year strategy are similar for 2SW salmon. Overall, 3.1% of 1SW salmon and 9.0% of 2SW salmon return to spawn for a second time. Within the 1SW component, all salmon returning to spawn for a third time are returning the year after their second spawning, whereas in the 2SW component, a greater proportion skip a year between their second and third spawning. Return rates between the second and third spawning averaged 18.0% and 39.2% for salmon originally spawning as 1SW and 2SW salmon, respectively.

2.4 POPULATION DYNAMICS: PAST AND PRESENT

Gibson and Bowlby (2013) describe the use of equilibrium analyses in evaluating the effects of human activities or natural perturbations on salmon populations as follows (figure numbers are edited to match this document):

“Human activities may affect some parts of a fish population’s life history, such as its average fecundity, survival from one age class or life stage to the next, age-at-maturity or the number of times an individual reproduces. These parameters in turn affect the population’s productivity, and one way to assess the effects of a human activity is to evaluate the expected change in productivity that results from changes in life history parameters.

Equilibrium modeling is one approach that can be used to assess impacts of human activities in this way. This kind of analysis begins by splitting the life cycle of salmon into two parts, and for a given set of life history parameters, determining the population size at which the rates in each part of the life cycle are balanced such that the population does not increase or decrease in size. This is the population equilibrium for that specific set of parameter values. By varying the life history parameters in a manner that represents the expected response to a human activity and examining the resulting change in equilibrium population size, the effects of the activity on the population can be evaluated. Equilibrium models are widely used for analyzing population dynamics (Moussalli and Hilborn 1986), for estimating biological reference points for fisheries management (Myers et al. 1994), for providing a basis for the estimation of the long-term consequences of mortality caused by pollution, dams or other human activities (Barnthouse et al. 1988) and for linking fish habitat and fish population dynamics (Hayes et al. 1996).

In the case of Atlantic Salmon, a natural split in the life cycle occurs at the smolt stage when fish are migrating to the marine environment. The first part of the model gives freshwater production (the number of smolt produced as a function of egg deposition). The second part is the lifetime egg-per-smolt relationship, which gives the rate at which smolts are expected to produce eggs during their entire life. This approach is illustrated in Figure 2.4. In this example, the commonly used Beverton-Holt function is used to model smolt production in fresh water (Figure 2.4a). This model has two parameters: the slope of the function at the origin which gives the maximum rate at which eggs survive to become smolts. This is based on the idea that survival is greatest when population sizes are very low because competition between fish, which can result in reduced growth and increased mortality, is low. The other parameter is the carrying capacity of the river, which is the number of smolts that would be produced if egg deposition was extremely high. Changes in habitat quantity, say as a result of providing fish passage to areas that were previously inaccessible, have the effect of changing carrying capacity. Changes in habitat quality, say as a result of improving or reducing water quality, has the effect of changing the slope at the origin, but may also change carrying capacity as well (Hayes et al. 1996). Although only two parameters are used here, they combine the effects of egg-to-fry survival, fry-to-parr survival, parr-to-smolt survival and age-at-smoltification (Trzcinski et al. 2004).

The lifetime egg-per-smolt (EPS) relationship (Figure 2.4b) is assumed to be density independent. This is to say that the rate at which smolts produce eggs throughout their lives does not depend on the number of smolts that are produced. This is the equivalent of assuming resource availability in the marine environment is not limiting population growth, and therefore mortality at sea is not density-dependent. In contrast, resource availability in freshwater (see above), which determines carrying capacity, was assumed to limit production of smolt. This paradigm is consistent with most population models for diadromous fish, and is further supported by a recent analysis of the timing of density dependence in Atlantic Salmon, which found strong evidence for density dependence in salmon populations within fresh water and little evidence for

> density dependence in salmon within the marine environment (Gibson 2006). The rate at which smolts produce eggs is calculated based on the survival of juvenile salmon in the marine environment, age-at-maturity, fishing mortality, fecundity, and the number of times a fish spawns throughout its life.
>
> The population equilibrium is derived by finding the abundance at which the production of smolts by eggs equals the reciprocal of the production of eggs by smolts (Figure 2.4c); note that graphically this is the equivalent of flipping the axes in Figure 2.4b, so that the plots can be overlain. The equilibrium is the population size at which the population will stabilize if all model parameters do not change. Effects of human activities or other changes to population dynamics are evaluated by examining how the equilibrium changes in response to changes in the life history parameters resulting from the activity. In the example shown in Figure 2.4c, a decrease in smolt-to-adult survival shifts the equilibrium point to a smaller population size. If smolt-to-adult survival decreases far enough, the equilibrium population size goes to zero and the population will become extinct in the absence of human intervention or a change in one or more of the vital rates. However, an equilibrium population size greater than zero does not necessarily mean that a population is viable, because no allowance is made for random variability in the life history parameters or for catastrophic events."

The mathematics underlying the equilibrium analysis, including the EPS calculations and the calculations for equilibrium population size, are provided in Appendix 1.

The lifetime egg-per-smolt models are a useful mechanism for evaluating how the changes in return rates described in Section 2.2 influence a smolt's contribution to subsequent salmon production. The results of these analyses for the Nashwaak population are summarized in Table 2.3. For this population, EPS values ranged between 117 and 732 eggs/smolt in the 1973-82 time period, and between 49 and 151 eggs/smolt in the 2000's. Similar to the return rate analyses, the difference in the values calculated using return rates to the river mouth and to spawning escapement is an indicator of the effects of the recreational fishery. In this case, EPS at spawning escapement was reduced to 65% its value at the river mouth during the early time period. Bayesian posterior probability densities for the aveage lifetime egg-per-smolt estimates for the two time periods (Figure 2.5) indicate that the maximum likelihood estimates reasonably match the modes of the posterior densities. The 95% Bayesian credible intervals for the two time periods do not overlap for either population (an indicator of the probability that the rates are different).

Gibson et al. (2009) estimated that the EPS value for the Tobique River population (using 1989 to 2005 data) was 38.5 eggs/smolt, a value which is low. Given their freshwater production estimate described above, one smolt would have to produce more than 217 eggs (the inverse of 0.0046 smolts per egg) in order for the population to replace itself.

The results of the equilibrium analyses for the Nashwaak River salmon population are summarized in Table 2.4 and Figure 2.6. As mentioned, the equilibrium is the population size at which the population will stabilize if all model parameters do not change. To see how this works, consider Figure 2.6, top panel. Start at an egg deposition of 5 million eggs, and use the curved freshwater production line to determine how many smolts would be produced (slightly over 20,000 smolts). Then use the average EPS line (the middle dashed line) to determine how many eggs would be produced by that number of smolts (in the vicinity of 9 to 10 million eggs). Then, for that number of eggs, determine how many smolts would be produced, and so on. One should rapidly approach the equilibrium (the point where the freshwater production curve intersects the EPS line). Then try the same thing in Figure 2.6, bottom panel, starting at 20 million eggs. It should become evident that the equilibrium is an attractor towards which the population moves, and that recovery planning is about shifting the attractor to a place that causes the population to increase in size.

As shown in Figure 2.6, the equilibrium population size for the Nashwaak River population varied substantially in the 1970's and early 1980's because of the variability in the return rates during that time period. At the minimum return rates observed during that time period, the equilibrium population size was zero. During the 2000's, the mean equilibrium for the Nashwaak River population was 1.7 million eggs (Table 2.4), Although this value is greater than zero, but is low enough that the population is expected to be at risk of extirpation due to the effects of random environmental variability, as shown in the next section. The maximum lifetime reproductive rate for the Nashwaak population (Table 2.4, Figure 2.7) has decreased from averages of 2.49 in the 1973-82 time period, to an average of 1.13 during the 2000's. These values mean that during the 2000's, at low abundance and in the absence of density dependence (which further lowers reproductive rates), a salmon in the Nashwaak River produces on average a total of 1.13 replacement salmon throughout its life, indicating that the population has almost no capacity to rebuild if environmental events such as floods or droughts lower survival at some point in time. Note that the minimum rate indicates that there are years of low survival, which is why this population is at risk from environmental stochasticity.

Table 2.4 also provides a comparison of the equilibrium population size of the Nashwaak River population with and without the recreational fishery in the past. This metric is better for evaluating the effects of the fishery than the return rate and EPS comparisons presented earlier, because in addition to showing how egg production is decreased, the equilibrium analyses also take into account the capacity of the population to compensate for this reduction. In the case of the Nashwaak River population, this capacity is not high due to the low freshwater productivity already described. Although the recreational fishery had the effect of reducing the lifetime egg production per smolt to about 65% of its value without the fishery, its effect on the average equilibrium egg deposition was to reduce it to 40% its value without the fishery. Similarly, the maximum lifetime reproductive rate was reduced from an average of 2.49 to 1.60, reducing the population's resiliency to low survival events.

Gibson et al. (2009) conducted an equilibrium analysis for the Tobique River population to examine the combined effects of low freshwater productivity, downstream passage mortality for smolts and low at-sea survival. They used the following scenarios to explore the scope for management to facilitate recovery and to model the potential outcomes of future management strategies:

1. increased at-sea survival for immature salmon,
2. increased survival for migrating smolts (intended to represent improvements to downstream fish passage at all three hydroelectric facilities), and
3. increased freshwater productivity, resulting in greater smolt production within the river (intended to represent freshwater habitat restoration activities).

For scenario 1, they compared three levels of at-sea survival:

1. the current state using the mean return rates for salmon in the Nashwaak River (3.2% for 1SW and 0.9% for 2SW),
2. the most beneficial observed state on the St. John River system (6.4% for 1SW and 1.6% for 2SW), and
3. a hypothetical reduction in at-sea mortality, in which return rates for 1SW and 2SW fish were assumed to equal 8% and 3%, respectively.

For scenario 2, they evaluated the effect of increasing survival of emigrating smolts by comparing two levels of passage survival:

1. the status quo in which fish passage survival of smolt is 54.7%, and
2. a hypothetical scenario in which passage survival of smolt was increased to 100%.

These values were thought to bracket the range of possible outcomes to improved fish passage. For scenario 3, they explored two levels of freshwater productivity to examine its importance on equilibrium population size. The first represented present conditions and used the parameter values determined from the freshwater production model (Table 2.1). The second was a hypothetical situation in which the asymptotic recruitment level and maximum survival rate of age 1 parr were doubled, such that M_{parr} = 0.49 and R_{asy} = 18.6. Gibson et al. (2009) describe their results as follows (figure and table numbers are changed to match this document):

> “Under present freshwater conditions in the Tobique River, the maximum number of smolts produced per egg ($\breve{\alpha}$) was estimated to be 0.0046 with an asymptotic population size ($\breve{R}_{asy}$) of 27,009 smolts. This latter estimate is quite low, equating to 0.34 smolts per 100 m^2 (using the habitat area estimate obtained from orthophoto maps). For comparison, 3.8 smolts per 100 m^2, calculated for Atlantic Salmon in the 1970’s (Symons 1979) is sometimes used as a general reference value. The low production estimate for smolt in the Tobique is surprising given past salmon abundance in this river. Under the hypothetical scenario of improved freshwater production, the maximum number of smolts produced per egg ($\breve{\alpha}$) is 0.012 with an asymptotic population size ($\breve{R}_{asy}$) of 141,733 smolts. This value is equivalent to 6.53 smolts per 100 m^2 of habitat.
>
> At the average observed smolt-to-adult return rates, but at a hypothetical fish passage survival rate of 1.0, lifetime egg production per smolt was calculated to be 77 eggs. Presently, the estimated passage survival rate of Tobique River Atlantic Salmon is 0.547 (AMEC 2005; Carr 2001), which reduces the expected lifetime egg production of a smolt to 38.5 eggs. Given current freshwater production estimated above, one smolt would have to produce more than 217 eggs (the inverse of 0.0046 smolts per egg) throughout its lifetime in order to have an equilibrium population size greater than zero. As such, the population would be expected to decline to extinction in the absence of human intervention or a change in the vital rates.
>
> Consistent with the above values, the scenario analyses (Table 2.5, Figure 2.8) indicate that addressing fish passage mortality alone, with no change in either freshwater production or survival at-sea, is not expected to be sufficient to create a viable population. Similarly, if treated in isolation, increasing marine survival rates to 6.4% and 1.6% for 1SW and 2SW fish, respectively, or even to 8% and 3%, does not result in an equilibrium population size greater than 0 (Figure 2.8, top panel). Similarly, the population equilibrium remains at zero if only the capacity of the freshwater environment to produce smolts is increased (Figure 2.8, top panel). However, a small equilibrium population size exists if fish passage and freshwater production are increased concurrently and at-sea survival remains at its average level (Figure 2.8, lower panel). This equilibrium is well below the conservation requirement for this population (Table 2.5).
>
> Equilibrium population sizes that were very close to or above the conservation requirements for the Tobique River could be produced under three scenarios: (1) when

> freshwater production was improved and marine survival was increased to the highest modeled value and fish passage was at current levels (Table 2.5), (2) when freshwater production and fish passage were increased and marine survival was equal to the highest observed values in the Nashwaak River (Table 2.5), and (3) when fish passage survival and freshwater production were increased and at-sea survival was equal to 8% and 3% for 1SW and 2SW Atlantic Salmon, respectively (Table 2.5). The effect of at-sea survival on egg deposition is evident in Figure 2.8: under the improved freshwater production scenario, increasing at-sea survival from the maximum observed rate to the maximum hypothetical rate nearly doubles the equilibrium egg deposition with only a small increase in smolt production. Taken together, these scenarios show that although improvements to fish passage seem to have little influence on the equilibrium size of the population given current levels of freshwater production and at-sea survival, low fish passage survival does have the potential to limit the effectiveness of recovery actions focused on other life stages."

3.0 POPULATION VIABILITY ANALYSIS USING PRESENT LIFE HISTORY PARAMETERS

The long term population projections are carried out using a population viability analysis (PVA) following the approach of Gibson and Bowlby (2013) for Southern Upland Atlantic Salmon. They describe the role of PVA in conservation biology as follows:

> "PVAs are used extensively in conservation biology to predict both the risk of extinction for populations and species and to evaluate management strategies to recover at-risk populations. In a PVA, a population dynamics model is used to determine how the probability of persistence is affected by current conditions and future perturbations (Beissinger and McCullough 2002), and models are often used to identify threats to the persistence of a population and to evaluate how future management actions or environmental changes may influence the probabilities of extinction or of achieving recovery goals (Reed et al. 2002). Using simulations of population trajectories, PVA allows one to explore the logical implications of current knowledge and assumptions (Bowlby and Gibson 2011).
>
> Although some authors have cautioned against the use of PVAs because the predictions, typically time to extinction, are almost always quite uncertain (e.g. Taylor 1995; McCarthy et al. 1996; Ludwig 1999), many authors believe that PVA's can be used to assess relative risk (e.g. Akçakaya and Raphael 1998; Beissinger and Westphal 1998; McCarthy et al. 2001). Reed et al. (2002), argue that these relative evaluations are the most appropriate use of PVAs and can be used as a basis for choosing the most effective management strategy from a given set of possibilities (Lindenmayer and Possingham 1996).
>
> Some relatively simple PVA models are possible. For example, the Dennis-type PVA (Dennis et al. 1991) can be used to evaluate extinction risk if the assumption is made that future trends will be similar to past trends, although this type of model is of limited utility for evaluating recovery actions because the underlying biology is not considered. Models which track abundance at multiple life stages are therefore preferred when evaluating population persistence (O'Grady et al. 2004), provided adequate data are available and uncertainties are accounted for (Holt and Peterman 2008, Legault 2005, McCarthy et al. 2001). [text deleted]
>
> With respect to selecting recovery strategies, McCarthy et al. (2003) used a simulation study and found that they were able to identify the better of two management

> strategies 67-74% of the time using 10 years of data, and 92-93% of the time with 100 years of data."

The PVA model used here is described in detail in Appendix 3. It is an adaptation of the population dynamics model used to estimate life history parameter values (described in Appendix 1), and is also adapted from the PVA model developed by Gibson and Bowlby (2013) to reflect differences in the available information about repeat spawning dynamics between the LaHave River population and the Nashwaak River population (also described in Appendix 1). The model is fully age- and stage-structured and tracks abundance of all life stages and age classes for the duration of the forward projection. As described in Appendix 3, auto-correlated random variability is added at each age class or life stage transition. As the strength of this autocorrelation increases, good years are increasingly likely to be followed by good years (and bad followed by bad). Here the same autocorrelation and variances were used as were used by Gibson and Bowlby (2013) for the Southern Upland populations.

As was done by Gibson and Bowlby (2013), two versions of the PVA were run in parallel using values for two time periods: the past (the 1973-1982 time period) and the present (2000 to 2009) when the dynamics of the populations were very different. Comparison of the PVA results from these periods helps to ensure that the conclusions about present population viability are not simply a modeling artifact in the PVA.

Similar to Gibson and Bowlby (2013) extreme environmental events were included in the PVA:

> "Additionally, because Atlantic Salmon occupy habitats that are periodically subject to extreme conditions (e.g. floods and droughts), the effects of extreme events (which can lead to very high mortality) were included in the model. The life history of Atlantic Salmon, which distributes the reproductive effort of a cohort over multiple years, likely evolved in part as a strategy to cope with this kind of variability, and changes in their dynamics can affect the age structure in a population. In this analysis, extreme events affect survival between the egg and the fry life stages, thereby allowing density-dependent compensation to occur which would partially offset some of the mortality. The effect of the extreme events would be greater if it was incorporated after density dependence (i.e. after age 1) in the life cycle.
>
> The frequency and magnitude of extreme environmental events has not been determined for these populations. In most model runs, the probability that an extreme event occurred was set to 0.1 and the effect of the extreme event was assumed to reduce fry abundance by 80%. This means that on average, 10 events reducing the abundance of fry by 80% from the expected value would occur every 100 years. As modeled, a greater or lesser number of extreme events could occur in any simulated population trajectory, and their distribution through time is random (see Appendix 3 for details). The sensitivity of the results to these assumed values was evaluated by running scenarios with no extreme events as well as scenarios with more extreme environmental events. As shown in Section 2, the population growth rates of Outer Bay of Fundy Atlantic Salmon are currently very low, making it difficult for populations to recover after low survival events. Although the values used to simulate extreme mortality events are assumed, this loss of resiliency is illustrated via this analysis."

The rationale for including extreme environmental events in the PVA for the Nashwaak population is the same.

Analyses were carried out for Nashwaak River salmon populations using both the “past” and “present” dynamics. The population is modeled as a closed population, meaning that it is not affected by either immigration or emigration.

Gibson and Bowlby (2013) further describe the model:

> “For each scenario analyzed with the PVA, 2000 population trajectories were simulated and the extinction and recovery probabilities were calculated as the proportion of populations that go extinct by a specified time. For both the past and present scenarios, the population was projected forward from a starting abundance based on the estimated adult population size in 2010. The population was assumed to be at equilibrium, and the numbers of eggs, parr, smolt and adults, as well as their age, sex and previous spawning structure, at the start of each simulation were calculated from the adult abundance using the life history parameter values specific to the dynamics being simulated. To evaluate extinction probabilities, a quasi-extinction threshold of 15 females was assumed. This means that annual egg deposition is given a value of zero if the abundance drops below 15 females. A population must be below this value for two consecutive years to be assumed extinct in a given year. However, if the female abundance is higher in the next year, the egg deposition is calculated as per the model. A population can therefore sit on the quasi-extinction threshold for a number of years and can theoretically recover unless there are several sequential years where the female spawner abundance is less than 15. When evaluating recovery probabilities, the conservation requirement was used as the recovery target. The probability of recovery is calculated as the proportion of the simulated populations that are above the recovery target in a given year. Abundance in a population may be above the recovery target for a period of time, but is no longer considered recovered if its abundance subsequently drops to a level below the recovery target.
>
> Abundances for each life stage were projected forward for 100 years even though there is considerable uncertainty about what the dynamics of these populations will be at that time. However, the reason for using these long term projections is not to estimate what abundance will be at some time, but rather to evaluate longer term viability for each scenario (i.e. does the projection go to zero or not). In other words, the longer term projections are used to determine whether the populations are viable for each combination of life history parameters, random variability and extreme events included in the scenario.”

3.1 POPULATION VIABILITY IN THE PAST AND AT PRESENT

Abundance trajectories for the Nashwaak River salmon population (Figure 3.1) indicate that, given the present (2000’s) population dynamics, this population is expected to decline towards extirpation and has zero probability of reaching its recovery target (Table 3.1, Figure 3.2). The probability of extirpation (Table 3.1, Figure 3.2) increases after about 40 years, with 28% of the simulated populations being extirpated within 100 years. None of the 2000 simulated population trajectories met the recovery target within 100 years (Table 3.1). In contrast, abundance trajectories using the past (1973-82) dynamics (Figure 3.1) indicate rapid population growth. None of the simulated population trajectories extirpate within 100 years (Table 3.1, Figure 3.2), but only abount 55% of the simulated populations are above the recovery target, in any given year, 50 years in the future.

3.2 EFFECTS OF EXTREME ENVIRONMENTAL EVENTS

The effects of environmental variability and extreme events were investigated using the Nashwaak River population model. For each scenario described below, 2000 simulated population trajectories were run, each starting at the 2008-2012 average adult abundance. The Nashwaak River population has an equilibrium population size greater than zero, and therefore would not become extinct in the absence of environmental variability, as shown by the deterministic model in Figure 3.3 (top left panel) (refer also to Section 2.4). However, when random variability is added to the projections (using the same life history parameter values as in the base model), populations begin to become extinct towards the end of the simulations (Figure 3.3 – left column, second panel from the top). When extreme events are added, extinction risk increases further (Figure 3.3 – left column, third panel from the top). Changing the frequency and magnitude of the extreme events changes the extinction probabilities as expected (Figure 3.3 – left column, bottom two panels). However, when the same random variability and extreme event scenarios are modeled using the 1973-1982 dynamics, none of the 10,000 simulated population trajectories become extinct. This highlights the resiliency that these salmon populations had in the past to environmental variability as a result of higher maximum lifetime reproductive rates.

3.3 EFFECT OF TIME (STARTING POPULATION SIZE)

The analyses indicate that in the absence of human intervention or a change in survival for some other reason, abundance of these two Outer Bay of Fundy populations salmon will continue to decline. To examine the effect of delaying recovery activities, the population viability analysis (base model) for the Nashwaak River population was re-ran starting at 100%, 50%, 25% and 10% of the 2008-2012 mean abundance estimates used in the base model simulations (896 small salmon and 263 large salmon). Using the present dynamics, further reductions in population size have the effect of shortening time to extinction (Figures 3.4 and 3.5). A reduction in starting population size of 50% reduces the time to extinction, although greater than 50% of the simulated populations have not extirpated within the 100 year time period, whereas a reduction in size of 75% reduces the time to which 50% of the simulated populations are extinct to about 90 years. Using the 1973-1982 dynamics, time to recovery is similarly increased (Figures 3.4 and 3.5). When the analysis is run using the 2012 abundance (29 small salmon and 63 large salmon) using the 2000's dynamics, the extinction risk is much higher, about 25% in 50 years.

3.4 SENSITIVITY TO THE QUASI-EXTINCTION THRESHOLD

The results presented in this section are derived using a quasi-extinction threshold of 15 female salmon. Population viability analyses are known to be sensitive to the assumed threshold. This value is very low relative to the past abundances of salmon in these rivers. If depensatory dynamics exist, populations may not be able to recover from low abundances, even ones that are higher than this threshold. To evaluate the effects of the assumed quasi-extinction threshold on the estimated extinction probabilities, the analysis using the Nashwaak River population model was re-ran with the quasi-extinction threshold increased to 30, 50 and 100 female salmon (values that are still very low relative to past abundance). When scenarios are run using the 2000's dynamics, time to extinction decreases markedly as the threshold is increased (Figure 3.6). However, this threshold has nearly no effect on rebuilding times when the 1973-1982 dynamics are used.

4.0 POPULATION VIABILITY ANALYSIS OF RECOVERY AND ALLOWABLE HARM SCENARIOS

4.1 RECOVERY SCENARIOS

To evaluate how the probability of extinction and probability of meeting the recovery target would be expected to vary with increased freshwater productivity and increased lifetime egg production per smolt, 24 scenarios were evaluated using the Nashwaak River population PVA model. Information about the nature and magnitude of recovery actions required for the recovery actions for the Tobique River salmon populations was provided in Section 2.4.

The values used in the EPS component of the model were derived using the 1973-19822 and 2000-2009 parameter values (fecundity, return rates and sex ratio) as upper and lower estimates, respectively (Table 4.1), with two intermediate scenarios evenly spaced between these (i.e. at 1/3rd and 2/3rd the difference between past and present values).

Increased freshwater production was modeled by increasing smolt production by factors of 1.0 (no increase), 1.2 (20% increase), 1.5 (50% increase) and 2.0 (double or 100% increase).

Each combination of increased freshwater productivity and at-sea survival was modeled for a total of 16 scenarios (see results below). In addition, eight other scenarios are presented to investigate the effects of extreme events. In these, freshwater productivity was increased by a factor of 1.5 and simulations were carried out for all four at-sea survival values. For each scenario, the probabilities of extinction and recovery were evaluated using 2000 simulated population trajectories.

Abundance trajectories, extinction probabilities and recovery probabilities for the Nashwaak River population for each scenario are provided in Figures 4.1, 4.2 and 4.3, respectively. Associated numerical values are provided in Tables 4.2. The results of these analyses indicate how close the Nashwaak River Atlantic Salmon are to the threshold between becoming extirpated and being viable. Panel “A” in each figure shows the results using the current dynamics; as previously described, the population is expected to extirpate in the absence of human intervention or a change in vital rates for some other reason. Panel “B” shows the effect of increasing freshwater productivity by 20%. This improvement is not large, but does markedly reduce extinction risk, even if marine mortality rates remain unchanged. None of the 2000 simulated populations became extinct within 100 years under this scenario. Similarly, relatively small changes to the marine component of the model lead to viable populations; none of the simulated populations extirpated in the 1/3rd increase scenarios (Panel G) and a small proportion of the populations reached their recovery targets. Although small, numerically-viable populations are produced in both of these examples, greater changes are necessary in order for populations to consistently be above their conservation requirements, and likely require both a change in freshwater productivity and in lifetime production of eggs by smolts, as shown by the increase in proportion reaching the recovery target as freshwater productivity increases (Figure 4.3; compare Panels G to J). Additionally, a recovery probability of 84% in 30 years is achieved in the 2/3rd increase in the marine component when combined with a freshwater productivity increase of 50% (Panel O). In contrast, increases in the EPS component to the past levels without an increase in freshwater productivity only produce an 8% probability of meeting the recovery target (Panel M). Within limits, these conclusions are robust to how the frequency of extreme events is modeled (Panels E, K, Q, W, F, L, R, X). The probability of recovery increases when the frequency of the extreme events is reduced (e.g. compare Panels R and O).

4.2 ALLOWABLE HARM SCENARIOS

The effects of increased levels of harm to the population were also evaluated using the PVA for the Nashwaak River population. Here, decreased freshwater production was modeled by decreasing smolt production by factors of 1 (no decrease), 0.9 (10% decrease), 0.8 (20% decrease), 0.7 (30% decrease) and 0.5 (50% decrease). Decreases in at-sea survival were modeled as decreases of 0%, 10%, 30% and 50%. Extinction probabilities for each scenario are shown in Figure 4.4 and the associated numerical values are provided in Table 4.3.

None of the simulated populations recovered in these scenarios and extinction probability increased rapidly with relatively low levels of harms. For example, the 10% decrease in either at-sea survival or freshwater productivity increased the 100 year extinction probability to about 80%. Although under these scenarios the extinction risk is low on the time scale of a couple decades (Table 4.3), this result is because of the starting population size. The population would be in decline over the entire time period, and the use of a lower starting population size would decrease the time to extinction.

5.0 DISCUSSION, UNCERTAINTIES AND CONCLUSIONS

The dynamics of two Outer Bay of Fundy Atlantic Salmon populations were analyzed: the Tobique River population (thought to be representative of populations upstream of Mactaquac Dam) and the Nashwaak River population (thought to be representative of populations in the Saint John River downstream of Mactaquac Dam). A major difference in the present dynamics between these populations is the effect of hydroelectric development on the upstream populations. One of the effects of this development is a reduction in the smolt-to-adult return rates by 45.7% as a result of reduced survival of downstream migrating smolts. As such, extinction risk for populations above Mactaquac Dam is much higher than for those populations downstream.

In order to carry out the recovery scenario analyses, a decision had to be made about whether to only vary at-sea survival, or whether to also vary the sex ratio and fecundity values. As shown in Table 4.1, one of the changes is a near doubling of the proportion female in the 1SW component of the population. Assuming there is a genetic component to age-at-maturity, this could be the result of either genetic drift, or due to selective pressure as a result of increased at-sea mortality. If the former, then in the recovery scenarios, modeling changes in at-sea survival only would be appropriate, whereas if the change is due to selective pressure, then having the sex ratios revert back in the increased at-sea survival scenarios would be appropriate, as was done. Preliminary analyses (not shown) indicated that if the decision was made not to allow the sex ratios to change, extinction risk would be slightly lower than as modeled with the sex ratio change.

Although population projections were not done for the Tobique River population, the allowable harm analysis for the Nashwaak may be partially informative with respect to the effect of hydroelectric development on the populations. As already shown, the equilibrium population size under current conditions for this population is zero and the population is expected to extirpate in the absence of human intervention or a change in survival for some other reason. If all other factors were the same (starting population size, all life history parameters), a comparison of scenario A and scenario E in Figure 4.4 could be considered to approximately show how extinction risk differs as a result of this development.

Although times to extinction and recovery are presented in this document, readers are cautioned not to interpret these values too literally, given that prediction of extinction times using PVA is known to be highly uncertain (Taylor 1995; McCarthy et al. 1996; Ludwig 1999). As mentioned in the introduction to the PVA analyses, the most appropriate use of PVA's is to assess relative risk (e.g. Akçakaya and Raphael 1998; Beissinger and Westphal 1998; McCarthy et al. 2001) which can be used as a basis for choosing the most effective management strategy from a given set of possibilities (Lindenmayer and Possingham 1996). When comparing scenarios for Outer Bay of Fundy Atlantic Salmon, the important information is how much the time to extinction or recovery changes with when survival changes, not that extinction is predicted in (for example) 40 years.

The maximum survivals from egg to smolt (the slope at the origin of the spawner-recruit curve) for both populations (0.007 and 0.005 for the Nashwaak and Tobique populations, respectively) appear low relative to the values estimated for some populations, but not others. Using a similar model, Gibson and Bowlby (2013) estimated values of 0.017 and 0.034 for the LaHave River (above Morgans Falls) and the St. Mary's River (West Branch) populations, respectively, two populations thought to be partially effected by river acidification. In contrast, Gibson et al. (2008b) report a value of 0.005 for the Big Salmon River. O'Connell et al. (2006) compared egg to smolt survival for nine Atlantic Salmon populations in eastern Canada. The Nashwaak River population had the lowest survival in the comparison after adjusting for density dependent effects by standardizing by river size and egg deposition (their Figure 17).

The results presented here highlight some important differences between the dynamics Outer Bay of Fundy Atlantic Salmon and those of the Inner Bay of Fundy and Southern Upland. At-sea survival estimates for Outer Bay of Fundy salmon (as inferred by from the Nashwaak River population) are roughly ten times higher than those for Inner Bay of Fundy salmon (Gibson et al. 2008a). For the Inner Bay of Fundy populations, at-sea survival is low enough that it cannot be offset by recovery actions focused on other threats. Although at-sea survival of both Outer Bay of Fundy and Southern Upland Atlantic Salmon has decreased, it remains high enough that small increases in freshwater productivity are predicted to lead to small but viable populations, an outcome that is not possible within the Inner Bay. However, for both Outer Bay of Fundy and Southern Upland salmon, increases in at-sea survival will be needed if populations are to be recovered to abundances above their respective conservation requirements. Readers are cautioned that, because of the fundamental differences in the dynamics of salmon populations in the regions (both in fresh water and in the marine environment), extrapolating about the limiting effects of at-sea survival from one region to another could lead to erroneous conclusions about its overall effects on recovery potential and population viability. In addition, within the Outer Bay of Fundy designatable unit, extrapolation from populations below Mactaquac Dam to those found in tributaries above Mactaquac Dam requires a correction for the effects of reduced downstream survival.

Gibson and Bowlby (2013) provided the following guidance on interpreting the recovery targets and probability of recovery when describing the PVA for Southern Upland salmon:

> "Throughout these analyses, the conservation requirement was used as the recovery target when assessing the probability of recovery, consistent with its definition when it was developed and its use as the critical-cautious boundary in the precautionary fisheries framework (see Bowlby et al. 2013a). In the analyses here, small increases in productivity and survival led to populations that were viable (conditional on model assumptions) at levels well below the conservation requirement. However, it is not known whether these populations would truly be viable in the longer term because the smaller populations may be at risk due to declining fitness caused by genetic effects, including inbreeding and loss of genetic variation (Frankham 2008).
>
> Lande and Barrowclough (1987) showed that an effective population size (Ne) of approximately 500 individuals can maintain most genetic variability, although there are

> estimates of the number required that are lower. For Atlantic salmon, Elliott and Reilly (2003) found that an effective breeding population of 80-90 individuals was sufficient to maintain most of the genetic variability in populations introduced in Australia and Tasmania. Census population sizes (Ncensus) would be expected to be larger. Although overlapping generations, iteroparity and straying make the estimatation of Ne difficult, there are several studies that have estimated Ne and Ne/Ncensus ratios for salmonids (reviewed in Trzcinski et al. 2004). The average of the lower and upper limits of Ne/Ncensus ratios across taxa and studies give a range from 0.26 to 0.88. If we use the range of 0.26-0.88 and assume that a minimum of 80-100 individuals are necessary to maintain genetic variability, then the minimum total population size should be between 91-385. If Lande and Barrowclough's (1987) more conservative estimate of a minimum effective population size of 500 individuals is used, then minimum census population size should be between 568-1,923 individuals. These values assume a closed population. If straying exists between rivers, the minimum census population size required to maintain genetic diversity would be lower."

As discussed in Jones et al. (2014), the recovery target should be revisited once populations are selected for recovery, recovery actions are identified, and information about the expected dynamics of the recovered population is obtained. In this document, the probability of recovery should be interpreted in the context of reaching the conservation requirement rather than in the context of preventing extinction. The sensitivity analyses with respect to the quasi-extinction threshold indicates how extinction risk increases if larger population sizes are required for longer term viability.

Overall, the retrospective examination of the recreational fishery on the Nashwaak River population indicated that the fisheries did reduce population size, and that this reduction was great enough to have been a contributing factor to the overall population decline: a 60% reduction in equilibrium egg depositions in the 1973-1982 time period when retention fishers were open for both large and small salmon. Although not shown here similar analysis for the 1980-1989 time period show a 53% reduction in the equilibrium egg deposition during a period when the transition to hook-and-release fisheries was occurring.

These analyses of the population dynamics are not adjusted for the effects of commercial fisheries, either locally in the past, or presently in international or distant waters. Without this adjustment, the effect is to underestimate maximum lifetime reproductive rates and hence viability. This effect would have been greater in the past when local commercial fisheries were operating and when landings in international and distant waters were higher, and as a result, the changes in the dynamics between the past and present scenarios may be underestimated, with higher levels of productivity in the past.

The population viability analyses illustrate the expected population trajectories for a specific set of life history parameter values, but do not include linkages among the various survival rates. For example, it has been suggested that survival of fish in large schools is enhanced when traveling through predator fields compared to fish traveling singly or in small schools (Cairns 2001). As another example, the age-of-smoltification may be linked to the growth rates of parr, which can be density dependent.

The information on population dynamics presented here indicate that abundance of Outer Bay of Fundy Atlantic Salmon is expected to continue to decline in the absence of human intervention or a change in life history parameters for some other reason. As shown by the population viability section (Section 4), both the probability of extinction and the time to recovery increase if abundance decreases further before recovery actions are initiated. For example, extinction risk in the nearer term was much higher when the 2012 abundance estimates were used to initialize the population, which are much lower than the 5-year average used in most model runs. As was noted for Southern Upland salmon (Gibson and Bowlby 2013), it is expected that there are risks to the population of being at low population size, such as the loss of genetic variation and inbreeding depression. The scenarios do not fully explore the uncertainties in how small a population may get before it cannot be recovered (i.e. the models do not include depensatory processes). However, the sensitivity analysis with respect to the quasi-extinction threshold shows that the risk of extinction increases as this threshold is increased. Irrespective of the absolute value of this threshold, further reductions in abundance do take Outer Bay of Fundy salmon populations closer to this limit. As discussed above, there are risks associated with being at low population size that are expected to lead to lower fitness and thus a reduced capacity for population increase. Recovery is therefore expected to become more difficult if abundance continues to decline, as is expected for these populations with the continued passage of time.

6.0 ACKNOWLEDGMENTS

The authors thank the many people without whose hard work collecting the data these analyses would not have been possible; and Gerald Chaput for helpful comments on an earlier draft of this manuscript.

7.0 REFERENCES

Akçakaya H.R., and M.G. Raphael. 1998. Assessing human impact despite uncertainty: viability of the Northern Spotted Owl metapopulation in northwestern USA. Biodiversity and Conservation 7: 875-894.

AMEC. 2005. Conceptual Facility to Bypass Atlantic Salmon Smolts at the Tobique Narrows Dam: Final Report. Report to the Saint John Salmon Recovery Committee of the New Brunswick Salmon Council. AMEC Earth and Environmental, Fredericton, N.B.

Barnthouse, L. W., R. J. Klauda, D. S. Vaughan and R. L. Kendall. 1988. Science, law, and the Hudson River power plants: a case study in environmental assessment. American Fisheries Society Monograph 4. American Fisheries Society, Bethesda, Maryland.

Beissinger, S.R., and S.R. McCullough. 2002. Population viability analysis. The University of Chicago Press, Chicago, Illinois.

Beissinger, S.R., and M.I. Westphal. 1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management 62: 821-841.

Bowlby, H.D., A.J.F. Gibson and A. Levy 2013. Recovery Potential Assessment for Southern Upland Atlantic Salmon: Status, Past and Present Abundance, Life History and Trends. DFO Canadian Science Advisory Secretariat Research Document 2012/005.

Bowlby, H.D. and A.J.F. Gibson. 2011. Reduction in fitness limits the useful duration of supplementary rearing in an endangered salmon population. Ecological Applications 21:3032-3048.

Carlin, B.P., and T.A. Louis. 1996. Bayes and empirical Bayes methods for data analysis. Volume 69 of *Monographs on Statistics and Applied Probability*. Chapman and Hall, London, UK.

Carr, J. 2001. Downstream movements of juvenile Atlantic salmon (Salmo salar) in the dam-impacted Saint John River drainage. Canadian Manuscript Report in Fisheries and Aquatic Sciences 2573, Ottawa.

Cairns, D.K. 2001. An evaluation of possible causes of the decline in pre-fishery abundance of North American Atlantic salmon. Canadian Technical Report of Fisheries and Aquatic Sciences No. 2722, Ottawa.

Chaput G., and R. Jones 2007. Reproductive rates and rebuilding potential for two multi-sea-winter Atlantic salmon (*Salmo salar* L.) stocks of the Maritime provinces. DFO Canadian Science Advisory Secretariat Research Document 2006/027.

Chib, S., and E. Greenberg. 1995. Understanding the Metropolis-Hastings algorithm. Am. Stat. 49: 327-335.

Chilcote, M. W., K. W. Goodson, and M. R. Falcy. 2011. Reduced recruitment performance in natural populations of anadromous salmonids associated with hatchery-reared fish. Canadian Journal of Fisheries and Aquatic Sciences 68:511-522.

COSEWIC. 2011. COSEWIC assessment and status report on the Atlantic Salmon *Salmo salar* Nunavik population, Labrador population, Northeast Newfoundland population, South Newfoundland population, Southwest Newfoundland population, Northwest Newfoundland population, Quebec Eastern North Shore population, Quebec Western North Shore population, Anticosti Island population, Inner St. Lawrence population, Lake Ontario population, Gaspe-Southern Gulf of St. Lawrence population, Eastern Cape Breton population, Nova Scotia Southern Upland population, Inner Bay of Fundy population Outer Bay of Fundy population in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. 136pp. (www.sararegistry.gc.ca/status/status_e.cfm).

Dennis, B., P. Munholland, and J.M. Scott. 1991. Estimation of growth and extinction parameters for endangered species. Ecological Monographs 61, 115–143.

Deriso, R.B., T.J. Quinn II and P.R. Neal. 1985. Catch-age analysis with auxiliary information. Canadian Journal of Fisheries and Aquatic Sciences 42: 815-824.

DFO. 2011. Status of Atlantic Salmon in Salmon Fishing Areas (SFAs) 19-21 and 23. DFO Canadian Science Advisory Secretariat Science Response 2001/005.

Efron, B. and R.J. Tibshirani. 1993. An Introduction to the Bootstrap. International Thomson Publishing. New York.

Elliott, N.G., and A. Reilly. 2003. Likelihood of bottleneck event in the history of the Australian population of Atlantic Salmon (*Salmo salar* L.). Aquaculture 215: 31-44.

Fournier, D. 1996. An introduction to AD Model Builder for use in nonlinear modelling and statistics. Otter Research Ltd., Nanaimo, BC, Canada.

Fournier, D.A. and C.P. Archibald. 1982. A general theory for analyzing catch at age data. Canadian Journal of Fisheries and Aquatic Sciences 39: 1195-1207.

Frankham, R. 2008. Genetic adaptation to captivity in species conservation programs. Molecular Ecology 17:325–333.

Francis, A.A. 1980. Densities of juvenile Atlantic salmon and other species, and related data from electroseining studies in the Saint John River system, 1968-78. Canadian Data Report of Fisheries and Aquatic Sciences No. 178.

Gamerman, D. 2000. Markov Chain Monte Carlo: Stochastic Simulation for Bayesian Inference. CRC Press, London, UK.

Gibson, A.J.F. 2006. Population Regulation in Eastern Canadian Atlantic Salmon (Salmo salar) populations. DFO Canadian Science Advisory Secretariat Research Document 2006/016.

Gibson, A.J.F., and P.G. Amiro. 2003. Abundance of Atlantic Salmon (*Salmo salar*) in the Stewiacke River, NS, from 1965 to 2002. DFO Canadian Science Advisory Secretariat Research Document 2003/108.

Gibson, A.J.F. and P.G. Amiro. 2007. Predicting the Effects of Recovery Activities on Atlantic Salmon Populations. pp 15-23. *in* Brylinsky, M. and L. Hinks, editors. Proceedings of the Acid Rain Mitigation Workshop, Bedford Institute of Oceanography, May 26th and 27th, 2006.

Gibson, A.J.F., H.D.Bowlby, J.R. Bryan and P.G. Amiro. 2008a. Population viability analysis of Inner Bay of Fundy Atlantic Salmon with and without Live Gene Banking. DFO Canadian Science Advisory Secretariat Research Document 2008/057.

Gibson, A.J.F., R.A. Jones, and P.G. Amiro. 2008b. Equilibrium analyses of the recovery feasibility of four Atlantic Salmon (*Salmo salar*) populations in Nova Scotia and Southwest New Brunswick. DFO Canadian Science Advisory Secretariat Research Document 2008/010.

Gibson, A.J.F., R.A. Jones and H.D. Bowlby. 2009. Equilibrium analyses of a population's response to recovery activities: a case study with Atlantic Salmon. North American Journal of Fisheries Management 29: 958-974.

Gibson, A.J.F., and H.D. Bowlby. 2013. Recovery Potential Assessment for Southern Upland Atlantic Salmon: Population Dynamics and Viability. Fisheries and Oceans Canada. DFO Canadian Science Advisory Secretariat. Research Document. 2012/142.

Hayes, D. B., C. P. Ferreri and W. W. Taylor. 1996. Linking fish habitat to their population dynamics. Canadian Journal of Fisheries and Aquatic Sciences 53 (supplement 1): 383-390.

Hilborn, R. 2001. Risk analysis for salmon spawning reference levels. p. 177-193 *In*: E. Prevost and G. Chaput [eds.], Stock, Recruitment and Reference Points, Assessment and Management of Atlantic Salmon. INRA, Paris.

Hilborn, R., and M. Mangel. 1997. The ecological detective: Confronting models with data. Princeton University Press, New Jersey.

Holt, C. A., and R. M. Peterman. 2008. Uncertainties in population dynamics and outcomes of regulations in sockeye salmon (*Oncorhynchus nerka*) fisheries: implications for management. Canadian Journal of Fisheries and Aquatic Sciences 65:1459-1474.

Jones, R.A., L. Anderson, and T. Goff. 2004. Assessments of Atlantic salmon stocks in southwest New Brunswick, an update to 2003. Can. Sci. Advis. Sec. Res. Doc. 2004/019.

Jones, R.A., L. Anderson, and C. Clarke. 2014. Assessment of the Recovery Potential for the Outer Bay of Fundy Population of Atlantic Salmon (*Salmo salar*); Status, Trends, Distribution, Life History Characteristics and Recovery Targets. Can. Sci. Advis. Sec. Res. Doc. 2014/008.

Jonsson, B., and N. Jonsson. 2006. Cultured Atlantic Salmon in nature: a review of their ecology and interaction with wild fish. ICES Journal of Marine Science 63:1162-1181.

Lande, R., and G.F. Barrowclough. 1987. Effective population size, genetic variation, and their use in population management. Pages 87 – 123 in Viable populations for conservation, M.E. Soule (ed.). Cambridge University Press, New York.

Lande, R., S. Engen and B.-E. Saether. 2003. Stochastic Population Dynamics in Ecology and Conservation. Oxford University Press, New York.

Legault, C.M. (2005). Population viability analysis of Atlantic Salmon in Maine, USA. Transactions of the American Fisheries Society 134; 539-562.

Lindenmayer, D.B. and H.P. Possingham. 1996. Ranking conservation and timber management options for Leadbeater’s possum in southeastern Australia using population viability analysis. Conservation Biology 10: 235-251.

Ludwig, D. 1999. Is it meaningful to estimate a probability of extinction? Ecology 80:293-310.

Marshall, T.L. 1987. Assessment of Atlantic salmon of the Saint John River, N.B. 1986. Canadian Aquatic Fisheries Science Advisory Council Research Document 87/55. Ottawa.

Marshall, T. L., and G. H. Penney. 1983. Spawning and river escapement requirements for Atlantic salmon of the Saint John River, New Brunswick. Canadian Aquatic Fisheries Science Advisory Council Research Document 83/66. Ottawa.

Marshall, T.L., R.A. Jones, and L. Anderson. 2000. Assessment of Atlantic Salmon stocks in southwest New Brunswick, 1999. DFO Canadian Stock Assessment Secretariat Research Document 2000/010. Ottawa.

Marshall, T.L., Clarke, C.N., Jones, R.A., and Ratelle, S.M. 2014. Assessment of the Recovery Potential for the Outer Bay of Fundy Population of Atlantic Salmon (Salmo salar): Habitat Considerations. DFO Canadian Stock Assessment Secretariat Research Document 2014/007. Ottawa.

McCarthy, M.C., M.A. Burgman, and S. Ferson. 1996. Logistic sensitivity and bounds on extinction risks. Ecological Modelling 86:297-303.

McCarthy M. A., H. P. Possingham, J. R. Day, and A. J Tyre. 2001. Testing the accuracy of population viability analysis. Conservation Biology 15: 1030–1038.

McCarthy, M.A., S.J. Andelman, and H.P. Possinham. 2003. Reliability of Relative Predictions in Population Viability Analysis. Conservation Biology 17: 982–989.

McAllister, M.K., and G.P. Kirkwood. 1998. Bayesian stock assessment: a review and example application using the logistic model. ICES Journal of Marine Sciences 55: 1031-1060.

McAllister, M.K., E.K. Pikitch, A.E. Punt, and R. Hilborn. 1994. A Bayesian approach to stock assessment and harvest decisions using the sampling/importance resampling algorithm. Canadian Journal of Fisheries and Aquatic Sciences 51: 2673-2687.

Merritt, M.F., and T.J. Quinn, II. 2000. Using perceptions of data accuracy and empirical weighting information: assessment of a recreational fish population. Canadian Journal of Fisheries and Aquatic Sciences 57: 1459-1469.

McGinnity, P., P. Prodohl, N. O’Maoileidigh, R. Hynes, D. Cotter, N. Baker, B. O’Hea, and A. Ferguson. 2004. Differential lifetime success and performance of native and non-native Atlantic Salmon examined under communal natural conditions. Journal of Fish Biology 65(Suppl. A):173-187.

Moussalli, E., and R. Hilborn. 1986. Optimal stock size and harvest rate in multistage life history models. Canadian Journal of Fisheries and Aquatic Sciences 43: 135-141.

Myers, R. A., A. A. Rosenberg, P. M. Mace, N. J. Barrowman and V. R. Restrepo. 1994. In search of thresholds for recruitment overfishing. ICES Journal of Marine Science 51: 191-205.

Myers, R.A., J. Bridson, and N.J. Barrowman. 1995. Summary of worldwide stock and recruitment data. Canadian Technical Report of Fisheries and Aquatic Sciences 2024.

O'Connell,M. F.,J. B. Dempson, and G. Chaput, 2006. Aspects of the Life History, Biology, and Population Dynamics of Atlantic Salmon (*Salmo salar* L.) in Eastern Canada. DFO Canadian Stock Assessment Secretariat Research Document. 2006/014. Ottawa.

O'Grady, J. J., D. H. Reed, B. W. Brook, and R. Frankham. 2004. What are the best correlates of predicted extinction risk? Biological Conservation 118:513-520.

O'Neil, S.F., M. Bernard, P. Gallop and R. Pickard. 1987. 1986 Atlantic salmon sport catch statistics, Maritime Provinces. Canadian Data Report of Fisheries and Aquatic Sciences No. 663.

O'Neil, S.F., K. Newbould and R. Pickard. 1989. 1987 Atlantic salmon sport catch statistics, Maritime Provinces. Canadian Data Report of Fisheries and Aquatic Sciences No. 770.

O'Neil, S.F., D.A. Stewart, K. Newbould and R. Pickard. 1991. 1988 Atlantic salmon sport catch statistics, Maritime Provinces. Canadian Data Report of Fisheries and Aquatic Sciences No. 852.

O'Neil, S.F., D.A. Stewart, K. Newbould and R. Pickard. 1996. 1989 Atlantic salmon sport catch statistics, Maritime Provinces. Canadian Data Report of Fisheries and Aquatic Sciences No. 999.

Penney, G.H., and T.L. Marshall. 1984. Status of Saint John River, N.B., Atlantic Salmon in 1983 and forecast of returns in 1984. Canadian Aquatic Fisheries Science Advisory Council Research Document 84/47. Ottawa.

Punt, A.E. and R. Hilborn. 1997. Fisheries stock assessment and the Bayesian approach. Rev. Fish. Bio. Fish. 7: 35-63.

Quinn, T.J., and R.B. Deriso. 1999. Quantitative Fish Dynamics. Oxford University Press. New York, NY, USA.

Reed, J.M., L.S. Mills, J.B. Dunning, E.S. Menges, K.S. McKelvey, R. Frye, S. R. Beissinger, M.-C. Anstett and P. Miller. 2002. Emerging issues in Population Viability Analysis. Conservation Biology 16:7-19.

Ricker, W.E. 1975. Computation and Interpretation of Biological Statistics of Fish Populations. Bull. Fish. Res. Can. 191, 382 p.

Shelton, P.A., B. Best, A. Cass, C. Cyr, D. Duplisea, J. Gibson, M. Hammill, S. Khwaja, M. Koops, K. Martin, B. O'Boyle, J. Rice, A. Sinclair, K. Smedbol, D. Swain, L. Velez-Espino, and C. Wood. 2007. Assessing recovery potential: long-term projections and their implications for socio-economic analysis. DFO Canadian Science Advisory Secretariat Research Document 2007/045.

Symons, P.E.K. 1979. Estimated escapement of Atlantic Salmon (*Salmo salar*) for maximum smolt production in rivers of different productivity. Journal of the Fisheries Research Board of Canada 36: 132–140.

Taylor, B.L. 1995. The Reliability of Using Population Viability Analysis for Risk Classification of Species. Conservation Biology 9 (3), 551–558.

Trzcinski, M.K., A.J.F. Gibson, P.G. Amiro and R.G. Randall. 2004. Inner Bay of Fundy Atlantic Salmon critical habitat case study. DFO Canadian Science Advisory Secretariat Research Document 2004/114. 77p.

Walters, C., and D. Ludwig. 1993. Calculation of Bayes posterior distributions for key population parameters. Canadian Journal of Fisheries and Aquatic Sciences 51: 713-722.

Weir, L. K., J. A. Hutchings, I. A. Fleming, and S. Einum. 2004. Dominance relationships and behavioural correlates of individual spawning success in farmed and wild male Atlantic Salmon, *Salmo salar*. Journal of Animal Ecology 73:1069-1079.

8.0 TABLES

Table 2.1. Maximum likelihood estimates (M.L.E.) and their standard deviations (std. dev.) for the life history parameters characterizing freshwater productivity for the Atlantic Salmon populations in the Nashwaak River (using data from 1970 to 2011) and in the Tobique River (using data from 1989 to 2005). Also shown are the estimates of the maximum survival from egg to smolt and the asymptotic recruitment level derived for smolts. The conservation requirements are provided as indicators of the potential relative size of the populations. Estimates for the Tobique River population are from Gibson et al. (2009).

	Nashwaak River		Tobique River	
Parameter	M.L.E.	std. dev.	M.L.E.	std. dev.
Conservation Requirement:	12.80 million eggs		19.50 million eggs	
Age and Stage specific parameter values:				
$M_{Egg:}$	0.96	0.01	0.96	0.02
α	0.54	0.07	0.93	0.28
R_{asy}	28.01	8.79	9.31	2.39
M_{Parr}	0.53	0.10	0.75	0.56
j_2	0.61	0.06	0.37	0.07
j_3	0.99	0.01	0.97	0.03
Egg to smolt dynamics:				
$\breve{\alpha}$	0.007	0.001	0.005	n/a
$\breve{R}_{asy}$	104,430	36,178	27,009	n/a

Table 2.2. A summary of the average return rates (percent) of one sea-winter and two sea-winter wild Atlantic Salmon for the 1973 to 1982 (Past) and 2000 to 2009 (Present) time periods in the Nashwaak River. The values are the maximum likelihood estimates from the life history models. Two sets of values are provided: return rates to the river mouth and return rates through to spawning, including the recreational fishery removals. The difference between the two sets of values is an indicator of the effect of the recreational fishery on the proportion of the population surviving to spawn in each time period.

	Time Period	
Parameter	1973-1982	2000-2009
Return rates to river mouth (%):		
1SW mean	6.18	4.95
1SW minimum	3.02	1.38
1SW maximum	12.14	11.48
2SW mean	4.04	1.10
2SW minimum	1.30	0.41
2SW maximum	9.07	1.96
Return rates to spawning - including recreational fishery removals (%):		
1SW mean	3.87	4.95
1SW minimum	1.97	1.38
1SW maximum	7.50	11.48
2SW mean	2.62	1.10
2SW minimum	0.82	0.41
2SW maximum	5.64	1.96

Table 2.3. A summary of the number of eggs produced per smolt throughout its life (EPS) for wild Atlantic for the 1973 to 1982 (Past) and 2000 to 2009 (Present) time periods in the Nashwaak River. The values are the maximum likelihood estimates from the life history models. Two sets of values are provided: EPS derived using return rates to the river mouth, and using survival through to spawning during the fall. The difference in the values is an indicator of the effect of the recreational fishery on the lifetime egg production per smolt in each time period.

Parameter	Time Period	
	1973-1982	2000-2009
EPS using return rates to river mouth:		
mean	333	151
minimum	117	49
maximum	732	312
EPS using survival through to spawning - including recreational fishery removals:		
mean	215	151
minimum	75	49
maximum	454	312

Table 2.4. A summary of the equilibrium population sizes and maximum lifetime reproductive rates for wild Atlantic Salmon in the Nashwaak River for the 1973 to 1982 and 2000 to 2009 time periods. The values are the maximum likelihood estimates from the life history model. Two sets of values are provided: those derived using return rates to the river mouth and those derived based on survival through to spawning escapement. The difference is an indicator of the effect of the recreational fishery (Nashwaak only) on the population dynamics in each time period.

Parameter	Time Period	
	1973-1982	2000-2009
Values based on returns to mouth:		
Equilibrium egg deposition		
mean	20,805,000	1,761,400
minimum	0	0
maximum	62,444,000	18,550,000
Equilibrium smolt abundance:		
mean	62,433	11,674
minimum	0	0
maximum	85,312	59,523
Max. lifetime reproductive rate:		
mean	2.49	1.13
minimum	0.88	0.37
maximum	5.46	2.33
Values based on survival to spawning escapement (includes the effect of the recreational fishery:		
Equilibrium egg deposition:		
mean	8,408,700	1,761,400
minimum	0	0
maximum	33,463,000	18,550,000
Equilibrium smolt abundance:		
mean	39,195	11,674
minimum	0	0
maximum	73,635	59,523
Max. lifetime reproductive rate:		
mean	1.60	1.13
minimum	0.56	0.37
maximum	3.39	2.33

Table 2.5. Equilibrium population sizes of Atlantic Salmon in the Tobique River, New Brunswick, for combinations of two smolt passage survival scenarios, two freshwater production scenarios and three levels of at-sea survival. Equilibrium population sizes above one half the conservation requirement are marked with an asterisks; those above the conservation requirement are denoted with two asterisks (adapted from Gibson et al. 2009).

		At-sea survival[c]					
		Average		Maximum		Hypothetical	
Passage survival[a]	Freshwater production[b]	Eggs ($x10^6$)	Smolt ($x10^3$)	Eggs ($x10^6$)	Smolt ($x10^3$)	Eggs ($x10^6$)	Smolt ($x10^3$)
current	current	0.00	0.00	0.00	0.00	0.00	0.00
current	improved	0.00	0.00	7.34	53.97	18.69*	86.14
improved	current	0.00	0.00	0.85	3.40	4.84	12.21
improved	improved	7.29	53.73	23.15**	93.06	43.90**	110.65

Notes:

[a] smolt passage survival: current = 0.547, improved = 1.00;
[b] freshwater production: current = parameter values in Table 5, improved = doubling survival of age 1+ parr and R_{asy};
[c] average at-sea survival = return rates of 3.2% and 0.9%, maximum = 6.4% and 1.6%, and hypothetical = 8% and 3%, for 1SW and 2SW salmon respectively.

Table 3.1. Probabilities of extinction and of recovery within 1 to 10 decades for the Nashwaak River Atlantic Salmon population. Two scenarios are shown, one based on the 1973-1982 dynamics (past dynamics) and one based on the 2000's dynamics (present dynamics). The same random numbers are used for each scenario to ensure they are comparable. Probabilities are calculated as the proportion of 2000 Monte Carlo simulations of population trajectories that either went extinct or are above the recovery target in the given year.

	Probability of Extinction		Probability of Recovery	
Year	Present	Past	Present	Past
10	0.00	0.00	0.00	0.05
20	0.00	0.00	0.00	0.29
30	0.00	0.00	0.00	0.42
40	0.00	0.00	0.00	0.50
50	0.01	0.00	0.00	0.52
60	0.03	0.00	0.00	0.54
70	0.06	0.00	0.00	0.56
80	0.11	0.00	0.00	0.54
90	0.19	0.00	0.00	0.54
100	0.28	0.00	0.00	0.55

Table 4.1. At-sea survival rates used in the recovery scenario analyses for the Nashwaak River 1973-1982 dynamics (past) and one based on the 2000's dynamics (present). The intermediate fraction is the proportionate increase in at-sea survival between the past and present scenarios.

	Time Period			
Life History Parameter	Present	Intermediate 1/3	Intermediate 2/3	Past
1SW return rate (%)	4.95	5.29	5.62	5.95
2SW return rate (%)	1.29	2.31	3.33	4.35
Fecundity (small)	3,430	3,357	3,285	3,212
Fecundity (large)	7,387	7,305	7,224	7,142
Proportion female (small)	0.408	0.349	0.290	0.231
Proportion female (large)	0.796	0.817	0.838	0.858

Table 4.2. Proportions of 2000 simulated population trajectories that either go extinct or meet the recovery target within 10, 20, 30 and 50 year time horizons based on recovery scenarios for the Nashwaak River Atlantic Salmon population. The marine scenarios reflect changes from the present levels (2000's) of at-sea survival to those in the past (1973-1982). The freshwater scenarios reflect increases in freshwater productivity from the present level (1) to 2 times the present level. The lettering for the runs corresponds to those in Figures 4.1 – 4.3. Extreme event scenarios are the average frequency of extreme events and the reduction in egg to fry survival corresponding to the event.

Run	Marine Scenario	Freshwater Scenario	Extreme Event Scenario	Proportion Extinct				Proportion Recovered			
				10 yr	20 yr	30 yr	50 yr	10 yr	20 yr	30 yr	50 yr
a	present	1	10 yr; 0.2	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00
b	present	1.2	10 yr; 0.2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
c	present	1.5	10 yr; 0.2	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.01
d	present	2	10 yr; 0.2	0.00	0.00	0.00	0.00	0.03	0.18	0.25	0.32
e	present	1.5	20 yr; 0.1	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.01
f	present	1.5	none	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.03
g	intermediate 1/3	1	10 yr; 0.2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
h	intermediate 1/3	1.2	10 yr; 0.2	0.00	0.00	0.00	0.00	0.00	0.02	0.04	0.06
i	intermediate 1/3	1.5	10 yr; 0.2	0.00	0.00	0.00	0.00	0.05	0.25	0.34	0.44
j	intermediate 1/3	2	10 yr; 0.2	0.00	0.00	0.00	0.00	0.33	0.84	0.89	0.92
k	intermediate 1/3	1.5	20 yr; 0.1	0.00	0.00	0.00	0.00	0.05	0.30	0.42	0.52
l	intermediate 1/3	1.5	none	0.00	0.00	0.00	0.00	0.05	0.38	0.54	0.63
m	intermediate 2/3	1	10 yr; 0.2	0.00	0.00	0.00	0.00	0.01	0.06	0.08	0.14
n	intermediate 2/3	1.2	10 yr; 0.2	0.00	0.00	0.00	0.00	0.05	0.28	0.39	0.49
o	intermediate 2/3	1.5	10 yr; 0.2	0.00	0.00	0.00	0.00	0.25	0.75	0.84	0.88
p	intermediate 2/3	2	10 yr; 0.2	0.00	0.00	0.00	0.00	0.71	0.98	0.99	1.00
q	intermediate 2/3	1.5	20 yr; 0.1	0.00	0.00	0.00	0.00	0.27	0.80	0.88	0.91
r	intermediate 2/3	1.5	none	0.00	0.00	0.00	0.00	0.30	0.89	0.94	0.96
s	past	1	10 yr; 0.2	0.00	0.00	0.00	0.00	0.05	0.29	0.42	0.52
t	past	1.2	10 yr; 0.2	0.00	0.00	0.00	0.00	0.20	0.68	0.78	0.83
u	past	1.5	10 yr; 0.2	0.00	0.00	0.00	0.00	0.52	0.94	0.96	0.97
v	past	2	10 yr; 0.2	0.00	0.00	0.00	0.00	0.89	1.00	1.00	1.00
w	past	1.5	20 yr; 0.1	0.00	0.00	0.00	0.00	0.56	0.95	0.97	0.98
x	past	1.5	none	0.00	0.00	0.00	0.00	0.61	0.99	1.00	1.00

Table 4.3. Proportions of 2000 simulated population trajectories that either go extinct or meet the recovery target within 10, 20, 30 and 50 year time horizons based on harm scenarios for the Nashwaak River Atlantic Salmon population. The marine scenarios reflect changes from the present levels (2000's) of at-sea survival, by decreasing this 10%, 30%, and 50%. The freshwater scenarios reflect decreases in freshwater productivity from the present level by 10%, 20%, 30% and 50%. The lettering for the runs corresponds to those in Figure 4.4. Extreme events are set to occur once every 10 years with a bad year scalar of 0.2.

Run	Marine Harm	Freshwater Harm	Proportion Extinct				Proportion Recovered			
			10 yr	20 yr	30 yr	50 yr	10 yr	20 yr	30 yr	50 yr
a	present	Present	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00
b	present	10%	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00
c	present	20%	0.00	0.00	0.02	0.38	0.00	0.00	0.00	0.00
d	present	30%	0.00	0.00	0.11	0.85	0.00	0.00	0.00	0.00
e	present	50%	0.00	0.20	0.88	1.00	0.00	0.00	0.00	0.00
f	10%	Present	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00
g	10%	10%	0.00	0.00	0.02	0.34	0.00	0.00	0.00	0.00
h	10%	20%	0.00	0.00	0.08	0.78	0.00	0.00	0.00	0.00
i	10%	30%	0.00	0.02	0.31	0.98	0.00	0.00	0.00	0.00
j	10%	50%	0.00	0.41	0.98	1.00	0.00	0.00	0.00	0.00
k	30%	Present	0.00	0.00	0.11	0.85	0.00	0.00	0.00	0.00
l	30%	10%	0.00	0.02	0.31	0.98	0.00	0.00	0.00	0.00
m	30%	20%	0.00	0.07	0.64	1.00	0.00	0.00	0.00	0.00
n	30%	30%	0.00	0.24	0.91	1.00	0.00	0.00	0.00	0.00
o	30%	50%	0.01	0.90	1.00	1.00	0.00	0.00	0.00	0.00
p	50%	present	0.00	0.03	0.45	1.00	0.00	0.00	0.00	0.00
q	50%	10%	0.00	0.10	0.74	1.00	0.00	0.00	0.00	0.00
r	50%	20%	0.00	0.28	0.93	1.00	0.00	0.00	0.00	0.00
s	50%	30%	0.00	0.57	0.99	1.00	0.00	0.00	0.00	0.00
t	50%	50%	0.02	0.98	1.00	1.00	0.00	0.00	0.00	0.00

9.0 FIGURES

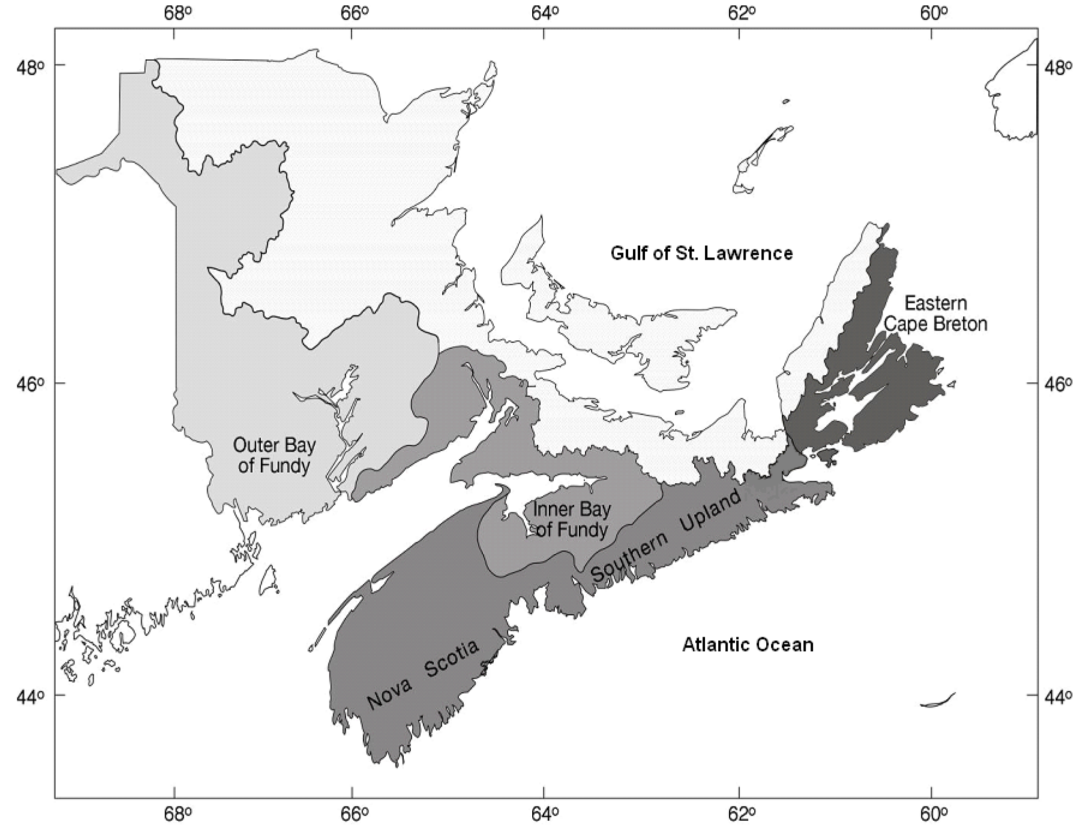

Figure 1. Map showing the location of the Outer Bay of Fundy relative to the three other designatable units for Atlantic Salmon in the Maritimes.

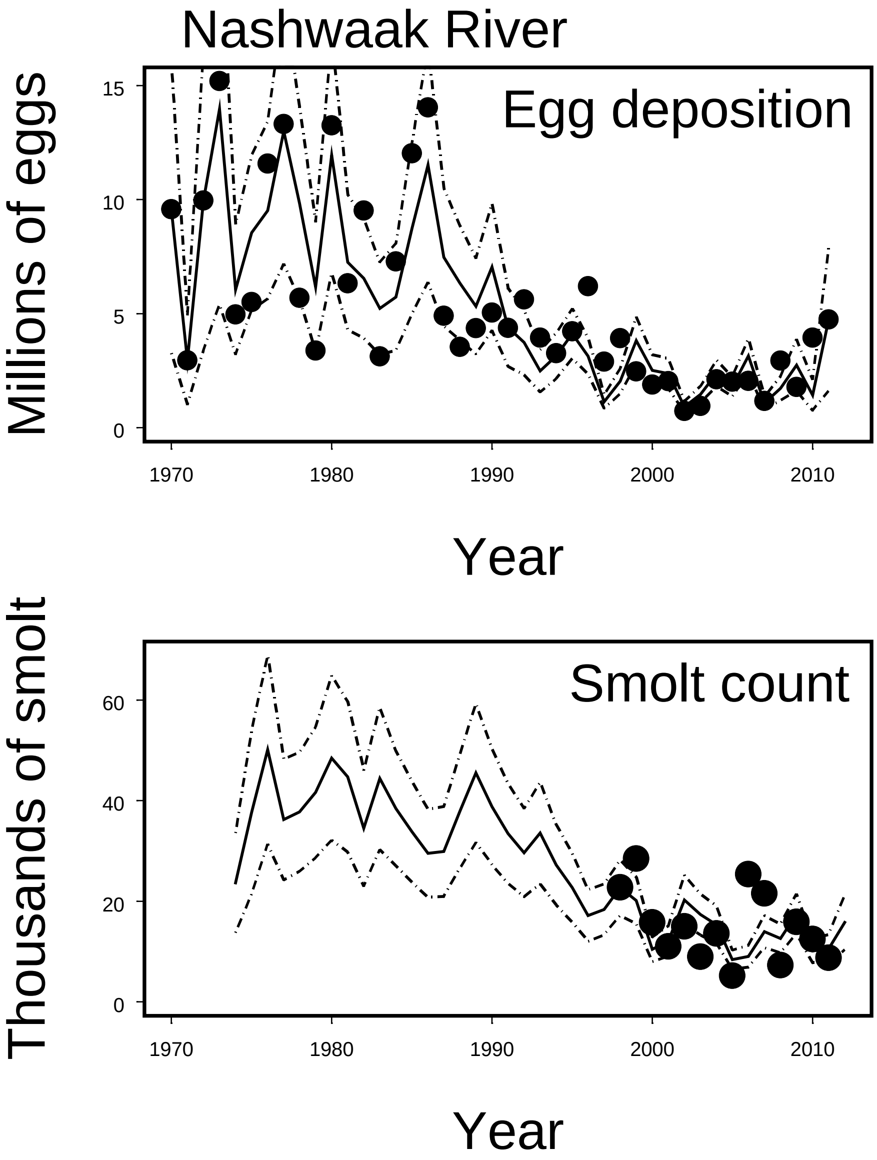

Figure 2.1. Observed (points) and fitted (lines) of egg depositions and smolt counts for the Atlantic Salmon populations in the Nashwaak River as estimated with the life history model. The broken lines show 95% confidence intervals based on normal approximations.

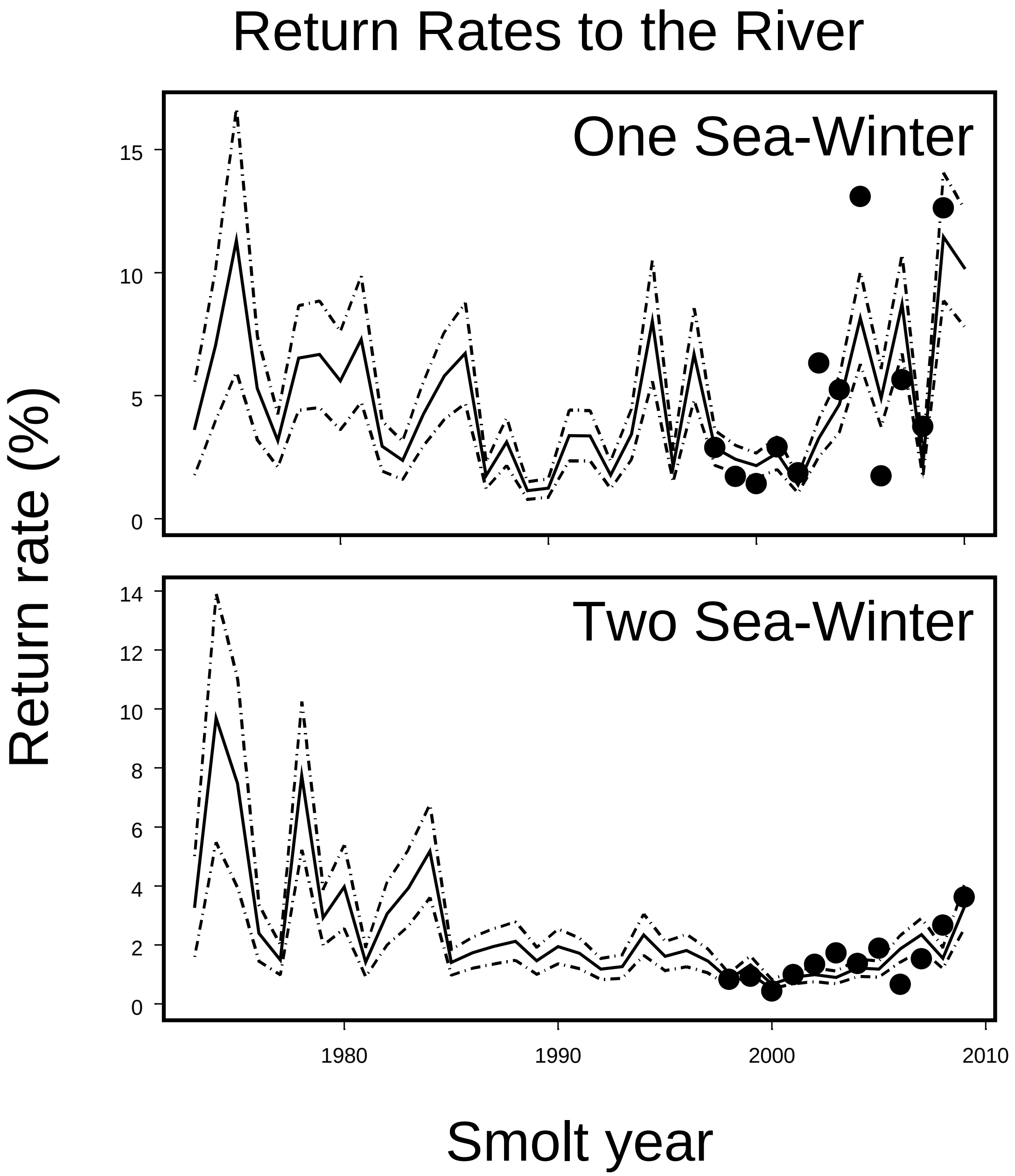

Figure 2.2. Observed (points) and estimated (lines) return rates for one sea-winter and two sea-winter wild Atlantic Salmon for the Nashwaak population, as estimated with the life history model. The broken lines show 95% confidence intervals based on normal approximations.

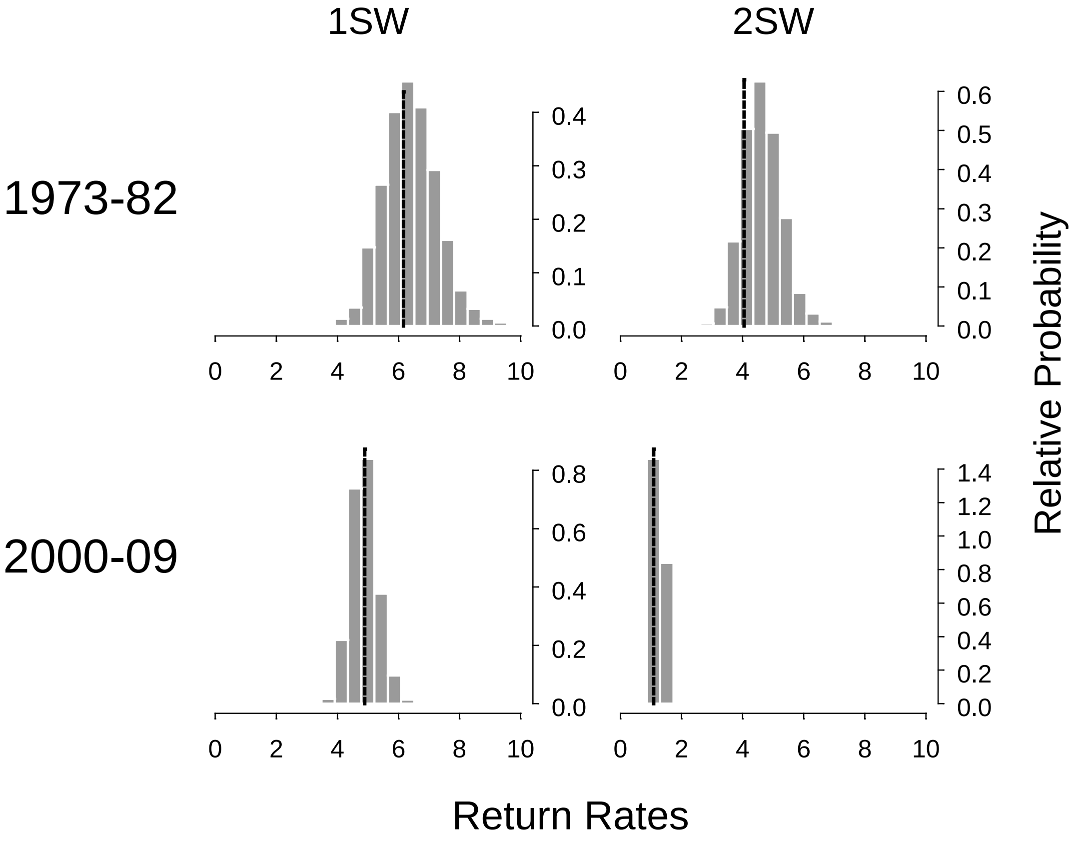

Figure 2.3. Bayesian posterior probability densities for the average return rates of one sea-winter (left column) and two sea-winter (right column) during the 1973 to 1982 (top row) and 2000 to 2009 (bottom row) time periods for the Nashwaak River wild Atlantic Salmon population. The return rates are as estimated to the mouth of the river. The time periods refer to the years of smolt production.

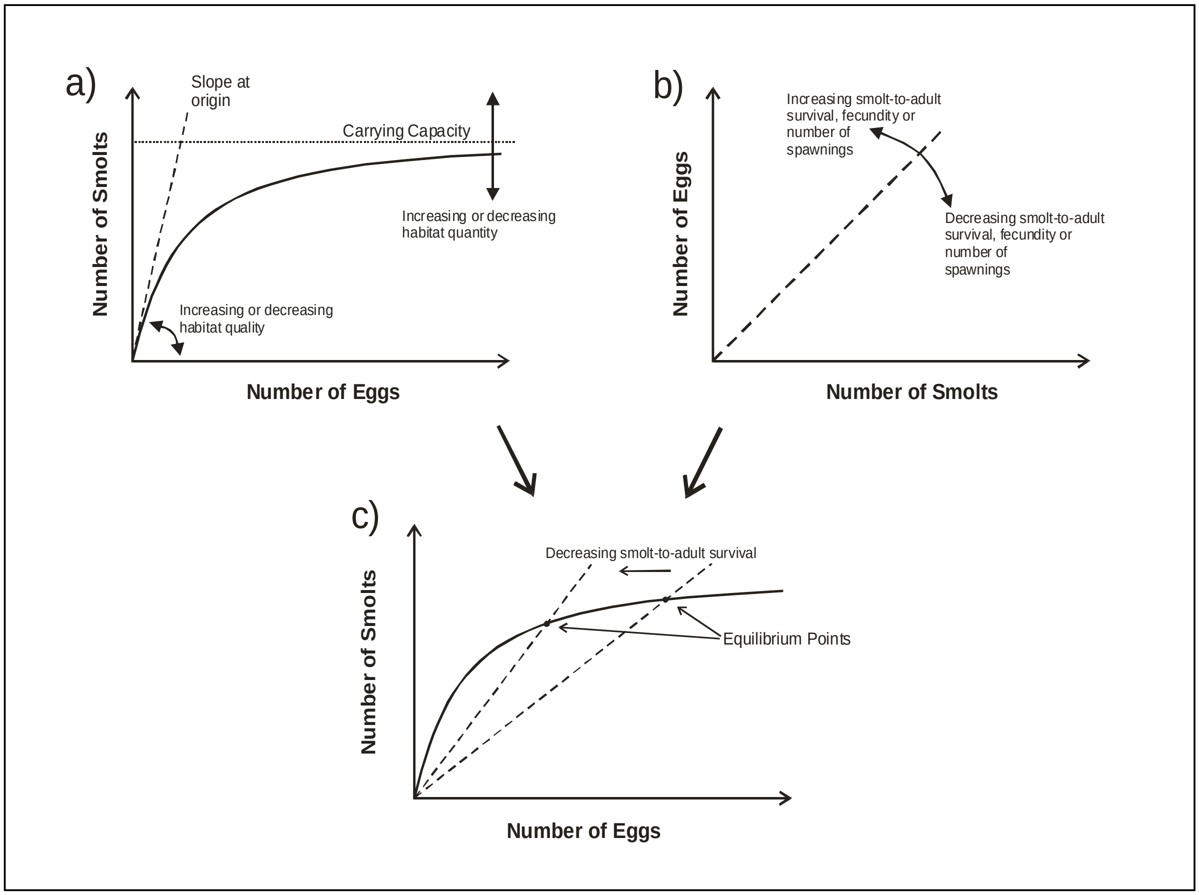

Figure 2.4. Conceptual diagram showing how an equilibrium model can be used to analyze the dynamics of a fish population and to determine how a population will respond to either changes in life history parameter values or recovery actions. A Beverton-Holt model (a) is used to model the density-dependent relationship for survival from eggs to smolt. The slope at the origin of this model, which is the maximum number of smolts produced per egg in the absence of density dependent effects, changes as habitat quality changes, whereas changes in the amount of habitat changes the carrying capacity. The number of eggs produced per smolt throughout its life (b) changes with smolt-to-adult survival, fecundity, age-at-maturity or the number of time a fish spawns throughout its life. The population equilibrium (c) occurs at the population size where the production of smolts by eggs is equal to the production of eggs by smolts throughout their lives, and is the size at which the population will stabilize if all life history rates and the habitat carrying capacity remain unchanged. The population equilibrium changes as the values of the life history parameters change (from Gibson and Amiro 2007).

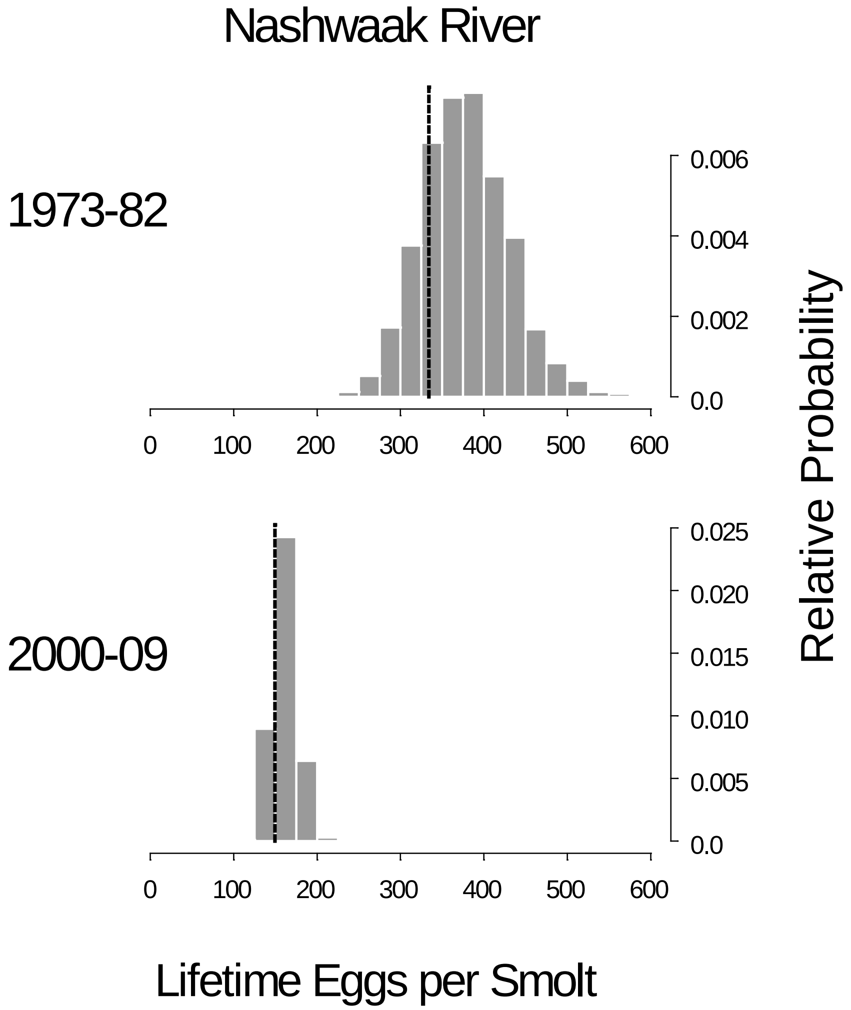

Figure 2.5. Bayesian posterior probability densities for the average numbers of eggs produced by a smolt throughout its life during the 1973 to 1982 (top row) and 2000 to 2009 (bottom row) time periods for the Nashwaak River wild Atlantic Salmon population. The vertical dashed lines show the maximum likelihood estimates from the model. The return rates are as estimated to the mouth of the river. The time periods refer to the years of smolt production.

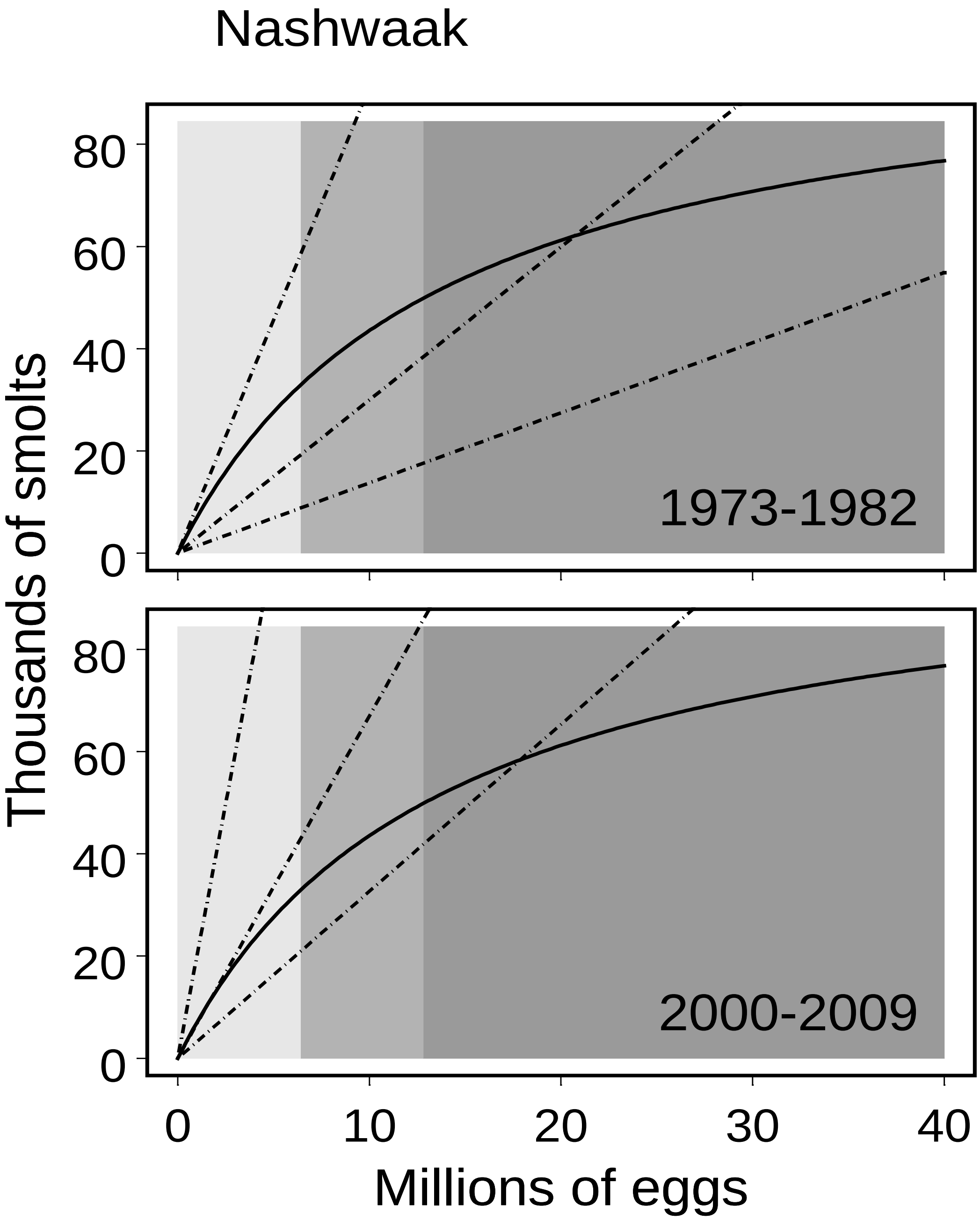

Figure 2.6. Equilibrium analysis of the dynamics of the Atlantic Salmon population in the Nashwaak River. The points are the observed egg depositions and smolt production for the 1973 to 1982 (top panel) and the 2000 to 2009 (lower panel) egg deposition years. The curved, solid line represents freshwater production. The straight, dashed lines represent marine production as calculated at the minimum observed return rates, the mean observed return rates, and the maximum observed return rates for 1SW and 2SW adults during the two time periods. Dark shading indicates egg depositions above the conservation egg requirement, medium shading is between 50% and 100% the egg requirement, and the light shading is below 50% of the requirement.

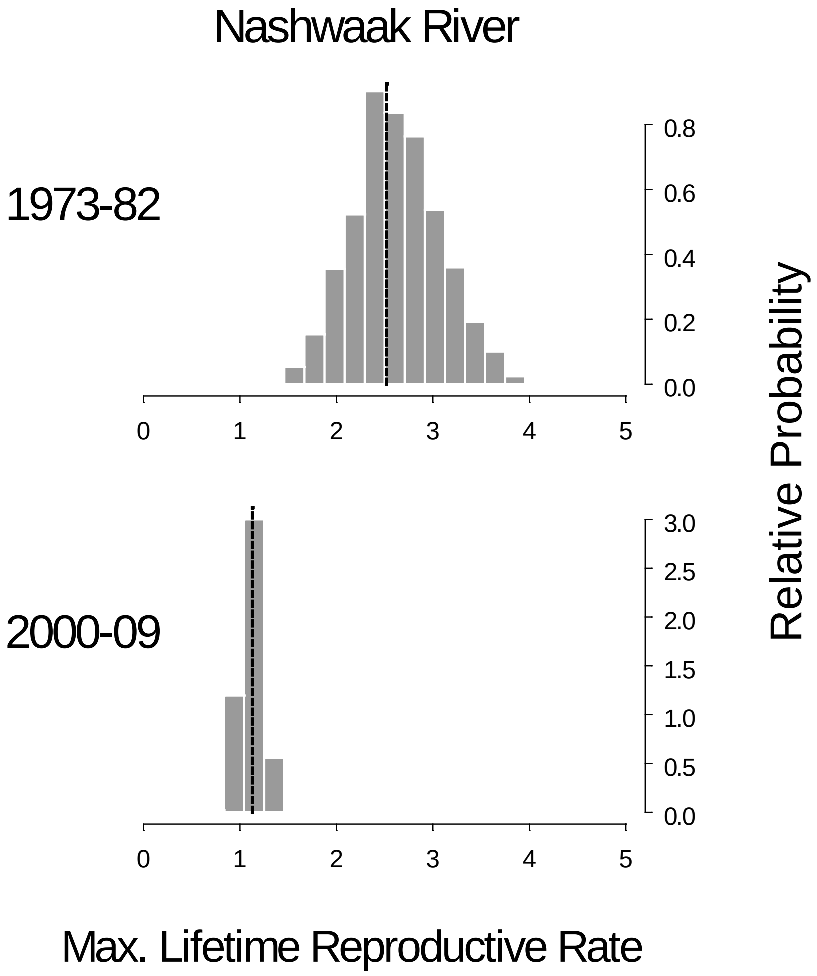

Figure 2.7. Bayesian posterior probability densities for the maximum lifetime reproductive rate during the 1973 to 1982 (top row) and 2000 to 2009 (bottom row) time periods for the Nashwaak River wild Atlantic Salmon population. The vertical dashed lines show the maximum likelihood estimates from the model. The return rates are as estimated to the mouth of the river. The time periods refer to the years of smolt production.

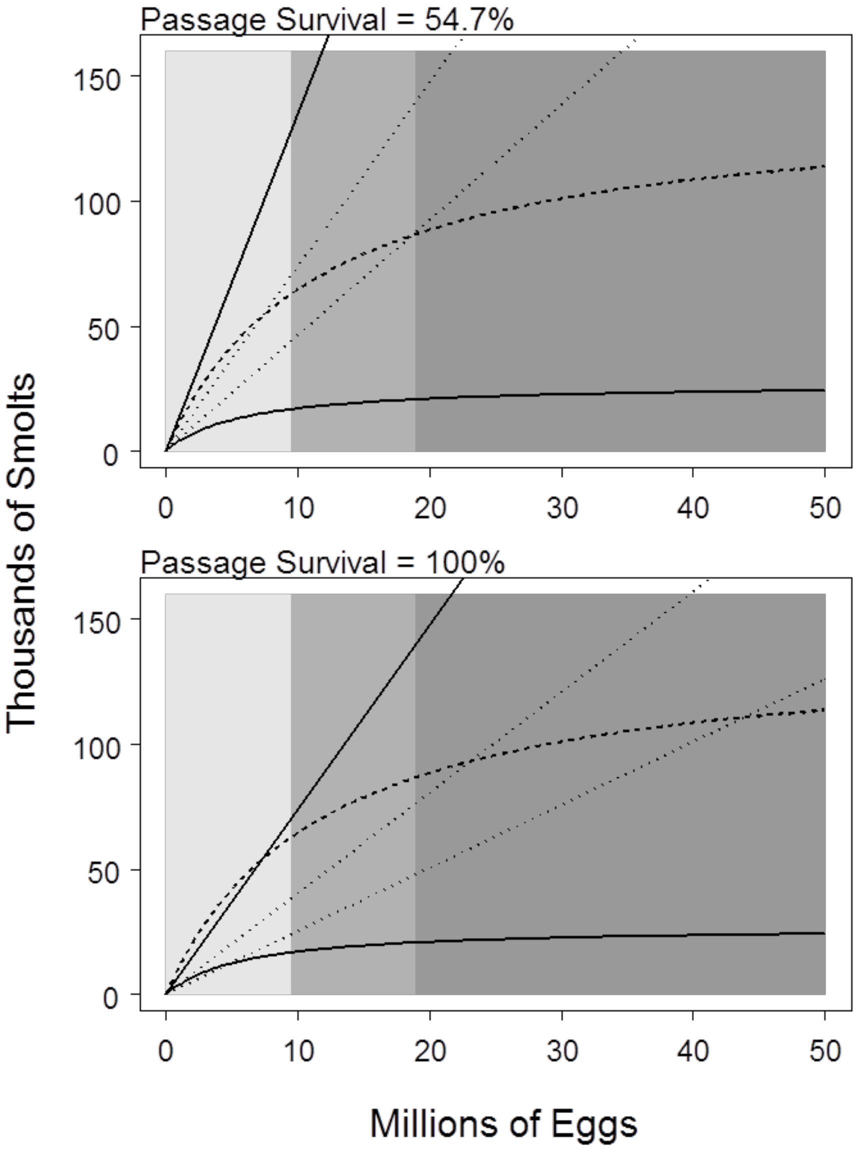

Figure 2.8. Equilibrium analysis of the Tobique River salmon population dynamics given current fish passage mortality of 45.3% (top panel) and a scenario if fish passage mortality was reduced to zero (lower panel). In both panels, the curved, solid line is the current estimated freshwater production and current marine production is shown by the solid straight line. The curved dashed line is the hypothetical scenario where the carrying capacity for age 1 parr is doubled to 18.6 parr/100 m^2 and survival of parr age 1 and older is doubled to 0.49 per year. The middle dashed line is the marine production calculated using the maximum observed rates on the Nashwaak, and the dashed line on the right represents a hypothetical scenario of 8% and 3% return rates for 1SW and 2SW salmon respectively. Dark shading indicates egg depositions above the conservation egg requirement, medium shading is between 50% and 100% the egg requirement, and the light shading is below 50% of the requirement (from Gibson et al. 2009).

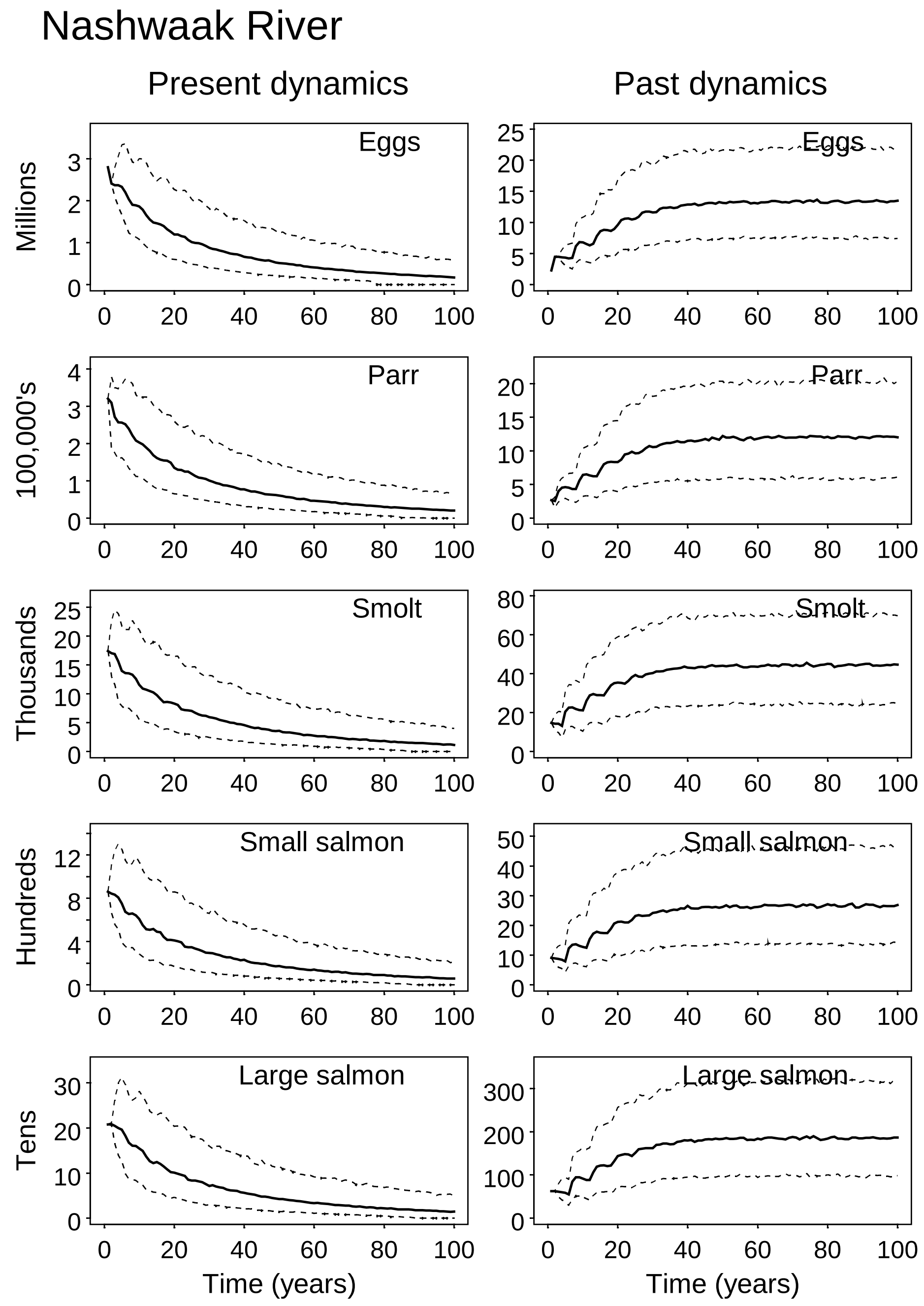

Figure 3.1. Simulated median abundance (solid line) with the 10th and 90th percentiles (dashed lines) for each of five life stages from Monte Carlo simulations of the Nashwaak River Atlantic Salmon population viability model. Two scenarios are shown, one based on the 1973-1982 dynamics (right panels) and one based on the 2000's dynamics (left panels). The graphs summarize 2000 simulations for each scenario.

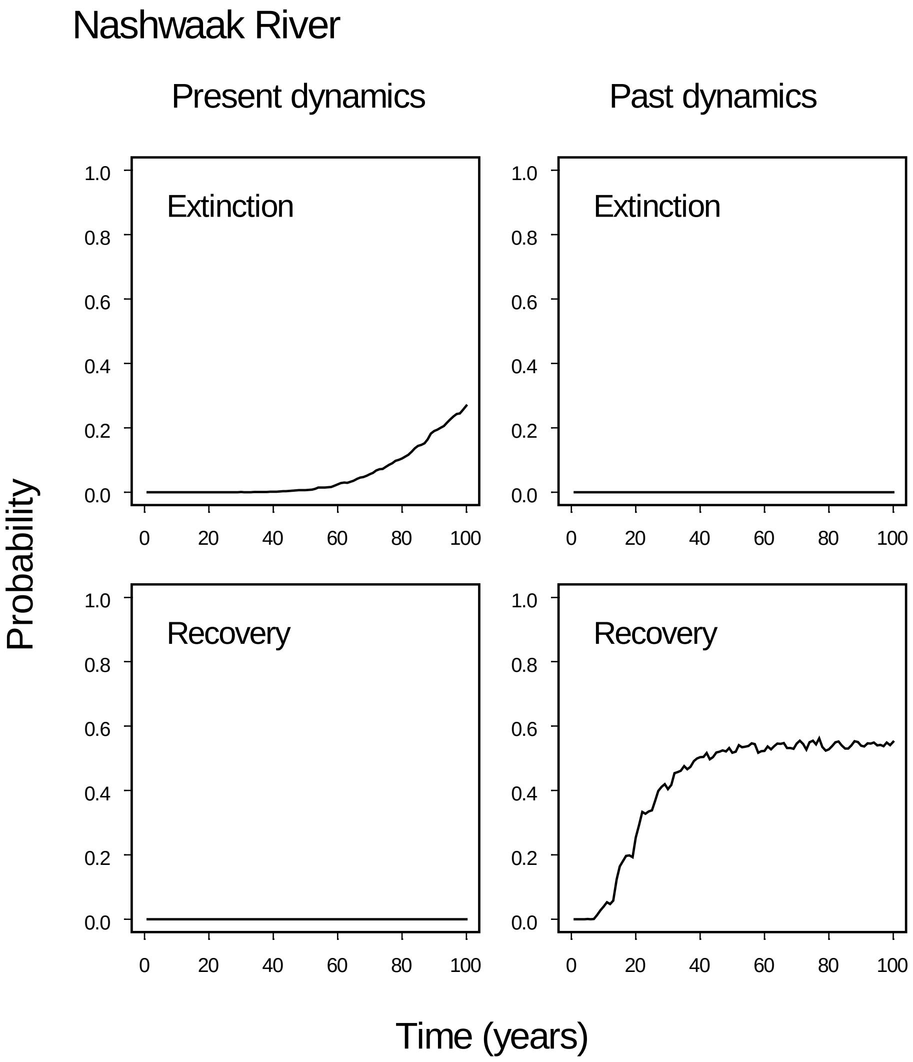

Figure 3.2. The probability of extinction and the probability of recovery as a function of time for the Nashwaak River Atlantic Salmon population. Two scenarios are shown, one based on the 1973-1982 dynamics (right panels) and one based on the 2000's dynamics (left panels). Probabilities are calculated as the proportion of 2000 Monte Carlo simulations of population trajectories that either when extinct or met the recovery target.

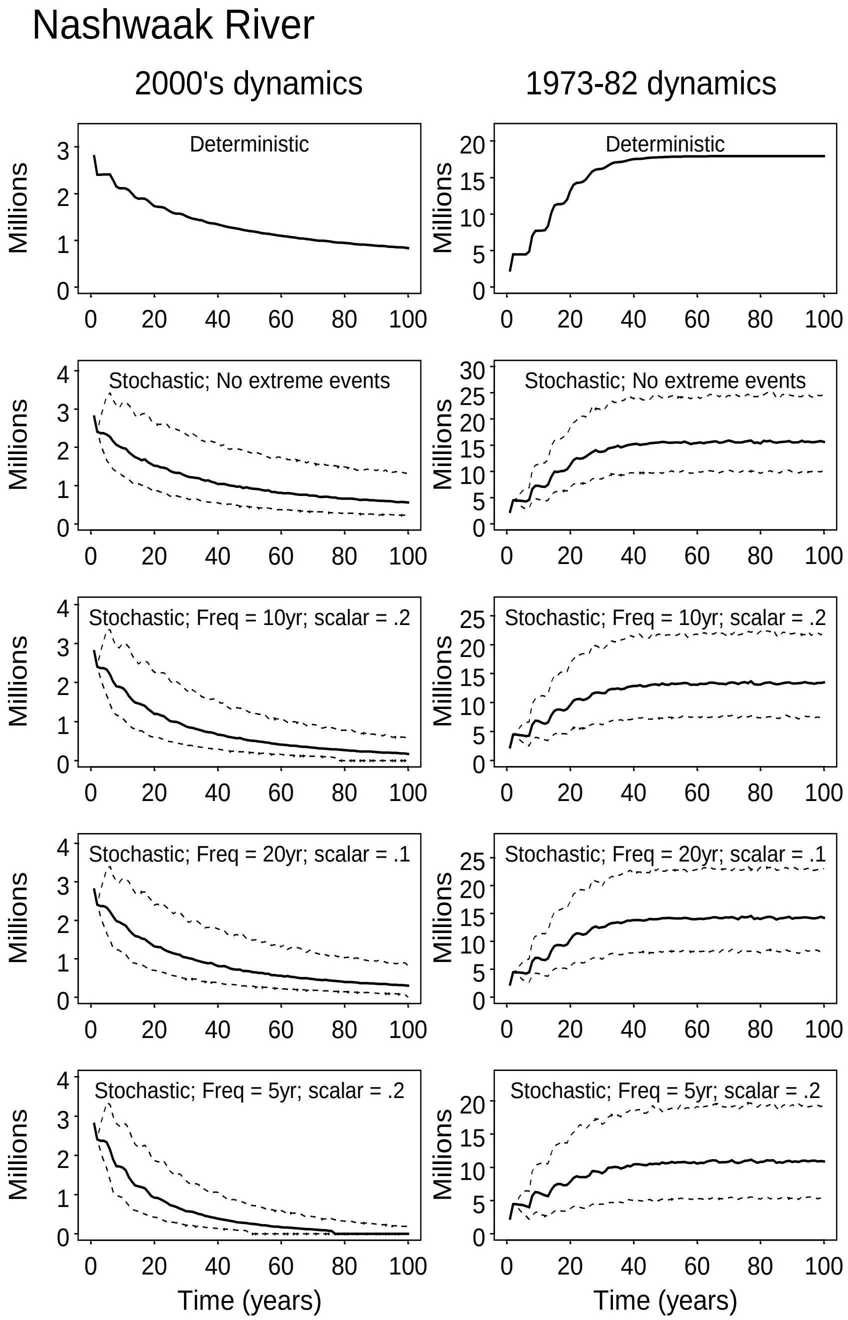

Figure 3.3. Sensitivity analysis of the effects of extreme events on the viability of Nashwaak River Atlantic Salmon. The graphs summarize 2000 simulations for each scenario. The median abundance (solid line), and the 10th and 90th percentiles (dashed lines) are shown. Panels on the right and left are based on the 1973-1982 dynamics and 2000's dynamics respectively. The top row shows a deterministic run without extreme events, the second row a stochastic run without extreme events, the third row a stochastic run with extreme events (the base model), the fourth row the effect of decreasing the frequency and increasing the magnitude of extreme events, and the bottom row the effect of increasing the frequency of extreme events.

Nashwaak River

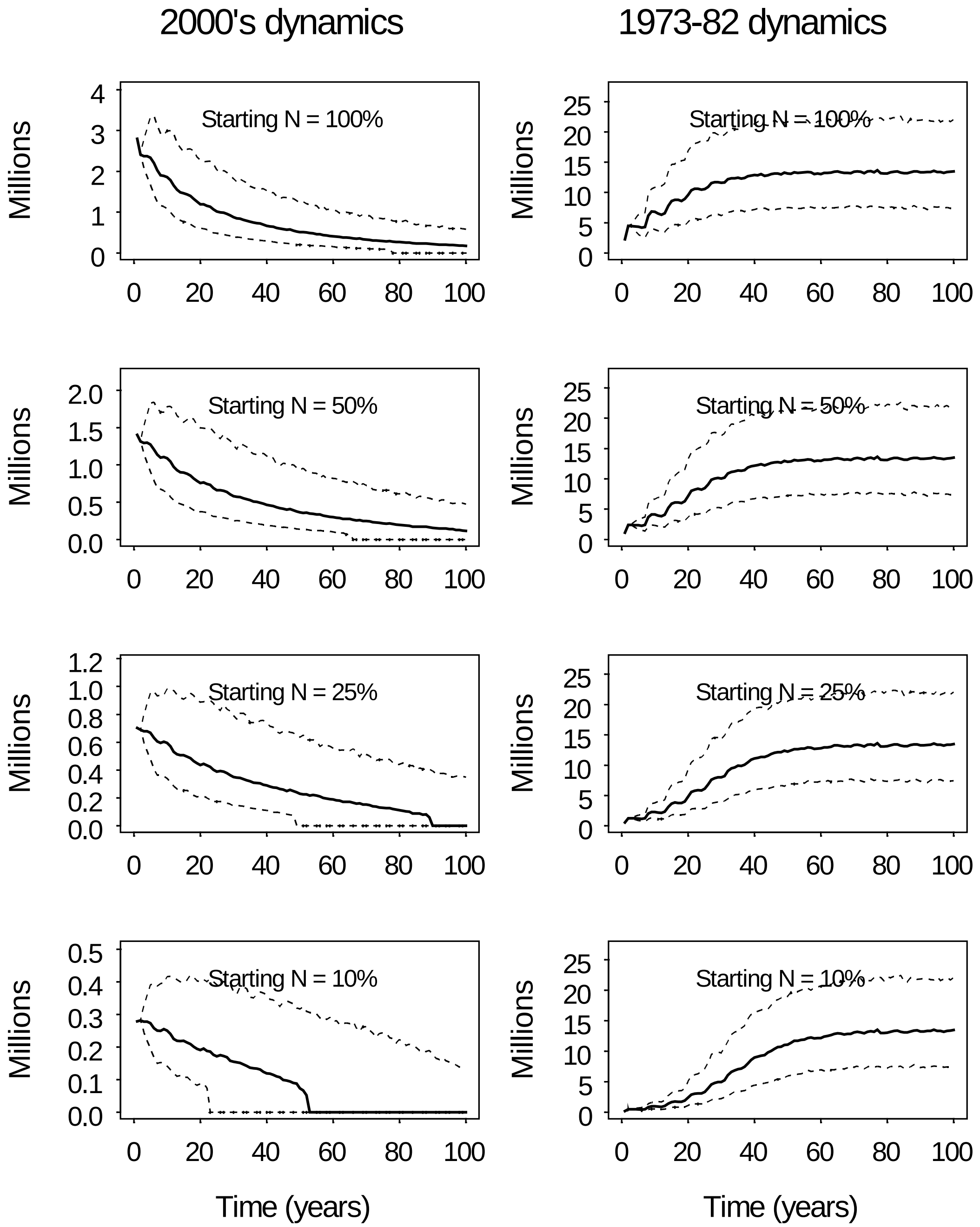

Figure 3.4. The effect of further reductions in population size on the abundance trajectories using base model for the Nashwaak River Atlantic Salmon population. The graphs summarize 2000 simulations for each scenario. The median abundance (solid line), and the 10th and 90th percentiles (dashed lines) are shown. Panels on the right and left are based on the 1973-1982 dynamics and 2000's dynamics respectively. The top row shows the trajectories using the 2008-2012 average abundance estimate (896 small salmon and 263 large salmon) as the starting population size. The other rows show the effects of starting at 50%, 25% and 10% of the starting abundance respectively.

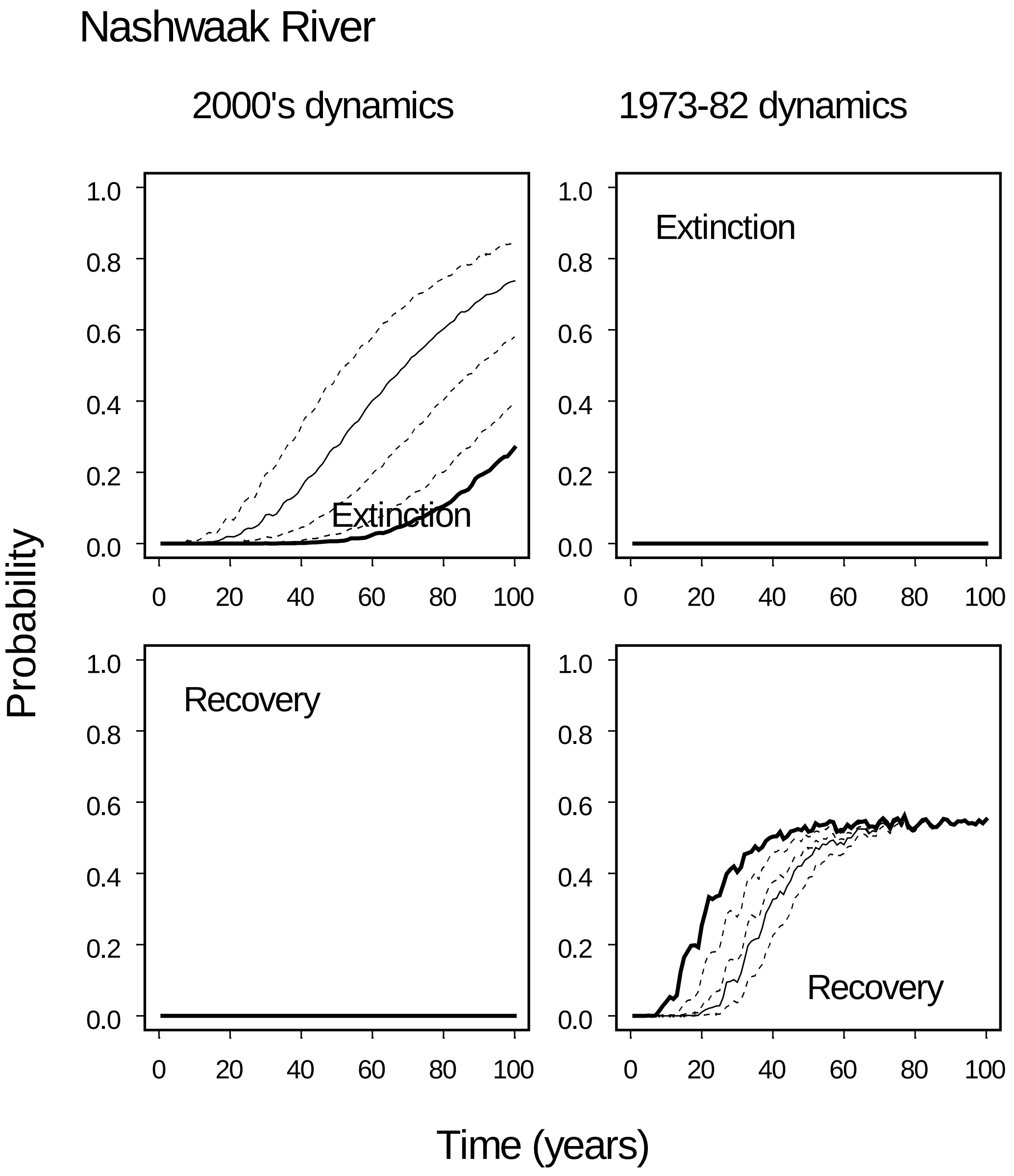

Figure 3.5. The probability of extinction (top row) and the probability of recovery (bottom row) as a function of time for the Nashwaak River Atlantic Salmon population showing the effects of further reductions in population size. Scenarios are based on the 1973-1982 dynamics (right panels) and on the 2000's dynamics (left panels). The thick solid lines show the probabilities when the starting population size is the 2008-2012 average abundance estimate (896 small salmon and 263 large salmon). The other lines show the effects of starting at 50%, 25% and 10% of this abundance (moving away from the solid line, respectively). The 2012 abundance estimates (29 small salmon and 63 large salmon) produce the thin solid line.

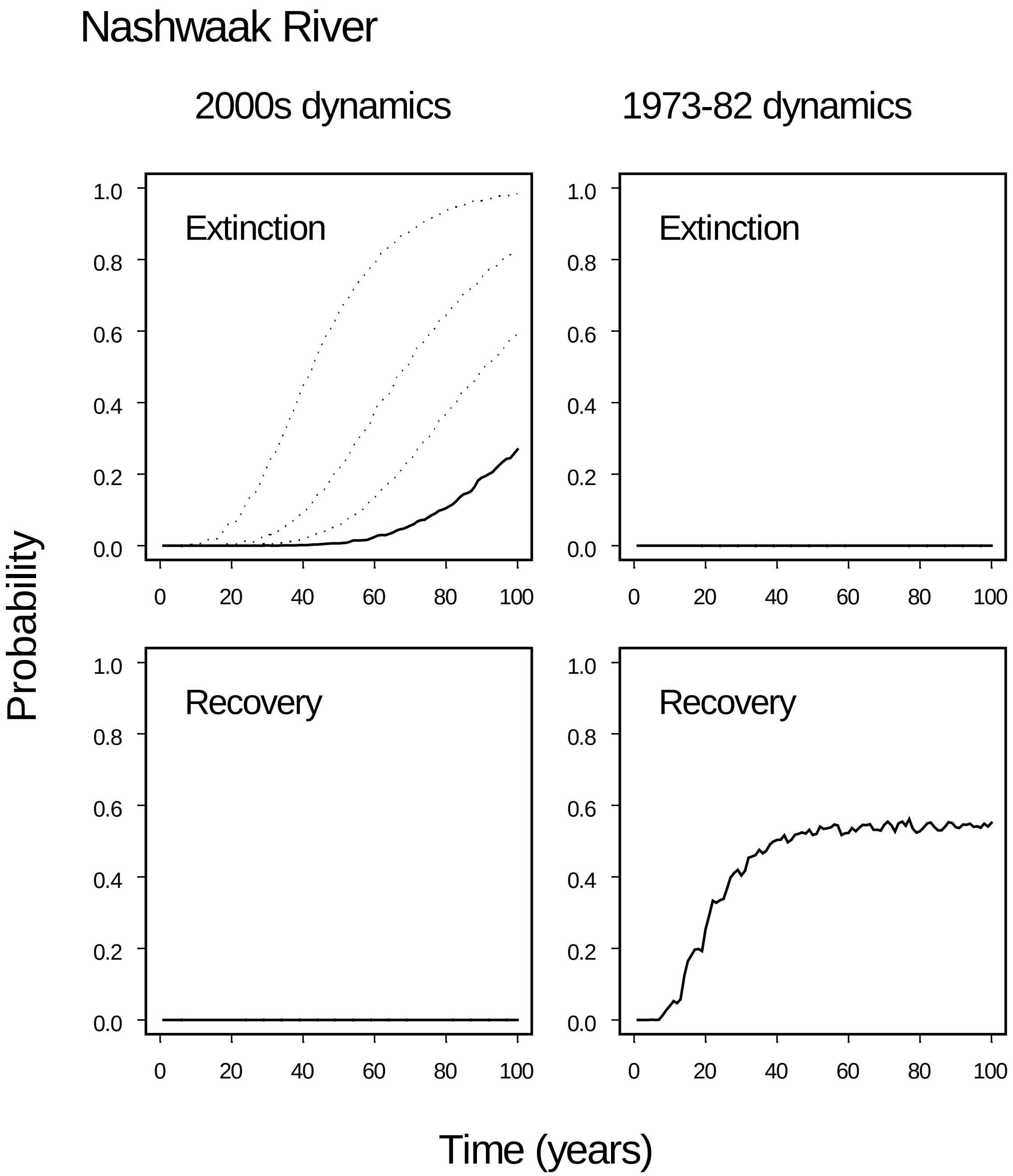

Figure 3.6. Sensitivity analyses showing the effect of the quasi-extinction threshold on the probability of extinction (top row) and the probability of recovery (bottom row) for the Nashwaak River Atlantic Salmon population. Scenarios are based on the 1973-1982 dynamics (right panels) and on the 2000's dynamics (left panels). The solid lines show the probabilities when the quasi-extinction threshold is set at 15 females. The dashed lines show the effects of setting the threshold at 30, 50 and 100 females (moving away from the solid line, respectively).

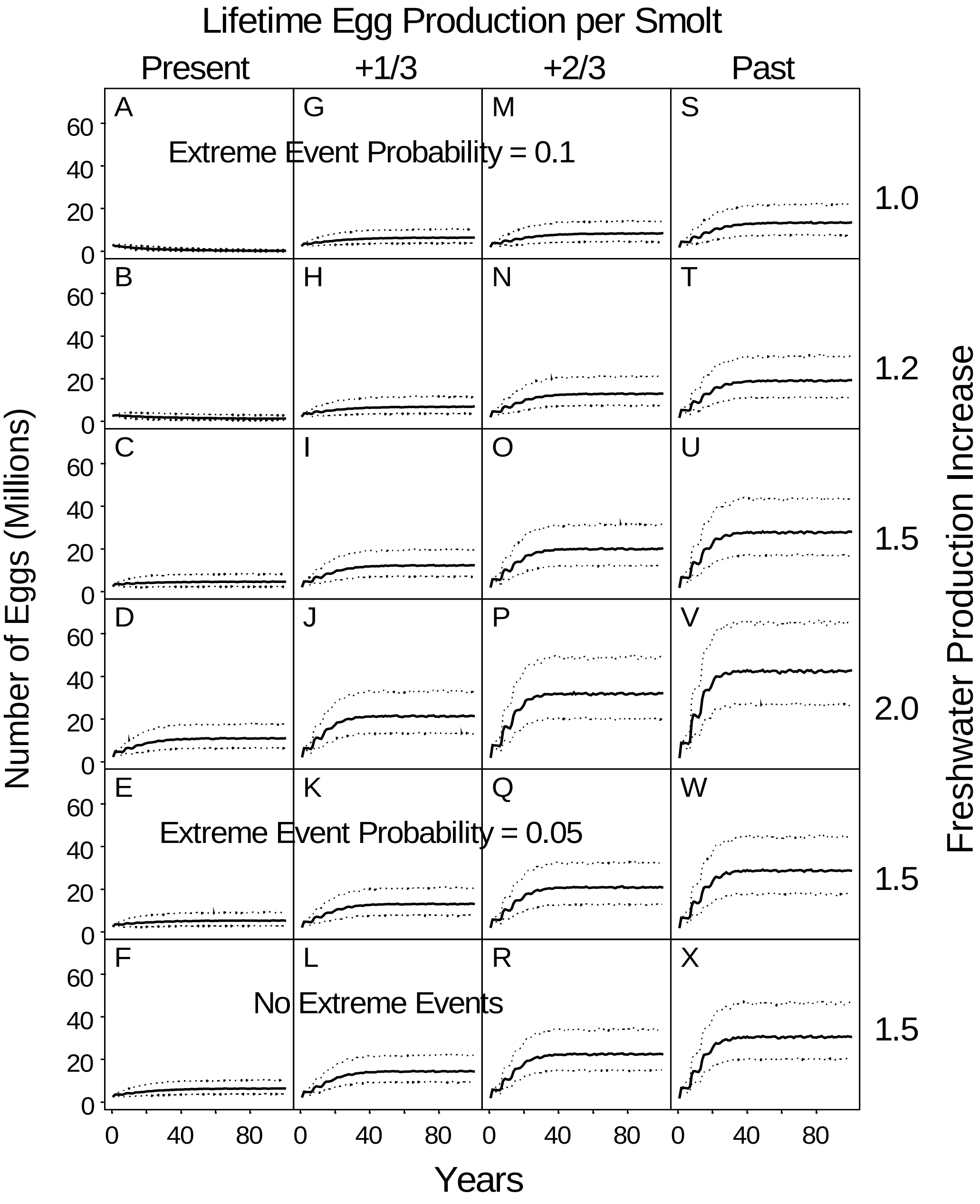

Figure 4.1. The effects of increasing at-sea survival and freshwater productivity on the simulated abundance of eggs for the Nashwaak Atlantic Salmon population. The graphs summarize 2000 simulations for each scenario. The median abundance (solid line), and the 10th and 90th percentiles (dashed lines) are shown. Panels on the right and the left are based on the 1973-1982 (past) and 2000's (present) at-sea survival respectively, and the two middle panels show scenarios using survivals increased by 1/3 and 2/3's of the difference in these values. The return rates of 1SW and 2SW salmon and survival between repeat spawning events are increased. The 2000's freshwater production is used in all scenarios. The top four rows show the effect of increasing freshwater productivity by factors of 1 (no change), 1.2 (20% increase), 1.5 (50% increase) and 2.0 (100% increase). The bottom two rows show the effect of changing the frequency of event events to an average of 1 every 20 years (5th row) and to no extreme events (bottom row).

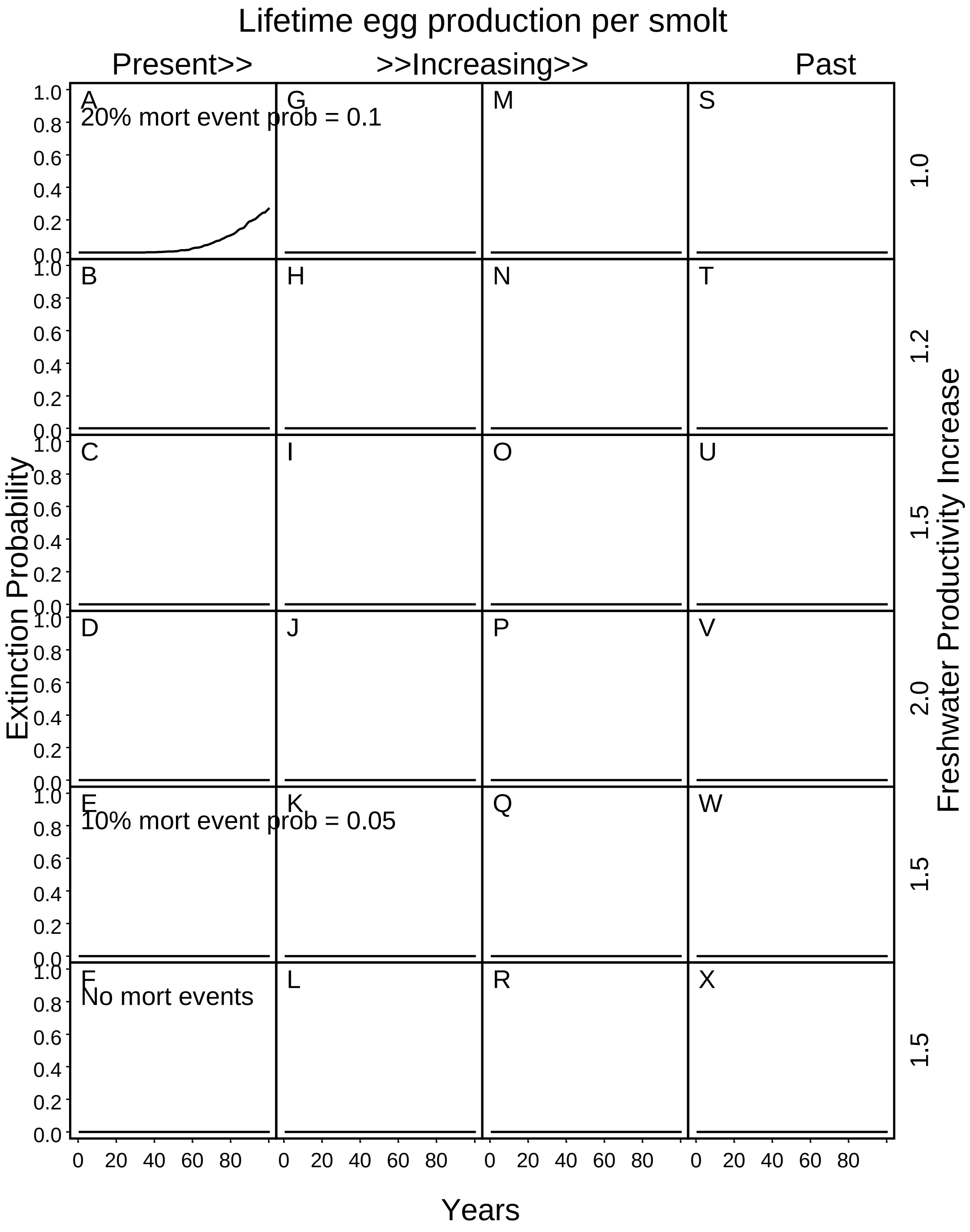

Figure 4.2. The effects of increasing at-sea survival and freshwater productivity on the probability of extinction for the Nashwaak River Atlantic Salmon population. Panels are described in the caption for Figure 4.1.

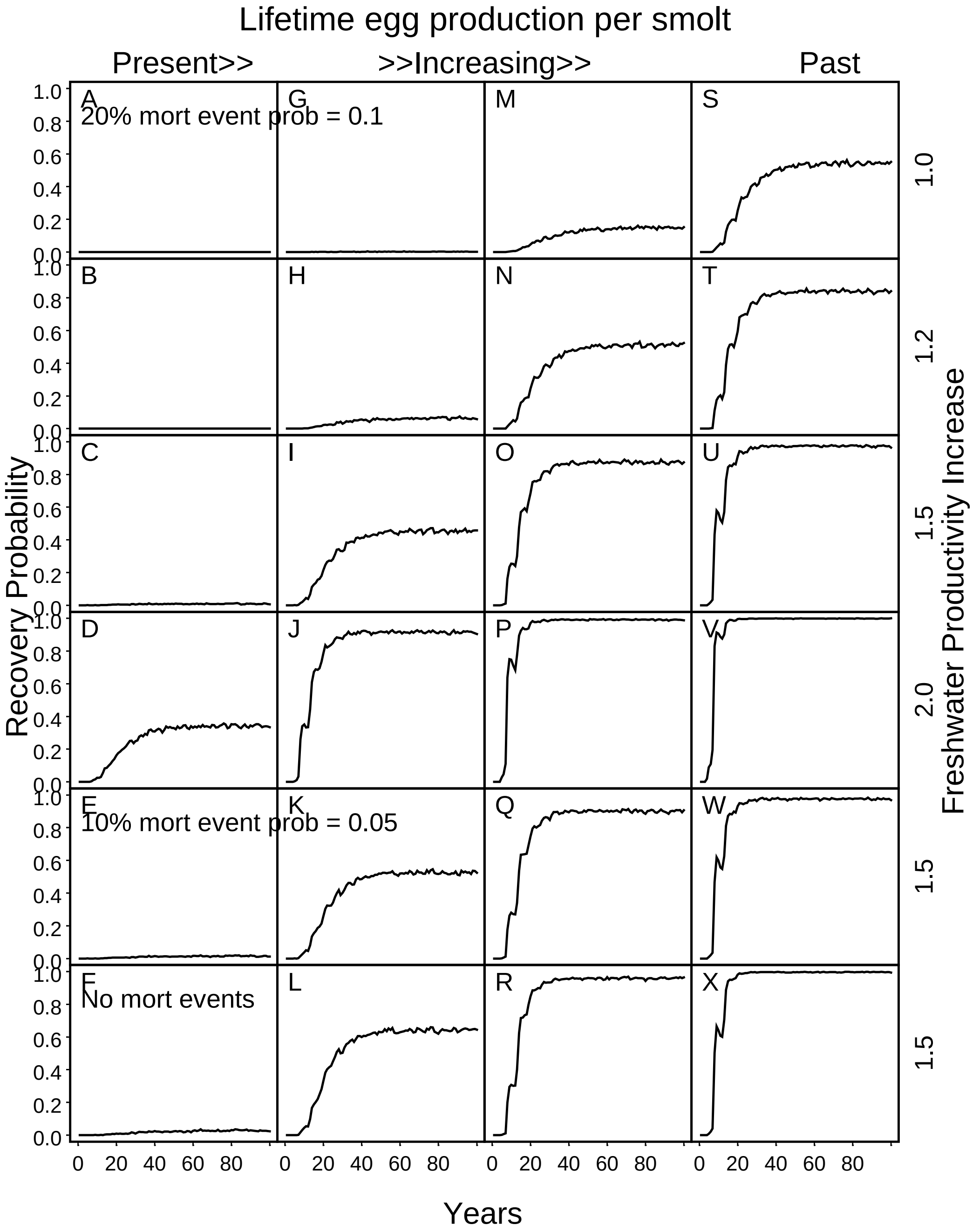

Figure 4.3. The effects of increasing at-sea survival and freshwater productivity on the probability of meeting the recovery target for the Nashwaak River Atlantic Salmon population. Panels are described in the caption for Figure 4.1.

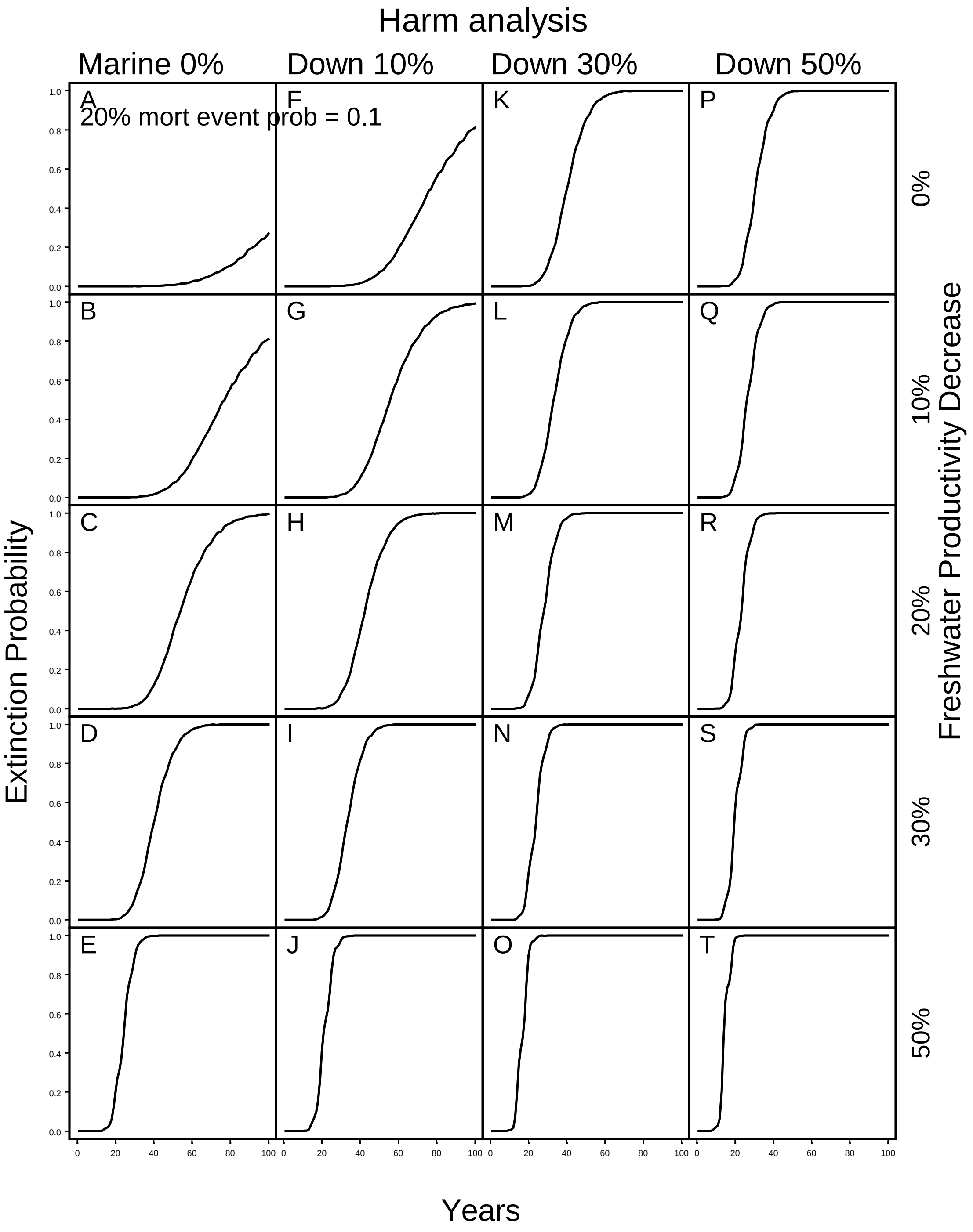

Figure 4.4. The effects of decreasing at-sea survival and freshwater productivity on the extinction probability for the Nashwaak Atlantic Salmon population. The graphs summarize 2000 simulations for each scenario. The median abundance (solid line), and the 10th and 90th percentiles (dashed lines) are shown. Panels on the left are based on the 2000's (present) at-sea survival, and the panels moving to the right show the effects of decreasing survival by 10%, 30% and 50% from the present values. The rows show the effect of decreasing freshwater productivity by factors of 1 (0% decrease), 0.9 (10% decrease), 0.8 (20% decrease), 0.7 (30% decrease) and 0.5 (50% decrease).

10.0 APPENDICES

APPENDIX 1. THE STATISTICAL, LIFE-HISTORY BASED POPULATION DYNAMICS MODEL USED FOR ESTIMATING LIFE HISTORY PARAMETER VALUES AND ANALYSING THE DYNAMICS OF THE NASHWAAK RIVER ATLANTIC SALMON POPULATION

This model is very similar to that developed by Gibson and Bowlby (2013) for analyzing the dynamics of the LaHave River and St. Mary's River Atlantic Salmon populations. Here, the model is modified to reflect differences in the available information about survival between spawning events for the Nashwaak River population and for the LaHave River population. The text in this Appendix is taken almost verbatim from Gibson and Bowlby (2013).

The life history parameter estimates provided in Section 2.2, as well as the information on population dynamics in Section 2.5, were derived using a statistical, life-history-based population dynamics model developed by Gibson et al. (2008b; 2009). The method follows the general theory developed by Fournier and Archibald (1982) and Deriso et al. (1985) for statistical catch-at-age models for stock assessment that allows auxiliary data to be incorporated for model fitting. The approach used here is similar in that multiple indices (auxiliary data) are used to derive estimates of the age- and stage-specific abundances and survival rates required to analyze the dynamics of these populations. The life history parameter estimates are then used to determine how recovery actions may be expected to change population size and viability.

As described in Section 2, the population dynamics model consists of two parts: a freshwater production model that provides estimates of the expected smolt production as a function of egg deposition, and a lifetime egg-per-smolt model that provides estimates of the rate at which smolts produce eggs throughout their lives. These components are combined via an equilibrium analysis that provides estimates of the abundance at which the population would stabilize if the input parameters remained unchanged. This combined model is then used to evaluate how equilibrium population size has changed through time, as well as how the population would be expected to change in response to changes in carrying capacity, survival, or life stage transition probabilities, as described in Section 2.5.

The structure of the population dynamics model (freshwater production model component and the lifetime egg-per-smolt model component) is described in Sections A1.1 and A1.2. The equilibrium model is described in Section A1.3. The statistical procedures used for parameter estimation and model fitting are described in Section A1.4.

A1.1 Freshwater Component of Life Cycle (Eggs to Smolts)

Model indices and parameter definitions for the freshwater component of the model are provided in Table A1.1 and the equations for characterizing dynamics in fresh water are provided in Table A1.2. A description of this model follows below.

The number of age-0 juveniles (or fry), at the time of the electrofishing surveys in the summer, is a function of egg deposition in the previous fall (calculated from total adult escapement in each year) multiplied by the egg to age-0 survival rate (Equation 1, Table A1.2).

Density dependence was incorporated into the model via survival from age-0 to age-1 using a Beverton-Holt function, based on the results of Gibson (2006). Abundance of age-1 juveniles is a function of the maximum survival rate between age-0 and age-1, α , the asymptotic density of age-1 parr (maximum number per 100 m^2 habitat units, R_{asy}), an electrofishing catchability coefficient or scalar, h, and the probability that a fish emigrates as a smolt at age-1, j_1 (Equation 2, Table A1.2). The product, $R_{asy}h$ is the carrying capacity of the river for age-1 parr.

The model is formulated this way because the electrofishing data, used to estimate the number of parr, is reported as a density (number per 100 m^2), whereas the interest here is in the total number of parr in the river. The parameter *h*, which can be estimated within the model, is used to scale the parr density to the total abundance. Estimating the parameter, rather than using the measured number of habitat units, corrects for potential issues that would arise if the electrofishing sites fished each year were not representative of the entire river (Gibson et al. 2009).

An implicit assumption made here is that the density of all age classes of parr can be scaled up to their respective abundances using a single value of h. This assumption is made because a set of age-specific catchabilities and mortalities would be identifiable (covariance of 1) in the model without some sort of auxiliary information about one parameter or the other (*sensu* Quinn and Deriso 1999). If the electrofishing sites are selected such that one age class is over- or under-represented in the sampling, the resulting age-specific mortality estimates would be biased, although the overall freshwater production curve would likely remain representative because the annual egg depositions and smolt abundance estimates do not have the same catchability issues.

The number of age-2 and older parr is determined by the number of parr in the cohort in the previous year ($P_{t-1,a-1}$), density-independent survival of parr (M_{Parr}), and the age-specific probability of smoltification, j_a (Equation 3, Table A1.2). The number of smolt in each age and year class, $S_{t,a}$, is determined similarly (Equation 4, Table A1.2). The maximum age at smoltification was assumed to be four for the Nashwaak River population, based on the observed ages of smolts during smolt monitoring in these rivers.

By combining the life stage-specific parameter estimates into a two parameter Beverton-Holt spawner recruitment function, it is possible to describe smolt production as a function of egg deposition. This is particularly convenient for the equilibrium population size calculations below to calculate overall freshwater productivity. Both parameters, the slope of the function at the origin (the maximum number of smolts produced per egg in the absence of density dependence) and the asymptotic recruitment level for smolts (the number of smolts that would be produced in a cohort if the egg deposition was infinite), can be calculated directly from the estimated parameters (Equations 5 and 6, Table A.1.2).

A1.2 Lifetime Egg-per-smolt Model

The freshwater component of the life history model is used to characterize survival, productivity and stage-transition probabilities from the egg to the smolt stage, whereas the second part of the life history model characterizes the manner in which smolts produce eggs throughout their lives, abbreviated as EPS (for eggs-per smolt). Model indices and parameter definitions for the EPS component of the model are provided in Table A1.3 and the equations characterizing these dynamics are provided in Table A1.4.

An important demographic parameter for evaluating the potential for population recovery is the rate at which smolts return to spawn for the first time, either as 1SW or 2SW salmon. One of the limitations of Atlantic Salmon data is that the smolt abundance time series available for calculating return rates of adults are relatively short (1998- present for the Nashwaak River population) and the data are only available in recent years when abundance is low and decreasing. This makes it impossible to directly calculate return rates (indicative of marine survival) in earlier years, when abundance was higher, and to determine how population dynamics have changed such that populations are no longer viable. To address this issue, the estimated smolt abundance from the freshwater production model (described above) was used to extend the time series available for calculating return rates. One of the data inputs for the

Nashwaak River population is the estimated annual spawning escapement of large and small salmon in the Nashwaak River. Estimates of the survival from the smolt life stage through to spawning escapement for 1SW and 2SW salmon are calculated (Equation 1, Table A.1.4) by dividing the number of salmon in each sea-age class by the estimated number of smolts emigrating either one or two years earlier (using smolt abundance estimates from the freshwater production model).

The sum of the mortalities associated with the recreational fishery (as a result of either retention or hook-and-release mortality) and the escapement for each sea-age group of adults provides an estimate of the returns of 1SW and 2SW fish to the river in a given year. Exploitation rates in fresh water are calculated from these values (Equation 2, Table A.1.4). Return rates to the mouth of the river for each sea-age group are calculated from the group-specific exploitation rates and escapement estimates, as well as the estimated number of smolts emigrating either one year or two years earlier (Equation 3, Table A.1.4).

The EPS is the sum of the lifetime egg production of 1SW and 2SW salmon multiplied by their respective return rates (Equation 4, Table A.1.4). The lifetime egg production for each sea-age category is a function of their size-specific fecundity, annual survival between spawning events, maximum number of spawnings, their sea-age specific (1SW or 2SW) return rates as either alternate-year or consecutive-year repeat spawners (Equations 4.1 and 4.2, Table A.1.4) and their subsequent probabilities of returning to spawn a third time.

A1.3 Equilibrium Calculations

As discussed in Section 2.5, equilibrium models are a useful way of evaluating the effects of human activities and life history changes on fish populations. The equilibrium egg deposition and number of smolts are denoted with asterisks to differentiate them from parameters in the freshwater life history model. Similarly, $\breve{\alpha}$ and $\breve{R}_{asy}$ represent the maximum rate of population growth and equilibrium population size for smolts in freshwater, respectively. The egg and smolt equilibrium values are calculated as follows:

Equilibrium egg deposition ($Eggs*$) is:

$$Eggs* = \frac{\left(\breve{\alpha}EPS - 1\right)\breve{R}_{asy}}{\breve{\alpha}},$$

and the equilibrium number of smolts ($S*$) is found by substituting $Eggs*$ into the freshwater production model:

$$S* = \frac{\breve{\alpha}Eggs*}{1 + \dfrac{\breve{\alpha}Eggs*}{\breve{R}_{asy}}}.$$

For the full derivation, see Gibson et al. (2009).

A1.4 Parameter Estimation and Statistical Considerations

Maximum Likelihood

Parameter estimates for the freshwater production model were obtained by simultaneously fitting the model to the observed data using maximum likelihood by minimizing the value of an objective function, *OFV,* (Quinn and Deriso 1999). The *OFV* equals the sum of the negative log likelihoods for the juvenile electrofishing data ($\ell_{electro}$), the smolt age-frequency data (ℓ^{smolt}_{age}), the egg deposition data (ℓ_{egg}) and the smolt count data (ℓ_{smolt}). Lognormal error structures (Myers et al. 1995) were used for all likelihoods except the smolt age-frequency data, for which a multinomial likelihood (Quinn and Deriso 1999) was used. The objective function and the likelihood equations are provided in Table A1.5.

Estimating the variance for multiple lognormal likelihoods is problematic without other information about their variability. Gibson and Amiro (2003) had similar issues estimating the mean standard deviation of the likelihood estimates (σ) for all components of a similar model. Following their approach, the average values obtained by Myers et al. (1995) from spawner-recruit relationships of 15 populations of Atlantic Salmon were used in this analysis. For recruitment ages of 0, 1 and 2, σ equaled or averaged 0.33 (n = 1), 0.33 (n = 4) and 0.58 (n = 1), respectively, where n is the number of populations considered. Based on these results, σ was set equal to the median value, 0.33, for all age categories.

The parameters for the freshwater production model are listed in Table A1.1. The specific demographic parameters estimated are egg and parr mortality (M_{Egg} and M_{Parr}), the maximum rate of population increase for age 1 parr (α), the carrying capacity for age 1 parr (R_{asy}), and the smoltification probabilities at age (j_a). In addition, annual egg depositions for each year, and the electrofishing scalar were estimated in the model. Attempts were made to estimate parameter values for two time periods (by splitting the model and data into an earlier and later time periods), but these were not successful, likely due to a lack of contrast in the data for the two time periods.

The freshwater production model was programmed using AD Model Builder (ADMB) (Fournier 1996), which uses the C++ auto-differentiation library for rapid fitting of complex non-linear models, has Bayesian and profile likelihood capabilities, and is designed specifically for fitting these and similar types of models. The change in the Akaike Information Criterion (AIC) was used to help assess the trade-off between model fit to the data and the number of parameters in the candidate model (Hilborn and Mangel 1997). The model with the lowest AIC was generally selected as the preferred model. A similar approach was used to help select the best data input (for example, to choose whether to use standardized or un-standardized electrofishing data). The model was run using several different sets of starting values to ensure that the model was not converging at a local (rather than a global) minimum. Standard errors for parameter estimates were calculated from the variance and correlation matrix generated by the Delta method (Efron and Tibshirani 1993). This is standard output from ADMB.

Bayesian Analyses

Bayesian methods provide a powerful tool for assessing uncertainty in fisheries models (McAllister et al. 1994). Punt and Hilborn (1997) and McAllister and Kirkwood (1998) have reviewed their fisheries applications. The posterior probability distributions resulting from Bayesian analyses show the uncertainty in model or policy parameters including both estimation uncertainty, as well as prior information about their values (Walters and Ludwig 1993). ADMB uses a Markov Chain Monte Carlo (MCMC) algorithm (Carlin and Louis 1996) to approximate the posterior distribution for parameters of interest. MCMC is a stochastic simulation method used to evaluate complex integrals in order to derive posterior distributions. ADMB uses the Metropolis Hastings algorithm (Chib and Greenberg 1995) to generate the Markov chain, using a multivariate normal distribution based on the variance-covariance matrix for the model parameters as the proposal function. If the chain is long enough, the posteriors will be reasonably well approximated.

Uniform bounded priors were assumed for all model parameters. Bounds were wide enough so as not to influence the fit. The posterior distribution was derived by sampling every 4,000th iteration from a chain consisting of 4,000,000 iterations of the MCMC algorthim after a burn in of

400,000 iterations. This level of thinning was sufficient to ensure that autocorrelation in the chain was not problematic. Convergence of the Markov chain was inferred informally by comparing the similarity of the 10th and 90th percentiles of the posterior densities based on the first 2,000,000 iterations with those based on the second 2,000,000 iterations, and by comparison of the posterior densities from several chains (Gamerman 2000).

Table A1.1. Parameters and indices used in the freshwater production model. Indices are used as subscripts for years and age classes; estimated parameters are those that are estimated by the model using maximum likelihood; and derived parameters are those values calculated from the estimated parameters. From Gibson and Bowlby (2013).

Model Parameter	Description	Type
t	Time in years	index
a	Juvenile age	index
Egg_t	Egg deposition in year *t*	estimated
M_{Egg}	Egg-to-fry mortality rate	estimated
$P_{t,0}$	Abundance of fry (age-0) in year *t*	derived
$D_{t,0}$	Density of fry (age-0) in year *t*	derived
α	Maximum survival from age-0 to age-1 (slope at the origin of the Beverton-Holt model)	estimated
R_{asy}	Asymptotic age-1 density (N/100 m^2)	estimated
h	Electrofishing scalar (habitat area in m^2)	constant or estimated
$P_{t,a}$	Abundance of parr of age *a* in year *t*	derived
$D_{t,a}$	Density of parr of age *a* in year *t*	derived
M_{Parr}	Parr mortality rate (age-1 and older)	estimated
j_a	Probability of smolting at age *a*	estimated
$S_{t,a}$	Abundance of smolt of age *a* in year *t*	derived
σ	Standard deviation for the likelihood functions	constant (0.33)
$\breve{\alpha}$	Maximum number of smolts produced per egg (slope at the origin of the Beverton-Holt model)	derived
$\breve{R}_{asy}$	Carrying capacity for smolts	derived

Table A1.2. Model equations for the freshwater production component of the population dynamics model. From Gibson and Bowlby (2013).

Equation Number	Description	Equation
1	Abundance of fry (age-0) in year *t*	$P_{t,0} = Egg_{t-1}(1-M_{Egg})$
2	Abundance of age-1 parr in year *t* (incorporating density dependence)	$P_{t,1} = \dfrac{\alpha P_{t-1,0}}{1+\dfrac{\alpha P_{t-1,0}}{R_{asy}h}}(1-j_1)$
3	Abundance of age-2 and older parr in year *t*	$P_{t,a} = P_{t-1,a-1}(1-M_{Parr})(1-j_a)$
4	Smolt abundance in year *t* of age *a*	$S_{t,a} = \left\{P_{t-1,a-1}(1-M_{Parr})(j_a) \qquad a=1,2,3,4\right\}$
5	Maximum survival from egg to smolt	$\breve{\alpha} = \alpha(1-M_{Egg})\left[j_1 + \sum_{a=2}^{4}\left[j_a\left(\prod_{k=1}^{k=a-1}(1-j_k)\right)(1-M_{Parr})^{a-1}\right]\right]$
6	Carrying capacity of the river for smolts	$\breve{R}_{asy} = R_{asy}h\left[j_1 + \sum_{a=2}^{4}\left[j_a\left(\prod_{k=1}^{k=a-1}(1-j_k)\right)(1-M_{Parr})^{a-1}\right]\right]$

Table A1.3. Parameters and indices used in the lifetime egg-per-smolt model. Indices are used as subscripts for years and age classes, derived parameters are those values calculated from the estimated parameters, and data are values such as counts that are used as model inputs for calculations (assumed known without error). Modified from Gibson and Bowlby (2013).

Model Parameter	Description	Type
c	Number of years as an immature salmon at sea	index
r	repeat spawning strategy: alt - alternate year repeat spawner; $cons$ - consecutive year repeat spawer	index
$Esc_{t,c}$	Spawning escapement of salmon of sea-age *c* in year *t*	data
$C_{t,c}$	Number of salmon of sea-age *c* removed by the recreational fishery in year *t*	data
$u_{t,c}$	Exploitation rate of salmon of sea-age *c* in year *t*	derived
$p_{c,r}$	Probability that a salmon of sea age *c* returns as a repeat spawner utilizing strategy *r*	data
$p_{c,r,r}$	Probability that a salmon of sea age *c* that has used strategy *r* utilizes strategy *r* when spawning for a third time	data
f_1	Fecundity of 1SW females (# of eggs)	data
f_2	Fecundity of MSW females (# of eggs)	data
RR_c^{river}	Return rates of salmon of sea-age *c* to the mouth of the river	derived
$RR_c^{escapement}$	Return rates of salmon of sea-age *c* to spawning escapement	derived
EPS	Lifetime egg production per smolt	derived

Table A1.4. Model equations for the lifetime egg-per-smolt component of the population dynamics model. Parameter definitions are provided in Tables A1.3 and A1.1. MModified from Gibson and Bowlby (2013).

Equation Number	Description	Equation
1	Return rates to the assessment facility/location for salmon of sea-age *c* in smolt year class *t*	$RR_{t,c}^{escapement} = \frac{Esc_{t+c,c}}{\sum_{a=2}^{4} S_{t,a}}; \quad c = 1,2$
2	Exploitation rates for salmon of sea-age *c* in smolt year class *t*	$u_{t,c} = \frac{C_{t+c,c}}{C_{t+c,c} + Esc_{t+c,c}}; \; c = 1,2$
3	Return rates to the mouth of the river for salmon of sea-age *c* in smolt year class *t*	$RR_{t,c}^{river} = \frac{Esc_{t+c,c} /(1-u_{t,c})}{\sum_{a=2}^{4} S_{t,a}}; \; c = 1,2$
4	Lifetime egg deposition per smolt in smolt year class *t*	$EPS_t = \sum_{c=1}^{2} RR_{t+c,c}^{escapement} Egg_c$, where:
4.1	Lifetime egg deposition for a 1SW salmon	$Egg_1 = f_1 + p_{1,alt} \; f_2 \; (1+ p_{1,alt,alt} + p_{1,alt,cons}) + p_{1,cons} f_2 \; (1+ p_{1,cons,alt} + p_{1,cons,cons})$
4.2	Lifetime egg deposition for a 2SW salmon	$Egg_2 = f_2 (1+ p_{2,alt} \; (1+ p_{2,alt,alt} + p_{2,alt,cons}) + p_{2,cons} \; (1+ p_{2,cons,alt} + p_{2,cons,cons}))$

Table A1.5. Likelihood functions and the objective function used for fitting the freshwater component of the population dynamics model. From Gibson and Bowlby (2013).

Equation Number	Description	Equation
1	Egg likelihood	$\ell_{egg} = -n \ln \sigma_{egg} \sqrt{2\pi} - \sum_t Egg_t^{obs} - \frac{1}{2\sigma_{egg}^2} \sum_t (\ln Egg_t^{obs} - \ln Egg_t)^2$
2	Electrofishing likelihood	$\ell_{electrofishing} = \sum_a \left(-n \ln \sigma_{elect} \sqrt{2\pi} - \sum_t D_{t,a}^{obs} - \frac{1}{2\sigma_{elect}^2} \sum_t (\ln D_{t,a}^{obs} - \ln(P_{t,a} / h))^2 \right)$
3	Smolt likelihood	$\ell_{smolt} = -n \ln \sigma_{smolt} \sqrt{2\pi} - \sum_t S_t^{obs} - \frac{1}{2\sigma_{smolt}^2} \sum_t (\ln S_t^{obs} - \ln S_t)^2$
4	Smolt age-frequency likelihood	$\ell_{age}^{smolt} = \sum_t \log \left(\frac{n_{smolt,t}!}{(x_{smolt,t,1}!)(x_{smolt,t,2}!).....(x_{smolt,t,r}!)} p_{smolt,t,1}^{x_{smolt,t,1}} ...... p_{smolt,t,r}^{x_{smolt,t,r}} \right)$
5	*O.F.V.* value	$O.F.V. = -(\ell_{egg} + \ell_{smolt} + \ell_{electrofishing} + \ell_{age}^{smolt})$

APPENDIX 2. LIFE HISTORY PARAMETER ESTIMATION FOR THE NASHWAAK RIVER ATLANTIC SALMON POPULATION

The life history parameter estimates for the Nashwaak River Atlantic Salmon population, provided in Section 2.2, as well as the information on the population's dynamics in Section 2.4, were derived using the statistical, life-history-based population dynamics model presented in Appendix 1. The application of the model to this population is described in this appendix. Included is a description of the data series used in the analyses, alternate model runs, interpretation of results, as well as the reasoning and biological justification for selecting the preferred model.

A2.1 Data

Recreational Fishery Statistics

Catch and effort data from the annual recreational Atlantic Salmon fishery (Table A2.1) have been collected using two methods: DFO collated statistics were used from 1970 until 1989 (Penny and Marshall 1984; Marshall 1987; O'Neil et al. 1987; 1989; 1991; 1996) and provincial catch data for small (1984-97) and large released (1990-94) have been adjusted using previous ratios when both datasets existed; for example, DFO/Prov stats. Effort data from 1970 to 1997 is tabled from provincial license database (O'Neil et al. 1996; K. Collet pers. comm.). Effort was estimated in rod days where any portion of a day fished by one angler was recorded as one rod day (effort is reported here but not used in the model). Nashwaak River has been closed to all recreational salmon fishing since 1998 (Jones et al. 2014).

The recreational fishery statistics are used in this analysis to estimate the smolt-to-adult return rates through to spawning escapement (i.e. after any removals by the recreational fishery). This was done by subtracting the number of virgin 1SW and 2SW salmon estimated to have been removed by the fishery from the numbers of adult returns, under the assumption that virtually all fishing occurs upstream of the counting fence. The numbers of virgin 1SW and 2SW wild salmon removed by the fishery are calculated from the number of large and small salmon reported in the recreational fishery statistics using the biological characteristics of the population sampled (Table A2.2). A hook-and-release mortality estimate of 4% was applied in the analysis, consistent with recent assessments (e.g. DFO 2011). Additionally the recreational fishery statistics were used to estimate abundance for those years in which the counting fence was not operated (and 1975), as described below.

Adult Abundance and Biological Characteristics

From 1993 until 2012, most adult salmon captured at a counting fence, operated by DFO and local First Nations, were counted, measured for fork length, categorized as either small or large salmon, externally sexed (male, female), classified as hatchery or wild, and marked with a hole punch. Further sampling details (including proportions and exceptions to general protocols), annual dates of operation, counts, return estimates, spawning escapement estimates and assessment method have been tabled in Jones et al. (2014). The counting fence was also operated for three years during the 1970's and annual assessments were completed in 1972-1973 but not in 1975 (Penny and Marshall 1984). Adult abundance and spawning escapement were estimated for small and large salmon from 1970 until 1993 (except 1972-1973) using recreational catch data. Catch rates (small: 0.48, 0.28 and large: 0.49, 0.27) derived by Penny and Marshall (1984) were averaged and then applied to recreational catch data to estimate annual small and large salmon abundance (Table A2.3). Biological characteristic data (mean length, proportion female, etc.) collected from salmon captured at the counting fence was applied to spawner escapement estimates to estimate annual egg depositions. In years when the fence not operated average biological characteristics information from other years was used; the mean values from 1970's were used for the 1970-1983 time period while the mean values from 1993 to 2011 were applied to the 1984 to 1992 time period. This break year of 1984 was used because of management changes that occurred that year, including closure of commercial fishery and mandatory hook and release of all large salmon.

The adult counts and biological characteristics are used for three purposes. First, these data are used to estimate annual egg deposition (described below). Second, the data are used to determine the number of 1SW and 2SW first-time spawning salmon that return to the river to spawn. Lastly, these estimates are then used to calculate the smolt-to-adult return rates to the spawning escapement as described in Appendix 1.

The repeat spawning dynamics of salmon in the Nashwaak River population includes both alternate and consecutive spawners that represent less than 10% of the total returns in all years since the counting fence resumed operation in 1993. Mean survival of 1SW and 2SW salmon from 1st spawning to 2nd spawning were 3.1% and 9.0% (alternate and consecutive combined), respectively based on the maiden recruits from 1993 to 2009 (Table A2.4).

Egg Deposition Time Series

The annual egg depositions in the Nashwaak were calculated using the estimates of small (1SW) and large (MSW) salmon, their biological characteristics, and a length-fecundity relationship for female salmon destined for tributaries upriver of Mactaquac Dam (Marshall and Penny 1983). The fecundities of 1SW and MSW salmon have changed through time as a result of changes in the mean length and sex ratios in the two size classes (Tables A2.5, A2.6). The average fecundities from the 1970's was used to calculate the annual egg depositions for the years 1970 to 1983, and the average fecundities from 1993 to 2011 were used for the years 1984 to 1992. The annual fecundity estimates (Table A2.5) were used from 1993 to 2011. The egg deposition time series used in the model is provided in Table A2.3.

A key decision in calculating this series was whether the series should include both cultured and wild salmon, or whether only wild salmon should be used in its derivation. Although cultured salmon have been found to have lower spawning success relative to wild individuals (McGinnity et al. 2004; Weir et al. 2004; Jonsson and Jonsson 2006; Chilcote et al. 2011), they are still expected to contribute to subsequent juvenile production. At present, there is no way to identify whether juvenile salmon in this population are the progeny of cultured or wild salmon spawning in the wild. Therefore, the decision was made to include cultured salmon when calculating the number of eggs deposited each year (Table A2.3). The proportion of the MSW salmon component (which includes repeat-spawners) that are 2SW salmon, as well as the proportions of 1SW and MSW salmon that are of wild origin, based on sampling of the adult population. If no adult sampling occurred (no fence operation) then hatchery returns were estimated using smolt-to-adult return rates from Mactaquac.

Smolt Abundance and Biological Characteristics

The annual smolt migration for the Nashwaak salmon population was monitored from 1998 to 2012. A collaborative project between DFO, Nashwaak Watershed Association Inc. (NWAI) and Atlantic Salmon Federation to estimate the wild smolt production of the Nashwaak River has been ongoing since 1998. Oromocto First Nation has also participated since 2010. One or two American constructed rotary screw traps (RST) have been installed on an annual basis usually from mid-April until late-May in the main stem of the Nashwaak River just downriver of Durham Bridge (Chaput and Jones 2007). The RSTs were usually checked once daily from throughout the peak migration period and less frequently (every other day) at start of operation and as the daily catches decreased. All unmarked smolts were identified for origin (wild or hatchery). From 1998 until 2001 smolts (mostly wild origin) were captured, marked and released from a counting fence that was operated on the Tay River in order estimate the efficiency of the RST. Starting in

2001, a portion of the smolts were marked with either numbered streamer tags or caudal clip and released upriver in order to estimate the capture efficiencies of the RST(s).

Biological characteristics are collected from a sample of smolts captured during the assessment. The fork length and weight of sampled individuals is recorded and a scale sample is collected to determine the smolt's age. Abundance estimates and the number of sampled smolts in each age category are provided in Table A2.7 (the numbers in category are provided, rather than the proportions because this is how the data are input into the model). The time period over which smolt data has been collected is relatively short in comparison with the adult times series.

Abundance of Fry and Parr

The relative abundance of age-0 (fry), age-1 and age-2+ (collectively known as parr) juvenile salmon is determined by electrofishing. Note that here, the notation 2+ is used to denote a plus group consisting of all parr age 2 and older, consistent with the way this notation is used for other species (Quinn and Deriso 1999).

Four key decisions had to be made about the juvenile density data in the model: how to best estimate site-specific juvenile abundance from the data; how to best determine abundance-at-age; whether data from all sites should be included in the analysis; and whether the data should be standardized to correct for changes in the location of sites from year-to-year.

With the exception of 1980, densities of juvenile salmon have been monitored at least three of the ten index sites on the Nashwaak River on an annual basis since 1968. Densities prior to 1980 along with site characteristics and locations were reported by Francis (1980). Densities (number of fish per 100 m^2 of habitat) of age-0 and older parr at these sites were derived using three methods: removal method using multiple sweeps and barrier nets (Francis 1980), mark-recapture techniques (Jones et al. 2004) using the adjusted Petersen method (Ricker 1975) or a mean probability of capture derived in Jones et al. (2004). In most years, the numbers of parr by age were determined from stratified sampling of large parr in 0.5 cm length intervals. Generally, one parr was scale sampled for each interval. If scale sampling was not completed in a particular year, then a length frequency distribution plot was used to partition the catches into age classes. When mark-recapture techniques were used, the number of age-0 parr or fry for the site was determined by applying the capture efficiency for age-1 and older parr to the number of fry captured during the marking pass. Also a mean probability of capture was applied if zero parr were marked or recaptured or if only the marking pass was completed (Jones et al. 2004). The densities from seven of the ten barrier sites have been adjusted to account for the expanded sites for when mark and recapture techniques were initiated in 1990 and 1991 (Marshall et al. 2000; updated in Jones et al. 2014). In addition to these 10 sites, the Nashwaak River juvenile surveys were expanded from 2004 to 2008, when an additional 10 or so sites were electrofished, but this data is not used in this analysis.

The densities presented are for wild (or adipose fin present) parr only. For the most part, prior to 1998 all fall fingerling and unfed fry were released unmarked (Jones et al. 2014) and suspected hatchery origin parr captured during electrofishing surveys were determined through observations made by field staff of fin erosion or condition. Between 1999 and 2006, most fall fingerlings released were adipose clipped and there were fewer unfed fry releases, thereby making the identification of wild parr easier and more accurate. From 2008 onwards, unclipped hatchery origin parr were determined by field staff based on fin erosion or condition.

The number of sites electrofished has varied from year-to-year. Variation in sample locations can bias the resulting density estimates if the proportions of high and low density sites change annually or systematically over time, and this in turn can bias survival estimates (Gibson et al. 2008b). Therefore, a generalized linear model (GLM) was used to reduce overall variation in the time series of estimated age-class densities, and to investigate how such variation in sample locations influences the predictive capacity of juvenile data. Following the approach in Gibson et al. (2008b), mean density of a given age class was estimated for each year using ‘site’ and ‘year’ as factors in the GLM, assuming a Poisson error distribution. Gibson et al. (2008b) found that the standardized data provided better estimates of survival and a significantly better model fit when modeling the dynamics of Tobique River Atlantic Salmon using methods similar to those presented here, a result consistent with those of Gibson and Bowlby (2013) for the LaHave River (above Morgans Falls) and the St. Mary’s River (West Branch) Atlantic Salmon populations. The standardized and un-standardized electrofishing time series are provided in Table A2.8. Initial model explorations indicated that using the standardized series consistently provided a better model fit than when the un-standardized series was used, so in the model runs below, only results with the standardised series are presented.

A2.2 Model Formulations

The model was set up using data from 1970 to 2011. The estimated demographic parameters for the freshwater production model are listed in Table A2.9. These include the annual mortality rates of eggs and parr (M_{Egg} and M_{Parr}), the maximum rate of population increase for age 1 parr (α), the carrying capacity for age 1 parr (R_{asy}), and the smoltification probability at age-2 (j_2). Similar to the analysis for the Tobique River (Gibson et al. 2009), the annual egg deposition was estimated, and a step function was used to split the model into two parts so that changes in freshwater productivity could be evaluated (the timing of the split was evaluated by profiling over the years to find the most probable split). Using the step function, reasonable model fits and parameter estimates could not be obtained. In particular, R_{asy} for the recent period could not be estimated when abundance is low (this is evident in Model 2, see below). In addition, when profiling over the years to find the most probable break year, the 1999 consistently was identified as the most probable year, which is problematic because there would only be one year in the earlier time period for which smolt data would be available if this year was chosen.

Consistent with Gibson et al. (2009) and Gibson et al. (2013), estimating the annual egg depositions, rather than using the data as constants in the model, improved the model fit.

The relative contribution of each likelihood to the objective function value (OFV) can be controlled using a set of weighting values. These values may be selected to keep any one part of the objective function from dominating the fit, or alternatively, to reflect perceptions of data accuracy (Merritt and Quinn 2000). For the base model for the Nashwaak River population, all weights were set equal to one, an approach that has the advantage that the OFV can be interpreted as the likelihood.

A base model run was selected that consisted of using the standardized electrofishing time series and estimating a single set of parameters considered representative of the average dynamics of the entire time period. Examples of other model runs are shown in Table A2.9, including:

1. Model 2: similar to the base model but attempting to use a step function to estimate separate parameter values for two time periods – carrying capacity (R_{asy}) could not be estimated for the more recent time period using this approach;
2. Model 3: similar to Model 2 with separate model parameters for the two time periods except for R_{asy} for which a single value was estimated – standard errors were larger this run than for other runs, otherwise a plausible model but with limited smolt data for fitting in the earlier time period;

3. Model 4: similar to Model 3 but with greater weight placed on fitting to the egg and smolt time series – carrying capacity could not be estimated with this formulation;
4. Model 5: similar to Model 3 but with lesser weight placed on fitting to the egg and smolt time series – plausible model but with limited smolt data for fitting in the earlier time period;
5. Model 6: using only data from 1996 to 2011 – unable to estimate carrying capacity.

Several other data combinations and likelihood weighting combinations were evaluated as well. Although the parameter estimates varied slightly in each case, none of these other model runs altered the conclusion that the dynamics exhibited in the base model are a reasonable approximation of the dynamics of the Nashwaak River population at this time.

A2.3. Results

Parameter estimates from the model are given in Table A2.9 and model fits and diagnostic plots for the base model are shown in Figure A2.1 to A2.9. Overall, the model fits to the data appear reasonable and, in the case of the base model, the parameter estimates are plausible. Fits to both the egg deposition data and the smolt counts (Figure A2.1) and to the electrofishing data (Figure A2.2) capture the general pattern in the data. The estimated abundance of smolts in the 1970’s and 1980’s is 2 to 5 times the estimated smolt abundance in the late-2000’s. Scatterplots of the abundance of parr within a cohort in sequential age classes (Figure A2.3) illustrate the asymptotic behaviour (characteristic of density dependence) for age-1 at relatively low densities of both age-0 and age-1 fish. Although the estimated relationships appear to characterize the overall pattern in the data reasonably, the data do show scatter around the fitted relationships. Additionally, there are negative residuals at higher abundance for both the age-0 and age-2+ age classes, potentially indicating that density dependence could be occurring in more age classes than just between age-0 and age-1.

The observed and estimated return rates of 1SW and 2SW salmon to the river mouth are shown in Figure A2.4. The differences between the estimated rates and the observed rates result only from the different smolt abundance values (observed versus estimated) going into the model because the number of adults is the same in the both cases. The return rates for 1SW salmon declined during the late-1970’s and 1980’s, but have increased to higher levels during the 2000’s. In contrast, return rates for 2SW salmon have not increased to the same extent as for 1SW salmon. Return rates to spawning escapement (Figure A2.5) were lower than to the river mouth in the earlier time periods, showing the effect of retention recreational fisheries, but the rates are more similar in recent years. However, the magnitude of the difference in the return rates to the river mouth and to spawning escapement depends on the assumed exploitation rates used to estimate past abundance, a model assumption rather than an analytical result.

MCMC diagnostic plots for estimated and derived model parameters are shown in Figures A2.6 to A2.9. In general, the trace plots (second column from left) appear reasonable, and do not show significant autocorrelation (third column from left). Minima appear reasonably defined by the OFV for all model parameters (right columns). The comparisons of the marginal probability densities with the maximum likelihood estimates (left columns) indicate very good agreement between these measures of central tendency.

Maximum lifetime reproductive rates are relatively similar among model runs (Table A2.9). For the 1970’s, the estimated rates vary from a value of 1.67 to 2.74 spawners per spawner. For the 2000’s, they vary from 1.03 to 1.33. The lowest value came from Model 2, in which freshwater parameter values were allowed to change once through time. All values are low enough that populations would have little to no capacity to compensate for the effects of environmental perturbations (floods, droughts, years of lower at-sea survival), leading to the conclusion that this population is expected to extirpate in the absence of human intervention or environmental change.

Overall, the base model produces parameter estimates than are roughly similar to the parameter values produced by the other five model runs shown here (Table A2.9). One potentially important difference is that Model 1 has a higher mortality estimate for eggs and a lower mortality estimate for older parr than the other models, suggesting that the timing of mortality is earlier than that suggested by the other models. Although cases could be made to select Model 2 or Model 5 as the base model, Model 1 was choosen as the base model primarily because of the limited smolt abundance data in the earlier time period, but also because the parameter estimates are fairly similar, particularly when rolled up to the egg-to-smolt or maximum lifetime reproductive rate levels.

Further discussion of the parameter values and their implications for recovery planning is provided in the main body of the text (Sections 2.2. and 2.5).

Table A2.1. Recreational catches for the Nashwaak River. Effort data is estimated from provincial licenses. Small (1984-97) and large released (1990-94) catch data have adjusted using previous ratios of DFO officer/Prov stats from 1984-1989.

Year	Season	Catch (small)	Retained (small)	Released (small)	Catch (large)	Retained (large)	Released (large)	Effort (rod days)
1970	open	811	811	0	854	854	0	5,967
1971	open	733	733	0	205	205	0	4,171
1972	open	581	581	0	926	926	0	5,843
1973	open	408	408	0	923	923	0	8,597
1974	open	495	495	0	433	433	0	6,345
1975	open	663	663	0	467	467	0	8,985
1976	open	1,746	1,746	0	941	941	0	10,293
1977	open	1,096	1,096	0	1,190	1,190	0	12,062
1978	open	451	451	0	511	511	0	11,625
1979	open	960	960	0	221	221	0	9,843
1980	open	1,107	1,107	0	1,183	1,183	0	14,659
1981	open	1,085	1,085	0	498	498	0	12,896
1982	open	1,278	1,278	0	792	792	0	19,287
1983	open	420	420	0	260	260	0	14,340
1984	open	439	434	5	410	0	410	6,339
1985	open	719	654	65	673	0	673	3,233
1986	open	982	751	231	750	0	750	8,995
1987	open	886	750	136	177	0	177	6,282
1988	open	249	201	48	190	0	190	6,687
1989	open	465	448	17	214	0	214	9,335
1990	open	206	196	10	298	0	298	12,218
1991	open	228	186	42	248	0	248	12,254
1992	open	535	426	109	278	0	278	13,429
1993	open	213	137	76	82	0	82	9,063
1994	open	22	0	22	12	0	12	1,496
1995	open	n/a	0	n/a	0	0	0	11
1996	closed	0	0	0	0	0	0	n/a
1997	open	14	0	14	0	0	0	224
1998	closed	0	0	0	0	0	0	n.a.
1999	closed	0	0	0	0	0	0	n.a.
2000	closed	0	0	0	0	0	0	n.a.
2001	closed	0	0	0	0	0	0	n.a.
2002	closed	0	0	0	0	0	0	n.a.
2003	closed	0	0	0	0	0	0	n.a.
2004	closed	0	0	0	0	0	0	n.a.
2005	closed	0	0	0	0	0	0	n.a.
2006	closed	0	0	0	0	0	0	n.a.
2007	closed	0	0	0	0	0	0	n.a.
2008	closed	0	0	0	0	0	0	n.a.
2009	closed	0	0	0	0	0	0	n.a.
2010	closed	0	0	0	0	0	0	n.a.
2011	closed	0	0	0	0	0	0	n.a.

Table A2.2. Proportions of wild origin adult Atlantic Salmon that are virgin one sea-winter (1SW), virgin two sea-winter (2SW), and repeat spawning salmon based on samples collected at a counting fence on the Nashwaak River for three years in the 1970's and from 1993 to 2011. The proportion of 2SW salmon in the large component of the population, used to split the large component of the recreational catch in the population model, is also shown.

		Proportion			
Year	Total abundance	1SW	2SW	Repeat spawners	2SW in the large component
1972	3,095	0.344	0.622	0.031	0.948
1973	4,093	0.234	0.628	0.130	0.819
1975	2,969	0.541	0.377	0.065	0.822
1993	1,233	0.672	0.224	0.090	0.682
1994	972	0.635	0.291	0.074	0.797
1995	1,315	0.684	0.302	0.015	0.954
1996	2,223	0.738	0.204	0.058	0.779
1997	671	0.495	0.422	0.083	0.836
1998	1,552	0.809	0.122	0.069	0.639
1999	936	0.706	0.199	0.085	0.677
2000	701	0.726	0.217	0.052	0.794
2001	513	0.472	0.463	0.065	0.877
2002	415	0.824	0.109	0.067	0.621
2003	396	0.725	0.250	0.026	0.907
2004	777	0.739	0.244	0.018	0.932
2005	856	0.814	0.166	0.020	0.892
2006	852	0.777	0.204	0.019	0.917
2007	561	0.822	0.141	0.037	0.793
2008	1,384	0.876	0.114	0.009	0.926
2009	607	0.465	0.448	0.087	0.837
2010	2,166	0.911	0.069	0.020	0.779
2011	1,544	0.637	0.341	0.022	0.938

Table A2.3. Spawning escapement for 1SW and multi sea-winter (MSW) Atlantic Salmon, and the egg deposition time series used for inputs for the Nashwaak River population dynamics models. The proportion of the MSW salmon component (which includes repeat-spawners) that are 2SW salmon, and the proportions of 1SW and MSW salmon that are of wild origin based on sampling of the adult population, are also shown.

				Proportions		
Year	1SW	MSW	Egg deposition (millions)	2SW in MSW component	1SW that are wild	MSW that are wild
1970	1,312	1,402	9.606	0.863	1.000	1.000
1971	1,186	337	2.960	0.863	1.000	1.000
1972	624	964	9.957	0.948	1.000	1.000
1973	1,039	2,533	15.190	0.819	1.000	1.000
1974	801	711	4.972	0.863	1.000	1.000
1975	1,072	767	5.520	0.822	1.000	1.000
1976	2,824	1,545	11.620	0.863	1.000	1.000
1977	1,773	1,954	13.344	0.863	1.000	1.000
1978	729	839	5.706	0.863	0.992	1.000
1979	1,553	363	3.397	0.863	0.970	0.993
1980	1,790	1,942	13.287	0.863	0.905	0.981
1981	1,755	818	6.345	0.863	0.939	0.947
1982	2,067	1,300	9.547	0.863	0.980	0.974
1983	679	427	3.135	0.863	0.861	0.964
1984	702	1,067	7.264	0.820	0.836	0.899
1985	1,058	1,751	11.792	0.820	0.874	0.937
1986	1,215	1,951	13.191	0.820	0.952	0.957
1987	1,213	461	4.409	0.820	0.974	0.888
1988	325	494	3.366	0.820	0.887	0.973
1989	725	557	4.293	0.820	0.963	0.947
1990	317	775	5.010	0.820	0.984	0.976
1991	301	645	4.221	0.820	0.887	0.994
1992	689	723	5.224	0.820	0.953	0.970
1993	866	555	3.948	0.682	0.868	0.730
1994	610	349	3.264	0.797	0.933	0.915
1995	940	436	4.222	0.954	0.956	0.954
1996	1,804	641	6.203	0.779	0.897	0.887
1997	364	362	2.888	0.836	0.897	0.926
1998	1,238	309	3.917	0.639	0.998	0.940
1999	658	269	2.468	0.677	0.994	1.000
2000	489	189	1.887	0.794	1.000	1.000
2001	224	266	2.034	0.877	0.992	0.996
2002	320	69	0.725	0.621	0.997	0.924
2003	280	109	0.950	0.907	0.966	0.965
2004	569	201	2.116	0.932	0.973	0.981
2005	712	155	2.007	0.892	0.953	0.981
2006	681	186	2.045	0.917	0.925	0.974
2007	442	98	1.166	0.793	0.983	0.943
2008	1,217	168	2.932	0.926	0.981	0.988
2009	274	328	1.780	0.837	0.949	0.967
2010	2,008	195	3.942	0.779	0.979	0.980
2011	1,033	575	4.739	0.938	0.951	0.974

Table A2.4. Average probabilities that 1SW and 2SW salmon return to spawn as either alternate-year or consecutive-year repeat spawners for a second or third spawning event based on biological data corresponding to cohorts spawning for the first time in the years 1993 to 2009.

Variable	Probabilities
To a second spawning:	
$p_{1,alt}$	0.023
$p_{1,cons}$	0.008
$p_{2,alt}$	0.046
$p_{2,cons}$	0.044
To a third spawning:	
$p_{1,alt,alt}$	0.000
$p_{1,alt,cons}$	0.108
$p_{1,cons,alt}$	0.000
$p_{1,cons,cons}$	0.072
$p_{2,alt,alt}$	0.202
$p_{2,alt,cons}$	0.046
$p_{2,cons,alt}$	0.113
$p_{2,cons,cons}$	0.031

Table A2.5. Mean fork length, proportion female and estimated fecundity for 1SW and MSW Atlantic Salmon in the Nashwaak River based on samples collected at the counting fence. Expected number of eggs per fish (sexes combined) for each age class during two time periods are also shown.

Year	Mean Fork Length (cm)		Proportion Female		Estimated Fecundity	
	1SW	MSW	1SW	MSW	1SW	MSW
1972	57.5	76.3	0.328	0.822	3,419	6,733
1973	54.7	78.2	0.212	0.835	3,091	7,211
1975	55.1	79.3	0.152	0.918	3,136	7,503
Mean:	55.8	77.9	0.231	0.858	3,215	7,149
1993	57.1	77.8	0.279	0.858	3,370	7,108
1994	58.8	78.7	0.517	0.850	3,583	7,342
1995	57.2	78.3	0.363	0.983	3,382	7,237
1996	57.1	78.7	0.437	0.759	3,370	7,342
1997	57.0	79.8	0.440	0.861	3,358	7,639
1998	57.0	80.4	0.518	0.723	3,358	7,806
1999	58.4	79.8	0.459	0.679	3,532	7,639
2000	56.9	79.8	0.360	0.899	3,346	7,639
2001	58.4	78.1	0.393	0.900	3,532	7,185
2002	57.7	82.2	0.304	0.672	3,444	8,329
2003	58.1	77.5	0.273	0.900	3,494	7,031
2004	58.2	78.7	0.440	0.852	3,506	7,342
2005	57.8	78.3	0.433	0.862	3,456	7,237
2006	57.4	79.9	0.410	0.780	3,407	7,667
2007	57.7	79.1	0.419	0.747	3,444	7,449
2008	57.7	77.2	0.496	0.736	3,444	6,956
2009	56.9	77.0	0.289	0.663	3,346	6,906
2010	57.0	77.7	0.440	0.702	3,358	7,082
2011	57.8	79.5	0.478	0.707	3,456	7,557
Mean:	57.6	78.9	0.408	0.796	3,431	7,394

Table A2.6.Expected number of eggs per fish (sexes combined) for one 1SW and MSW Atlantic Salmon in the Nashwaak River for two time periods are also shown. Values are calculated using the mean values shown in Table A2.5.

Age group	Time period	Expected number of eggs
1SW	1970’s	742
1SW	1993 - 2011	1,399
MSW	1970’s	6,136
MSW	1993 - 2011	5,890

Table A2.7. Smolt abundance in the Nashwaak River from 1998 to 2012; and the number of smolts sampled by age class.

Year	Abundance estimate	Number sampled	Number age-2	Number age-3	Number age-4
1998	22,750	204	162	42	0
1999	28,500	287	193	87	7
2000	15,800	208	117	89	2
2001	11,000	194	166	28	0
2002	15,000	230	209	21	0
2003	9,000	137	103	34	0
2004	13,600	154	118	36	0
2005	5,200	59	35	24	0
2006	25,400	401	340	61	0
2007	21,550	343	301	42	0
2008	7,300	359	265	94	0
2009	15,900	420	353	67	0
2010	12,500	426	221	205	0
2011	8,750	166	136	30	0
2012	11,060	154	93	61	0

Table A2.8. Annual mean densities of juvenile Atlantic Salmon by age class in the Nashwaak River used as inputs for the population dynamics model. Two variants of the data are used: “Un-standardized” values are annual means calculated directly from the data, whereas the “Standardized” values are estimated with a generalized linear model with “site” and “year” as factors to correct for the effects of changes in the sites included in the survey each year.

		Un-standardized			Standardized		
Year	N	Age-0	Age-1	Age-2	Age-0	Age-1	Age-2
1970	3	15.00	3.83	7.03	18.86	3.49	7.66
1971	10	46.35	5.72	7.07	46.35	5.72	7.07
1972	10	21.00	1.82	13.82	21.00	1.82	13.82
1973	10	27.33	0.09	10.17	27.33	0.09	10.17
1974	10	54.54	1.88	8.88	54.54	1.88	8.88
1975	10	50.64	14.05	9.52	50.64	14.05	9.52
1976	10	33.86	7.94	2.16	33.86	7.94	2.16
1977	10	24.75	10.87	2.21	24.75	10.87	2.21
1978	8	48.86	6.85	3.53	46.65	6.86	3.49
1979	5	71.96	16.04	4.42	47.18	13.78	4.55
1981	7	53.71	14.17	4.21	44.78	11.57	3.49
1982	8	40.45	9.88	2.99	37.56	9.12	2.79
1983	8	20.76	7.83	2.91	19.28	7.23	2.72
1984	8	33.69	5.38	1.85	31.28	4.97	1.73
1985	8	35.30	6.29	2.56	32.78	5.81	2.39
1986	8	38.64	7.06	2.09	35.87	6.52	1.95
1987	8	52.40	11.70	0.78	48.65	10.81	0.72
1988	8	48.25	9.80	1.06	44.80	9.05	0.99
1989	8	46.39	11.35	1.91	43.07	10.49	1.79
1990	8	33.61	10.49	1.64	31.21	9.69	1.53
1991	8	32.46	8.68	1.51	30.14	8.01	1.41
1992	9	26.70	12.90	1.04	24.31	11.85	0.96
1993	9	12.72	7.00	1.98	11.58	6.43	1.83
1994	10	3.99	3.29	0.76	3.99	3.29	0.76
1995	8	10.46	8.64	1.58	9.71	7.98	1.47
1996	9	8.10	3.31	0.63	8.35	3.36	0.65
1997	9	12.93	5.33	0.86	13.33	5.42	0.87
1998	10	3.61	3.85	0.98	3.61	3.85	0.98
1999	10	7.91	4.37	1.18	7.91	4.37	1.18
2000	9	11.83	4.02	0.10	12.20	4.09	0.10
2001	9	11.92	9.38	1.19	10.85	8.61	1.10
2002	8	15.43	5.51	1.26	14.32	5.09	1.18
2003	9	4.79	5.01	0.72	4.36	4.60	0.67
2004	9	5.21	2.41	0.67	4.74	2.21	0.62
2005	9	6.79	4.49	0.42	6.18	4.12	0.39
2006	8	3.03	4.64	0.71	2.97	3.81	0.59
2007	9	5.53	4.04	0.50	5.04	3.71	0.46
2008	9	8.02	4.86	0.89	7.30	4.46	0.82
2009	9	6.04	2.89	0.64	5.50	2.65	0.59
2010	9	19.58	4.74	0.96	17.82	4.36	0.88
2011	8	2.33	4.79	0.49	1.92	3.94	0.40

Table A2.9. Maximum likelihood estimates (standard errors) for life history parameter estimates for the Nashwaak River Atlantic Salmon population obtained from six versions of the dynamics model. The objective function value (OFV) for Model 6 is not comparable to others because the data series is shorter; similarly the OFV for models 4 and 5 are not comparable to the others because of the weights. "NA" values occur where a model parameter is not applicable (e.g. if only a single survival is estimated as in models without a break year). "Past" dynamics are those for the 1973-82 cohorts, whereas "present" refers to the 2000-09 cohorts, except in models without a break year in which the "past" estimates are used for the full time period.

Model:	Base	Model 2	Model 3	Model 4	Model 5	Model 6
Assumptions and Fit:						
Time period:	1970-2011	1970-2011	1970-2011	1970-2011	1970-2011	1993-2011
Standardized electrofishing data:	yes	yes	yes	yes	yes	yes
Break year	None	1999	1999 but 1 Rasy	1999 but 1 Rasy	1999 but 1 Rasy	None
Weights: eggs, smolt, smolt age	1	1	1	10	1/10	1
OFV	719.5	697.3	698.5	7,312.5	0.42	2,165.4
Freshwater production:						
Electro q	10,044 (2,929)	17,376 (4,733)	17,562 (4,813)	17,795 (3,937)	17,845 (7,972)	17,009 (5,567)
M_{Egg} (past)	0.96 (0.01)	0.94 (0.02)	0.94 (0.02)	0.94 (0.01)	0.93 (0.04)	0.95 (0.02)
M_{Egg} (present)	*NA*	0.93 (0.02)	0.93 (0.02)	0.91 (0.02)	0.94 (0.03)	*NA*
α (past)	0.54 (0.07)	0.44 (0.09)	0.36 (0.07)	0.38 (0.04)	0.49 (0.12)	0.63 (0.07)
α (present)	*NA*	0.62 (0.08)	0.65 (0.1)	0.53 (0.06)	0.79 (0.15)	*NA*
R_{asy} (past)	28.01 (8.79)	28.2 (28.2)	69.427 (95)	infinite	20.143 (8)	infinite
R_{asy} (present)	*NA*	infinite	*NA*	*NA*	*NA*	*NA*
M_{Parr} (past)	0.53 (0.1)	0.59 (0.08)	0.58 (0.08)	0.61 (0.06)	0.59 (0.15)	0.70 (0.07)
M_{Parr} (present)	*NA*	0.73 (0.06)	0.73 (0.06)	0.75 (0.04)	0.73 (0.07)	*NA*
j_2 (past)	0.61 (0.06)	0.49 (0.07)	0.5 (0.07)	0.47 (0.05)	0.50 (0.16)	0.5 (0.06)
j_2 (present)	*NA*	0.49 (0.05)	0.49 (0.05)	0.47 (0.04)	0.48 (0.08)	*NA*
j_3 (past)	0.99 (0.01)	0.91 (0.04)	0.50 (0.07)	0.90 (0.02)	0.91 (0.01)	0.98 (0.01)
j_3 (present)	*NA*	1.00 (<0.01)	1.00 (<0.01)	1.00 (<0.01)	1.00 (<0.01)	*NA*
Egg to smolt dynamics:						
$\breve{\alpha}$ (past)	0.007 (0.001)	0.008 (0.002)	0.007 (0.002)	0.005 (0.001)	0.01 (0.006)	0.006 (0.001)
$\breve{\alpha}$ (present)	*NA*	0.007 (0.001)	0.007 (0.001)	0.007 (0)	0.009 (0.004)	*NA*
$\breve{R}_{asy}$ (past) (thousands)	104.4 (36.2)	140.1 (84.5)	353.2 (499.7)	infinite	101.3 (70.9)	infinite
$\breve{R}_{asy}$ (present) (thousands)	*NA*	infinite	206.0 (282.8)	Infinite	61.2 (30.5)	*NA*

Model:	Base	Model 2	Model 3	Model 4	Model 5	Model 6
Return Rates (%):						
1SW average (past)	6.18 (0.85)	5.27 (1.34)	4.84 (1.27)	5.53 (0.58)	5.22 (3.22)	*NA*
1SW average (present)	4.95 (0.45)	5.02 (0.48)	5.00 (0.48)	4.97 (0.16)	5.23 (1.54)	*NA*
2SW average (past)	4.04 (0.56)	3.44 (0.87)	3.20 (0.84)	3.73 (0.38)	3.36 (2.1)	*NA*
2SW average (present)	1.10 (0.1)	1.11 (0.11)	1.11 (0.11)	1.12 (0.04)	1.11 (0.33)	*NA*
Lifetime egg prod. per smolt:						
average EPS (past)	333.2 (45.8)	284.1 (72.2)	263.4 (69.0)	306.4 (31.5)	278.3 (173.1)	*NA*
average EPS (present)	150.9 (13.7)	153 (14.7)	152.4 (14.7)	152.8 (4.8)	156.06 (45.7)	*NA*
Max. lifetime reproductive rate:						
average (past)	2.49 (0.42)	2.3 (0.58)	1.79 (0.42)	1.67 (0.16)	2.74 (0.97)	*NA*
average (present)	1.13 (0.12)	1.03 (0.11)	1.11 (0.15)	1.09 (0.06)	1.33 (0.43)	*NA*
Equi. egg abundance (millions):						
average (past)	20.81 (4.04)	22.49 (7.28)	41.41 (43.96)	infinite	17.90 (5.90)	*NA*
average (present)	1.76 (1.13)	infinite	3.17 (3.33)	infinite	2.39 (2.21)	*NA*
Equi. Smolt abundance (thousands):						
average (past)	62.4 (15.9.)	79.2 (35.0)	156.2 (179.90)	infinite	64.3 (41.1)	*NA*
average (present)	11.6 (7.4)	infinite	20.8 (21.9)	infinite	15.3 (14.8)	*NA*

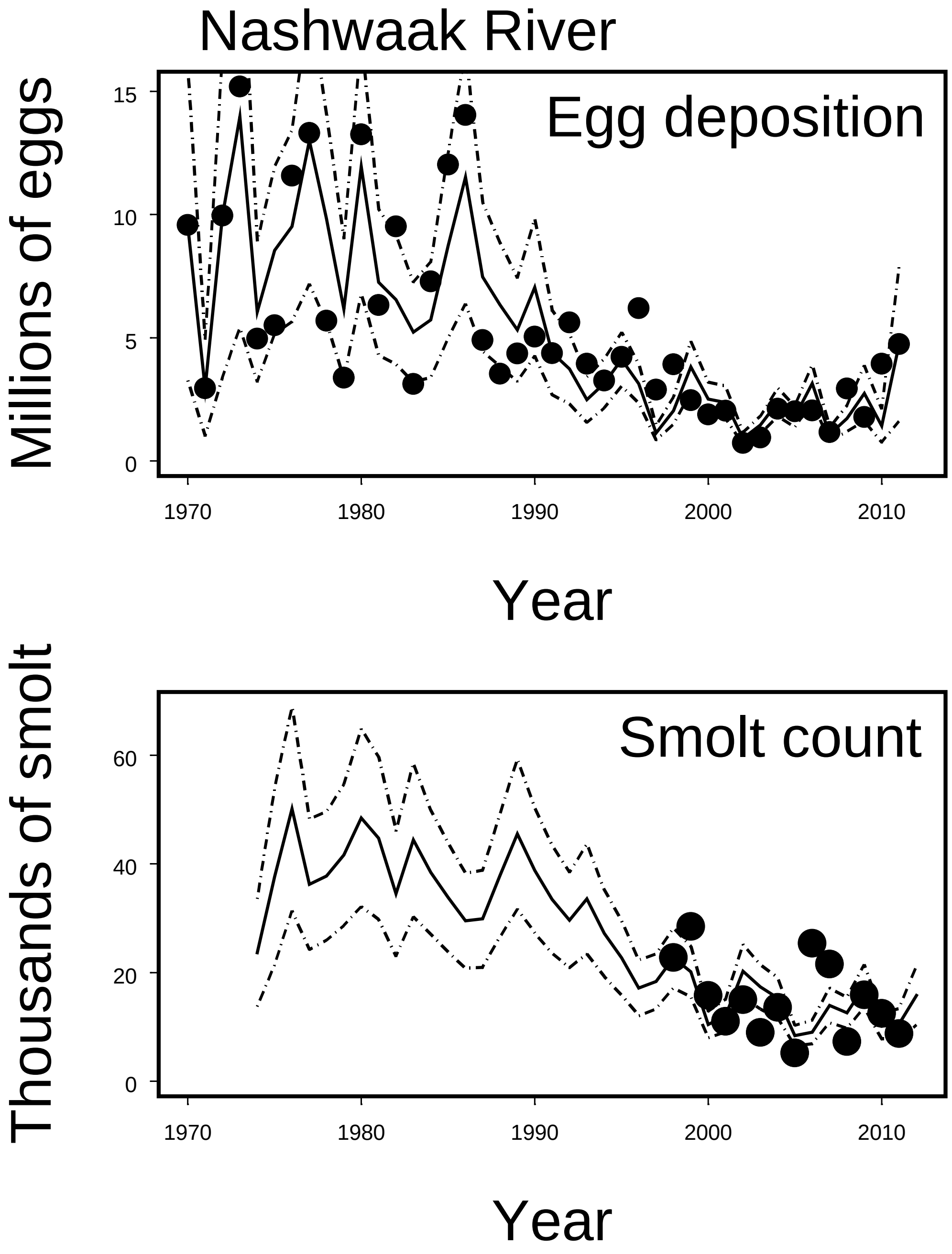

Figure A2.1. Observed (points) and estimated (solid lines) egg depositions (top panel) and smolt counts (bottom panel) from the base population dynamics model for the Nashwaak River Atlantic Salmon population. The dashed lines show 95% confidence intervals based on normal approximations.

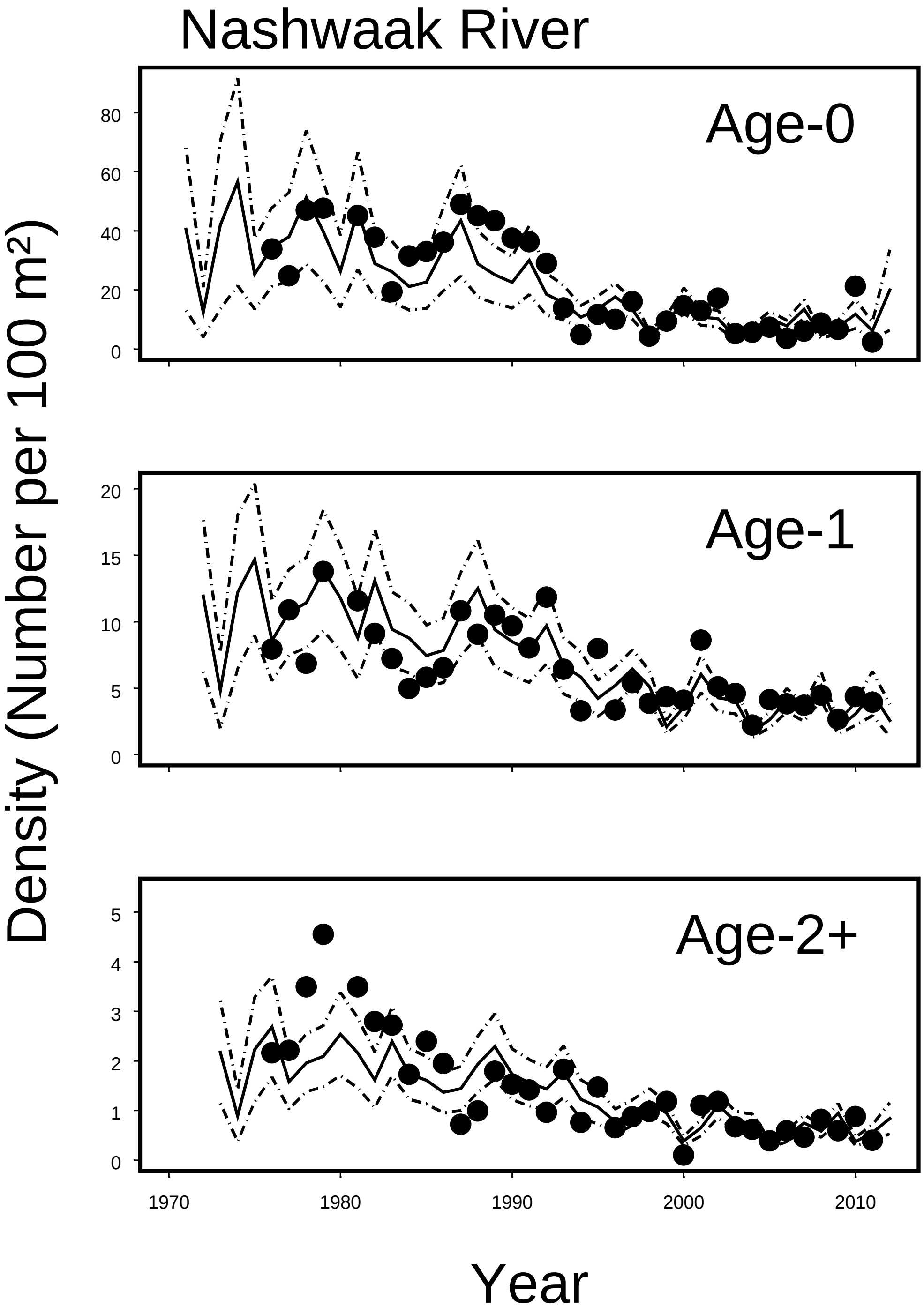

Figure A2.2. Observed (points) and estimated (solid lines) age-0 (top panel), age-1 (middle panel) and age-2+ (bottom panel) juvenile salmon densities from the base population dynamics model for the Nashwaak River Atlantic Salmon population. The dashed lines show 95% confidence intervals based on normal approximations.

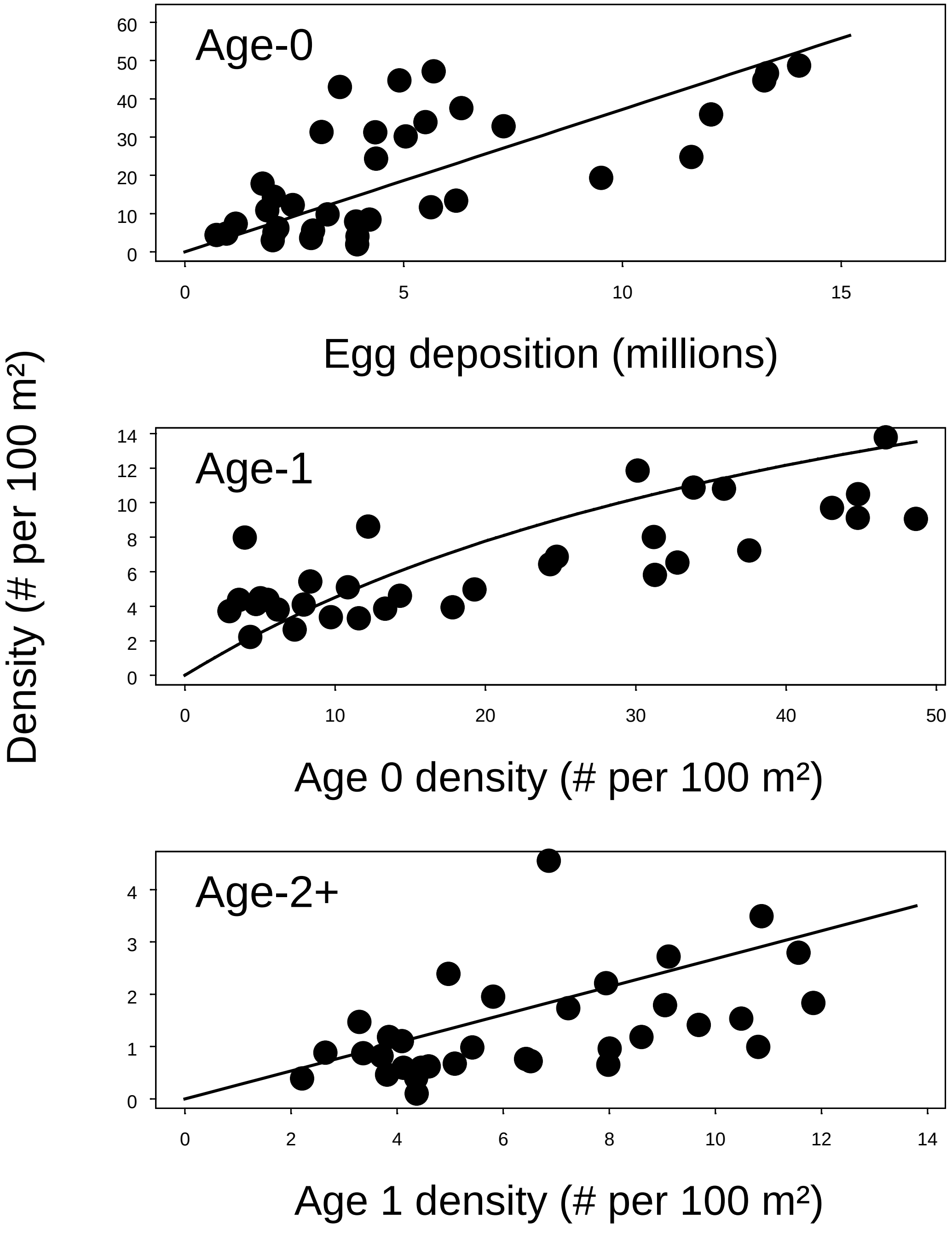

Figure A2.3. Functional relationships between the abundance of eggs, and the densities of age-0 , age-1 and age-2+ juvenile salmon from the base population dynamics model for the Nashwaak River Atlantic Salmon population. The points show the data and the lines show the fitted relationships between age classes.

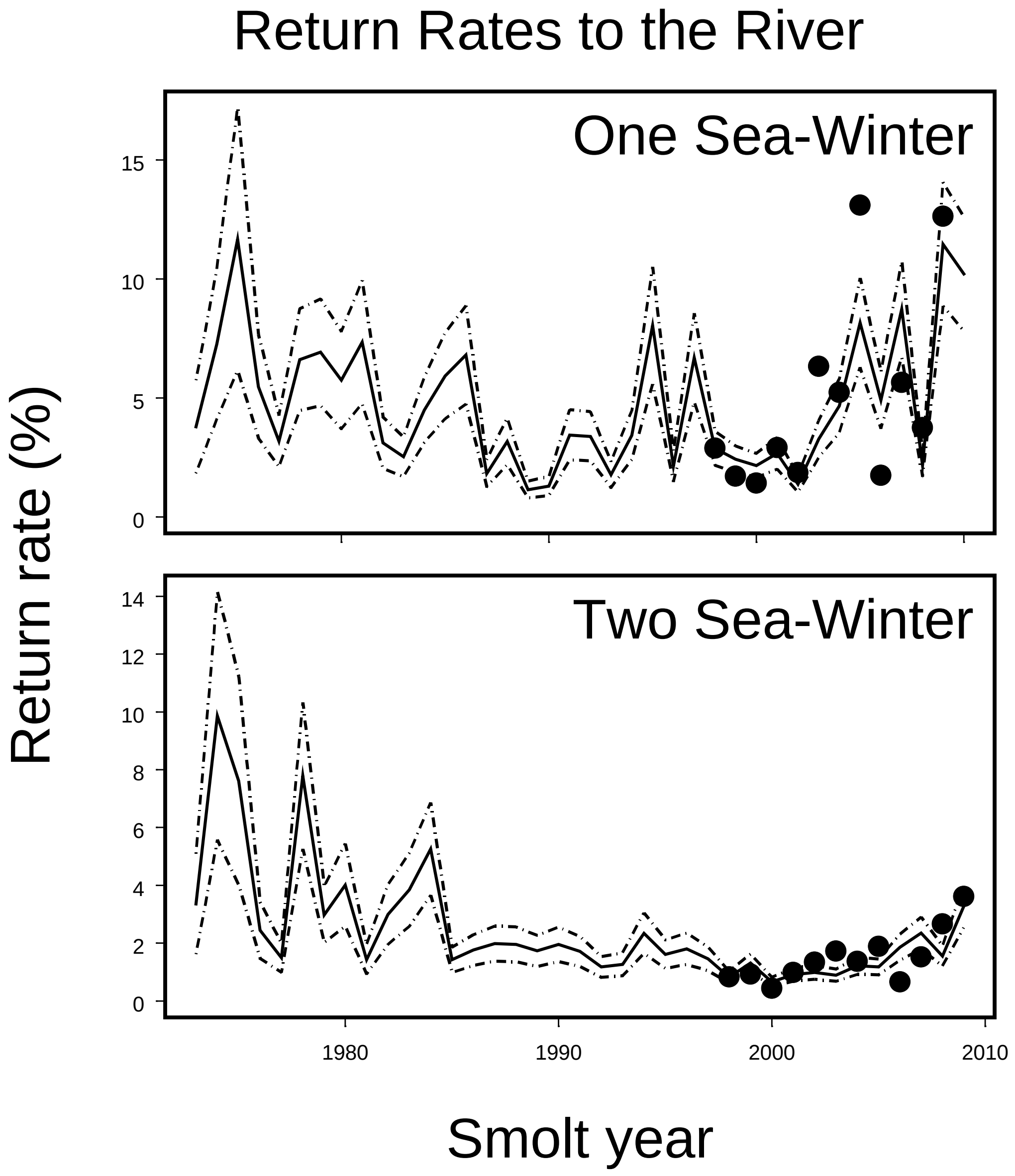

Figure A2.4. Observed (points) and estimated (solid lines) smolt-to-adult return rates to the river mouth (indicative of at-sea survival) for salmon returning as one sea-winter (top panel) and two sea-winter (bottom panel) adults. Estimates are obtained from the base population dynamics model for the Nashwaak River Atlantic Salmon population. The dashed lines show 95% confidence intervals based on normal approximations.

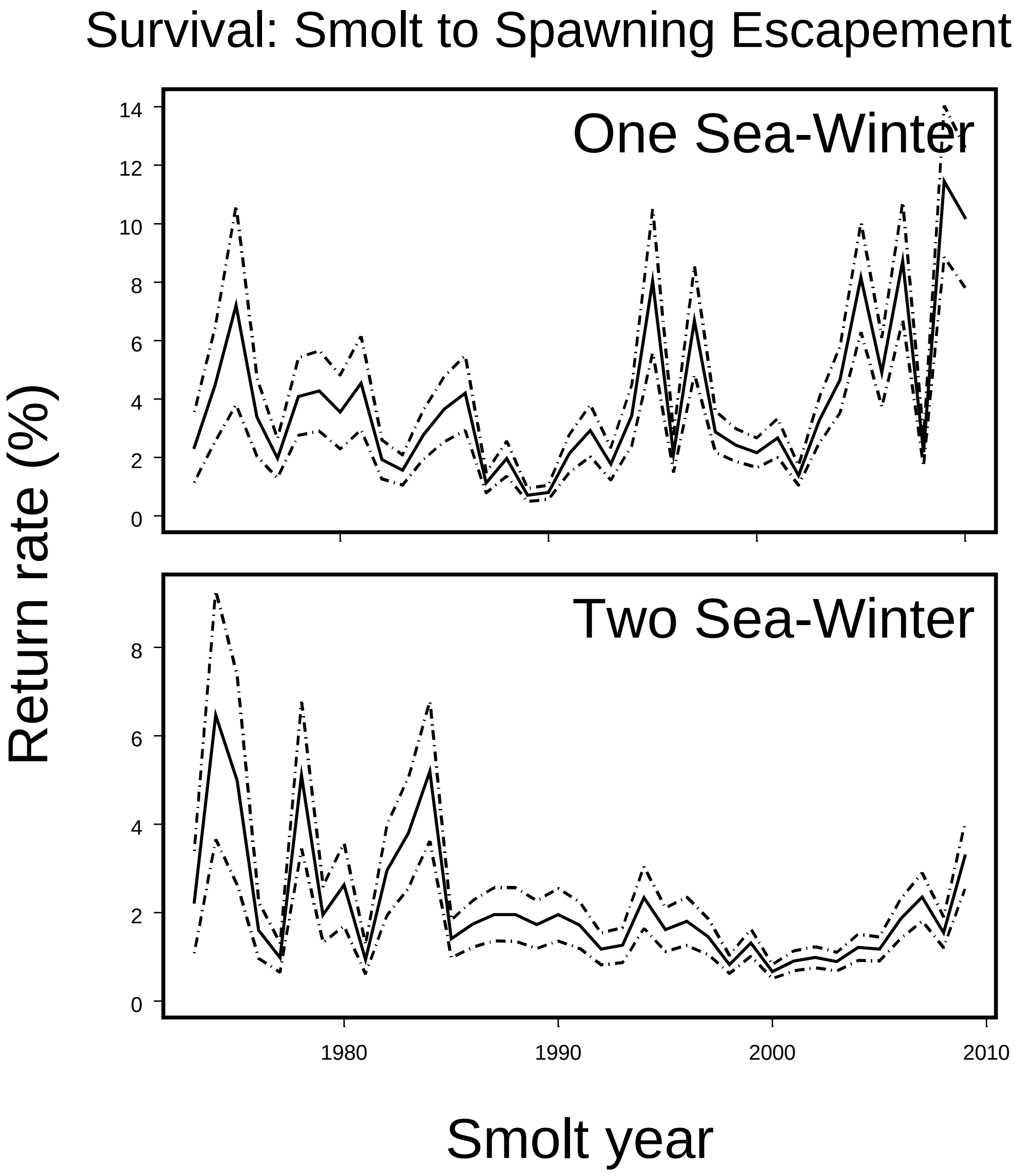

Figure A2.5. Estimated (solid lines) smolt-to adult return rates to spawning escapement (includes the effects of both at-sea survival and the recreational fishery) for salmon returning as one sea-winter (top panel) and two sea-winter (bottom panel) adults. Estimates are obtained from the base population dynamics model for the Nashwaak River Atlantic Salmon population. The dashed lines show 95% confidence intervals based on normal approximations.

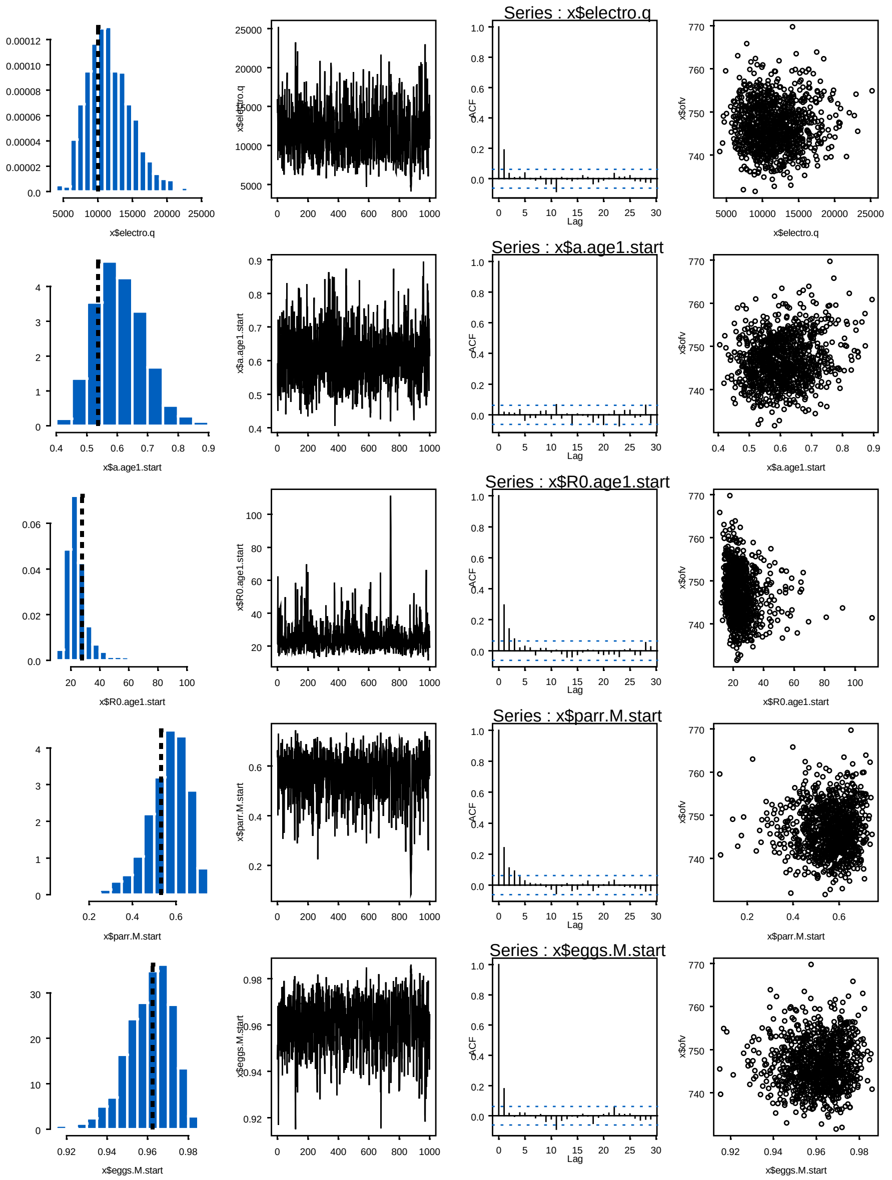

Figure A2.6. Markov Chain Monte Carlo (MCMC) results for the freshwater production model survival parameters for the Nashwaak River base model run. The first column shows the probability density (the dashed lines are the maximum likelihood estimates), the second column shows the thinned chain, the third column shows the auto-correlation in the chain, and the fourth column is a plot of the objective function value versus the parameter value for each step in the MCMC chain.

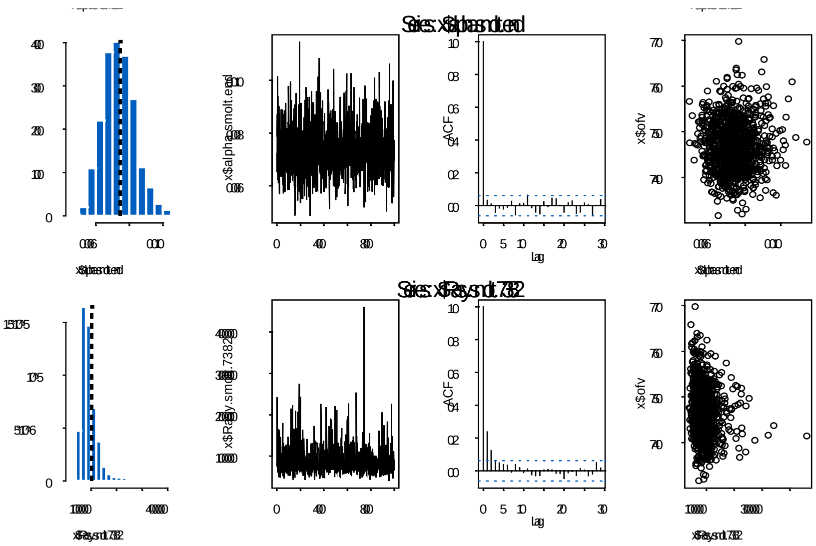

Figure A2.7. MCMC results for two derived model parameters for the Nashwaak River base model run: the egg-to-smolt Beverton-Holt alpha and asymptotic recruitment level and the mean lifetime egg-per smolt values for. A single set of two estimated values were used for the entire time period. The first column shows the probability density (the dashed lines are the maximum likelihood estimates), the second column shows the thinned chain, the third column shows the auto-correlation in the chain, and the fourth column is a plot of the OFV versus the parameter value for each step in the MCMC chain.

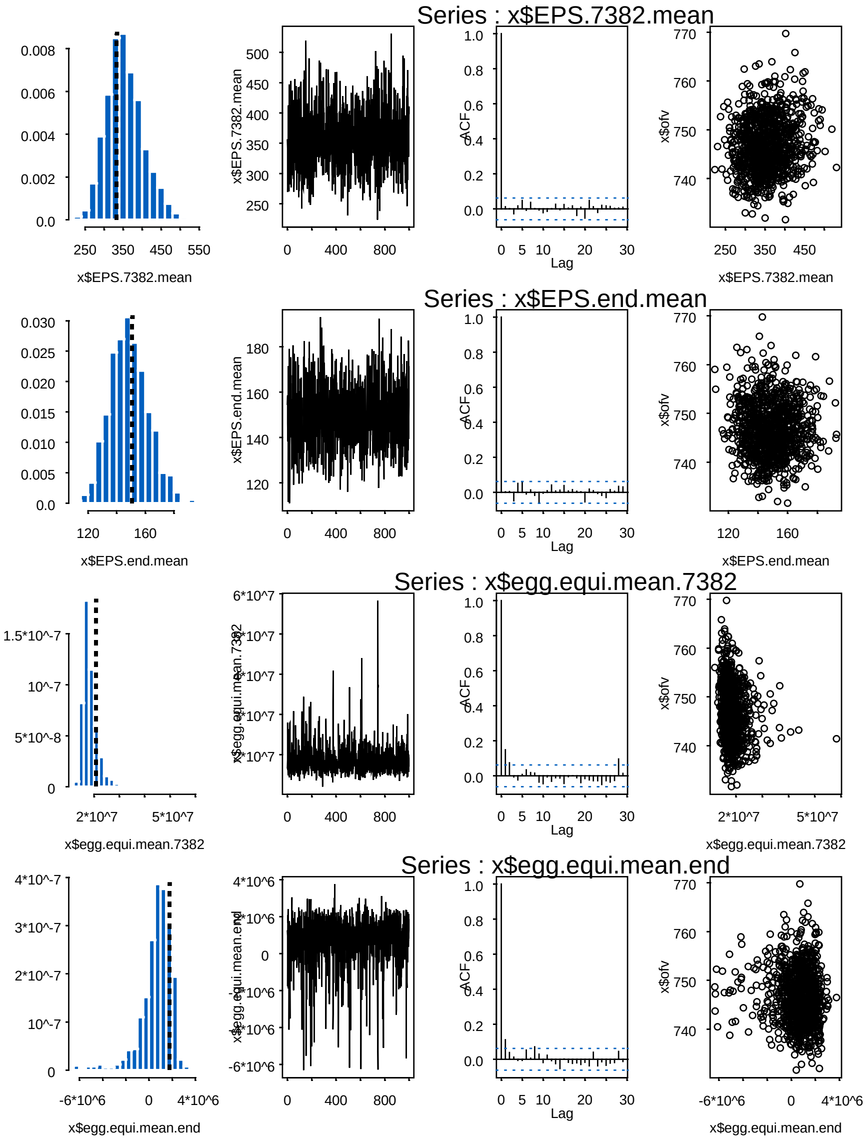

Figure A2.8. MCMC results for three derived model parameters (mean lifetime egg production per smolt (EPS) and the equilibrium abundance of eggs) for two time periods (start = 1973-1982; end = 2000-2009) for Nashwaak River base model. The first column shows the probability density (the dashed lines are the maximum likelihood estimates), the second column shows the thinned chain, the third column shows the auto-correlation in the chain, and the fourth column is a plot of the OFV versus the parameter value for each step in the MCMC chain.

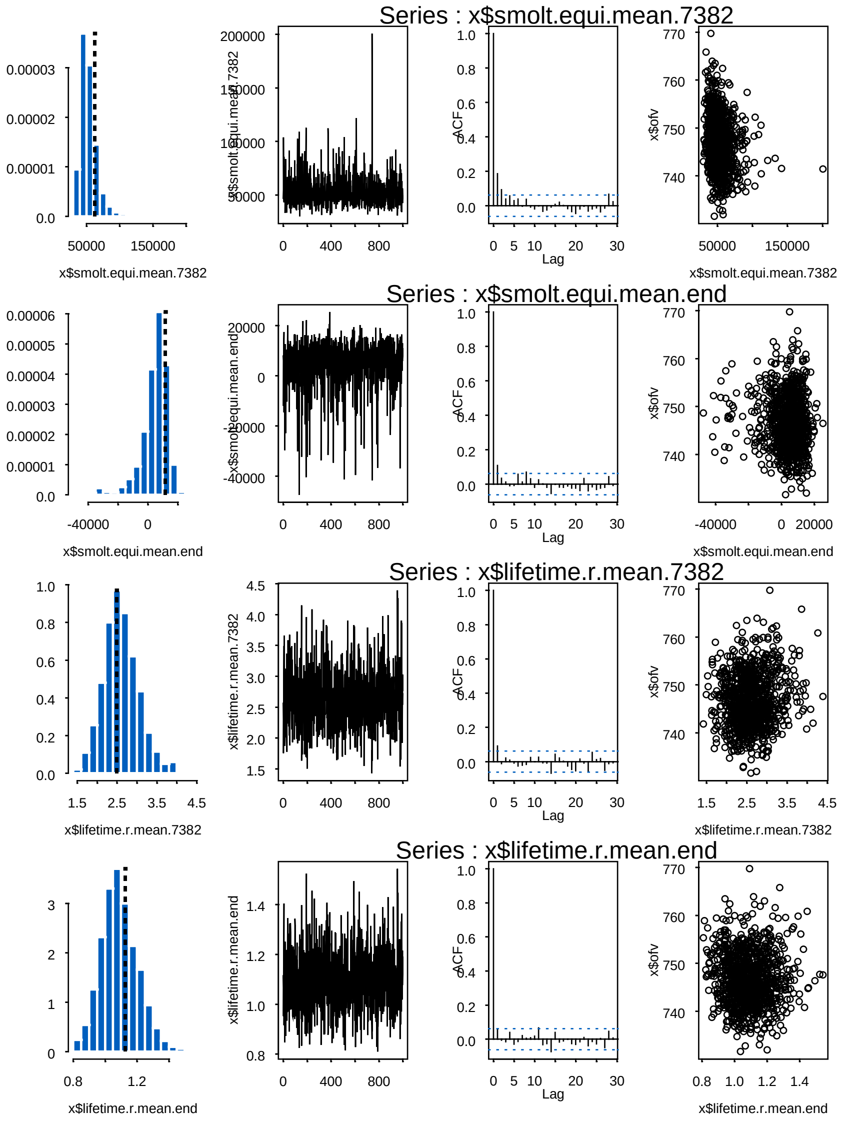

Figure A2.9. MCMC results for two derived model parameters (mean equilibrium abundance of smolts, and maximum lifetime reproductive rate) for two time periods (start = 1973-1982; end = 2000-2009) for Nashwaak River base model. The first column shows the probability density (the dashed lines are the maximum likelihood estimates), the second column shows the thinned chain, the third column shows the auto-correlation in the chain, and the fourth column is a plot of the OFV versus the parameter value for each step in the MCMC chain.

APPENDIX 3. DESCRIPTION OF MODELS USED FOR THE POPULATION VIABILITY ANALYSES FOR OUTER BAY OF FUNDY ATLANTIC SALMON

The following text is slightly modified from Gibson and Bowlby (2013).

The population viability analyses presented in Sections 3 and 4 were carried out using a forward projecting population model developed specifically for the life history of Outer Bay of Fundy Atlantic Salmon. Population viability analysis (PVA) is a powerful tool to explore current conditions, assess risks and simulate how future management actions or environmental changes could influence the abundance of a population (Reed et al. 2002).The dynamical model (i.e. describing the life history) is identical to the one described in Appendix 1, with a slight variation in that all repeat spawning salmon are assumed to be consecutive year spawners after their second spawning (they may be alternate or consecutive year spawners between their first and second spawning). This variation has very little effect on the model output because it only slightly modifies the probability of surviving between spawning events and very, very few salmon survive to spawn a third time.

The dynamical equations used to project the population forward through time are the same as those for the life history model presented in Appendix 1, with the addition of random variability in survivals and stage transition probabilities (described below). A starting abundance equal to the average population size for the years 1997 to 2011 are used to initialize the model.

Two sets of life history parameter values are used in the simulations: those that are representative of the 1973 to 1982 cohorts (past dynamics) and those that are representative of the 2000 to 2009 cohorts (present dynamics). These values are those estimated using the statistical model and are provided in Section 2.

For both the past and present scenarios, the numbers of eggs, parr, smolt and adults, as well as their age, sex and previous spawning structure, are calculated using the mean life history parameter values specific to the simulation, corresponding to the starting population size.

Random variability was incorporated into future mortality rates, sex ratio, and maturity schedules for greater biological realism (Shelton et al. 2007). A lognormal distribution was used for the deviates around the mortality parameters (or functions), and a logistic distribution for the probabilities of smoltification, the proportion maturing after one winter or two winters at sea and the probability of being a consecutive or alternate year repeat spawner, as well as the sex ratio parameters. Lognormal distributions are often used to model the deviates around survival functions as survival is multiplicative in nature. Given that sex ratio and maturity are proportions, the logistic transformation better describes the binomial nature of their error distributions.

Deviates are expected to be temporally autocorrelated (Hilborn 2001) given that the effect of environmental variability on population vital rates tends not to be completely random (Lande et al. 2003). As the strength of this autocorrelation increases, good years are increasingly likely to be followed by good years (and bad followed by bad).

Example of how lognormal variability in survival parameters was incorporated:

Let $\overline{M}$ equal the average instantaneous rate of mortality affecting a life stage of salmon (as estimated using the life history model), σ_w equal the standard deviation of the residuals of the mortality rate (the amount of variability in the rate) and d be a constant describing the degree of autocorrelation. The instantaneous mortality used in the forward projection in year t is given by:

$$M_t = \overline{M} + w_t + \sigma_w^2 / 2$$

where

$$w_t = (w*_{t-1}\, d + w*_t)\sigma_w$$

and

$$w*_t \sim N(0,1).$$

The parameter M_t is then used to model the survival between 2 ages, following the general format of:

$$N_{t+1} = N_t \exp(-M_t)$$

Note that although the survival estimated by this algorithm is not strictly bounded to fall below one, for practical purposes it meets this criterion given the rates used in this analysis. In multiple model runs of 2000 simulated populations, each with several life history parameters, at no time was a survival value greater than one simulated with this algorithm.

For the population projection model used in the PVA (above), the random variability in the egg, parr and mature salmon mortality parameters M_t^{Egg}, M_t^{Parr}, M_t^{Mat} were modeled in this fashion after converting the annual mortality rates estimated with the statistical model to instantaneous rates. Additionally, the return rates for 1SW and 2SW salmon were converted to instantaneous mortality rates and modeled similarly, and random variability was included around the age-0 to age-1 survival function in the same way.

Example of how logistic variability was incorporated into stage transition probabilities and sex ratios:

Let p be the mean parameter value in the form of a proportion. The logit mean of the parameter (S) becomes:

$$S = \ln(p/(1-p))$$

Autocorrelated random deviates for t years are calculated as:

$$w_t = (w*_{t-1}\, d + w*_t)\sigma_w$$

where

$$w*_t \sim N(0,1),$$

where d and σ_w are as described above.

The annual probability becomes:

$$p_t = \exp(S + w_t)/(1 + \exp(S + w_t))$$

where p_t is the probability of transitioning from one life stage to another (e.g. smolting or not smolting at a given age, maturing or not maturing at a given age, etc.) in the given year.

Random variability and autocorrelation:

The values used for random variability and autocorrelation are those used by Gibson and Bowlby (2013). Based on their analysis of autocorrelation in the return rate time series for the LaHave River and St. Mary's River populations, the autocorrelation coefficient in the marine environment was set to 0.45, and a (instantaneous) return rate variance of 0.475 was used for the PVA's. The same autocorrelation coefficient value was assumed for the freshwater environment as was derived for the marine environment. However, random variability was assumed to be lower in the freshwater environment and values of $\sigma = 0.2$ were assumed for all other model parameters, except for the probability of smoltification for which a value of 0.3 was assumed. Within limits, the general extinction patterns are not overly sensitive to perturbations of the variances (i.e. higher or lower values for σ), although the time to extinction does vary as more or less variability is assumed. Examples of a simulated mortality rate and smoltification probability time series are shown in Figures A3.1 and A3.2 to illustrate how mortality varies in the PVA. Note that, because the same random numbers are used to generate the series, the pattern is the same for each parameter, but the values are re-scaled by the average rates.

Catastrophic events:

Atlantic Salmon occupy naturally variable habitats that are at times subject to extreme conditions. Floods and droughts in fresh water are examples of these, both of which can lead to very high mortality in one or many of the juvenile life stages. The effects of extreme events are included in the model using two parameters. The first parameter is the frequency parameter, ψ, which is the expected number of these events in 100 years. A random number, υ_t, is drawn from a uniform distribution [0,1] for each year in each simulated population trajectory, and the value $1/\psi$ is compared to υ_t. If $\upsilon_t < 1/\psi$, that year is considered an extreme event year. The second parameter, ϑ, is used to model the effect of the event. In this analysis, the effect of the event was included between the egg and the fry life stages, thereby allowing density-dependent compensation to occur which would partially offset some of the mortality (because the survival of age 0 to age 1 increases as population size decreases). The effect of the extreme events would be greater if it was incorporated after density dependence. The simulated number of fry, $P_{t,0}$, is then:

$$P_{t,0} = \left\langle \begin{array}{ll} Egg_t e^{-M_t^{Egg}} \vartheta & \text{if } \upsilon_t < 1/\psi, \quad \upsilon_t \sim \text{unif}[0,1] \\ Egg_t e^{-M_t^{Egg}} & \text{otherwise} \end{array} \right\rangle .$$

In the absence of specific information about the frequency and effects of extreme events, values of 10 and 0.2 were assumed for ψ and ϑ, respectively. This means that on average, 10 events reducing the abundance of fry by 80% from the expected value would occur every 100 years. As modeled, a greater or lesser number of extreme events could occur in any simulated population trajectory, and their distribution through time is random. To illustrate the effects of including extreme events, 1000 random survival values were generated assuming a mean survival of 0.5 and $\sigma = 0.2$. These values are compared to a set of random survivals including catastrophic events assuming values of 10 and 0.2 for ψ and ϑ, respectively (Figure A3.3). In this example, the median survival is reduced from 0.488 to 0.476 when extreme events are included.

Probability of extinction and recovery:

For each scenario analyzed with the PVA, 2000 population trajectories were simulated and the extinction probabilities are calculated as the proportion of populations that go extinct by a specified time. A quasi-extinction threshold of 15 females is assumed and an egg deposition of zero is assigned if the abundance drops below this value. A population must be below this value for two consecutive years to be assumed extinct in a given year. If the female abundance is higher the next year, the egg deposition is calculated as per the model. A population can therefore sit on the quasi-extinction for a number of years and can theoretically recover. Recovery probabilities were calculated as the proportion of the simulated population trajectories that were above the recovery target in a given year. As such, a population could be in a recovered state for a period of time, and then cease to be considered recovered if its abundance subsequently declined to a level below the recovery target.

In instances where comparisons were made between scenarios, the same set of random numbers was used to generate variability in parameter values to ensure that the differences between the scenarios do not occur by chance (i.e. because a different set of numbers is used).

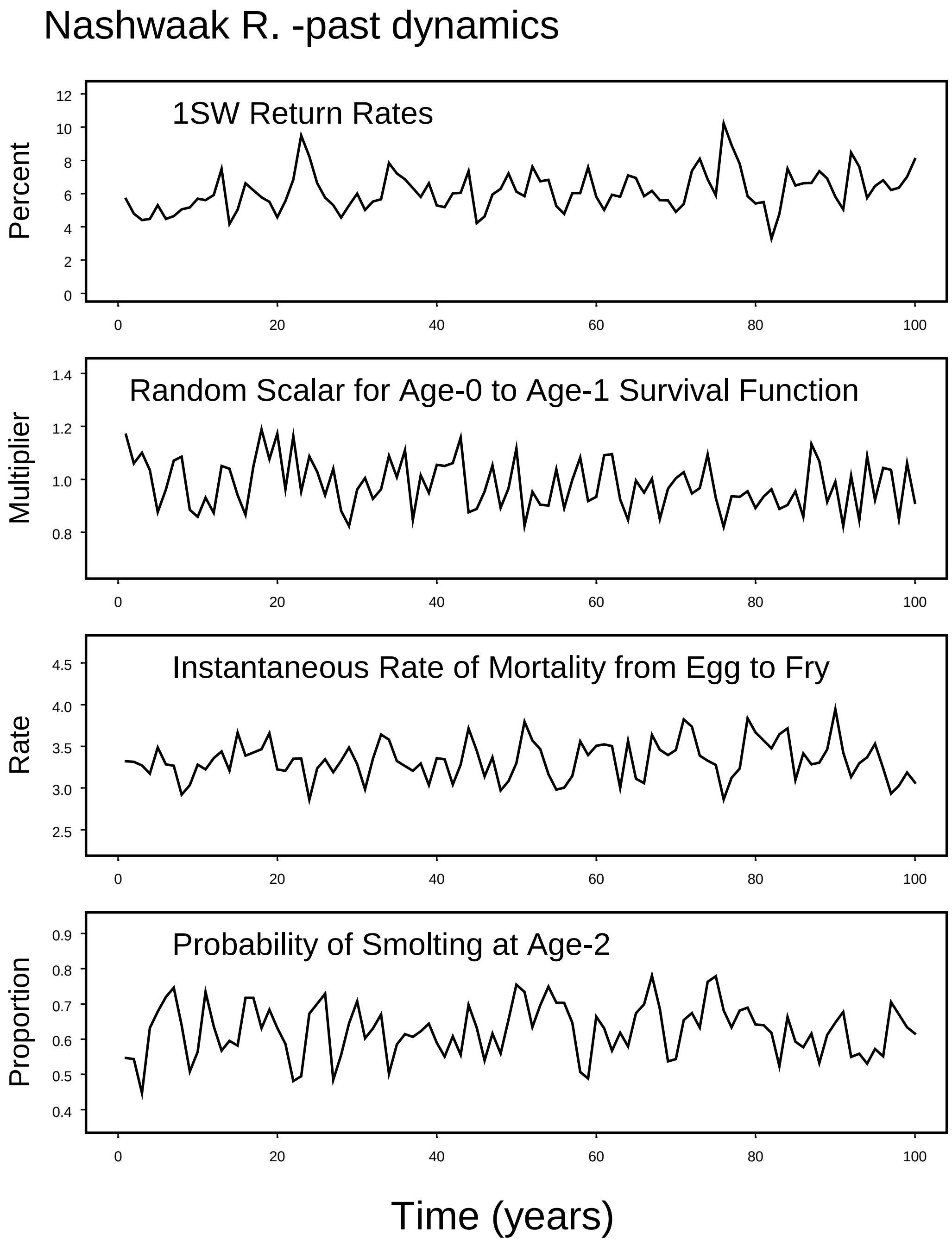

Figure A3.1. Examples of how the life history parameter values used change through time given the autocorrelation values and extent of random variability used in the analyses. Starting values for the parameters are from the base case population viability analysis for Nashwaak using past (1973-1982) population dynamics. Values are for a single 100-year stochastic projection.

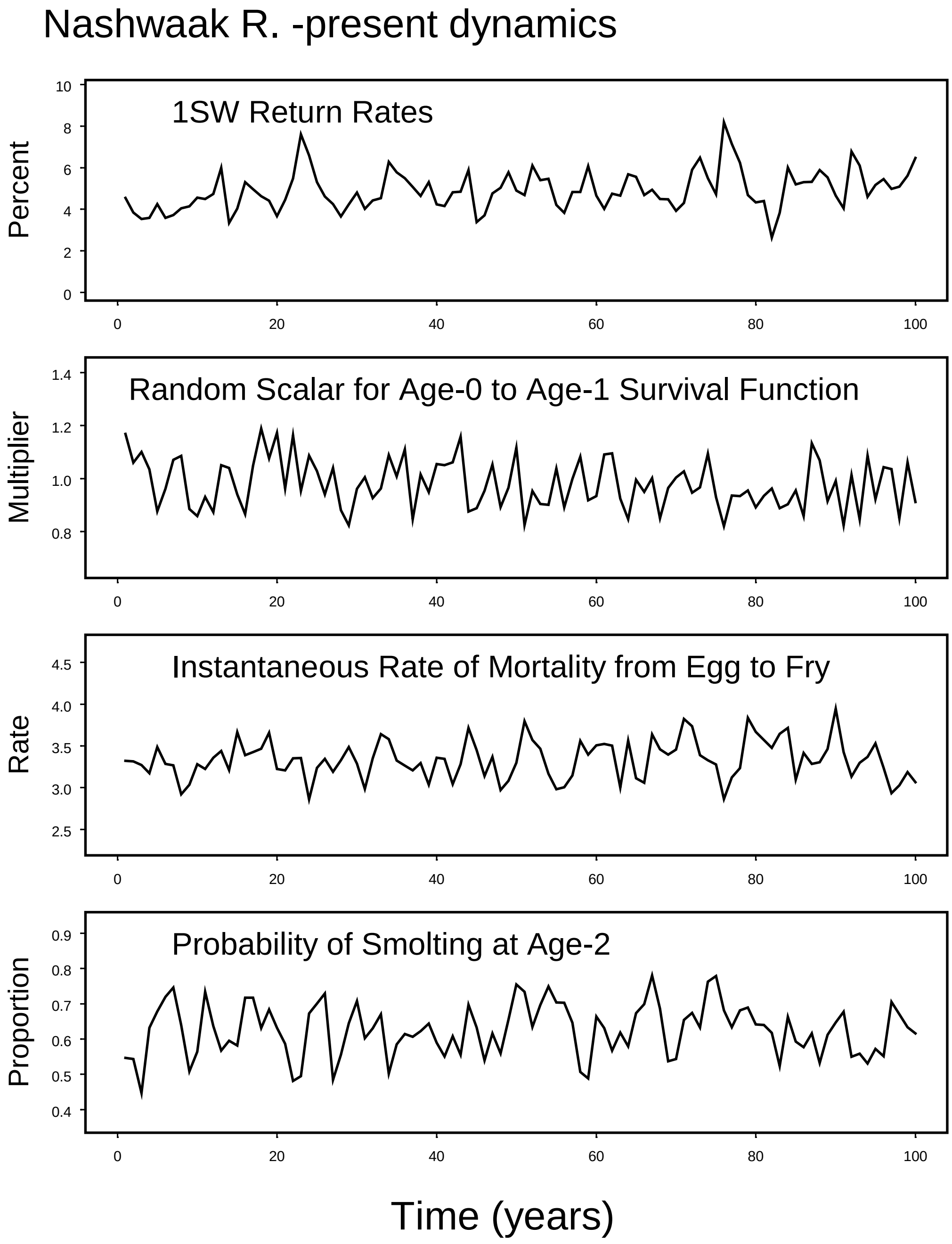

Figure A3.2. Examples of how the life history parameter values used change through time given the autocorrelation values and extent of random variability used in the analyses. Starting values for the parameters are from the base case population viability analysis for Nashwaak River using present (2000's) population dynamics. Values are for a single 100-year stochastic projection.

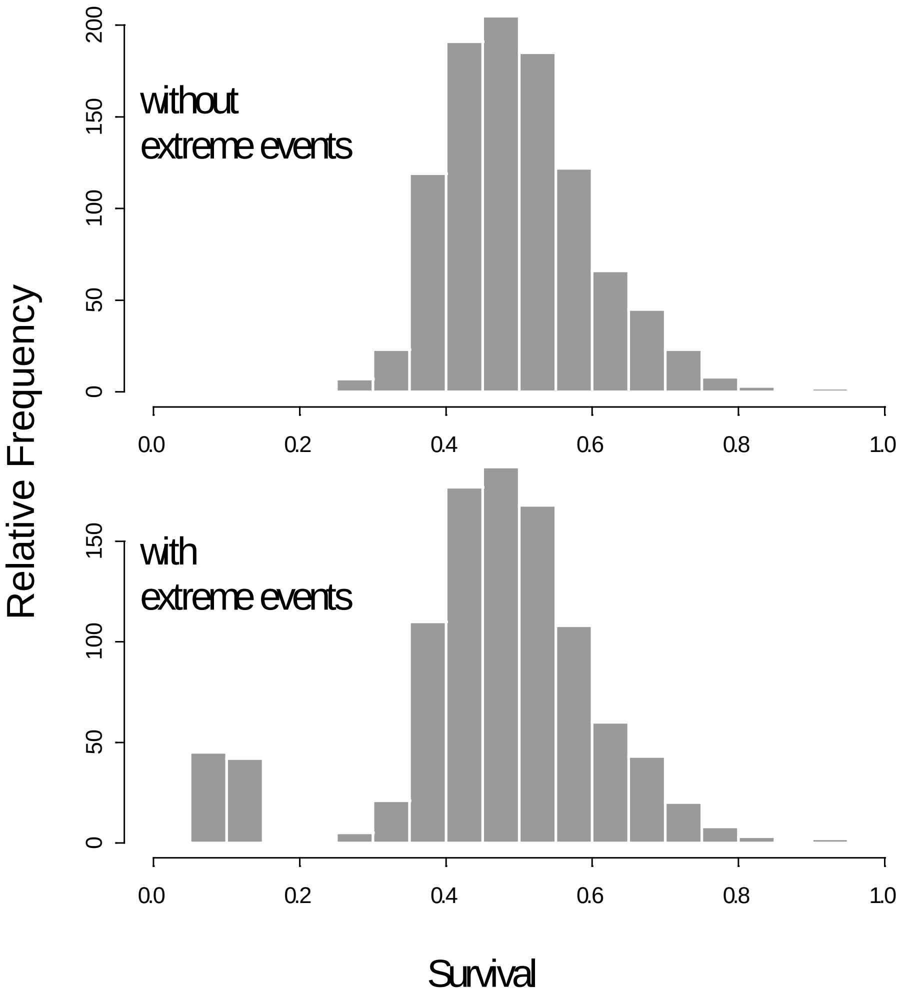

Figure A3.3. Comparison of two sets of 1000 random survival values generated assuming a mean survival of 0.5 and $\sigma = 0.2$ *with the distribution in the lower panel including catastrophic events. Values of 10 and 0.2 were assumed for* ψ *and* ϑ*, respectively (from Gibson and Bowlby 2013).*

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Research Document 2006/016

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Population Regulation in Eastern Canadian Atlantic salmon (*Salmo salar*) populations

Régulation des populations de saumon atlantique (*Salmo salar*) de l'est du Canada

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ISSN 1499-3848 (Printed / Imprimé)

FOREWORD

This document is a product from a workshop that was not conducted under the Department of Fisheries and Oceans (DFO) Science Advisory Process coordinated by the Canadian Science Advisory Secretariat (CSAS). However, it is being documented in the CSAS Research Document series as it presents some key scientific information related to the advisory process. It is one of a number of contributions first tabled at a DFO-SARCEP (Species at Risk Committee / *Comité sur les espèces en péril*) sponsored workshop in Moncton (February 2006) to begin the development of a 'Conservation Status Report' (CSR) for Atlantic salmon. When completed in 2007, the CSR could form the basis for a Committee on the Status of Endangered Wildlife in Canada (COSEWIC) status report, recovery potential assessment and recovery strategy, and most importantly, enable DFO to implement pre-emptive management measures prior to engagement in any listing process.

AVANT-PROPOS

Le présent document est issu d'un atelier qui ne faisait pas partie du processus consultatif scientifique du ministère des Pêches et des Océans, coordonné par le Secrétariat canadien de consultation scientifique (SCCS). Cependant, il est intégré à la collection de documents de recherche du SCCS car il présente certains renseignements scientifiques clés, liés au processus consultatif. Il fait partie des nombreuses contributions présentées au départ lors d'un atelier parrainé par le MPO-SARCEP (*Species at Risk Committee* / Comité sur les espèces en péril) à Moncton (février 2006) en vue de commencer l'élaboration d'un rapport sur la situation de la conservation du saumon atlantique. Lorsqu'il sera terminé, en 2007, ce rapport pourrait servir de base à un rapport de situation du Comité sur la situation des espèces en péril au Canada (COSEPAC), à une évaluation du potentiel de rétablissement et à un programme de rétablissement mais, avant tout, il permettra au MPO de mettre en œuvre des mesures de gestion anticipées avant même de s'engager dans un processus d'inscription.

Abstract

The timing and nature of density-dependent survival in Atlantic salmon populations was analysed using electrofishing data from nine populations in the Maritime Provinces to evaluate its role in freshwater environments, and smolt-to-adult return-rate data from 15 populations in eastern Canada to evaluate its importance in marine habitat. Three spawner-recruit models, a Beverton-Holt, a Ricker and a one-parameter density-independent model, were fit to each data series using maximum likelihood, and model fits were compared using likelihood ratio tests. Within fresh water, no single, unequivocal pattern was evident with respect to the timing of density dependence. Of the six egg-to-age-0 transitions, the addition of a second parameter for density dependence resulted in a statistically better fit in three cases. In six of the nine age-0-to-age-1 transitions, the addition of second parameter provided a statistically better fit, as was also the case in three of the nine age-1-to-age-2 comparisons. Of the nine populations, density dependence was not detected in two populations, was detected in only one transition in two populations, detected in two transitions in four populations and was detected in all three transitions in the remaining population. Overcompensation was not detected in these data. The Ricker model (which exhibits overcompensation) did not provide a statistically significantly better fit in any of the 25 comparisons, and when comparisons were made over all populations, the Beverton-Holt model (which does not exhibit overcompensation) provided a statistically better fit for the three age class transitions investigated here. In the marine environment, density dependence was potentially detected in three of the 15 return-rate data series for salmon maturing after one winter at sea, but was not detected in any of the nine return-rate data series for fish maturing after two winters at sea. Carrying capacity for age-1 salmon was found to be highly variable among populations. Using a mixed-effects model, the median carrying capacity was estimated to be 24.8 parr/100m^2 with 95% of the probability density falling between 3.8 and 165.9 parr/100m^2. The variability in both the timing of density dependence and carrying capacity for parr highlights the need for population-specific data for establishing reference points or when planning recovery or enhancement activities.

Résumé

Nous avons analysé les paramètres temporels et la nature de la survie dépendante de la densité chez les populations de saumon atlantique à l’aide de données provenant de l’électropêche menée sur neuf populations des provinces maritimes afin d’évaluer le rôle de cette espèce dans les environnements d’eau douce. Nous nous sommes également servis de données sur les taux de remonte saumoneaux-adultes de 15 populations de l’est du Canada pour évaluer l’importance de ce poisson dans l’habitat marin. Nous avons ajusté trois modèles géniteurs-recrues (modèle de Beverton-Holt, modèle de Ricker et modèle à un paramètre indépendant de la densité) à chaque série de données selon le maximum de vraisemblance, et nous avons ensuite comparé les ajustements des modèles au moyen de tests du ratio de vraisemblance. En eau douce, aucun profil univoque n’est ressorti en ce qui concerne les paramètres temporels de la dépendance à la densité. Dans trois des six transitions œuf-âge 0, l’ajout d’un deuxième paramètre pour la dépendance à la densité a résulté en un meilleur ajustement sur le plan statistique. Dans six des neuf transitions âge 0-âge 1, l’ajout d’un deuxième paramètre a également donné un meilleur ajustement sur le plan statistique, de même que dans trois des neuf comparaisons entre l’âge 1 et l’âge 2. Parmi les des neuf populations, nous n’avons relevé aucune dépendance à la densité chez deux populations, une dépendance à la densité dans une transition uniquement chez deux populations, dans deux transitions chez quatre populations et une dépendance à la densité dans chacune des trois transitions chez la les autres populations. Nous n’avons relevé aucune surcompensation dans ces données. Le modèle de Ricker (qui montre la surcompensation) n’a pas donné un ajustement vraiment meilleur sur le plan statistique dans aucune des 25 comparaisons, et lorsque nous avons comparé toutes les populations, le modèle de Beverton-Holt (qui ne montre pas la surcompensation) a donné un meilleur ajustement sur le plan statistique pour les trois transitions des classes d’âge présentement étudiées. En milieu marin, nous avons probablement relevé une dépendance à la densité dans trois des 15 séries de données sur les taux de remonte des saumons arrivant à maturité après un hiver en mer, mais nous n’avons pas relevé de dépendance à la densité dans aucune des neuf séries de données sur les taux de remonte des poissons arrivant à maturité après deux hivers en mer. La capacité biotique du saumon d’âge 1 s’est révélée grandement variable d’une population à l’autre. À l’aide d’un modèle à effets mixtes, nous avons estimé que la capacité biotique moyenne est de 24,8 tacons/100m2, alors que 95 % de la densité de probabilité descend entre 3,8 et 165,9 tacons/100m2. La variabilité des paramètres temporels de la dépendance à la densité et de la capacité biotique des tacons démontre bien que nous devons recueillir des données propres aux populations pour établir des points de référence ou planifier des activités de rétablissement ou de mise en valeur.

Introduction

The status of Atlantic salmon populations in the Atlantic Provinces varies regionally. Populations around the Bay of Fundy and Nova Scotia's Atlantic coast are in strong decline, whereas the status of Nova Scotia and New Brunswick Gulf of St. Lawrence populations range from declining to stable (DFO 2003, Gibson and Hubley 2006). In Newfoundland and Labrador, some populations are increasing, whereas others are decreasing or stable (Dempson et al. 2006, DFO 2005, Gibson and Hubley 2006). Atlantic salmon populations have a rich, complex life history that is highly variable (Hutchings and Jones 1998). Variability and plasticity in life history characteristics, such as size and growth in fresh water, biological characteristics of smolts, size and growth in marine water, survival in fresh water and at sea, and fecundity for eastern Canadian populations is summarised by O'Connell et al. (2006). The timing and nature of population regulation in juvenile salmon populations in Atlantic Canada may be correspondingly variable; a hypothesis investigated using meta-analysis in this document.

The concept of population regulation is closely tied to the concept of population persistence. Although population size may fluctuate widely through time, long-term persistence and a tendency not to grow unchecked implies a regulatory mechanism that controls population size (Royama 1992). Such regulation may be density-independent or density-dependent, termed fragile and robust regulation by Royama (1992). While it's possible for populations to persist without exhibiting continual growth in the absence of density dependence, it is extremely unlikely. Under these conditions, population size over time should behave as a random walk, a behavior that is inconsistent with the concepts of persistence and a bound on population size. So, while density-independent factors can markedly influence population size, questions about the timing and nature of population regulation are really questions about the timing and nature of density dependence within the population. Here, we focus on density-dependent survival, although density dependence can influence other life history characteristics such as growth and fecundity.

The nature of population regulation in salmonids is of more than theoretical interest. The question of when and how year class size is determined is a fundamental question in fisheries biology that spans a time period of nearly 100 years (e.g. Hjort 1914, Myers and Cadigan 1993), the answer to which has implications for many questions in fisheries management, including the effects of fishing on abundance, as well as being a key determinant of the effectiveness of stocking. It is also important for the development of models used for setting biological reference points (Clark 1991), determining the limits of exploitation (Myers and Mertz 1998a), and for evaluating the potential benefits of recovery activities for at-risk populations (Trzcinski et al. 2004).

In the context of population regulation, compensatory density dependence (where the rate of population increase is negatively correlated with population size) is implied. When compensatory density dependence occurs over the full range of population sizes, survival between life stages is a decreasing monotonic function of the population size such that the maximum survival rate occurs at a population just greater than zero. An alternative scenario, an abundance threshold below which survival switches to an increasing function of population size, is also possible and is termed depensation (Clark 1976). However, due to its de-stabilizing characteristics, it is not a regulatory mechanism (Rose et al. 2001).

The concept of depensation, which occurs at low abundance, is controversial in fish population biology, with most information coming from spawner-recruitment (SR) analyses. Myers et al. (1995b) did not find evidence of depensation in 125 of 128 spawner-recruit time series they examined. Liermann and Hilborn (1997) conducted a similar analysis with a different depensatory model and concluded that depensation may be more common that suggested by Myers et al. (1995b). Barrowman et al. (2003) did not find evidence of depensation for coho salmon, a similar result to Gibson and Myers (2003) for alewife. Both authors found that most SR data sets are not informative about the shape of the SR function at the origin. However, there is empirical evidence that depensation may exist based on population recovery. Populations that undergo large declines often do not rapidly recover (Hutchings 2000, Hutchings 2001), potentially indicating that depensatory population dynamics may be quite common.

There is also uncertainty about the shape of the SR curve at high abundance, but typically one of two options is used (Hilborn and Walters 1992). The first is the Beverton-Holt model, which is based on the assumption that competition within a cohort results in a mortality rate that is a linear function of the number of fish alive in the cohort at any time. The result is an SR curve that behaves asymptotically: recruitment increases as spawner abundance increases, albeit very gradually at high abundance. The other model is the Ricker model, which is based on the assumption that the mortality rate is dependent on the initial cohort size. Based on this assumption, increasing spawner abundance leads to increases in recruitment up to a maximum, after which further increases in spawner abundance have the effect of reducing the number of recruits. This phenomenon is known as overcompensation. Solomon (1985) reviewed the evidence for both dome-shaped and asymptotic curves for Atlantic salmon, and concluded that asymptotic curves were most appropriate. However, as acknowledged by the author, the possibility exists that many of the populations were at low enough levels that a descending right-hand limb might not be detectable.

While several other SR functions have been proposed, one that warrants mention for salmonids is the hockey-stick model (Barrowman and Myers 2000). This model is based on the concept of territoriality: some finite number of territories are available to a population. As abundance increases, survival is density-independent until all territories are filled, after which it is density-dependent.

Rose et al. (2001) provide a detailed review of compensation in fish populations. They suggest that density-dependent mortality can be caused by density-dependent responses by predators or via density-dependent growth, reproduction or movement affecting survival, and summarize considerable evidence for each case. The concept of density-dependent mortality is thus closely tied to that of carrying capacity and resource limitation (Beverton 1995). For diadromous species such as Atlantic salmon, the potential mechanisms for density-dependent mortality likely vary with environments. In fresh water, resource limitation (bottom-up control) may lead to density-dependent mortality, as implicated when growth rates are also density dependent. In the marine environment, resources are likely less limiting as fish are less concentrated, but density-dependent predation (top-down control) is a plausible mechanism. Density-dependent mortality as result of predation may occur as either a numerical (increase in the number of predators) or functional (behavioral responses that change predation rates) response to the prey density (Begon et al. 1990). In the case of salmon, it is unlikely that overall predator abundance increases on the relatively short time period that salmon remain in a particular environment. However, predators may aggregate in response to increased salmon abundance, a functional response similar to that suggested for predators foraging on reef fish (Hixon 1998).

There is considerable evidence for density dependence in the juvenile Atlantic salmon life stages in fresh water, although in some studies, such as those in highly productive areas, it was not observed (Gibson 1993). For example, O'Connell et al. (2006) show strong evidence of density dependence between the egg and smolt life stages in Western Arm Brook, Northeast Brook, Trepassey River, and Conne River in Newfoundland. However, while there is evidence of density dependence in salmon in freshwater habitat, the mechanism is less well understood and the point at which density dependence begins to operate, its intensity and its precise form are less clear (Milner et al. 2003). Density-dependent size-at-age has been demonstrated for salmon parr (Gibson 1993, Korman et al. 1994, Amiro et al. 2003) which could lead to density-dependent survival. Armstrong and Griffiths (2001) found that the proportion of parr sheltering in an indoor stream decreased with increasing density, an observation that may have consequences for over-wintering survival as well as carrying capacity. While these studies suggest that density dependence may occur at older life stages, other studies indicate that density dependence occurs only in the very early life stages during a 'critical period' (Elliott 2001). This latter position is commonly adopted. In a recent review of population regulation in salmon and trout populations, Milner et al. (2003) conclude that density dependence in salmon is likely sustained longer than in trout and may last at least through the first summer. Thereafter, up to the smolt stage, survival has generally been found to be density independent.

In the next two sections, the nature and timing of density-dependent survival in salmon populations in eastern Canada is evaluated for freshwater habitat using electrofishing data from nine rivers in the Maritime Provinces, and for the marine environment using smolt-to-adult return rate data for fifteen populations in Eastern Canada.

Density Dependence in Fresh water

While density dependence has been shown to occur for salmon in fresh water, less information is available about the timing and nature of the density dependence. As a first step to developing a population dynamics model for inner Bay of Fundy salmon, Trzcinski et al. (2004) fit models to electrofishing data sets for two inner Bay of Fundy populations to evaluate the timing of density dependence. They compared the fits of a density-independent model to a model with density dependence (a Beverton-Holt stock recruitment model) for three age-class transitions: egg-to-age-0, age-0-to-age-1 and age-1-to-age-2, and concluded there was only evidence of density dependence between age-0 and age-1. Here, their analysis is extended to include data from nine populations in the Maritime Provinces, in an attempt to answer the following questions:

1) When does density dependence occur in fresh water?
2) Is overcompensation characteristic of salmon populations?
3) Can density dependence be quantified from these data?

For this analysis, the annual mean density of age-0, age-1 and age-2 salmon, obtained by electrofishing, and the annual egg depositions estimated from stock assessments are used. A summary of the data series used in the analysis is provided in Table 1.

Methods

Spawner-recruit (SR) models are a widely accepted tool for analyzing population dynamics (Moussalli and Hilborn 1986, Myers et al. 1999). They provide a basis for estimating biological reference points for management (Myers and Mertz 1998b, Gibson and Myers 2004), a method for evaluating the effects of mortality caused by pollution, dams or other human activities (Barnthouse et al. 1988, Hayes et al. 1996), and are a suitable tool for investigating the timing and nature of density dependence (e.g. Myers and Cadigan 1993). Here, three SR models are fit to the egg and juvenile data from the nine rivers. The first of these models is a one-parameter model in which the number (or density) of fish of age *a* in year *t*, denoted $N_{t,a}$, is a density-independent function of the number (or density) in the preceding age class in the preceding year:

$$N_{t,a} = \alpha N_{t-1,a-1}.$$

The interpretation of α differs slightly depending on the data. When fit to the age-0-to-age-1 juvenile densities, it is the density-independent age-specific survival rate between age classes. However, due to the differences in the data reporting (eggs are reported as the estimated egg deposition in the river, whereas the age-0 data is the mean density in the river), when the model fit to the egg-to-age-0 data, α is scalar that is proportional to egg-to-age-0 survival, but differs from the actual survival by the number of effective habitat units within the river (or the ratio of the mean density of fish obtained by electrofishing to the number of fish in the population) which is unknown. Finally, when fit to the age-1-to-age-2 data, α is a composite parameter that combines both survival and probability of smoltification at age-2. In all cases, the true value of α would be between zero and one.

The other two models used here are the most commonly used two-parameter SR models: the Beverton-Holt and Ricker (Hilborn and Walters 1992). These models differ fundamentally in their assumptions of the underlying biology, the latter showing a decline in recruitment at higher spawner abundance, a phenomenon known as overcompensation. The Beverton-Holt model gives $N_{t,a}$, as a density-dependent function of $N_{t-1,a-1}$:

$$N_{t,a} = \frac{\alpha N_{t-1,a-1}}{1 + (\alpha N_{t-1,a-1}/R_{asy})} .$$

Here, α is the slope at the origin, and in the deterministic model is the maximum survival rate between age classes in the absence of density dependence at low population sizes (Myers et al. 1999) and R_{asy} is the asymptotic recruitment level. As $N_{t-1,a-1}$ approaches infinity, R_{asy} is the limit approached by R (Beverton-Holt models are often written in terms of the half saturation constant, K, which is related to R_{asy} by: $R_{asy} = \alpha K$), and is the carrying capacity for age-a fish expressed as a density. The same caveats apply to the interpretation of α when fit to the egg-to-age-0 data.

The Ricker model also gives $N_{t,a}$, as a density-dependent function of $N_{t-1,a-1}$:

$$N_{t,a} = \alpha N_{t-1,a-1} e^{-\beta N_{t-1,a-1}} .$$

Here, the interpretation of α is the same as for the Beverton-Holt model, as written above, and β is a shape parameter that determines how rapidly survival declines as abundance increases.

Parameter estimates for each population and model were obtained using maximum likelihood assuming a lognormal error structure for recruitment (Myers et al. 1995a). Denoting the Beverton-Holt spawner-recruit function as $g(N_{t-1,a-1})$, the log-likelihood is given by:

$$\ell(\alpha, R_0, \sigma) = -n \log \sigma \sqrt{2\pi} - \sum \log N_{t,a} - \frac{1}{2\sigma^2} \sum \log \left(\frac{N_{t,a}}{g(N_{t-1,a-1})} \right)^2$$

where σ is the shape parameter for a lognormal distribution and n is the number of paired observations. We used profile likelihoods to assess the plausibility of the individual parameter estimates given the data. The log profile likelihood for α, $\ell_{\mathrm{p}}(\alpha)$, is:

$$\ell_{\mathrm{p}}(\alpha) = \max_{R_0 \sigma} \ell(\alpha, R_{asy}, \sigma) .$$

The maximum likelihood estimate for α occurs where $\ell_{\mathrm{p}}(\alpha)$ achieves its maximum value. The plausibility of other possible values of α was evaluated by comparing their log likelihoods with the maximized log likelihood. A likelihood ratio based 95% confidence interval for α was calculated as:

$$\{\alpha : 2\left[\ell_p(\alpha^{\mathrm{MLE}}) - \ell_p(\alpha)\right] \leq \chi_1^2(0.95)\}.$$

The profile likelihood and the associated 95% confidence interval for R_{asy} were found similarly.

In isolation, many datasets are relatively uninformative about these parameters. Based on the idea that many populations of the same or similar species share similar life history strategies, Myers et al. (1999, 2001) developed methods that allow parameter estimates from several populations to be combined, providing a probability distribution for the parameter estimates at some higher organizational level such as the species. The resulting probability distributions can be combined with comparatively limited population-specific data to make inferences at the level of the specific population. This approach, known as meta-analysis, allows conclusions to be reached by drawing upon data from many populations. Hierarchical Bayesian methods (Carlin and Louis 1996) are one approach to meta-analysis that has been applied to salmon production (Prevost et al. 2001). Mixed-effects models are an alternate approach that have the advantage that a joint prior distribution for the fixed effects and variance components doesn’t have to be specified. Estimates are obtained using maximum likelihood, and are identical to empirical Bayes estimates in that the priors are obtained from the data (often referred to as MLE priors). As such, these priors can then be used as priors for Bayesian analyses of population dynamics for stocks where little data exists about the stock under investigation (Myers et al. 2002).

As will be seen in the results, the strongest evidence for density dependence occurs between age-0 and age-1, with little evidence for overcompensation, although parameter estimates are not completely satisfactory. Therefore, Beverton-Holt models were also fit to the age-0-to-age-1 data using a mixed-effects model to evaluate whether more plausible parameter estimates could be obtained. The models were fit using the approximate maximum likelihood algorithm of Lindstrom and Bates (1990), using the S-Plus nonlinear mixed-effects library of Pinheiro and Bates (1999).

To simplify notation we use F (for "fry") to denote age-0 and P (for "parr") to denote age-1 juvenile salmon. We have data for several salmon populations and for each population i, we have n_i observations of the form $(P_{ij}, F_{ij}),\ j = 1....n_i$. These observations are modelled as:

$$P_{ij} = \frac{\alpha_i F_{ij}}{1 + \alpha_i F_{ij} / \mathrm{R}_{asy\,i}} e^{\varepsilon_{ij}}$$

where $\alpha_i > 0$, $\mathrm{R}_{asyi} > 0$ and $\varepsilon_{ij} \sim N(0, \sigma_i^2)$. As specified, error variance differs among populations, and errors are multiplicative. Taking the natural logarithms of both sides yields:

$$\log P_{ij} = \log(\alpha_i) + \log(F_{ij}) - \log(1 + \frac{\alpha_i F_{ij}}{\mathrm{R}_{asy\,i}}) + \varepsilon_{ij}\,.$$

We define $\log \widetilde{\alpha}_i = a + b_i$ and $\log \widetilde{\mathrm{R}}_{0i} = c + d_i$:

$$\log P_{ij} = a + b_i + \log(F_{ij}) - \log(1 + \left(\frac{\exp(a + b_i) F_{ij}}{\exp(c + d_i)} \right) + \varepsilon_{ij}\,.$$

One approach to fitting this model is to treat a, b_i, c and d_i as fixed effects, which is the equivalent to fitting to each data series individually as was done earlier in this section when evaluating the timing of density dependence. The alternative method, used here, takes advantage of similarities among populations. We assumed that $\log \alpha_i$ and $\log \mathrm{R}_{asyi}$ are normally distributed random variables and fit the model treating a and c as fixed, and b_i and d_i as random effects. Here, a and c are the means of $\log \alpha_i$ and $\log \mathrm{R}_{asyi}$ respectively, and b_i and d_i are the random deviates for each population, such that:

$$\begin{pmatrix} b_i \\ d_i \end{pmatrix} \overset{iid}{\sim} \mathrm{N}\left[\begin{pmatrix} 0 \\ 0 \end{pmatrix}, \begin{pmatrix} \sigma^2{}_b & 0 \\ 0 & \sigma^2{}_d \end{pmatrix} \right].$$

Note that when estimated using this model, α_i and $\mathrm{R}_{\mathrm{asy}i}$ are the median survival rates and carrying capacities for an age class within each population.

Results

In total, models were fit to 24 data series, including six for egg-to-age-0 transitions and nine series each for the transitions from age-0-to-age-1 and from age-1-to-age-2 (Figure 1). No single, unequivocal pattern emerges. For example, in the case of the NW Miramichi River, the slope at the origin of the Beverton-Holt model fit to the eggs-to-age-1 data is infinite, suggesting no relationship between egg deposition and the abundance of age-0 fry. In contrast, the fit to the Tobique River egg-to-age-0 data suggests a strong relationship between these variables.

Similarly, no pattern is immediately evident for the timing of density dependence among populations. Of the six egg-to-age-0 transitions, in three cases the addition of a second parameter for density dependence resulted in a statistically better fit (Table 2). Of the nine age-0-to-age-1 transitions, in six cases the addition of second parameter provided a statistically better fit, as was also the case in three of the nine age-1-to-age-2 comparisons. Of the nine populations, density dependence was not detected in two populations, was detected in only one transition in two populations, was detected in two transitions in four populations and was detected in all three transitions in the remaining population. The two populations in which density dependence was not detected (Stewiacke River and Margaree River) had two of the highest observed age-1 densities, although the absence of density dependence suggests that carrying capacity may not have been reached for these populations. Overall, the results indicate that the timing of density dependence within fresh water is highly variable among populations.

Despite the lack of a consistent pattern in the timing of density dependence, the data are more informative about the nature of density dependence. As evidenced by the statistical comparison between the Ricker and Beverton-Holt models (Table 3), overcompensation does not appear to be characteristic of these Atlantic salmon populations. Statistically, the Ricker model did not provide a significantly better fit in any of the 25 comparisons (likelihood ratio test; 95% C.L.), whereas the fit of the Beverton-Holt model was significantly better in four of the individual comparisons. When comparisons were made over all populations combined, the Beverton-Holt model provided a statistically better fit (likelihood ratio test; 95% C.L.) for all three transitions investigated here.

One difference in the application of SR models used here from more traditional applications with marine populations (using spawning biomass and number of recruits) is that the α parameter has a direct interpretation as the maximum annual survival rate (at low population sizes in the absence of density dependence) between the stages. As such, the true value of α cannot exceed one, although in five of the Beverton-Holt age-0-to-age-1 models, and in two of the age-1-to-age-2 models, the estimate of α did exceed one (Table 2). Estimation of α requires extrapolation to the origin which can be problematic particularly when most data is at higher densities. Here, we use the mixed-effects model fit to the age-0-to-age-1 data to evaluate whether the same data series are simply uninformative about α, or whether the estimates are high for some other reason. If the data are uninformative, the estimates obtained from the mixed-effects model would be expected to be better (i.e. more plausible) than those from the models fit individually due to the influence of the data from other populations. If the estimates do not change, they would be assumed to be high for some other reason (e.g. sampling bias).

Figure 2 is a meta-analytic summary of the individual fits for each population and the mixed model results for salmon at the species level. The raindrop plots (Barrowman 2000) for each population (light grey shaded region) show the profile likelihood for each parameter, the width of which can be used to gauge the relative plausibility of different values. When estimated for each population individually, the maximum age-0-to-age-1 survival was well determined for only two of the eight populations, the Restigouche and the SW Miramichi rivers. These populations also have the lowest estimates. Bounds for the confidence intervals could be determined for all populations, although one population, the Tobique River, has a lower bound greater than one. With all populations combined, the mixed model random effects distribution for $\log \alpha$ has a mean of -0.09 and a standard deviation of 0.476. Forty-two percent of the mass of this probability distribution is within a range that is not biologically plausible ($\alpha > 1$).

Overall, the individual datasets contained more information about the habitat carrying capacity than the maximum age-0-to-age-1 survival, although carrying capacities varied widely between populations (Figure 2). Two exceptions were the Margaree and Stewiacke populations, for which the profile likelihoods were ramped. With all populations combined, the mixed model random effects distribution for $\log R_{asy}$ has a mean of 3.21 and a standard deviation of 0.97. The median habitat carrying capacity for age-1 salmon populations is therefore 24.8 parr/100m^2, with a 95% confidence interval of 3.8 to 165.9 parr/100m^2.

Estimates of α obtained from the mixed model are similar to those from the individual fits (Figure 3, Table 4). In contrast, the estimates of R_{asy} for three populations (Margaree, Stewiacke and SW Miramichi) dropped substantially when estimated using the mixed-effects model, consistent with the wide confidence intervals obtained for R_{asy} for these populations (Figure 2).

A comparison of the age-0-to-age-1 data series is shown in Figure 4. Data are plotted on the same scale, and models from both the individual and mixed model fits are shown. Differences in the fits are subtle, although since data are plotted on the same scale, some differences are evident. First, the two populations with the greatest change in the carrying capacity estimate appear to have different issues: the Margaree populations have no data near the origin as well as little contrast in the data, whereas the Stewiacke data is all near the origin but perhaps contain too little contrast to estimate carrying capacity. The rivers with the best estimates for both parameters (Restigouche and NW and SW Miramichi) are those with the greatest range of observed densities. Finally, with the exception of the Stewiacke River, the carrying capacity for age-1 parr of the Gulf of St. Lawrence rivers (Restigouche, Margaree and NW and SW Miramichi) appears higher than that of Bay of Fundy and Nova Scotia Atlantic coast rivers.

Density Dependence in the Marine Environment

Density dependence in Atlantic salmon populations in the marine environment is relatively unstudied. Here, we apply the approach used above to the smolt-to-adult returns data from 15 populations (Table 5) to determine whether density dependence can be detected in the marine environment. Three models (density independent, Beverton-

Holt and Ricker) were fit to data for the smolt-to-1SW return data and smolt-to-2SW return data individually. As such, the statistical comparisons do not distinguish between survival rates and age-at-maturity when testing for density dependence.

Results

Of the 15 smolt-to-1SW comparisons (Figure 5), density dependence was potentially detected in three populations: Campbellton, NE Trepassey, and St. Jean. However, both the Campbellton and NE Trepassey models produced biologically impossible parameter estimates (Table 6) from the Beverton-Holt model, although not so from the Ricker. For the other 12 populations, the density-dependent models produced infinite estimates of the carrying capacity in five cases, such that the fits were virtually identical to the density-independent models, a result that strengthens conclusions about the lack of density dependence in the marine environment for these populations.

Density dependence was not detected in any of the nine smolt-to-2SW returns (Figure 6). Similar to the 1SW results, five of the nine density-dependent models produced infinite estimates of the carrying capacity (Table 7), again strong evidence against density dependence in the marine environment.

Discussion

Several interesting factors have surfaced during this analysis of the electrofishing data. First, the timing of population regulation appears variable among salmon populations, but appears most frequently between age-0 and age-1. Additionally, in some populations, it appears to take place gradually over a couple of years, and in others it appears to be relatively rapid, occurring in only one age class. This result is in contrast with the position of Milner et al. (2003), in a recent review of population regulation in salmon and trout populations, that “density-dependent mortality only operates for comparatively short periods of the life cycle, during critical stages, when regulation of population size was achieved by competition for limited resources”. This position is partially based on the work of Elliot (1989, 1993a) for a brown trout population, in which density dependence took place during a critical period approximately 30 to 70 days from when fry dispersed from the spawning gravels. Additional evidence was derived from studies that have shown density dependence occurring throughout the first summer followed by density-independent survival through to the smolt stage (Gee et al. 1978, Egglishaw and Shackley 1977, Gardiner and Shackley 1991), again in contrast to the results presented herein. Armstrong et al. (2003) documents the differences in freshwater habitat requirements of salmon of different ages. If habitat requirements differ with age or size, and the availability of these habitats varies among rivers, then the timing of density dependence would also be expected to vary from population to population, consistent with our results. The results presented here are also consistent with the findings of Myers and Cadigan (1993) for Atlantic cod where density dependence can occur for 3 or more years.

The Ricker model has often been used for modeling Atlantic salmon population dynamics (e.g. Chaput et al. 1992b), but often without systematic approaches to model selection being applied prior to its use. This may have contributed to the conflicting results reported for various populations within the last 30 years. For example, Gee et al. (1978) working on an English river, proposed dome shaped curves for these populations. Buck and Hay (1984), working on a Scottish stream, did not find evidence that the number of migrating smolt decreased as egg depositions increased despite working at depositions well above Elson's norm. Jonsson et al. (1998) proposed an asymptotic model for salmon in a river in Norway. Chaput et al. (1992) fit both a Beverton-Holt model and Ricker model to data from the Margaree River and concluded that the Ricker model provided the better fit to the data. Kennedy and Crozier (1993) decided that a dome shaped curve provided a better fit to egg-to-smolt data on River Bush, but it was not clear what, if any, selection criteria were applied. Here, when taken on the whole, the Beverton-Holt model provided a better fit to the data for all age classes, and we did not find a single case where the Ricker model provided a statistically significant better fit than did the Beverton-Holt. As an alternative approach, Michielsens and McAllister (2004) advocated a Bayesian hierarchical approach to modeling salmon population dynamics which included both the Beverton-Holt and Ricker models. Posterior probability densities were developed from data for nine populations with the Beverton-Holt model providing a slightly better fit to the data.

As pointed out by Myers et al. (2001) and Gibson and Myers (2003), when the data for several populations are simply standardized and plotted on the same scale (Figure 4), patterns become evident and populations that are different are easily identified. One of the most interesting factors to come out of this analysis is the relative difference in the carrying capacity for age-1 parr between the Gulf of St. Lawrence populations and those on the Atlantic coast and Bay of Fundy. The one exception to this pattern is the high carrying capacity of the Stewiacke River population, one of the endangered inner Bay of Fundy salmon populations. The reason for these differences is unclear. Habitat in many Atlantic coast rivers has been impacted by acid rain, although pH problems are not known to be an issue for Bay of Fundy rivers.

Chadwick (1987) concluded that freshwater survival is comparatively less variable than marine survival, a result similar to that of Peterman (1981) for coho salmon, and of Jonsson et al. (1999) for a salmon population in Norway. Based on the individual analysis using the Beverton-Holt model herein, σ averaged 0.490 (std. dev. = 0.204) for egg-to-age-0 survival, 0.373 (std. dev. = 0.107) for the age-0-to-age-1 transition, and 0.488 (std. dev. = 0.206) for the age-1-to-age-2 transition. Based on the density-independent model σ for the smolt-to-1SW, returns averaged 0.383 (std. dev. = 0.148) and for the smolt-to-age-2, returns averaged 0.483 (std. dev. = 0.139). These results suggest that the variability in survival in freshwater is not less than that in the marine environment. Although density dependence in fresh water appears to be a more important regulatory mechanism than in the marine environment, variability in marine survival may be more important for determining the annual spawning run size given that density dependence in freshwater has the potential to buffer variability in survival in this environment.

The estimated carrying capacity for age-1 parr varied among populations by a factor of about 16 times. Grant and Kramer (1990) developed a relationship between body size and territory size for salmonids that they used to examine the hypothesis that territory size limits the maximum population density of salmonids. They concluded that body length explained 87% of the variation in territory size in juvenile salmonids despite variation in species. Here, factors other than body size (e.g. the environment, or number of territories with a habitat) must come into play to explain the differences in carrying capacity found among rivers.

The possibility that some of the among-population differences result from different sampling schemes for various rivers cannot be precluded. In some instances (e.g. the Stewiacke River) sites are selected using a stratified random method designed to sample habitat (using stream gradient as the habitat measure) in proportion to its availability. In others, sites are selected based on accessibility, and some habitat types are often not sampled due to confounding variables such as depth. However, for the most part, sites are sampled more or less consistently from one year to the next, so biases are likely to be consistent from year to year. If so, the resulting parameter estimates would be biased, however, the conclusions drawn about the timing and nature of density dependence should not be affected by these potential differences.

Meta-analysis has been touted as a way of improving biological and management parameters for fisheries (Myers and Mertz 1998b, Myers et al. 1999, Gibson and Myers 2004). However, our meta-analysis did not provide a plausible random effects distribution for the maximum survival rate from age-0-to-age-1. Given that α describes a stage-specific survival rate, rather than population growth rate (as in conventional SR models), the survival parameter has to range from zero to one in order to be biologically realistic. This allows the plausibility of the random effects distribution for survival to be evaluated. Such biological bounds are not known when estimating populations' maximum reproductive rates, and if similar estimation issues exist in those analyses, it is possible that they are only partially alleviated when these kinds of methods are applied. Note however, the model is only one part of the meta-analysis; the data standardizations allow for between population comparisons, and an examination of the effects of the priors (other data) on the resulting estimates for the individual populations.

When estimated using the mixed-effects model, habitat carrying capacity among rivers still varied by a factor of about 16. This variability is slightly less than that determined for alewife of about 25 times (Gibson and Myers 2003) and for Atlantic cod of more than 20 times (Myers et al. 2001), but is slightly higher than that determined for coho salmon of about 10 times (Barrowman et al. 2003). However, as discussed above, we cannot discount the possibility that some of this variability may result from differences in sampling design between some populations.

As outlined by Jonsson and Jonsson (2004), most salmon fisheries theory assumes that the mortality of salmon in the ocean is density-independent, a rationale based on the idea that the population density is far below the assumed carrying capacity for salmon in that habitat. However, other density-dependent effects are possible, such as density-dependent predation on migrating smolt in estuaries or adults prior to upstream migration for spawning. Beverton's (1995) concentration hypothesis states that the potential for density dependence should be greatest when organisms are most concentrated, which is potentially during migration near the mouth of the river for salmon in the marine environment.

Given the data used in the analysis, the tests for density dependence in the marine environment would not distinguish between density-dependent survival and density-dependent age-at-maturity. Friedland and Hass (1996) examined the circuli spacing of 1SW and 2SW salmon within a smolt year class, and found that, in general, circuli spacings were wider for 1SW than for 2SW returns of the same smolt cohort. The 1SW fraction was correlated with late summer growth, suggesting that growth during this season is pivotal in determining the proportion of a smolt class that matures as a 1SW. If growth during that time period is density dependent, then age at maturity would also be density dependent.

Given the nature of anadromy, salmon populations are segregated from other salmon populations while in fresh water, but can mix with other populations while in the marine environment. In our analysis, density dependence in the marine environment was modeled using the abundance from the river specific population. This formulation is appropriate for testing whether density dependence is occurring while populations are segregated. Examples of potential density-dependent mechanisms that would be detected using this formulation are aggregative responses of predators during smolt migration or when adults are in the estuaries prior to migrating upstream. While populations are mixed at sea, the potential exists for density dependence to occur as a function of the total number of smolts emigrating from all rivers, although selection of an appropriate spatial scale for the analysis may be problematic. Myers et al. (1997) found the correlation scale for recruitment of pink and sockeye salmon was less than 300 km, suggesting widespread phenomena do not have a dominant influence for these species. Evaluation of this kind of relationship is a topic for future research.

References

Amiro, P.G., A.J.F. Gibson, and K. Drinkwater. 2003. Identification and exploration of some methods for designation of critical habitat for survival and recovery of inner Bay of Fundy Atlantic salmon (*Salmo salar*). DFO Can. Sci. Advis. Sec. Res. Doc. 2003/120.

Armstrong, J.D., and S.W. Griffiths. 1995. Density-dependent refuge use among overwintering Atlantic salmon juveniles. J. Fish. Biol. 45: 1524-1530.

Armstrong, J.D., P.S. Kemp, G.J.A. Kennedy, M. Ladle, and N.J. Milner. 2003. Habitat requirements of Atlantic salmon and brown trout in rivers and streams. Fish. Res. 62: 143-170.

Barnthouse, L.W., R.J. Klauda, D.S. Vaughan, and R.L. Kendall. 1988. Science, law, and the Hudson River power plants: a case study in environmental assessment. Am. Fish. Soc. Mono. 4. American Fisheries Society, Bethesda, Maryland.

Barrowman, N.J. 2000. Nonlinear mixed effects models for meta-analysis. PhD thesis. Department of Mathematics and Statistics, Dalhousie University, Halifax, Nova Scotia.

Barrowman, N.J., and R.A. Myers. 2000. Still more spawner-recruitment curves: the hockey stick and its generalizations. Can. J. Fish. Aquat. Sci. 57: 665-676.

Barrowman, N.J., R.A. Myers, R. Hilborn, D.G. Kehler, and C.A. Field. 2003. The variability among populations of coho salmon in the maximum reproductive rate and depensation. Ecol. Appl. 13: 784-793.

Begon, M.J., L. Harper, and C.R. Townsend. 1990. Ecology: Individuals, Populations and Communities. Blackwell Scientific, Oxford.

Beverton, R.J.H. 1995. Spatial limitation of population size: the concentration hypothesis. Netherlands J. Sea Res. 34: 1-6.

Buck, R.J.G., and D.W. Hay. 1984. The relation between stock size and progeny of Atlantic salmon, *Salmo salar* L., in a small Scottish stream. J. Fish. Biol. 23: 1-11.

Chadwick, E.M.P. 1982. Stock–recruitment relationship for Atlantic salmon (*Salmo salar*) in Newfoundland rivers. Can. J. Fish. Aquat. Sci. 39: 1496–1501.

Chadwick, E.M.P. 1987. Causes of variable recruitment in a small Atlantic salmon stock. Am. Fish. Symp. 1: 390-401.

Chaput, G., and R. Jones. 1992. Stock–recruit relationship for multi-sea-winter salmon from the Margaree River, N.S. CAFSAC Res. Doc. No. 92/124.

Chaput, G., C. Mullins, and E.M.P. Chadwick. 1992. Stock–recruit relationship for Atlantic salmon from Western Arm Brook, Newfoundland. CAFSAC Res. Doc. No. 92/123.

Carlin, B.P., and T.A. Louis. 1996. Bayes and Empirical Bayes Methods for Data Analysis, Volume 69 of Monographs on Statistics and Applied Probability. Chapman and Hall, London.

Clark, C.W. 1976. Mathematical Bioeconomics. Wiley, New York.

Clark, W.G. 1991. Groundfish exploitation rates based on life history parameters. Can. J. Fish. Aquat. Sci. 48: 734-750.

Dempson, J.B., M.F. O'Connell, D.G. Reddin, and N.M. Cochrane. 2006. Stock status summary for Atlantic salmon from Newfoundland and Labrador.). DFO Can. Sci. Advis. Sec. Res. Doc. 2006/028.

DFO. 2003. Atlantic Salmon Maritime Provinces Overview for 2002. DFO Science Stock Status Report 2003/026.

DFO. 2005. Assessment of Newfoundland and Labrador Atlantic Salmon. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2005/052.

Egglishaw, H.J., and P.E. Shackley. 1977. Growth, survival and production of juvenile Atlantic salmon in a Scottish stream, 1966–1975. J. Fish Biol. 24: 647–672.

Elliot, J.M. 1989. Mechanisms responsible for population regulation in young migratory brown trout, *Salmo trutta*. 1. The critical time for survival. J. Animal Ecology 58: 45-58.

Elliott, J.M. 1993a. A 25-year study of production of juvenile sea trout, *Salmo trutta* in an English Lake District stream. Can. Spec. Publ. Fish. Aquat. Sci. 118: 109–122.

Elliott, J.M. 2001. The relative role of density in stock-recruitment relationships of salmonids. Pages 25 – 66 in Prevost and Chaput (eds). Stock, recruitment and reference points, assessment and management of Atlantic salmon. INRA editions, Fishereis and Oceans Canada, Ottawa.

Friedland, K.D., and R.E. Hass. 1996. Marine post-smolt growth and age at maturity of Atlantic salmon. J. Fish Biol. 48: 1-15.

Gardiner, R. and P. Shackley. 1991. Stock and recruitment and inversely density-dependent growth of salmon, *Salmo salar* L., in a Scottish stream. J. Fish Biol. 38: 691-696.

Gee, A.S., N.J. Milner and R.J.H. Hemsworth. 1978. The effect of density on mortality in juvenile Atlantic salmon (*Salmo salar* L.). J. Anim. Ecol. 47: 497-505.

Gibson, R.J. 1993. The Atlantic salmon in fresh water: spawning, rearing and production. Reviews in Fish Biology and Fisheries 3: 39-73.

Gibson, A.J.F., and B. Hubley. 2006. Status of Eastern Canadian Atlantic salmon populations based on trends.). DFO Can. Sci. Advis. Sec. Res. Doc. 2006/026 (in prep.).

Gibson, A.J.F., and R.A. Myers. 2003. Biological Reference Points for Anadromous Alewife (*Alosa pseudoharengus*) Fisheries in Atlantic Canada. Can. Tech. Rep. Fish. Aquat. Sci. No. 2468.

Gibson, A.J.F., and R.A. Myers. 2004. Estimating reference fishing mortality rates from noisy spawner-recruit data. Can. J. Fish. Aquat. Sci. 61: 1771-1783.

Grant, J.W.A., and D.L. Kramer. 1990. Territory size as a predictor of the upper limit to population density of juvenile salmonids in streams. Can. J. Fish. Aquat. Sci. 47: 1724-1737.

Hayes, D.B., C.P. Ferreri, and W.W. Taylor. 1996. Linking fish habitat to their population dynamics. Can. J. Fish. Aquat. Sci. 53 (suppl. 1): 383-390.

Hilborn, R., and C.J. Walters. 1992. Quantitative Fisheries Stock Assessment: choice, dynamics and uncertainty. Chapman and Hall, New York.

Hixon, M.A. 1998. Population dynamics of coral-reef fishes: Controversial concepts and hypotheses. Aust. J. Ecol. 23: 192-201.

Hjort, J. 1914. Fluctuation in the great fisheries of northern Europe reviewed in the light of biological research. Rapports et Proces-Verbaux des Reunions, Conseil International pour l'Exploration de la Mer 20: 1-228.

Hutchings, J.A. 2000. Collapse and recovery of marine fishes. Nature 406: 882-885.

Hutchings, J.A. 2001. Conservation biology of marine fishes: perceptions and caveats regarding assignment of extinction risk. Can. J. Fish. Aquat. Sci. 58: 108-121.

Hutchings, J.A., and M.E.B. Jones. 1998. Life history variation and growth rate thresholds for maturity in Atlantic salmon, *Salmo salar*. Can. J. Fish. Aquat. Sci. 55 (suppl. 1): 22-47.

Jonsson, B., and B. Jonsson. 2004. Factors affecting marine production of Atlantic salmon (*Salmo salar*). Can. J. Fish. Aquat. Sci. 61: 2369-2383.

Jonsson, N., B. Jonsson, and L.P. Hansen. 1998. The relative role of density-dependent and density-independent survival in the life cycle of Atlantic salmon *Salmo salar*. J. Anim. Ecol. 67:751-762.

Kennedy, G.J.A., and W.W. Crozier. 1993. Juvenile Atlantic salmon (*Salmo salar*)—production and prediction. Can. Spec. Publ. Fish. Aquat. Sci. 118: 179-187.

Korman, J., D.R. Marmorek, G.L. Lacroix, P.G. Amiro, J.A. Ritter, W.D. Watt, R.E. Cutting, and D.C.E. Robinson. 1994. Development and evaluation of a biological model to assess regional-scale effects of acidification on Atlantic salmon (*Salmo salar*). Can. J. Fish. Aquat. Sci. 51: 662-680.

Liermann, M., and R. Hilborn. 1997. Depensation in fish stocks: a hierarchic Bayesian meta-analysis. Can. J. Fish. Aquat. Sci. 54: 1976-1985.

Lindstrom, M.J., and D.M. Bates. 1990. Nonlinear mixed effects models for repeated measures data. Biometrics. 46: 673-687.

Michielsens, C.G.J., and M. K. McAllister. 2004. A Bayesian hierarchical analysis of stock–recruit data: quantifying structural and parameter uncertainties. Can. J. Fish. Aquat. Sci. 61: 1032-1047.

Milner, N.J., J.M. Elliott, J.D. Armstrong, R. Gardiner, J.S. Welton, and M. Ladle. 2003. The natural control of salmon and trout populations in streams. Fish. Res. 62: 111-125.

Moussalli, E., and R. Hilborn. 1986. Optimal stock size and harvest rate in multistage life history models. Can. J. Fish. Aquat. Sci. 43: 135-141.

Myers, R.A., J. Bridson, and N.J. Barrowman. 1995a. Summary of worldwide stock and recruitment data. Can. Tech. Rep. Fish. Aquat. Sci. 2024.

Myers, R.A., and N.G. Cadigan. 1993. Density-dependent juvenile mortality in marine demersal fish. Can. J. Fish. Aquat. Sci. 50: 1591-1598.

Myers, R.A., N.J. Barrowman, J.A. Hutchings, and A.A. Rosenberg. 1995b. Population dynamics of exploited fish stocks at low population levels. Science 269: 1106-1108.

Myers, R.A., K.G. Bowen, and N.J. Barrowman. 1999. The maximum reproductive rate of fish at low population sizes. Can. J. Fish. Aquat. Sci. 56: 2404-2419.

Myers, R.A., B.R. MacKenzie, K.G. Bowen, and N.J. Barrowman. 2001. What is the carrying capacity of fish in the ocean? A meta-analysis of population dynamics of North Atlantic cod. Can. J. Fish. Aquat. Sci. 58: 1464-1476.

Myers, R.A., and G. Mertz. 1998a. The limits of exploitation: a precautionary approach. Ecol. Applications 8: S165-S169.

Myers, R.A. and G. Mertz. 1998b. Reducing uncertainty in the biological basis of fisheries management by meta-analysis of data from many populations; a synthesis. Fish. Res. 37: 51-60.

Myers, R.A., B.R. MacKenzie, K.G. Bowen, and N.J. Barrowman. 1997. Spatial scales of interannual recruitment variations of marine, anadromous and freshwater fish. Can. J. Fish. Aquat. Sci. 54: 1400-1407.

Myers, R.A., N.J. Barrowman, R. Hilborn, and D.G. Kehler. 2002. Inferring Bayesian priors with limited direct data: applications to risk analysis. N. Am. J. Fish. Man. 22: 351-364.

Peterman, R.M. 1981. Form of random variation in salmon smolt-to-adult relations and its influence on production estimates. Can. J. Fish. Aquat. Sci. 38: 1113-1119.

Pinheiro, J.C., and D.M. Bates. 1999. Mixed effects methods and classes for S and S-Plus, Version 3. Bell Labs, Lucent Technologies and University of Wisconson, Madison.

Prévost, É., G. Chaput, and E.M.P. Chadwick. 2001. Transport of stock-recruitment reference points for Atlantic salmon. Pages 95 – 135 in Prévost and Chaput (eds). Stock, recruitment and reference points, assessment and management of Atlanics salmon. INRA editions, Fisheries and Oceans Canada, Ottawa.

O'Connell, M.F., J.B. Dempson, and G. Chaput. 2006. Aspects of the life history, biology, and population dynamics of Atlantic Salmon (*Salmo salar* L.) in Eastern Canada.). DFO Can. Sci. Advis. Sec. Res. Doc. 2006/014.

Quinn, T.J., and R.B. Deriso. 1999. Quantitative Fish Dynamics. Oxford University Press, New York.

Rose, K.A., J.H. Cowan Jr., K.O. Winemillar, R.A. Myers, and R. Hilborn. 2001. Compensatory density dependence in fish populations: importance, controversy, understanding and prognosis. Fish and Fisheries 2: 293-327.

Royama, T. 1992. Analytical Population Dynamics. Chapman and Hall, New York.

Solomon, D.J. 1985. Salmon stock and recruitment, and stock enhancement. J. Fish. Biol. 27 (suppl. A): 45-57.

Trzcinski, M.K., A.J.F. Gibson, P.G. Amiro, and R.G. Randall. 2004. Inner Bay of Fundy Atlantic salmon critical habitat case study.). DFO Can. Sci. Advis. Sec. Res. Doc. 2004/114.

Table 1. Data time series used to investigate the timing and nature of density dependence in salmon populations in fresh water.

River	First year	Last year	Number of years with electrofishing data	Number of years with egg depositions	Mean number of electrofishing sites per year
Big Salmon River	1967	2002	20	23	4.65
Stewiacke River	1984	2001	17	0	33.76
Tobique River	1979	2004	22	16	15.68
Restigouche River	1972	2004	33	0	34.57
NW Miramichi River	1971	2004	34	13	18.85
SW Miramichi River	1971	2004	34	13	32.17
Margaree River	1991	2000	10	0	4.90
Nashwaak River	1981	2005	25	11	6.96
St. Mary's River	1985	2004	17	10	15.77

Table 2. Tests for density dependence in freshwater life stages for nine Atlantic salmon populations. Fits of linear models (density independent) are compared with the fits of Beverton-Holt models (density dependent: pure compensation) and Ricker (density dependent: overcompensation) models. Preferred models were selected using likelihood ratio tests at a 95% confidence level and the principle of parsimony: density independence was chosen unless the addition of a second parameter for density dependence provided a statistically better fit.

River	Stock	Recruit	Model	α	*B* or R_{asy}	σ	NLL	Model Choice
Big Salmon River	eggs	age0	Dens. independent	18.008		1.16	80.074	den-dep
	eggs	age0	Beverton-Holt	58.291	19.409	0.723	71.087	B-H
	eggs	age0	Ricker	27.357	0.247	0.862	74.431	
	age0	age1	Dens. independent	0.762		0.538	47.076	den-dep
	age0	age1	Beverton-Holt	1.404	18.643	0.411	42.761	
	age0	age1	Ricker	1.137	0.026	0.429	43.445	
	age1	age2	Dens. independent	0.36		1.014	30.854	den-ind
	age1	age2	Beverton-Holt	15.239	1.674	0.961	30.007	
	age1	age2	Ricker	0.536	0.046	0.993	30.518	
Margaree River	age0	age1	Dens. independent	0.452		0.143	31.137	den-ind
	age0	age1	Beverton-Holt	0.543	313.258	0.138	30.859	
	age0	age1	Ricker	0.548	0.002	0.138	30.832	
	age1	age2	Dens. independent	0.288		0.365	27.1	den-ind
	age1	age2	Beverton-Holt	0.913	19.353	0.313	25.712	
	age1	age2	Ricker	0.586	0.014	0.316	25.811	
NW Miramichi River	eggs	age0	Dens. independent	2.452		0.732	64.58	den-dep
	eggs	age0	Beverton-Holt	infinity	71.864	0.252	51.768	B-H
	eggs	age0	Ricker	6.794	0.027	0.321	54.679	
	age0	age1	Dens. independent	0.575		0.57	119.354	den-dep
	age0	age1	Beverton-Holt	0.965	37.398	0.47	112.952	
	age0	age1	Ricker	0.872	0.011	0.483	113.901	
	age1	age2	Dens. independent	0.31		0.566	73.549	den-dep
	age1	age2	Beverton-Holt	0.809	6.449	0.397	61.813	
	age1	age2	Ricker	0.572	0.037	0.415	63.306	

Table 2 (con't).

River	Stock	Recruit	Model	α	B or R_{asy}	σ	NLL	Model Choice
Nashwaak River	eggs	age0	Dens. independent	4.248		0.573	33.091	den-dep
	eggs	age0	Beverton-Holt	9.244	15.048	0.464	30.773	
	eggs	age0	Ricker	7.346	0.215	0.474	30.991	
	age0	age1	Dens. independent	0.436		0.684	68.608	dep-dep
	age0	age1	Beverton-Holt	1.709	8.381	0.344	52.08	B-H
	age0	age1	Ricker	0.831	0.033	0.411	56.401	
	age1	age2	Dens. independent	0.193		0.642	23.941	den-indep
	age1	age2	Beverton-Holt	0.382	1.81	0.596	22.151	
	age1	age2	Ricker	0.303	0.066	0.605	22.528	
Restigouche River	age0	age1	Dens. independent	0.318		0.425	89.988	den-dep
	age0	age1	Beverton-Holt	0.502	26.677	0.322	81.045	
	age0	age1	Ricker	0.457	0.009	0.331	81.94	
	age1	age2	Dens. independent	0.286		0.423	45.159	den-ind
	age1	age2	Beverton-Holt	0.279	45.069	0.422	45.039	
	age1	age2	Ricker	0.304	0.006	0.422	45.039	
SW Miramichi River	eggs	age0	Dens. independent	1.082		0.471	61.736	den-dep
	eggs	age0	Beverton-Holt	infinity	88.158	0.265	54.857	
	eggs	age0	Ricker	2.714	0.01	0.26	54.59	
	age0	age1	Dens. independent	0.294		0.413	104.239	den-ind
	age0	age1	Beverton-Holt	0.352	71.08	0.387	102.027	
	age0	age1	Ricker	0.374	0.004	0.385	101.92	
	age1	age2	Dens. independent	0.238		0.582	61.614	den-dep
	age1	age2	Beverton-Holt	0.65	4.245	0.354	45.157	
	age1	age2	Ricker	0.442	0.043	0.367	46.396	

Table 2 (con't).

River	Stock	Recruit	Model	α	B or R_{asy}	σ	NLL	Model Choice
St. Mary's River	eggs	age0	Dens. independent	3.93		0.737	29.45	den-ind
	eggs	age0	Beverton-Holt	5.315	22.258	0.697	28.94	
	eggs	age0	Ricker	5.367	0.094	0.706	29.065	
	age0	age1	Dens. independent	0.51		0.613	35.38	den-dep
	age0	age1	Beverton-Holt	1.267	6.817	0.406	29.185	
	age0	age1	Ricker	0.905	0.054	0.41	29.349	
	age1	age2	Dens. independent	0.18		0.429	4.757	den-ind
	age1	age2	Beverton-Holt	0.241	2.593	0.406	3.929	
	age1	age2	Ricker	0.261	0.074	0.402	3.785	
Stewiacke River	age0	age1	Dens. independent	1.065		0.528	43.017	den-ind
	age0	age1	Beverton-Holt	1.027	126.761	0.521	42.84	
	age0	age1	Ricker	1.172	0.007	0.521	42.834	
	age1	age2	Dens. independent	0.297		0.356	21.165	den-ind
	age1	age2	Beverton-Holt	0.344	19.47	0.334	20.186	
	age1	age2	Ricker	0.354	0.013	0.335	20.237	
Tobique River	eggs	age0	Dens. independent	1.992		0.592	41.226	den-ind
	eggs	age0	Beverton-Holt	2.516	36.058	0.537	39.955	
	eggs	age0	Ricker	2.805	0.048	0.533	39.859	
	age0	age1	Dens. independent	0.459		0.658	46.275	den-dep
	age0	age1	Beverton-Holt	2.228	6.317	0.361	35.452	B-H
	age0	age1	Ricker	0.714	0.03	0.484	40.752	
	age1	age2	Dens. independent	0.237		0.738	17.866	den-dep
	age1	age2	Beverton-Holt	infinity	0.885	0.606	14.318	
	age1	age2	Ricker	0.696	0.223	0.608	14.382	

Table 3. Comparison of the negative log-likelihoods obtained by fitting Beverton-Holt and Ricker models to juvenile salmon data for nine populations.

Life Stage Transition	River	Beverton-Holt	Ricker
egg-to-age-0	Big Salmon River	71.087	74.431
	Nashwaak River	30.773	30.991
	NW Miramichi River	51.768	54.679
	St. Mary's River	28.940	29.065
	SW Miramichi River	54.857	54.590
	Tobique River	39.955	39.859
egg-to-age-0 total		**277.381**	**283.615**
age-0-to-age-1	Big Salmon River	42.761	43.445
	Margaree River	30.859	30.832
	Nashwaak River	52.080	56.401
	NW Miramichi River	112.952	113.901
	Restigouche River	81.045	81.940
	St. Mary's River	29.185	29.349
	Stewiacke River	42.840	42.834
	SW Miramichi River	102.027	101.920
	Tobique River	35.452	40.752
age-0-to-age-1 total		**529.201**	**541.374**
age-1-to-age-2	Big Salmon River	30.007	30.518
	Margaree River	25.712	25.811
	Nashwaak River	22.151	22.528
	NW Miramichi River	61.813	63.306
	Restigouche River	45.039	45.039
	St. Mary's River	3.929	3.785
	Stewiacke River	20.186	20.237
	SW Miramichi River	45.157	46.396
	Tobique River	14.318	14.382
age-1-to-age-2 total		**268.312**	**272.002**

Table 4. Comparison of the estimates of the maximum survival rates between age-0 and age-1 (α) and the age-1 habitat carrying capacities (R_{asy}) obtained by fitting Beverton-Holt models to juvenile salmon densities. Models were fit to the data for each river both individually and simultaneously using a mixed-effects model.

	Individual Estimates		Mixed Effects Model Estimates	
River	α	R_{asy}	α	R_{asy}
Big Salmon River	1.40	18.64	1.19	21.87
Margaree River	0.54	313.26	0.84	115.47
NW Miramichi River	0.97	37.40	0.97	36.81
Nashwaak River	1.71	8.38	1.34	9.13
Restigouche River	0.50	26.68	0.53	24.90
SW Miramichi River	0.35	71.08	0.40	51.39
St. Mary's River	1.27	6.82	1.03	7.97
Stewiacke River	1.03	126.76	1.12	56.30
Tobique River	2.22	6.31	1.36	7.46

Table 5. Data time series used to investigate the timing and nature of density dependence in salmon populations in the marine environment.

River	First year	Last Year	Number of data points 1SW	2SW
Narraguagus	1995	2002	5	6
Nashwaak	1998	2003	6	5
LaHave	1996	2004	9	8
NW Miramichi	1999	2003	5	2
SW Miramichi	2001	2003	3	0
a la Barbe	1990	1993	4	4
St Jean	1989	2003	14	13
BecScie	1988	1995	8	8
de la Trinite	1984	2003	19	19
Highlands	1980	2000	10	9
Conne	1987	2004	18	0
Rocky	1990	2004	15	0
NETrepassey	1986	2004	19	0
Campbellton	1993	2004	12	0
WAB	1985	2004	20	0

Table 6. Tests for density dependence in the marine environment (smolt-to-1SW spawners) for fifteen Atlantic salmon populations. Fits of linear models (density-independent) are compared with the fits of Beverton-Holt models (density dependent: pure compensation) and Ricker (density dependent: overcompensation) models. The density-dependent model was selected when the addition of a second parameter for density dependence resulted in a statistically significant better fit (likelihood ratio tests at a 95% confidence level).

River	Model	α	*B* or R_{asy}	σ	NLL	Model Choice
Narraguagus	Dind	0.001		0.268	-18.765	den. ind.
Narraguagus	BH	0.001	inf	0.268	-18.765	
Narraguagus	Rick	0.001	0	0.268	-18.765	
Nashwaak	Dind	0.029		0.477	-1.309	den. ind.
Nashwaak	BH	0.217	0.467	0.371	-2.821	
Nashwaak	Rick	0.055	0.039	0.398	-2.398	
LaHave	Dind	0.024		0.497	-3.721	den. ind.
LaHave	BH	1.16E+10	0.322	0.406	-5.552	
LaHave	Rick	0.090	0.087	0.398	-5.714	
NW Miramichi	Dind	0.043		0.34	13.300	den. ind.
NW Miramichi	BH	0.049	63.563	0.337	13.254	
NW Miramichi	Rick	0.050	0.001	0.338	13.258	
SW Miramichi	Dind	0.054		0.268	10.006	den. ind.
SW Miramichi	BH	0.450	28.852	0.158	8.423	
SW Miramichi	Rick	0.145	0.002	0.142	8.099	
a la Barbe	Dind	0.004		0.375	-16.535	den. ind.
a la Barbe	BH	0.004	inf	0.375	-16.535	
a la Barbe	Rick	0.004	0	0.375	-16.535	
St Jean	Dind	0.004		0.242	-12.835	den. dep.
St Jean	BH	0.008	0.932	0.203	-15.328	
St Jean	Rick	0.007	0.004	0.206	-15.141	
BecScie	Dind	0.014		0.201	-21.199	den. ind.
BecScie	BH	0.013	inf	0.201	-21.199	
BecScie	Rick	0.014	0	0.201	-21.199	
de la Trinite	Dind	0.014		0.696	10.616	den. ind.
de la Trinite	BH	0.011	inf	0.696	10.616	
de la Trinite	Rick	0.014	0	0.696	10.616	
Highlands	Dind	0.015		0.576	-11.638	den. ind.
Highlands	BH	0.086	0.154	0.513	-12.779	
Highlands	Rick	0.037	0.087	0.51	-12.848	

Table 6 (con't.)

River	Model	α	B or Rasy	σ	NLL	Model Choice
Conne	Dind	0.047		0.458	31.573	den. ind.
Conne	BH	0.191	3.942	0.437	30.733	
Conne	Rick	0.112	0.012	0.434	30.608	
Rocky	Dind	0.033		0.172	-23.543	den. ind.
Rocky	BH	0.041	1.477	0.157	-24.957	
Rocky	Rick	0.041	0.022	0.156	-25.015	
NE Trepassey	Dind	0.055		0.350	-41.948	den. dep.
NE Trepassey	BH	1.098	0.08	0.230	-49.906	
NE Trepassey	Rick	0.153	0.692	0.235	-49.509	
Campbellton	Dind	0.067		0.332	15.266	den. dep.
Campbellton	BH	inf	2.6	0.232	10.953	
Campbellton	Rick	0.176	0.024	0.236	11.192	
WAB	Dind	0.059		0.500	8.909	den. ind.
WAB	BH	0.052	inf	0.500	8.909	
WAB	Rick	0.059	0	0.500	8.909	

Table 7. Tests for density dependence in the marine environment (smolt-to-2SW spawners) for nine Atlantic salmon populations. Fits of linear models (density-independent) are compared with the fits of Beverton-Holt models (density dependent: pure compensation) and Ricker (density dependent: overcompensation) models. The density-dependent model was selected when the addition of a second parameter for density dependence resulted in a statistically significant better fit (likelihood ratio tests at a 95% confidence level).

River	Model	α	B or Rasy	σ	NLL	Model Choice
Narraguagus	Dind	0.007		0.585	-19.789	den. ind.
Narraguagus	BH	0.006	inf	0.585	-19.789	
Narraguagus	Rick	0.007	0	0.585	-19.789	
Nashwaak	Dind	0.008		0.506	-6.714	den. ind.
Nashwaak	BH	0.007	inf	0.506	-6.714	
Nashwaak	Rick	0.008	0	0.506	-6.714	
LaHave	Dind	0.005		0.488	-16.446	den. ind.
LaHave	BH	inf	0.063	0.392	-18.190	
LaHave	Rick	0.022	0.104	0.384	-18.370	
NW Miramichi	Dind	0.009		0.303	1.879	den. ind.
NW Miramichi	BH	0.008	inf	0.303	1.879	
NW Miramichi	Rick	0.009	0	0.303	1.879	
a la Barbe	Dind	0.004		0.375	-16.535	den. ind.
a la Barbe	BH	0.004	inf	0.375	-16.535	
a la Barbe	Rick	0.004	0	0.375	-16.535	
St Jean	Dind	0.009		0.392	2.612	den. ind.
St Jean	BH	0.010	4.787	0.388	2.486	
St Jean	Rick	0.011	0.002	0.388	2.480	
BecScie	Dind	0.009		0.531	-17.314	den. ind.
BecScie	BH	inf	0.052	0.479	-18.14	
BecScie	Rick	0.025	0.154	0.488	-17.991	
de la Trinite	Dind	0.010		0.771	5.568	den. ind.
de la Trinite	BH	0.007	inf	0.771	5.568	
de la Trinite	Rick	0.010	0	0.771	5.568	
Highlands	Dind	0.010		0.401	-16.743	den. ind.
Highlands	BH	0.033	0.132	0.336	-18.333	
Highlands	Rick	0.024	0.087	0.321	-18.750	

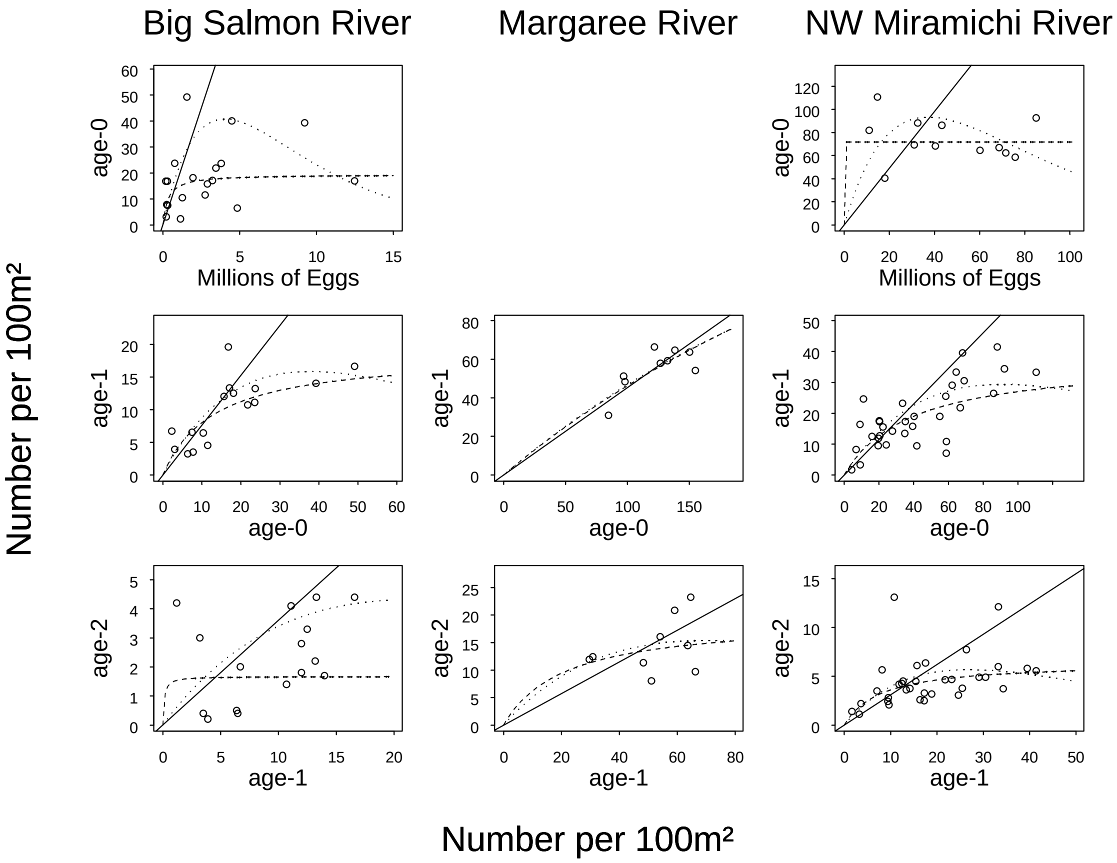


Figure 1. Observed (points) and predicted (lines) densities of Atlantic salmon obtained by fitting three models to the data. The data are the observed abundance or density within a cohort by age. The solid line is a one-parameter model that shows the fit obtained based on the assumption that survival is density independent. The dashed and dotted lines show the fits obtained from two-parameter Beverton-Holt and Ricker models respectively. The former model assumes that survival is purely compensatory, whereas the later model allows for overcompensation. Parameter estimates and statistical comparisons of the fits are provided in Table 2.

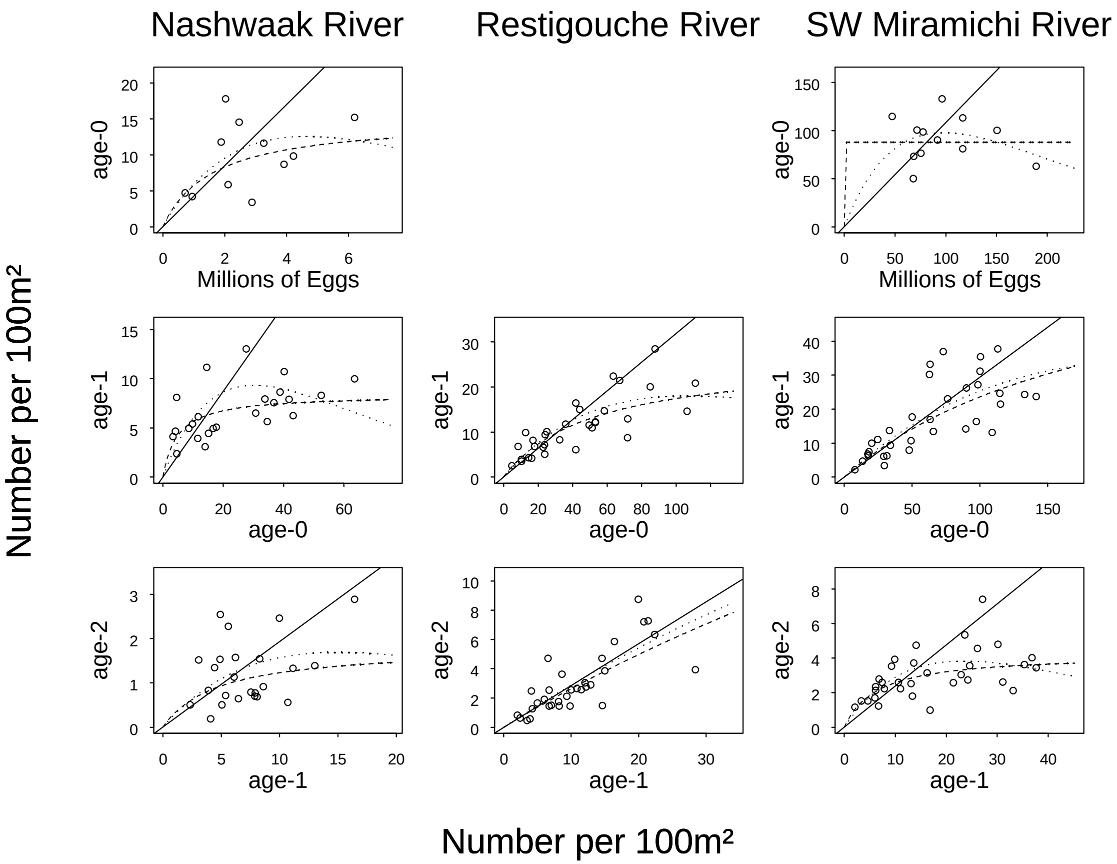
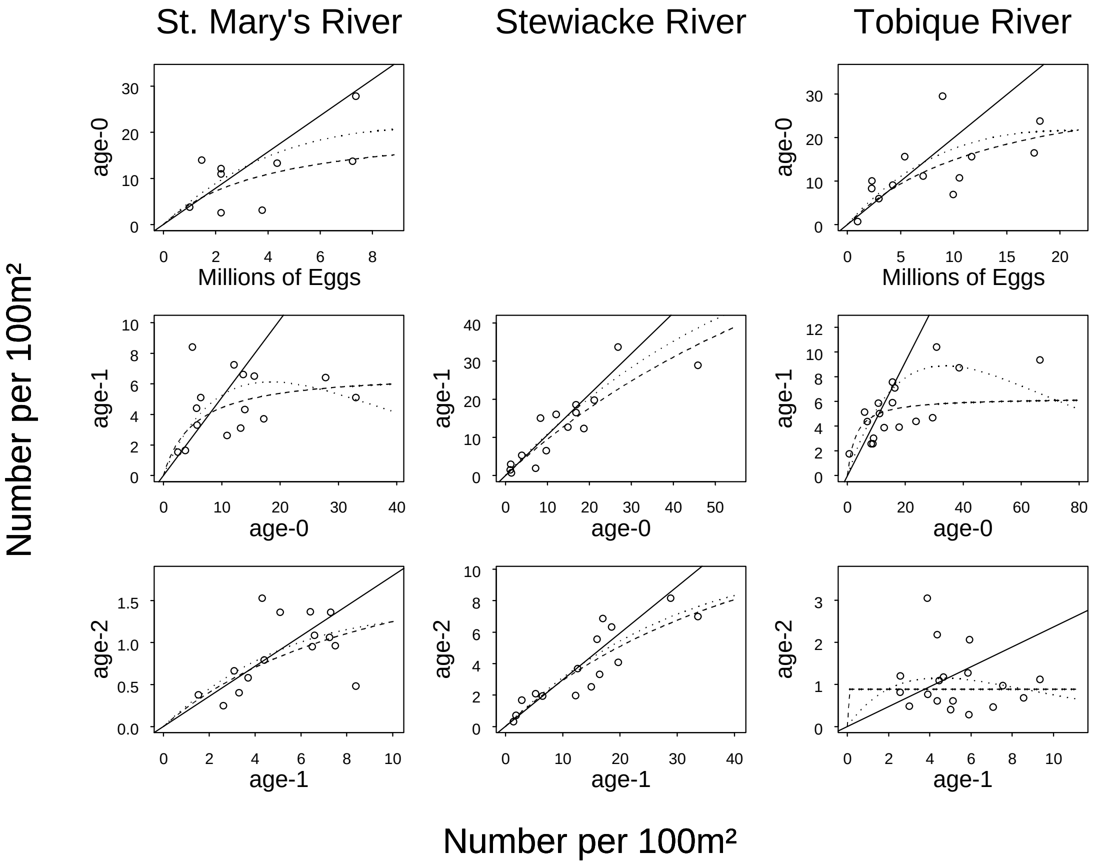


Figure 1 (con't).



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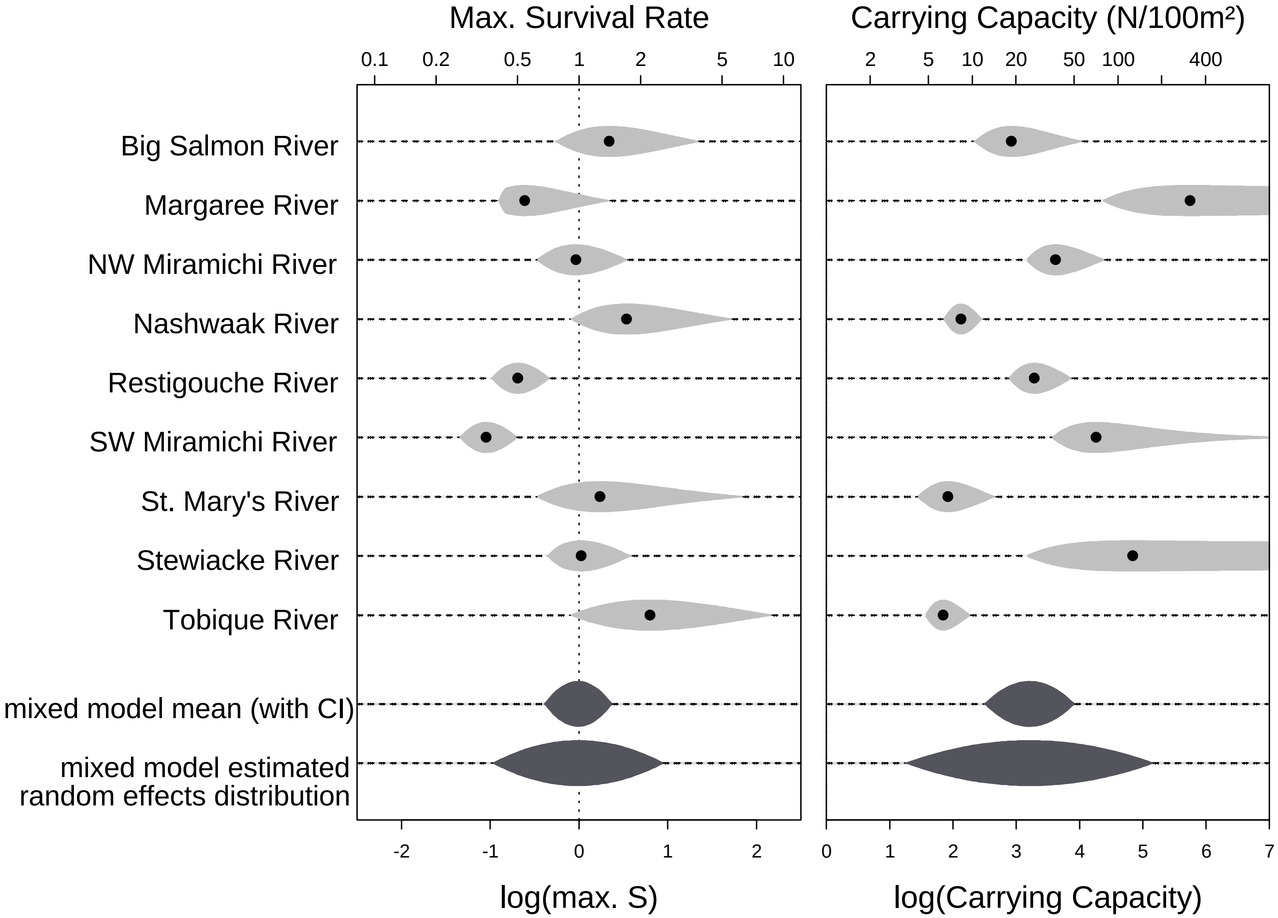

Figure 2. A meta-analytic summary of the maximium age-0-to-age-1 survival rate and the habitat carrying capacity for age-1 parr for nine salmon populations. The light grey shaded regions are individual fits that depict the profile likelihood for each parameter, truncated to show the 95% confidence interval. The height of the profile is used to gauge the relative plausibility of different values (greater height is more plausible). The black dot is the maximum likelihood estimate for each parameter. The dark grey shaded regions show summaries of the mixed model results. The "mixed model mean" represents the estimated mean of the logarithm of each parameter with a 95% confidence interval. The "mixed model estimated random effects distribution" is the normal distribution for the logarithm of each parameter based on its mean and variance estimated with the mixed-effects model.

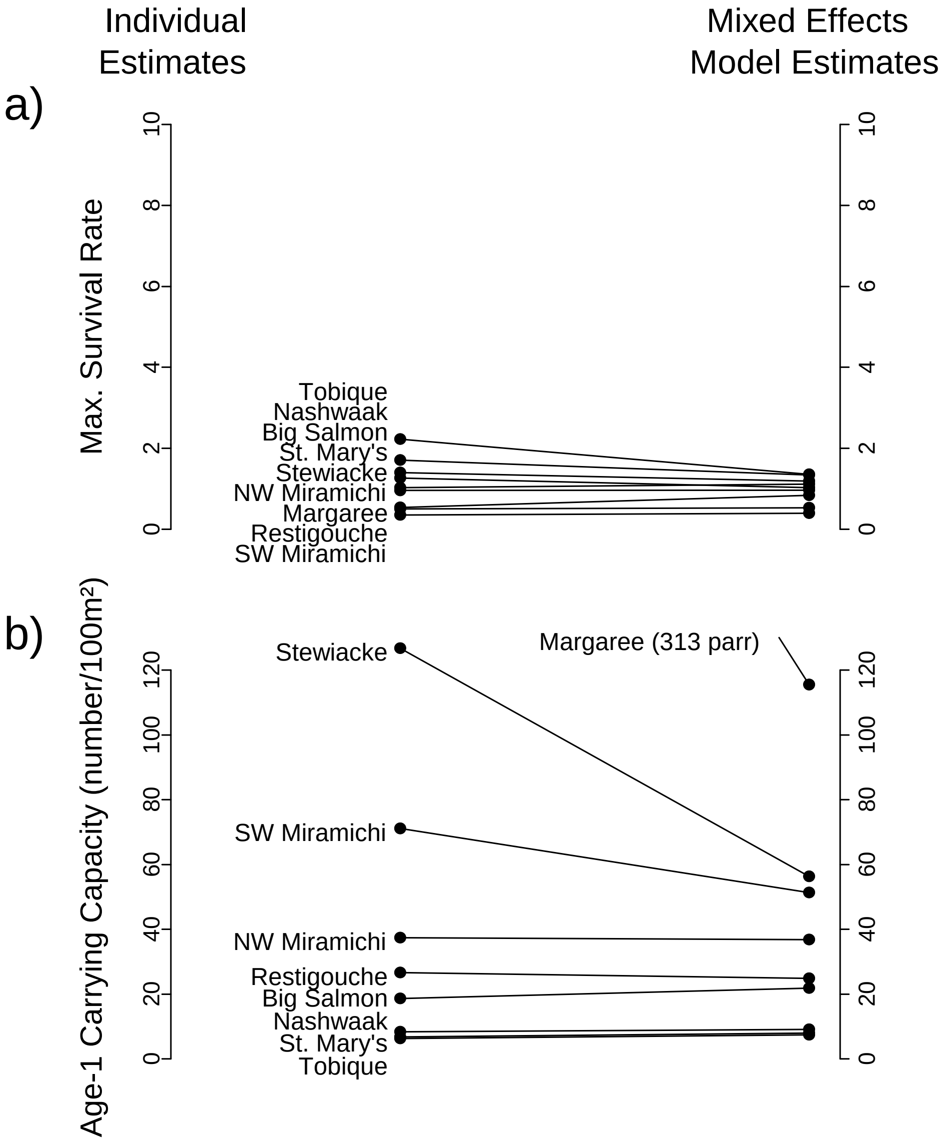

Figure 3. A comparison of the estimates of the maximum age-0-to-age-1 survival and the habitat carrying capacity for age-1 parr obtained from individual regressions on each salmon population and the empirical Bayes estimates obtained from the mixed model.

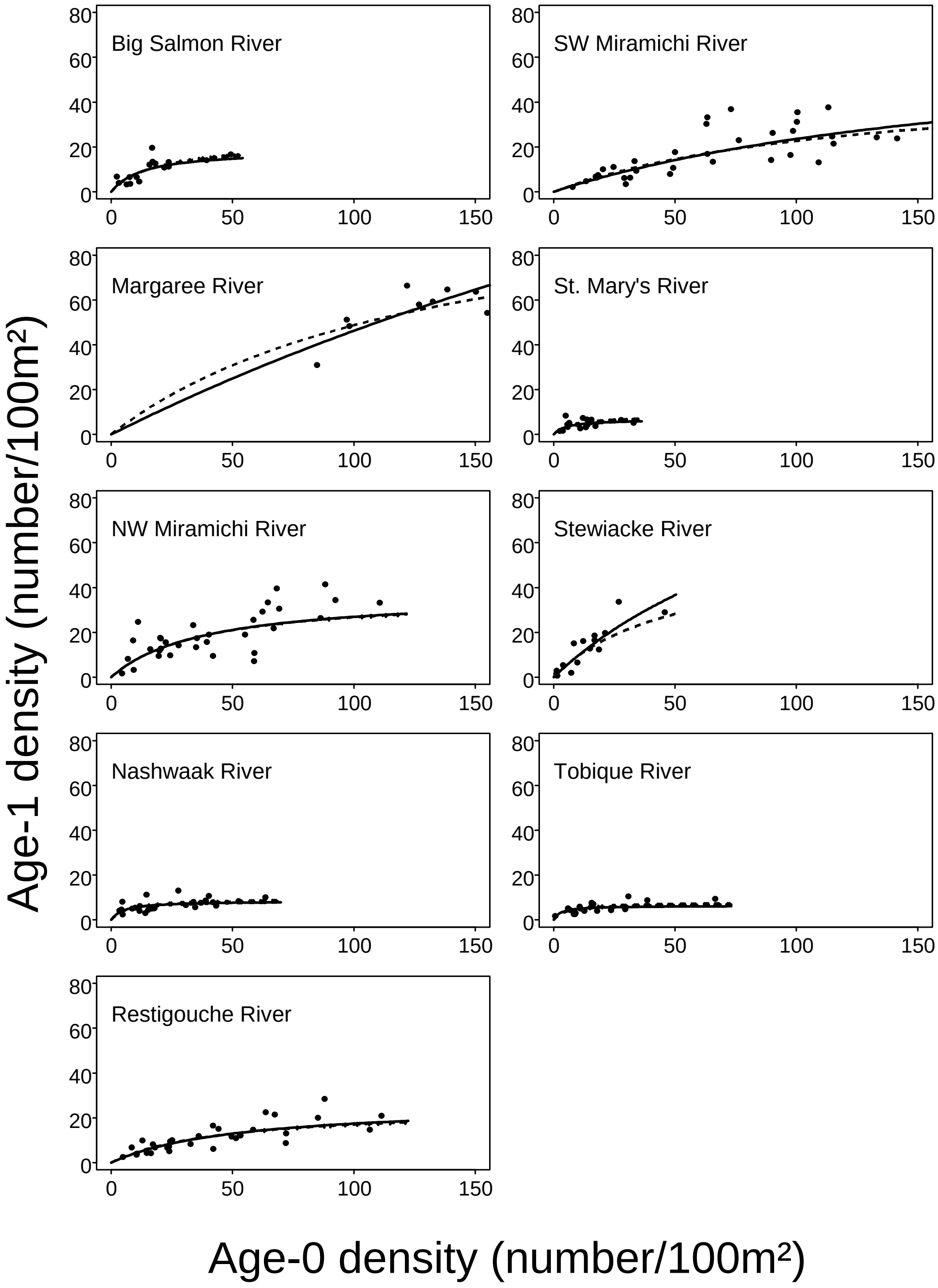

Figure 4. Beverton-Holt models fit to age-0 and age-1 densities for nine salmon populations. The solid line is the spawner-recruit relationship obtained for each population individually and the dashed line is the spawner-recruit relationship for each stock from the mixed-effects model.

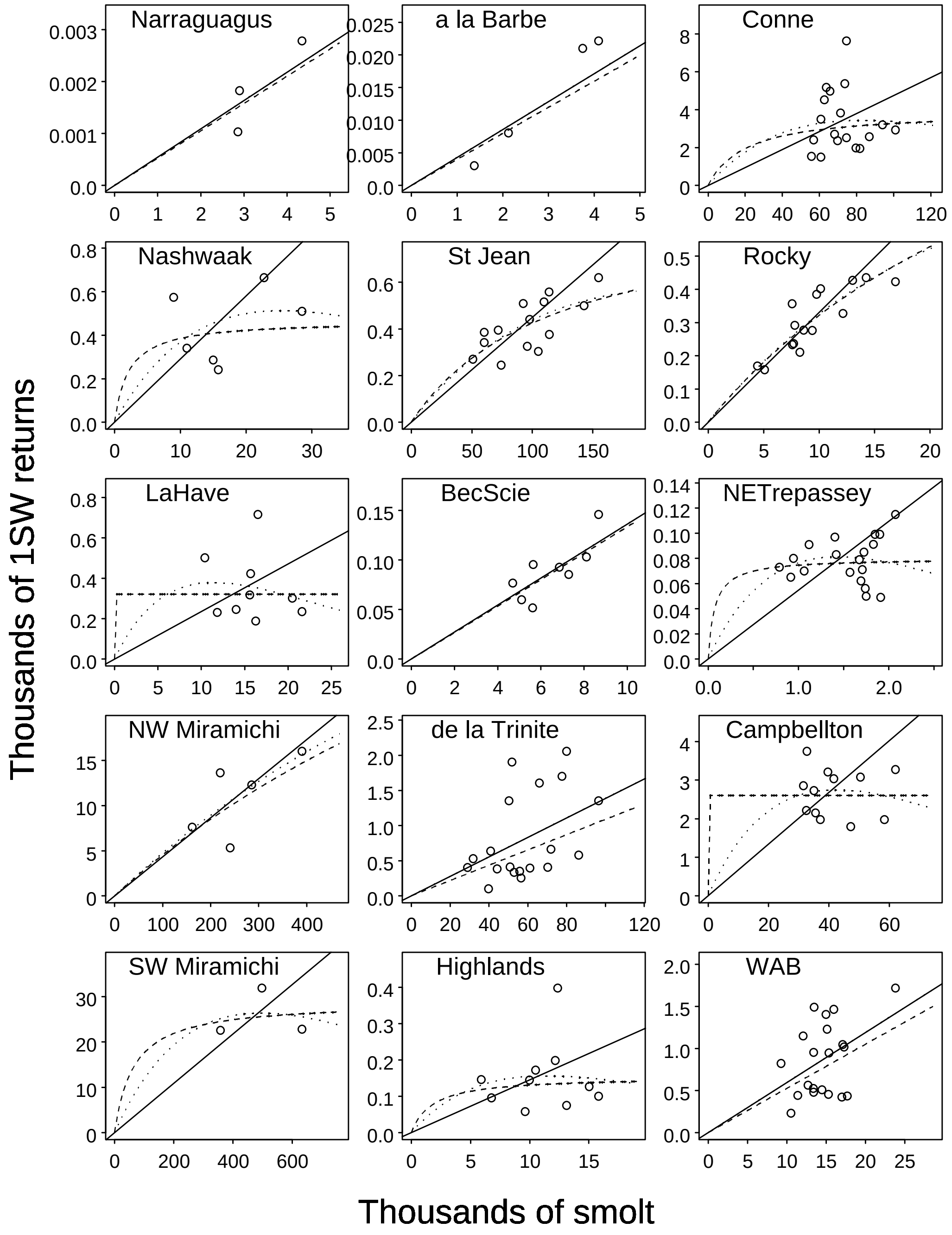

Figure 5. Observed (points) and predicted (lines) densities of Atlantic salmon obtained by fitting three models to the smolt-to-1SW spawner data. The data are the observed abundance or density within a cohort by age. The solid line is a one-parameter model that shows the fit obtained based on the assumption that survival is density independent. The dashed and dotted lines show the fits obtained from two-parameter Beverton-Holt and Ricker models respectively. Parameter estimates and statistical comparisons of the fits are provided in Table 6.

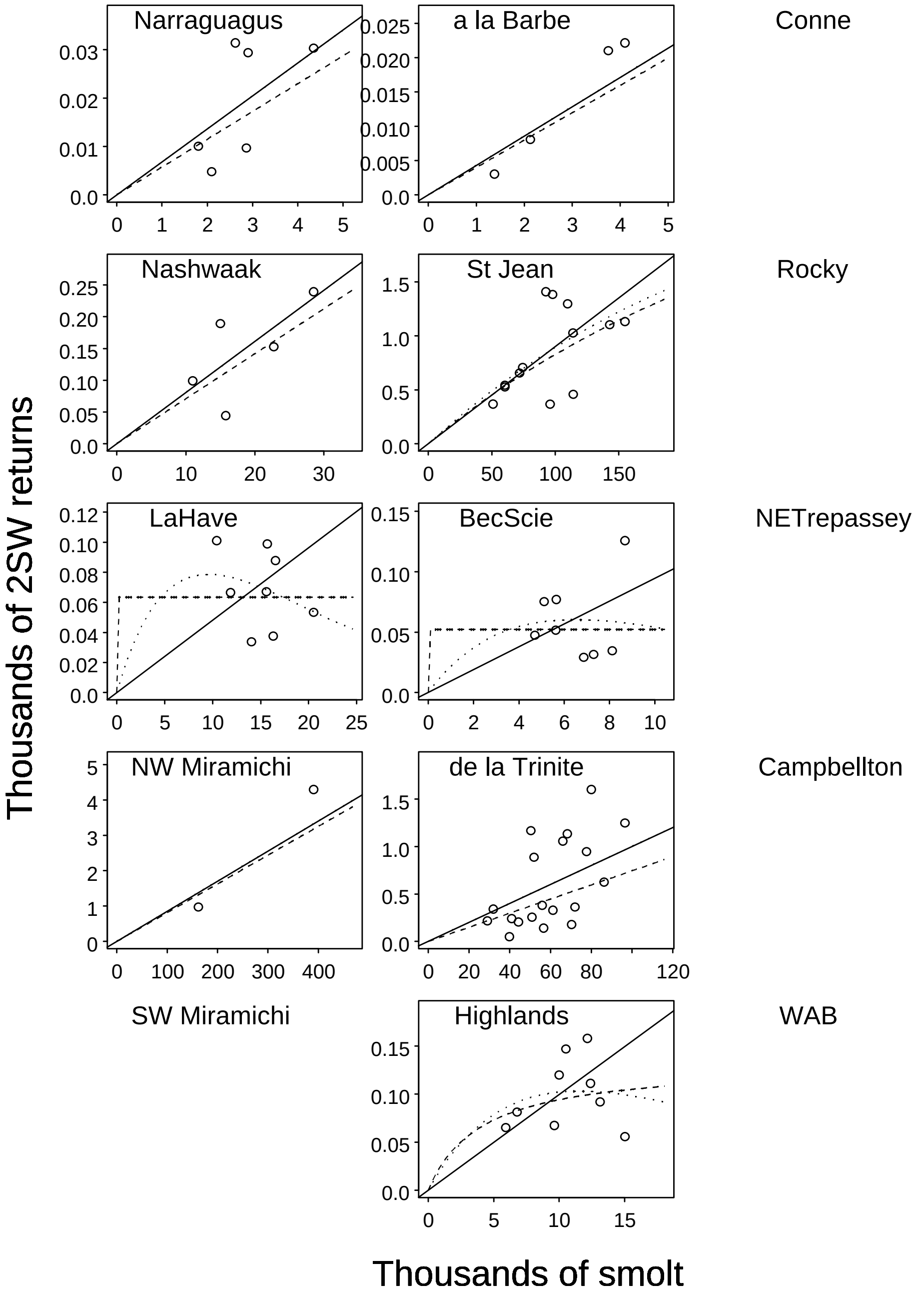

Figure 6. Observed (points) and predicted (lines) densities of Atlantic salmon obtained by fitting three models to the smolt-to-2SW spawner data. The data are the observed abundance or density within a cohort by age. The solid line is a one-parameter model that shows the fit obtained based on the assumption that survival is density independent. The dashed and dotted lines show the fits obtained from two-parameter Beverton-Holt and Ricker models respectively. Parameter estimates and statistical comparisons of the fits are provided in Table 7.

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Guidelines for writing rebuilding plans per the Fish Stocks Provisions and A Fishery Decision-making Framework Incorporating the Precautionary Approach

Table of Contents:

1.0 Introduction

This document describes (1) what a rebuilding plan for a prescribed major fish[1] stock must contain to meet the requirements of the Fish Stocks provisions s. 6.2 in the amended *Fisheries Act* (2019) and in the *Fishery (General) Regulations*, and (2) what a rebuilding plan for fish stock subject to the 2009 *Fisheries Decision-Making Framework Incorporating the Precautionary Approach* (PA Policy), must contain to align with the policy intent.

The 2019 amendments to the *Fisheries Act* include the Fish Stocks provisions (FSP), which introduced new legally-binding obligations on Fisheries and Oceans Canada (DFO) to:

- manage major fish stocks at or above levels necessary to promote sustainability (s. 6.1);
- develop and implement rebuilding plans for major fish stocks that have declined to or below their limit reference point to grow the stock above that point (s. 6.2); and

- prescribe in regulation the major fish stocks to which the provisions will apply (s. 6.3).

The *Fishery (General) Regulations* (FGR) list those stocks subject to the FSP, which are referred to in these guidelines as “prescribed major fish stocks.” In addition, the regulations also define the required contents of rebuilding plans for prescribed major fish stocks and the timelines for developing rebuilding plans for those stocks.

The 2009 PA Policy outlines DFO’s policy to apply the precautionary approach to make decisions respecting harvest levels in fisheries on stocks[2] subject to the policy but not the FSP. The policy states that “when a stock has reached the Critical Zone, a rebuilding plan must be in place with the aim of having a high probability of the stock growing out of the Critical Zone within a reasonable timeframe.” Under the 2009 PA Policy, a stock is considered to be in the Critical Zone when it is at or below its Limit Reference Point (LRP).

This document replaces the 2013 *Guidance for the Development of Rebuilding Plans under the Precautionary Approach Framework: Growing Stocks out of the Critical Zone*.

Rebuilding plans are standalone documents specifically intended to address statutory requirements, and the approved version will be made available on DFO’s website.

This document does not fetter the Minister's discretionary powers set out in the *Fisheries Act*, including those related to rebuilding plans.

Throughout this document, “must” is used to indicate when the guidance is mandatory to meet legislative, regulatory or policy requirements. “Should” is used to signal guidance that is strongly recommended.

2.0 The process to develop a rebuilding plan

This section provides guidance on four topics that are important parts of the process to develop a rebuilding plan:

1. Trigger for rebuilding plans;
2. Regulated timelines for a plan's development;
3. Transition out of a rebuilding plan to an Integrated Fisheries Management Plan (IFMP) when the rebuilding target is reached; and
4. Engagement with modern treaty partners, Indigenous groups, and stakeholders.

2.1 Trigger for rebuilding plans

For prescribed major fish stocks subject to the Fish Stocks provisions (FSP), the obligation to develop and implement a rebuilding plan is triggered when a stock declines to, or below, its LRP.

For stocks that are not subject to the FSP (i.e., they are not prescribed in the *Fishery (General) Regulations*), DFO's 2009 PA Policy still applies, and under the policy the requirement to develop a rebuilding plan is triggered once the stock declines to, or below, its LRP. However, in keeping with the 2009 PA Policy, if a fish stock is decreasing and approaching the LRP, management measures must encourage stock growth and arrest preventable declines, and the development of a rebuilding plan should be initiated sufficiently in advance to ensure that the plan is ready to be implemented if a stock declines to its LRP.

Determining whether a stock is at or below its LRP

Whether a stock is prescribed or not, it is considered to be at or below its LRP if the terminal year stock status indicator is estimated to be at or below the LRP with a greater than 50% probability or if the projected stock status indicator falls below the LRP with a greater than 50% probability under a zero catch scenario in a 1-year projection, unless an alternative method or probability is defined in stock-specific precautionary approach framework.

For situations where it is not possible to estimate the probability that the current or projected stock status indicator is below the LRP, a means of assigning status relative to limits should be pre-defined. This may include expert judgement or other methods.

2.2 Regulated Timelines for Plan Development

Under the FGR a rebuilding plan for a prescribed major fish stock must be developed within 24 months[3] of the day on which the Minister first has knowledge the stock has declined to or below its LRP (see section 2.1 above). If a stock is at or below its LRP when it is prescribed under the FSP, the 24-month timeline to develop a rebuilding plan for the stock starts the day the stock is prescribed in regulation.

To avoid being unable to meet the regulated timeline of 24 months, work to develop a rebuilding plan may have to start before the start of the 24-month period, if possible, and thus before a prescribed stock has declined below its LRP.

For fish stocks subject to the 2009 PA Policy, rebuilding plans should also be developed within 24 months from the start date of that period as noted above.

Interim management measures while a rebuilding plan is under development

During the development of a rebuilding plan for a prescribed stock that is at or below its LRP, the level of fishing, if any, must be consistent with rebuilding the stock above the LRP. This is a regulatory requirement for prescribed major fish stocks (subs. 70(5) of the FGR). “Level of fishing” refers to the total fishing mortality on the stock, including both directed fishing and bycatch.

This requirement comes into effect as soon as the 24-month timeline (described in the section 2.2 above) begins. There is no grace period until the next management decision on the stock.

To meet this requirement, management measures for the stock must meet the following criteria from the 2009 PA Policy:

- Conservation considerations must prevail;
- Removals from all sources are at the lowest possible level;
- There should be no tolerance for preventable decline, which is interpreted to mean that there is a very low likelihood (<5% probability; see table 1 of this guidance for details) of preventable decline; and
- Management actions must promote stock growth.

Management measures must meet the criteria even if projections are unavailable.

The scientific literature indicates that rapid reduction of fishing pressure for those stocks that need rebuilding generally results in greater rebuilding success (e.g., Murawski, 2010; NRC, 2014; Benson et. al., 2016).

If the Minister allows a level of fishing under 70(5) during the development of a rebuilding plan, the Minister has the discretion to decide on the level of fishing that will meet 70(5). Further, the decision about whether to permit fishing on a fish stock under 70(5) is distinct from any future decision about whether to permit fishing on the stock as part of its rebuilding plan.

Finalizing the plan to end the timeline

By the end of the 24-month period, the plan must be finalized and approved by the Minister. The date the plan was approved signals the end of the timeline and must be documented on the cover page of the rebuilding plan as noted in the template for rebuilding plans. The rebuilding plan should also indicate the date on which the implementation of the plan will start (e.g., for next fishing season).

If possible, the implementation of the rebuilding plan should start immediately following approval of the plan. To allow for necessary changes to licence conditions, variation or prohibition orders, for example, it may be necessary to delay implementation of a rebuilding plan until the start of the next fishing season for each fishery on the stock. If the full implementation of a rebuilding plan is delayed for these reasons, the interim management measures must continue and where possible additional measures from the plan introduced, until the full plan is in effect.

In accordance with subs. 70(7) of the FGR, the approved plan must be published on DFO’s website. The plan should be published within 120 days from the date of the plan’s approval by the Minister.

For fish stocks subject to the 2009 PA Policy, rebuilding plans should also be approved by the Minister and published on DFO’s website.

Extensions of the 24-month timeline to complete the plan

To complete the rebuilding plan, the Minister may extend the timeline up to an additional 12 months, bringing the total time to develop the plan to 36 months. Examples of why the timeline may need to be extended include:

- To collect and/or provide scientific information critical to developing the rebuilding plan;
- To provide additional time to seek feedback on the rebuilding plan from Indigenous peoples; or

- To discuss the management measures for a shared stock with other jurisdictions.

In accordance with subs. 70(4) of the FGR, the reasons the timeline has been extended must be published on DFO’s webpage on the FSP. The reasons should be published within 60 days of the decision.

2.3 Transition out of a rebuilding plan when rebuilding target is reached

“End point” of rebuilding plans

For a prescribed major fish stock subject to the FSP, the legal obligation of s. 6.2 to implement a rebuilding plan to rebuild the stock above its LRP only applies while the stock is at or below its LRP. However, to increase the likelihood that a stock will not decline back to or below its LRP and to be consistent with the 2009 PA Policy’s intent to grow depleted stocks to healthier levels, a stock’s rebuilding plan will remain in effect until the stock reaches its rebuilding target (for more information on the rebuilding target, see section 3.4a below). Thus the “start” and “end” points for a rebuilding plan will be asymmetric.

For a prescribed major fish stock subject to the FSP that is below its rebuilding target but above its LRP, the stock will be subject to either subs. 6.1(1) or subs. 6.1(2), which requires that management measures be implemented to:

a. Maintain the stock at or above levels necessary to promote sustainability (subs. 6.1(1)); or
b. Maintain the stock above the LRP (subs. 6.1(2)).

The rebuilding plan will be used to meet the s. 6.1 obligations (either subs. 6.1(1) or 6.1(2)) until the prescribed major fish stock reaches its rebuilding target. Once the stock reaches its rebuilding target, the rebuilding plan will come to an end and the fisheries on the stock will be subject to an IFMP or other management plan. The rebuilding target must be set at a level above the LRP so that there is a very low to low likelihood of the stock being below its LRP (<5-25% probability; see subsection on *Probability, Likelihood and Risk Tolerance* for more details, and in particular, Table 1). A rebuilding target has been reached when there is at least a 50% probability that the stock is at or above its rebuilding target.

For a fish stock subject to the 2009 PA Policy, continuing the rebuilding plan until the stock has reached its rebuilding target is consistent with the principles outlined in the 2009 PA Policy.

> ℹ The rebuilding target is not intended to be a “target reference point” as described in the 2009 PA Policy or the point at which stock rebuilding efforts cease. Instead, its function is to signal the transition from the dedicated rebuilding plan back to the standard fisheries management process along the stock’s growth trajectory above the upper stock reference point (USR) or to an established target reference point (TRP). In effect, it is a milestone.

Probability, likelihood and risk tolerance

Risk is inherent to fisheries management, including rebuilding, and is the effect of uncertainty on fishery objectives, measured in terms of the consequences of an event and the likelihood of their occurrence. The terms "probability", "likelihood", and "risk tolerance" are used throughout this guidance document (e.g., see paragraphs above on the rebuilding target), and require definition:

- **Probability**: The chance (statistical or relative frequency) that a given event or outcome has or will occur. It is typically used when uncertainty is associated with an outcome and can be quantified.
- **Likelihood**: Although often used to describe the extent to which a proposition (a hypothesis, or a model) explains available information (past events; e.g., the likelihood that habitat degradation or loss has contributed to a stock's decline), for the purposes of this guidance, likelihood is also the chance that a given event will happen, expressed or qualitatively (e.g., high likelihood). Annex 2B of DFO's 2009 PA Policy provides a likelihood scale of qualitative descriptors with their corresponding probability range (also provided as Table 1 below).
- **Risk:** In general, the possibility of something undesirable happening, for example, of harm or loss, or of failing to meet fisheries management objectives. Risk occurs as a result of uncertainty, and is measured in terms of the consequences of an event and the probability of its occurrence.
- **Risk Tolerance:** The tolerable, or acceptable, probability of an undesirable event occurring, such as a breach of a limit, or failure to achieve a target or other management objective.

The choice of risk tolerance when developing rebuilding plans must be guided by Annex 2B of DFO's 2009 PA Policy. For ease of reference, this table is also available below as Table 1. To provide greater clarity throughout this document, the title and column headers of Table 1 have been modified from the PA Policy to align with the definitions provided above.

Table 1: Likelihood scale to define DFO's risk tolerance in the 2009 PA Policy. Likelihood designations correspond to specific ranges of probability.

Probability of outcome	Likelihood designation
Less than 5%	Very low
5% - 25%	Low
25% - 50%	Moderate
~50%	Neutral
50% - 75%	Moderately high
75% - 95%	High
>95%	Very High

Preparing to transition out of rebuilding plan to IFMP or other management plan

Before transitioning a stock from a rebuilding plan to an IFMP or other management plan (e.g., a Conservation Harvesting Plan), the management measures planned for the stock under an IFMP should be evaluated and if necessary adjusted so that they have:

1. A low likelihood of the stock declining to its LRP in the short to medium term[4], taking into account the environmental conditions affecting the stock; and
2. A high likelihood of acceptably meeting the obligations under s. 6.1 of the FSP, including continuing the prescribed major fish stock's growth above the USR or to the TRP.

If appropriate or desired, the management measures used in the rebuilding plan can also be used in the IFMP to encourage further growth of the stock above its USR or to its TRP.

This evaluation should occur before the IFMP's management measures are implemented. If the stock is steadily growing, this evaluation may be undertaken in anticipation of the stock reaching its rebuilding target with the understanding that the rebuilding plan measures will remain in effect until the rebuilding target is reached. If the stock grows rapidly and reaches the rebuilding target unexpectedly, the rebuilding plan should remain in effect until the evaluation of the IFMP measures is completed.

2.4 Aboriginal and treaty rights and engagement with Indigenous peoples

DFO seeks to manage fisheries, including decisions flowing from the application of this guidance, in a manner consistent with the constitutional protection provided to Aboriginal and treaty rights by s. 35 of the *Constitution Act, 1982*. In practice, this may include:

- Engaging in consultations with Indigenous people when there is a legal duty to consult;
- Seeking and including Indigenous input and Indigenous Knowledge when developing rebuilding plans, where applicable; and
- Ensuring priority access for Food, Social, and Ceremonial (FSC) fisheries.

3.0 Elements of a rebuilding plan

This section explains how to fill in the rebuilding plan template (Annex A). Each rebuilding plan must contain sections A to I regardless of whether a stock is prescribed under the FSP. The **bolded** sections are legally required by regulation for major stocks subject to s. 6.2 under the FSP. All plans must meet the requirements of the FSP and the FGR by following the guidelines outlined in this section. However, once those obligations are met the plans may vary in complexity, scope and length.

A. Introduction and context;
B. **Stock status and stock trends;**
C. **Probable causes for the stock's decline;**
D. **Measurable objectives aimed at rebuilding the stock;**
    i. **Rebuilding target and timeline;**
    ii. **Additional measurable objectives and timelines;**
E. **Management measures aimed at achieving the objectives;**
F. Socio-economic analysis;
G. **Method to track progress towards achieving the objectives;**
H. **Periodic review of the rebuilding plan;**
I. References

The following provides guidance on the purpose of each section of the template, the expected content of each section, and in some cases, examples of how to complete the section depending on the amount and type of data. Clarifications of key elements of DFO's 2009 PA Policy are also provided.

> All rebuilding plans must be consistent with the intent of DFO's 2009 PA Policy.

3.1 Introduction and context

The purpose of this section of the rebuilding plan is to provide sufficient context on the stock and the fisheries on the stock to allow the non-expert reader to understand the remainder of the rebuilding plan. To do so, provide a brief overview of the stock and history of the fishery(ies) on the stock. At minimum, this should include:

- Biology of the stock;
- Relevant environmental conditions or ecosystem factors affecting the stock (if known), including any aquatic invasive or non-indigenous species;
- Directed fisheries and those with bycatch of the stock should have their current management measures and key participants described (include details regarding shared jurisdictions if applicable);

This section may also include:

- Socio-economic overview of landings (quantity and value) over the past 10 years and profiles of the affected fish harvesters;
- Cultural impacts of the declines of the stock to date;
- Fisheries management issues;
- Committee on the Status of Wildlife in Canada (COSEWIC) assessment status and the *Species at Risk Act* (SARA) listing decisions if applicable;
- Reference to any Indigenous Knowledge used in the development of the plan; and
- Description of the process and Indigenous groups and stakeholders involved in the development of the plan (alternatively, provide this information in an annex to the rebuilding plan).

If an IFMP for the stock is available, this section can be less than a page and direct readers to the relevant content in the IFMP. The rebuilding plan content should focus primarily on why it is important to promote the rebuilding of the stock, including for reasons such as legal, policy, conservation, and socio-economic. Ensure the IFMP clearly addresses the required content for the stock in question.

3.2 Stock status and stock trends

> ℹ This section provides information required by subs. 70(1)(a) of the FGR for prescribed major fish stocks.

The purpose of this section of the rebuilding plan is to describe the stock's status at the time of drafting the rebuilding plan, and its historical trajectory.

Table 1 of the rebuilding plan template describes the PA reference points for the stock. This section must indicate the date on which DFO determined that the stock was at or below its LRP (i.e., the start date of the 24-month rebuilding plan development time period). In addition, the section includes a summary of the stock's:

- Current status relative to PA reference points (LRP at a minimum, and USR[5], Removal Reference (RR), and TRP, if available) including its PA stock status zone,
- Stock status trends (retrospective for all stocks, even if based on proxies, and projected if available) and if applicable, any additional secondary indicators of importance to the stock assessment (e.g., fishing mortality, age structure, sex ratio, size distributions, etc.).

Include a reference to the most recent applicable science advisory or stock assessment document for the stock. During the periodic reviews of the rebuilding plan, this section should be updated if there are any changes to the stock's status (i.e., PA status zone) or trends as reported in a stock assessment. If applicable, comment on the following topics:

- Current COSEWIC assessment status (including relevant Designatable Unit) and/or SARA listing status (e.g., decision in progress, Special Concern) of the prescribed major fish stock.

- Indigenous knowledge.

3.3 Probable causes for the stock's decline

> ℹ This section provides information required by subs. 70(1)(b) of the FGR for prescribed major fish stocks.

The purpose of this section is to summarize the probable factors that have led to the decline of the stock as well as those that may affect rebuilding. The factors to cover include fishing mortality (from all sources), non-fishing anthropogenic factors, the biology of the stock, natural mortality, predator/prey interactions, environmental impacts (including climate, oceanographic and ecosystem factors), habitat limitations, and international issues. Where possible and relevant, this section may also include information regarding the relative contribution of the probable causes to the stock's decline, or those preventing recovery. In some cases, resolving the relative roles of the various contributing factors may be difficult. If there are identified knowledge gaps, these should also be acknowledged.

If the probable causes of a stock's decline have previously been documented in another scientific document (e.g., an IFMP, a COSEWIC assessment, a Recovery Potential Assessment, etc.), a summary of these factors must still be included in the rebuilding plan. Include a reference to the original document as well.

Habitat loss or degradation for prescribed major fish stocks

For prescribed major fish stocks, this section must describe whether habitat loss or degradation has occurred, and if it has, whether this loss or degradation contributed to the stock's decline. This is the first step to meet subs. 6.2(5) of the FSP, which requires that the rebuilding plan for the stock take into account whether there are measures in place to restore fish habitat, if habitat loss or degradation contributed to the stock’s decline. See section 3.5 for the guidance to meet the remaining requirements of subs. 6.2(5) within the rebuilding plan.

For stocks unlikely to rebuild under prevailing conditions

If the rebuilding prospects for a stock are negligible due to conditions outside DFO’s control such as high natural mortality or environmental conditions that are negatively affecting productivity or recruitment, this section should note these challenges. A stock will be identified as unlikely to rebuild when it is more likely to decline than grow even under conditions of no fishing and, where applicable, when other management measures (e.g., habitat restoration, hatchery enhancement, etc.) are also unlikely to result in stock growth.

3.4 Measurable objectives aimed at rebuilding the stock

> This section (including all subsections) provides information required by subss. 70(1)(c) and (d) of the FGR for prescribed major fish stocks.

3.4a Rebuilding target and timeline

> This section is required by subss. 70(1)(c) and 70(1)(d) of the FGR for prescribed major fish stocks.

The purpose of this section of the rebuilding plan is to describe the rebuilding target, which when reached, signals the transition point to a standard fisheries management process (see section 2.3 for more details).

This consists of defining the rebuilding target, the likelihood of attaining the rebuilding target, and the time frame to achieve the target (referred to as “timeline” in the regulations and through this guidance). Combined, the rebuilding target, timeline, and where possible, desired likelihood, provide a measurable objective against which the effectiveness of potential management measures can then be evaluated for how well they are likely to achieve this objective. Both a rebuilding target and the timeline to reach it are required in a rebuilding plan for a prescribed major fish stock.

The rebuilding target for the stock must, at a minimum, be set at a level above the LRP[6] such that there is a very low to low likelihood of the stock being below the LRP (<5-25% probability; as defined Table 1 of this document and Annex 2B of the 2009 PA Policy).

By setting the rebuilding target in accordance with this condition, the stock should be well-positioned to continue its growth above its USR, or to its TRP using the standard fisheries management process in accordance with the 2009 PA Policy. For some stocks, if the rebuilding target is expressed as a function of the LRP, the absolute value of the rebuilding target (e.g., expressed in tonnes) may be updated if the LRP changes, however the desired certainty of being above the LRP will remain constant. For some stocks, the target may be expressed solely as a deterministic estimate or empirical value (e.g., expressed in tonnes) instead of as a function of the LRP. In this case, the rebuilding target must be set high enough above the LRP that it is unlikely the stock is at or below its LRP when it is at the rebuilding target, given the uncertainty associated with the stock’s status.

The timeline to rebuild a stock to its rebuilding target must be between T_{min} and a maximum of two to three times T_{min}, where T_{min} is the time the stock would take to rebuild to that target in the absence of all fishing (F=0) under prevailing productivity conditions. To support the selection of a timeline while considering tradeoffs between likelihood of rebuilding success and socio-economic, and cultural impacts, measures to rebuild in T_{min}, $2×T_{min}$ and $3×T_{min}$ could be considered, taking into account uncertainties. If it is possible to estimate T_{min}, this maximum $3×T_{min}$ timeline must be used in rebuilding plans developed under subs. 6.2(1) of the FSP.

Where T_{min} cannot be calculated, estimates of generation time should be provided to inform rebuilding timelines. The 2009 PA Policy suggests that a “reasonable timeframe” for a stock to grow above its LRP should be between 1.5 to 2 times the generation time. However, for some stocks a longer time may be needed to reach its rebuilding target, for example due to a stock’s highly depleted state or its current productivity. To distinguish between a rebuilding plan under subs. 6.2(1) versus a rebuilding plan under 6.2(2), a timeline of up to two generations is recommended for a plan under subs. 6.2(1).[7] If generation time is used to set the rebuilding timeline, include the definition and calculation used to estimate generation time. For more information on generation time, see the Glossary.

If the generation time of the stock is unknown, the generation time should be estimated using expert judgement and the best available information on the life history characteristics of the stock, or a similar stock or species. Include a rationale for the estimated generation time.

Setting this timeline may include evaluating tradeoffs between conservation and socio-economic considerations; however, conservation objectives must prevail such that they seek to grow the stock above the LRP in a reasonable time.

Express the rebuilding objective, say $B_{rebuilt}$, and timeline using three components:

- Desired state (the desired management outcome) (e.g., $B > B_{rebuilt}$)

- Desired probability for achieving the desired state (e.g., in percentage)
- Time period (when we want the outcome) (in years)

For prescribed major fish stocks, providing a desired probability of achieving a rebuilding target within a certain timeline is not required by regulation. However, the desired likelihood or probability of achieving the rebuilding target should be stated explicitly, where possible. Doing so would be consistent with the 2009 PA Policy's criterion that a rebuilding plan must be in place with the aim of having a high probability (75–95%) of the stock growing out of the Critical Zone within a reasonable timeframe.

For stocks unlikely to rebuild under prevailing conditions

For a stock that is unlikely to rebuild (see section 3.3 for details), establish a rebuilding target as described above. Identify what the "end point" of the rebuilding plan would be in the event conditions change and rebuilding becomes feasible. Estimate the rebuilding timeline as described above, if possible. If it is not possible to establish a timeline to rebuild to the target, the rebuilding plan must explain the reasons why it is not feasible, in accordance with subs. 70(6) of the FGR. However, the inability to calculate T_{min} should not be used as a reason to exclude a timeline to the rebuilding target under subs. 70(6) of the FGR when generation time is known or can be estimated using expert judgement.

3.4b Additional measurable objectives and timelines

> This section provides information required by subss. 70(1)(c) and 70(1)(d) of the FGR for prescribed major fish stocks.

The purpose of this section of the rebuilding plan is to identify additional rebuilding objectives. Clearly articulated and measurable objectives will guide the selection and implementation of management measures and provide a means to measure progress to rebuild a stock.

These objectives and the management measures in the next section must first meet the requirements of s. 6.2 of the FSP (either subs. 6.2(1) or 6.2(2)), including taking into account stock biology and environmental conditions affecting the stock. Second, the objectives and management measures must be consistent with the following criteria from DFO’s 2009 PA Policy:

- Management actions must promote stock growth and removals from all sources must be kept to the lowest possible level until the stock has cleared this zone. The rebuilding plan must have a high likelihood of the stock growing to the rebuilding target within the plan’s rebuilding timeline.
    - Conservation considerations must prevail and the management measures must have a high likelihood (75-95% probability) of the stock growing to the rebuilding target within the plan’s rebuilding timeline.
- There should be no tolerance for preventable decline.
    - This means that total catch on the stock (both directed and bycatch) must be limited to where there is a very low likelihood of preventable decline (<5% probability) and that will allow the stock to grow under current environmental conditions.

If there is directed fishing or bycatch on the stock while it is below its LRP, the rebuilding plan must explain how the catch level respects these two PA Policy criteria. This analysis will help to demonstrate that DFO is meeting the key goal of s. 6.2 of the FSP, which is to implement a plan to rebuild the stock above its LRP.

Additional objectives must be established for each of the following categories, where applicable and feasible. If they are not applicable or feasible, provide a statement to this effect and explain why in the rebuilding plan.

- **Fisheries Management** – objectives related to the management of fisheries that harvest and/or intercept the stock. Where applicable, these objectives must be consistent with the Policy on New Fisheries for Forage Species, the Policy on Managing Bycatch, and/or the Policy for Managing the Impacts of Fishing on Sensitive Benthic Areas (Sensitive Benthic Areas Policy).
    - **Mixed stock and multispecies fisheries** – objectives should strive to find an appropriate balance between the rebuilding needs of the depleted stock with the socio-economic ramifications for those harvesters who primarily target other stocks that are at healthier levels. In such cases it may be necessary to limit catches of those other stocks in order to allow a depleted stock to rebuild unless there is a method to selectively target the healthy stock(s) or species while avoiding catches of the stock that requires rebuilding.
    - **Stock conservation** – objectives related to the life history characteristics and ecological function of the stock beyond the rebuilding target. These may also include shorter-term objectives (milestones) related to stock status other than the rebuilding target described in section 3.4a (e.g., stop further declines of the stock, have a high likelihood of a given rate of growth over shorter timeframes, etc.) These milestones should be distinguished from the rebuilding target. There may also be a desire to express longer-term stock status objectives beyond the rebuilding target. The rebuilding plan should clearly indicate these are long-term

    objectives that will be carried forward to the IFMP or management plan once the rebuilding target is achieved (see section 2.3 for more information on transitioning out of rebuilding plans.) Stock conservation objectives could also include restoring a stock to its "normal" or "near normal" life history characteristics (e.g., restoring age structure, size and age-at-maturity, genetic diversity, behavioural traits, distribution) and ecological function (e.g., restoring predator/prey relationships), to the extent possible.
  - **Socio-economic and cultural** – objectives related to socio-economic and/or cultural considerations of the fisheries.
  - **Shared jurisdictions and/or transboundary, straddling or highly migratory stocks** – where applicable, objectives related to the engagement (or promoting cooperation) with other jurisdictions to achieve the other rebuilding objectives in this plan (e.g., with regional fisheries management organizations or other countries for transboundary, straddling or highly migratory stocks, with Provinces, Territories, or other federal departments to address threats to the stock outside DFO's jurisdiction, etc.).
- **Habitat** – objectives related to habitat restoration if habitat loss or degradation is indicated as a probable cause for the stock's decline. Where applicable, these must be consistent with the Sensitive Benthic Areas Policy. Note these objectives do not have to be related to restoration, but could also include habitat protection, studies to identify or reduce knowledge gaps, monitoring of the remaining habitat, etc. DFO should consider the need, feasibility and costs and benefits of new or additional restoration measures and where possible and applicable, the efficacy of measures that are already in place.
- **Monitoring and Compliance** – objectives related to fisheries monitoring as well as those related to compliance and enforcement.

These must be designed to meet the requirements of the Fishery Monitoring Policy.

- **Knowledge Gaps** – goals related to ongoing or additional data collection and/or analysis to resolve knowledge gaps or suspected management challenges that affect or inform rebuilding plans, including any gaps related to the above categories.

Each objective must be measurable. As with the rebuilding target, the additional objectives should be expressed using three components: the desired state, desired probability to achieve the desired state (if uncertainty is an attribute of the state), and timeline to achieve the objective, where possible. Where the measurable objective includes a timeline (e.g., within 3 years), indicate the starting year of the timeline.

For data-poor or model-limited stocks, alternative means of expressing the measurable objective can be used, for example, omitting a probability. In some cases, the objective may not require an associated likelihood as success is binary (i.e., success is measured by achievement of the desired outcome). For some objectives, a desired state may be difficult to articulate (e.g., expansion of stock distribution to previously occupied habitats). In these instances, the measurable objective may be focused on improvements from the baseline over specified time intervals (e.g., increased abundance of stock in areas previously occupied in next five years).

As per the 2009 PA Policy, conservation objectives must be prioritized over socio-economic considerations. Specifically the policy states: “[For stocks] in the Critical Zone, conservation concerns are paramount and there is no tolerance for preventable declines.”

3.5 Management measures aimed at achieving the objectives

ℹ This section provides information required by subs. 70(1)(e) of the FGR for prescribed major fish stocks.

The purpose of this section of the rebuilding plan is to describe all of the fisheries management measures that will be in place under the rebuilding plan. Each measurable objective must have at least one associated management measure designed to achieve the objective. As stated in the previous section, these measures must be consistent with the criteria from the 2009 PA Policy noted above. Include measures for directed fisheries on the stock as well as any fisheries that incidentally catch the stock (i.e., bycatch) and may be a source of fishing-generated mortality (in keeping with DFO’s 2013 Policy on Managing Bycatch and informed by the 2019 Fishery Monitoring Policy and supporting tools).

Explicitly link each measure to at least one objective by briefly explaining how the measure is expected to contribute to achieving the objective(s). If applicable, also explain how the measures take into account the biology of the fish or the environmental conditions affecting the stock. Additional information on rebuilding best practices is available in NRC, 2014 and FAO, 2018. Table 2 gives examples of how to link management measures to measurable objectives. A narrative format is also acceptable.

Table 2: Examples of how to link management measures to measurable objectives and explain their intended outcomes.

Objective	Management measure(s)	Expected outcome	Biology or environmental conditions taken into account

Objective	Management measure(s)	Expected outcome	Biology or environmental conditions taken into account
$B > B_{rebuilt}$ with 50% probability in 20 years, starting in 2020	Total allowable catch set to ensure exploitation rate shall not exceed 10% of the biomass.	This level of fishing pressure is expected to result in net stock growth, and an expected probability of $B > B_{rebuilt}$ of 65% in 20 years.	This harvest decision rule is based on the stock assessment. For more details see CSAS XX/XXX.
Reduce the capture of undersized fish to less than 15% of the annual catch in 3 years, starting in 2020	• Minimum fish size is set to L50. • The small fish protocol will be implemented starting 2020.	A reduction in the fishing mortality of juvenile (small) fish aims to increase the productivity of the resource and encourage stock growth.	These measures take into account the length at which 50% the stock reaches reproductive maturity (L50).

Addressing habitat loss for prescribed major fish stocks

For prescribed major fish stocks, subs. 6.2(5) of the *Fisheries Act* requires that the rebuilding plan must take into account whether there are habitat restoration measures in place when habitat loss or degradation has been identified as a contributing factor to the stock's decline. For the purposes of meeting subs. 6.2(5), DFO must confirm whether restoration measures are in place. "In place" means a measure to restore the stock's habitat:

- has been implemented and has restored habitat and/or continues to contribute to restoring habitat;
- has recently been implemented and is expected to restore habitat; or
- is in the process of being implemented and is already contributing to restoring fish habitat.

While not required per subs. 6.2(5) of the FSP, the rebuilding plan should also include any habitat restoration measures that are proposed to meet any habitat objectives of the rebuilding plan as per section 3.4b.

3.6 Socio-economic analysis

The purpose of this section of the rebuilding plan is to outline a summary of the results of the socio-economic analysis conducted for the rebuilding plan. Provide a reference to the full socio-economic analysis once it is published.

3.7 Method to track progress towards achieving the objectives

> This section provides information required by subs. 70(1)(f) of the FGR for prescribed major fish stocks.

In this section of the rebuilding plan outline the performance metrics that will be used to measure progress towards the objectives, including the rebuilding target, and the frequency at which each metric will be evaluated against the objective (most metrics will likely be measured annually or during each stock assessment). Each measurable objective must be linked to at least one performance metric. Examples of performance metrics are provided in Table 3. Collectively, these performance metrics will provide

DFO with a means to transparently assess the progress of rebuilding plans and evaluate trade-offs of management outcomes in periodic reviews during the lifespan of the rebuilding plan.

This requirement is consistent with the 2009 PA Policy stipulation that the plan must be associated with appropriate monitoring and assessment of the condition of the stock to confirm the success of rebuilding.

Table 3: Example of performance metrics table to be included in rebuilding plan that links each measurable objective to at least one performance metric and indicates the frequency the metric will be measured. These examples are for illustrative purposes only.

Objective	Metric to measure progress	Frequency of measurement
$B > B_{rebuilt}$ with 50% probability in 20 years, starting in 2020	Biomass in the terminal year of the current stock assessment relative to $B_{rebuilt}$	Every stock assessment
Reduce the capture of undersized fish to less than 15% of the annual catch in 3 years, starting in 2020	Fishers logs and at-sea observer data demonstrate annual mean reduction in capture of undersized fish from 2020 to 2023 By 2023, the annual mean proportion of undersized fish is ≤ 15%	Annually using fisheries monitoring data
Develop stock assessment model within next 3 years, starting in 2020	Model is published in CSAS Research Document within 3 years.	Once

3.8 Periodic review of the rebuilding plan

> This section provides information required by subs. 70(1)(f) of the FGR for prescribed major fish stocks.

The purpose of this section is to (a) set out a schedule that will periodically review the rebuilding plan to assess progress towards achieving the plan's objectives, and (b) determine whether an adjustment to the plan is needed.

In this section of the rebuilding plan describe:

- The schedule to periodically review the rebuilding plan to evaluate the performance of the plan against its objectives,
- The process for conducting a review, indicating when Indigenous groups and stakeholders can participate and their roles (including the Terms of Reference for the review), and,
- The results of previous reviews and the action(s) taken in response to the reviews.

Reviews must be completed on a regular basis, with timelines determined based on factors such as the specifics of the stock in question, the science assessment cycle, and the schedule of fisheries advisory committee meetings. The reviews should be conducted sufficiently often such that rebuilding performance can be evaluated, or new information and data taken into account. The frequency of review may be adjusted based on stock trend (e.g., more frequent review if stock status continues to decline), the length of time the rebuilding plan has been in effect (e.g., more frequently at the beginning of the plan to assess whether measures are performing as expected), or exceptional circumstances (e.g., loss of key data used in rebuilding measures). This will allow the rebuilding plan to be revised as required. Intervals may be expressed as:

- Every X years (set at a frequency appropriate to the biology of the stock or acquisition of new data); or
- Following every X stock assessments (e.g., where X may be every stock assessment if they are conducted infrequently or only after a certain number of stock assessments in the case where assessments happen annually or biennially).

Include a rationale for the interval selected in the rebuilding plan (e.g., the stock assessment occurs every two years, so four years was selected to allow time for changes to be measured).

A review can also be planned outside the standard review interval, when necessary, if new information becomes available that necessitates a review. These exceptional circumstances to the schedule should be specified in the rebuilding plan to the extent possible. These may include:

- Changes to the stock assessment model(s);
- Large changes in stock status (positive or negative, such as due to a stock recruitment event);
- New information, data or analyses that significantly change understanding of the stock status or population dynamics;
- Change in status of predator or prey species;
- Major environmental events that may impact the stock (e.g., oil spill, migration barrier, etc.); and
- Changes in the COSEWIC assessment or SARA-listing status of the stock.

What a review should entail

The purpose of the review is to determine if the management measures are working to achieve the objectives and/or the objectives remain valid. This will involve evaluating progress toward each objective using the performance metrics produced in the “Measurable Objectives Aimed at Rebuilding the Stock” section of the rebuilding plan. Indigenous groups and stakeholders will be consulted by DFO. In addition, DFO will produce a report that evaluates rebuilding performance with accompanying evidence and may propose changes to the rebuilding plan, if necessary.

What to do with the results of the review

If the review determines that insufficient progress has been made towards the objectives and/or the management measures are not performing as expected, then changes may be required (e.g., additional catch restrictions or other appropriate management measures). Revisions to the rebuilding plan must include consultations with Indigenous groups and stakeholders.

Per subs. 70(7) of the FGR, the review report must be published on the DFO website. The review report should be published within 120 days after the approval of the review report by an appropriate DFO official.

If progress to rebuild the stock is occurring more quickly than anticipated, caution should be exercised regarding changing the management measures, such as increasing the allowable catch. Stock growth based on short-term monitoring results (e.g., the occurrence of an unexpectedly strong year class) should be seen as a rare opportunity to rebuild stock biomass and not a reason to increase catches or terminate a rebuilding plan before rebuilding is assured (FAO, 2005). International best practice is to continue the rebuilding plan until the stock has reached its rebuilding target to ensure that an early return to higher catch limits does not result in re-depletion of the stock (FAO, 2018).

3.9 References

In this section of the rebuilding plan reference any documents cited in the rebuilding plan. This should include relevant scientific documents from the primary literature and CSAS reports (e.g., Science Advisory Reports, Research Documents, Recovery Potential Assessments, etc.). Other documents to be cited may include the reports summarizing the plan’s socio-economic analyses, the stock’s IFMP, COSEWIC assessments, or grey literature (e.g., on habitat restoration measures undertaken by external parties). If possible, include the hyperlinks to the reports.

4.0 Subsection 6.2(2) of the Fish Stocks Provisions

Efforts should be made to mitigate identified adverse socio-economic and cultural impacts during the development of the rebuilding plan such that the plan is still consistent with the obligations of subs. 6.2(1) of the *Fisheries Act*, i.e., to grow the stock above its LRP, and within the guidelines outlined in section 3 of this document.

If it is not possible to address the identified adverse impacts and remain aligned with subs. 6.2(1), the Minister can invoke subs. 6.2(2) and adjust the management measures in the rebuilding plan to mitigate the adverse socio-economic and/or cultural impacts. Under subs. 6.2(2) if the management measures are adjusted – for example an increase in the allowable catch of the stock – this may make it necessary to extend the timeline to rebuild a stock above its LRP beyond the maximum timelines for rebuilding plans under subs. 6.2(1) described in section 3.4a of these guidelines.

In this case, to be consistent with subs. 6.2(2) of the FSP, the amended plan must minimize further decline of the stock. Thus, the management measures in a subs. 6.2(2) rebuilding plan must be consistent with the goal of rebuilding the stock above its LRP and minimizing decline of the stock. There must be a very low likelihood of preventable decline (< 5% probability) for the duration of the amendment of the rebuilding plan. This means that total fishing mortality on the prescribed major fish stock (both directed and bycatch) must be limited to levels that are expected to allow the stock to grow to the rebuilding target within the extended timeline.

5.0 Glossary

Bycatch: Any retained catch that includes species and specimens of the target species, such as specimens of a particular sex, size, or condition, that the harvester was not licensed to direct for but is required or permitted to retain; and all non-retained catch, including catch released from gear and entanglements, whether alive, injured or dead, and whether of the target species or the non-target species.

Cautious zone: The stock status zone above the Limit Reference Point (LRP) and below the Upper Stock Reference (USR) as described in DFO’s 2009 PA Policy.

Cost effectiveness analysis: A socio-economic analysis that evaluates the effectiveness of alternative means of accomplishing an objective relative to their cost. In the rebuilding plan context particularly, it would compare the relative costs of achieving the same outcome for the stock using different management tools.

Critical zone: The stock status zone for stocks at or below the Limit Reference Point (LRP) as described in the DFO’s 2009 PA Policy.

Fishery: Can refer to the sum of all fishing activities on a given resource, for example a hake fishery or shrimp fishery. It may also refer to the activities of a single type or style of fishing on a particular resource, for example a beach seine fishery or trawl fishery.

Harvest decision rules: Pre-agreed management actions to be taken under different stock status scenarios as described in DFO’s 2009 PA Policy. They are often described as a function of variables related to the status of the stock. For example, a decision rule can specify how fishing mortality (*F)* or yield should vary with biomass. Management acts on the rules using management measures. These measures are how the fishery's harvest levels and fishing activity are controlled or managed and include adjustments to Total Allowable Catch (TAC), effort levels or fishing time, gear modifications or usage, time and area closures, etc. Harvest decision rules are also sometimes referred to as harvest control rules or more infrequently, TAC decision rules.

Generation time: “Generation length is the average age of parents of the current cohort (i.e., newborn individuals in the population). Generation length therefore reflects the turnover rate of breeding individuals in a population. Generation length is greater than the age at first breeding and less than the age of the oldest breeding individual, except in taxa that breed only once. Where generation length varies under threat, such as the exploitation of fishes, the more natural, i.e., pre-disturbance, generation length should be used.” (IUCN 2001, 2012) For more information on calculating generation time, see the *Guidelines for Using the IUCN Red List Categories and Criteria*.

Healthy zone: The stock status zone above the Upper Stock Reference (USR) as described in DFO’s 2009 PA Policy.

Likelihood: Although often used to describe the extent to which a proposition (a hypothesis, or a model) explains available information (past events; e.g., the likelihood that habitat degradation or loss has contributed to a stock's decline), for the purposes of this guidance, likelihood is also the chance that a given event will happen, expressed or qualitatively (e.g., high likelihood). Annex 2B of DFO's 2009 PA Policy provides a likelihood scale of qualitative descriptors with their corresponding probability range (also provided as Table 1 in this guidance).

Limit Reference Point (LRP): The stock status below which serious harm is occurring to the stock. At this stock status level, there may also be resultant impacts to the ecosystem, associated species and a long-term loss of fishing opportunities, as described in DFO's 2009 PA Policy.

Maximum Sustainable Yield (MSY): The maximum average annual catch, or yield, that can be removed from a stock over an indefinite period under prevailing environmental conditions. The maximum use that a fishery resource can sustain without impairing its renewability through natural growth or replenishment.

Precautionary Approach (PA): Being cautious when scientific information is uncertain, unreliable or inadequate and not using the absence of adequate scientific information as a reason to postpone or fail to take action to avoid serious harm to the resource. (See DFO's 2009 PA Policy.)

Prescribed major fish stock: A stock that has been made subject to the Fish Stocks provisions in the *Fisheries Act* (ss. 6.1–6.3) by prescribing the stock in the *Fishery (General) Regulations* (s. 69).

Probability: The chance (statistical or relative frequency) that a given event or outcome has or will occur. It is typically used when uncertainty is associated with an outcome and can be quantified.

Removal Reference (RR): The limit fishing removal rate for the stock as described in DFO’s 2009 PA Policy. It is normally expressed in terms of fishing mortality (*F*) or harvest rate; but could also be described in other ways (ex. number of traps-hauls). It includes mortality from all fishing pressures. To comply with the United Nations Fish Stocks Agreement, it must be less than or equal to the fishing mortality associated with maximum sustainable yield.

Risk: In general, the possibility of something undesirable happening, for example, of harm or loss, or of failing to meet fisheries management objectives. Risk occurs as a result of uncertainty, and is measured in terms of the consequences of an event and the probability of its occurrence.

Risk tolerance: The tolerable, or acceptable, probability of an undesirable event occurring, such as a breach or a limit, or failure to achieve a target or other management objective.

Socio-economic analysis: A broad concept which covers several different types of analysis. There is a wide array of methodological approaches to socio-economic analysis. The validity and usefulness of each type of approach depends on the issues and decisions being analyzed. Socio-economic analysis includes socio-economic profile, cost-benefit analysis, cost effectiveness analysis, multiple account evaluation or regional economic impact analysis.

Stock: A population of individuals of an aquatic species found in a particular area. *Alternatively*: the living resources in the community or population from which catches are taken in a fishery. Use of the term *fish stock* usually implies that the particular population is more or less isolated from other stocks of the same species and hence self-sustaining.

Target Reference Point (TRP): Represents the overall stock level target for the stock as described in DFO’s 2009 PA Policy. It is determined by productivity objectives for the stock, broader ecological considerations and socio-economic objectives for the fishery. The TRP is typically set at or above the Upper Stock Reference (USR) and is unlikely to be the same level as the rebuilding target in a rebuilding plan.

Upper Stock Reference (USR): The stock level threshold below which the removals must be progressively reduced in order to avoid, with high probability, reaching the LRP as described in DFO’s 2009 PA Policy. Here the USR is acting as an operational control point, and its selection depends on other components of the management procedure such as the target harvest rate, as well as the risk tolerance for an LRP breach.

Uncertainty: The incompleteness of knowledge about the state or processes (past, present, and future) of a natural system. Uncertainty can be divided into six types: including process, observation, model, estimation, institutional, and implementation. For a more detailed discussion, see Francis and Shotton (1997).

6.0 References and further reading

Benson A.J., Cooper A.B., Carruthers, T.R. 2016. An evaluation of rebuilding policies for U.S. fisheries. PLoS ONE 11(1): e0146278. https://doi.org/10.1371/journal.pone.0146278.

DFO. 2006. Impacts of Trawl Gears and Scallop Dredges on Benthic Habitats, Populations and Communities. DFO Canadian Science Advisory Secretariat (CSAS) Science Advisory Report 2006/025

DFO. 2009. A fishery decision-making framework incorporating the precautionary approach. Last modified 2009-03-23. http://www.dfo-mpo.gc.ca/reports-rapports/regs/sff-cpd/precaution-eng.htm.

DFO. 2010. Potential impacts of fishing gears (excluding mobile bottom-contacting gears) on marine habitats and communities. DFO Canadian Science Advisory Secretariat (CSAS) Science Advisory Report 2010/003.

DFO. 2021. Science Guidelines to Support Development of Rebuilding Plans for Canadian Fish Stocks. DFO DFO Canadian Science Advisory Secretariat (CSAS) Science Advisory Report 2021/006.

FAO. 2005. World Inventory of Fisheries. Depleted Stocks Recovery: a Challenging Necessity. Issues Fact Sheets. http://www.oceansatlas.org/subtopic/en/c/1433/.

FAO. 2018. Rebuilding of marine fisheries Part 1: Global review. Fisheries and Aquaculture Technical Paper No 630/1, 294 p.

Francis, R.I.C.C., Shotton, R. 1997. “Risk” in fisheries management: a review. Can. J. Fish. Aquat. Sci. 54(1699-1715).

IUCN. 2001. IUCN Red List Categories and Criteria: Version 3.1. IUCN Species Survival Commission. IUCN, Gland, Switzerland and Cambridge, U.K.

IUCN. 2012b. IUCN Red List Categories and Criteria: Version 3.1. Second edition. Gland, Switzerland and Cambridge, UK: IUCN. https://www.iucnredlist.org/resources/categories-and-criteria

Murawski, S.A. 2010. Rebuilding depleted fish stocks: the good, the bad, and, mostly, the ugly. ICES J. Mar. Sci. 67: 1830-1840.

NRC. 2014. Evaluating the Effectiveness of Fish Stock Rebuilding Plans in the United States. Washington, DC: National Research Council, The National Academies Press. 143 p.

UNCOVER. 2010. Final Activity Report – Executive Summary (2010). The UNCOVER project: Understanding the mechanisms of fish stock/fishery recovery. UNCOVER (FP6- 2004-SSP4)

Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., Fogarty, M. J., Fulton, E. A., Hutchings, J. A., Jennings, S., Jensen, O. P., Lotze, H. K., Mace, P. M., McClanahan, T. R., Minto, C., Palumbi, S. R., Parma, A. M., Ricard, D., Rosenberg, A. A., Watson, R., and Zeller, D. 2009. Rebuilding Global Fisheries. Science 325: pp. 578- 585.

Annex A: Rebuilding plan template

Rebuilding plans should follow the template provided. Directions have been included throughout the template to facilitate its completion. These should be deleted before finalizing the rebuilding plan.

Directions are noted in the following ways:

- Info panels;
- As bulleted lists in normal font;
- Standardized language is in blockquotes; and
- Where stock-specific language must be included, it is shown by square brackets, i.e., [insert X information].

In addition, certain rebuilding plan sections contain additional directions for stocks where habitat loss or degradation contributed to the stock’s decline, or where stocks are unlikely to rebuild under prevailing conditions.

Rebuilding plan

Cover page

Include the species common and scientific names, stock name or area, species image or illustration, DFO region, date the stock was determined to be at or below its LRP, and the date the rebuilding plan was approved.

Foreword

ℹ Standard text that must be included in every rebuilding plan.

In 2009, Fisheries and Oceans Canada (DFO) developed A Fisheries Decision-Making Framework Incorporating the Precautionary Approach (PA Policy) under the auspices of the Sustainable Fisheries Framework. It outlines the departmental methodology for applying the precautionary approach (PA) to Canadian fisheries. A key component of the PA Policy requires that when a stock has declined to or below its limit reference point (LRP), a rebuilding plan must be in place with the aim of having a high probability of the stock growing above the LRP within a reasonable timeframe.

In addition, under section 6.2 of the Fish Stocks provisions (FSP) in the amended *Fisheries Act* (2019), rebuilding plans must be developed and implemented for prescribed major fish stocks that have declined to or below their LRP. This legislated requirement is supported by section 70 of the *Fishery (General) Regulations* (FGR), which set out the required contents of those rebuilding plans and establish a timeline for each rebuilding plan’s development.

The purpose of this plan is to identify the main rebuilding objectives for [name of stock(s)] in [identify area(s) covered by the plan], as well as the management measures that will be used to achieve these objectives. This plan provides a common understanding of the basic “rules” for rebuilding the stock(s). This stock is [prescribed/not prescribed] in the *Fishery (General) Regulations* (section 69) and thus [is/is not] subject to section 6.2 of the *Fisheries Act* and regulatory requirements.

The objectives and measures outlined in this plan are applicable until the stock(s) has reached its rebuilding target. Once the stock is determined to be at the target, the stock(s) will be managed through the standard Integrated Fisheries Management Plan (IFMP) or other fishery management process in order to fulfill the requirements of the FSP. Management measures outlined in this rebuilding plan are mandatory, and may be modified or further measures added if they fail to result in stock rebuilding.

This rebuilding plan is not a legally binding instrument which can form the basis of a legal challenge. The plan can be modified at any time and does not fetter the Minister's discretionary powers set out in the Fisheries Act. The Minister can, for reasons of conservation or for any other valid reasons, modify any provision of the rebuilding plan in accordance with the powers granted pursuant to the Fisheries Act.

Decisions flowing from the application of this rebuilding plan must respect the rights of Indigenous peoples of Canada recognized and affirmed by section 35 of the *Constitution Act* (1982), including those through modern treaties. Where DFO is responsible for implementing a rebuilding plan in an area subject to a modern treaty, the rebuilding plan will be implemented in a manner consistent with that agreement. The plan should also be guided by the 1990 *Sparrow* decision of the Supreme Court of Canada, which found that where an Aboriginal group has a right to fish for food, social and ceremonial purposes, it takes priority, after conservation, over other uses of the resource.

Contents

Introduction and context

ℹ This section is recommended but not required by regulation for prescribed major fish stocks.

- Provide a brief overview of the:
  - Relevant biology of the stock;
  - Relevant environmental conditions or ecosystem factors affecting the stock; and
  - Directed fisheries and those with bycatch of the stock should have their current management measures and key participants described (include details regarding shared jurisdiction if applicable). A map of the stock's extent and fishing areas may be included as well.
- This section may also include:
  - Socio-economic overview of landings (quantity and value) over the past 10 years and profiles of the affected fish harvesters;
  - Cultural impacts of the declines of the stock to date;
  - Fisheries management issues (including historical context if appropriate);

  - COSEWIC assessment status and SARA listing decisions if applicable;
  - Reference to Indigenous Knowledge used in the development of the plan; and
  - Description of the process and Indigenous groups and stakeholders involved in the development of the plan (alternatively, provide this information in an annex to the rebuilding plan).

Stock status and stock trends

> ℹ This section provides information required by subs. 70(1)(a) of the FGR for prescribed major fish stocks.

Required Components

- Complete the following table (if a reference point is unavailable indicate N/A or In Development as appropriate). State the role of the USR as well:

Table 1: Summary of the Precautionary Approach Framework reference points for [insert stock name].

PA reference point	Stock-specific value of the reference point	Source
Limit Reference Point (LRP)		
Upper Stock Reference (USR)		
Target Reference Point (TRP)		

PA reference point	Stock-specific value of the reference point	Source
Removal Reference (RR)		

Stock status and trends

- Note the date that DFO determined that the stock was at or below its LRP.
- Provide a summary of :
    - A current estimate of stock abundance and/or biomass, or relative measures for either (an index or a proxy), as used in the stock assessment, including reference to the stock's PA stock status zone (i.e., likely Critical or Cautious Zone).
    - Characterize historical, and if available, projected trends in estimated abundance, biomass or appropriate indices of either. If applicable, include any additional secondary indicators of importance to the stock assessment (e.g., fishing mortality, age structure, sex ratio, size distributions, etc.)
- Provide a reference to the most recent applicable science advisory or stock assessment document to refer readers to additional information on the stock.

Optional Elements

- If applicable, note current COSEWIC assessment status (include relevant Designatable Unit), SARA listing status, or Do Not List decision. Note any applicable difference between the geographic delineation of the stock and of the Designatable Units to ensure the conservation statuses relevant to the stock are fully described.
- Indigenous knowledge.

Probable causes for the stock's decline

> This section provides information required by subs. 70(1)(b) of the FGR for prescribed major fish stocks.

Required Components.

- Provide an overview of the probable factors that led to the decline of the species as well as those that may limit rebuilding.
- This section must address whether or not loss or degradation of the stock’s fish habitat has contributed to the stock’s decline.
    - If it is a probable cause, identify the type of habitat loss, its historical and current location(s), and how the stock uses the habitat (e.g., spawning grounds, nursery habitat, etc.). Note who has jurisdiction to manage activities that affect the habitat.
    - If habitat loss or degradation is not attributed to the stock’s decline, include the following sentence:

> Loss or degradation of the stock’s fish habitat is unlikely to have contributed to the stock’s decline given the current understanding of the best available evidence.

- If rebuilding prospects are currently negligible due to conditions outside DFO’s control, select the most appropriate opening sentence, and then complete the paragraph with the remaining standardized language:.

> Rebuilding appears to be unlikely under current ecosystem conditions, even with no fishing mortality (e.g., F=0 in projections).

> Rebuilding appears to be unlikely at current [insert appropriate factor] (e.g., predator abundance), even with no fishing (e.g., projections with zero fishing mortality and strong evidence that the lack of recovery is due to predation).

> It is likely that the stock will continue to decline at current [insert appropriate factor(s)] (e.g., predator abundance, environmental conditions), and surplus production is not expected. Local extinction is possible (or likely), (e.g., evidence that the stock is experiencing a strong Allee effect).

Remainder of standardized paragraph:

> This rebuilding plan has been developed to minimize, to the extent possible, further declines of the stock. This is to preserve the stock such that, should the prevailing conditions limiting the stock's recovery change, the stock retains the potential to rebuild.

Optional Elements

- Identify knowledge gaps and their potential impact on the rebuilding plan.
- Indigenous knowledge.

Measurable objectives aimed at rebuilding the stock

> This section (including all subsections) provides information required by subs. 70(1)(c) and (d) of the FGR for prescribed major fish stocks.

Rebuilding target and timeline

Required Components

- Provide the desired rebuilding target and timeline to achieve this target.
    - Where possible, also include the probability associated with this target.
    - Include a link (if available) to the report that calculated these figures or contact information if there are questions on how these were calculated.
- Note which year will be considered year 1 of the timeline (e.g., effective date of rebuilding plan).

For stocks unlikely to rebuild:

- Establish a rebuilding target as per usual. The intention here is to identify what the rebuilding plan's "end point" would be in the event conditions change and rebuilding becomes feasible.
- Calculate the rebuilding timeline as per usual, if possible. If it is not possible, per subs. 70(6) of the FGR, include the following paragraph:

> A timeline to the rebuilding target is not feasible to establish for this stock due to the [insert reason why the stock is unlikely to rebuild] limiting the likelihood of stock growth even in the absence of fishing. During each review, the factors limiting the stock's potential for growth will be re-assessed to determine if they are still influencing the stock and whether a rebuilding timeline can be calculated.

Additional measurable objectives and timelines

Required Components.

- Provide additional measurable objectives for each of the following categories, where applicable and feasible:

  - Fisheries management, including where appropriate:
    - Mixed stock and multispecies fisheries,
    - Stock conservation,
    - Socio-economic and cultural, and/or
    - Shared jurisdictions;
  - Habitat (required if habitat loss or degradation was identified as a factor in the stock's decline);
  - Monitoring and compliance;
  - Knowledge gaps.
- For stocks unlikely to rebuild, these objectives may be aimed at preserving the stock such that should the prevailing conditions limiting the stock's recovery change, the stock retains the potential to rebuild (e.g., monitoring the limiting external factors, or exploring whether there are means to mitigate the external limiting factor, etc.)
- Include timelines for each of these objectives.
  - For stocks unlikely to rebuild, these timelines should not be indefinite, but may reflect a duration of time for which they will be enacted and then assessed in a periodic review.
- To the extent possible, include the probability associated with the objective (this may not be possible for data- or model-poor stocks, or where success of the objectives is binary).

> To more easily track objectives, management measures and performance metrics, number the objectives. The rebuilding target should be numbered 1, and all other measurable objectives would follow.

Management measures aimed at achieving the objectives

> This section provides information required by subs. 70(1)(e) of the FGR for prescribed major fish stocks.

Required Components

- Outline all of the management measures that will be in place under the rebuilding plan and that are aimed at achieving the objectives set out in the previous section.
  - Include measures for directed fisheries on the stock as well as for any fisheries that incidentally catch the stock (i.e., bycatch) and may be a source of fishing-generated mortality.
  - If there is directed fishing or bycatch on the stock while it is at or below its LRP explain how the catch level has a high likelihood (≥75% probability) of stock growing to the rebuilding target within the plan's rebuilding timeline and a very low likelihood (<5% probability) of preventable decline as per section 3.4b above.
- Provide a brief explanation of how each measure is expected to contribute to achieving at least one of the objectives and the evidence for those expectations.
- If relevant, also explain how the measures take into account the biology of the fish or the environmental conditions affecting the stock.

The above information may be presented in tabular format, as shown in Table 2 below, or in narrative form if that is preferred. Include the number and description of each objective in the *Objective* column for ease of reference.

Table 2: Summary of management measures aimed at achieving the rebuilding plan objectives.

Objective	Management measure(s)	Expected outcome	Biology or environmental conditions taken into account

If habitat loss or degradation was listed as a probable cause of the stock's decline, habitat restoration measures must be addressed in this section of the rebuilding plan as follows. The information can be included in Table 2 above or in narrative format:

- List what habitat restoration measures are already in place or are planned (if any).
- If restoration measures are planned/in place, detail how the measures are expected to promote stock growth and reduce the risk of further decline.
    - At a minimum, this should be a qualitative statement of the expected or realized benefits of the restoration measures;
    - If possible, include a quantitative description of the expected or realized benefits of the restoration measures; or
- If there are no restoration measures in place, describe why measures may not be in place and what measures will be considered for the future (if any).
    - If no restoration measures are feasible, describe how this may impact the rebuilding objectives and how the rebuilding plan has taken this perspective into account.

Optional Elements

- Include the high level (qualitative or quantitative) results of past economic analysis of rebuilding measures (if available).
- Discuss the impacts or trade-offs of the proposed measures on other stocks/species if applicable.

Socio-economic analysis

> This section is strongly recommended, but not required by regulation for prescribed major fish stocks.

- Include a summary of the results of the socio-economic analysis conducted for the rebuilding plan. Provide a reference to the full socio-economic analysis once it is published.
- If completed, a summary of the cost-effectiveness analysis of potential management measures should be included or a link provided to the analysis.
- For stocks unlikely to rebuild, long-term economic benefits of rebuilding are unlikely to be calculated, but it may be possible to calculate the benefits of preventing or slowing further decline of the stock.

Method to track progress towards achieving the objectives

> This section provides information required by subs. 70(1)(f) of the FGR for prescribed major fish stocks.

Required Components

- Include the paragraph below and then complete the following table to describe the metrics and how often each will be measured to track the progress to achieving each of the objectives. Include the number and description of each objective in the *Objective* column for ease of reference.

> Performance metrics provide DFO with a means to assess the progress of the rebuilding plan towards the plan's objectives. For each objective, table 3 below outlines how and when progress will be measured.

Table 3: Summary of the performance metrics and frequency of measurement associated with each objective in this rebuilding plan.

Objective	Metric to Measure Progress	Frequency of Measurement

Periodic review of the rebuilding plan

> This section provides information required by subs. 70(1)(g) of the FGR for prescribed major fish stocks.

Required Components

- Include the paragraphs below to establish a schedule for the periodic review of the rebuilding plan and fill in the required additional information.

The rebuilding plan will be reviewed every [insert schedule here, expressed as either every X years or following every X stock assessments] to determine whether progress towards the plan's objectives, including the rebuilding target, is being made and whether revisions to the rebuilding plan are necessary in order to achieve those objectives. [Provide a brief rationale for the interval selected].

Additional reviews may also conducted outside the schedule stated above due to exceptional circumstances. For [insert stock name], exceptional circumstances are defined as:

- X
- Y
- Z

> Note, there is no minimum or maximum number of exceptional circumstances that can be listed here. If desired, a general exceptional circumstance made be included (e.g., any other circumstance that warrants a review of the rebuilding plan).

The review will be based on the data gathered using the metrics identified in the Method to Track Progress Towards Achieving the Objectives section of this plan. It will assess the progress of the implementation of management measures and evidence of their effectiveness, as well as the status of the stock and recent trends. In addition, the review will include opportunities for consultation with Indigenous groups and stakeholders on their views of the stock's progress towards rebuilding. See Annex [X] for the proposed Terms of

> Reference of the review. This Terms of Reference will be assessed at the beginning of each review to ensure the pre-defined terms and scope of the review remain appropriate.
>
> The review process will generate a report that evaluates progress towards each management objective against their timelines with accompanying evidence and may propose adjustments to the rebuilding plan if necessary to achieve the objectives.
>
> Stock rebuilding is not always a slow and steady, or even predictable process. Stocks may fluctuate and/or persist at low levels for years until conditions promote surplus production, resulting in rapid growth of the population. Thus lack of progress towards rebuilding may not be an indication that the rebuilding plan’s objectives or management measures are insufficient or ineffective.

References

This section is not required by regulation for prescribed major stocks.

Required Components

- Provide references to any documents cited in the rebuilding plan.

Annex A: [Insert heading]

This section is not required by regulation for prescribed major stocks.

Optional Elements

- Include any annexes as needed. Suggested possible annexes include:

  - The process and Indigenous groups and stakeholders involved in the rebuilding plan’s original development;
  - The proposed Terms of Reference for the rebuilding plan’s periodic reviews; and/or
  - The reports prepared following each periodic review.

Footnotes

[1] Section 2.1 of the *Fisheries Act* defines fish as (a) parts of fish, (b) shellfish, crustaceans, marine animals and any parts of shellfish, crustaceans or marine animals, and (c) the eggs, sperm, spawn, larvae, spat and juvenile stages of fish, shellfish, crustaceans and marine animals.

[2] The 2009 PA Policy applies to fish stocks managed by DFO; that is, those stocks that are the specific and intended targets of a fishery, whether in a commercial, recreational or subsistence fishery.

[3] Extensions of up to an additional 12 months are possible to this timeline. See the section on *Extensions of the 24 month Timeline to Complete the Plan* for more details.

[4] Short to medium term is not defined as this will vary depending on the availability of projections for each stock. The intent is to ensure that during the transition from the rebuilding plan to the standard IFMP management process, the fishing pressure on the stock is not increased so rapidly as to result in the stock declining again to the LRP.

[5] In addition, state the role of the USR, for example as a stock status target, a threshold for progressive reductions in the fishing mortality rate to avoid the stock declining to its LRP, and/or a threshold to delineate between the Healthy and Cautious Zones.

[6] Per the 2009 PA Policy, the LRP may be set in terms of biomass, abundance, or other units (such as escapement for salmon, or yield for effort controlled fisheries).

[7] Generation time is not equivalent to an estimate of T_{min}. Generation time does not take into account the stock’s state of depletion or likelihood of growth and thus may not be reflective of the stock’s ability to regrow during that timeframe.

Date modified:
2022-06-07

The relative role of density-dependent and density-independent survival in the life cycle of Atlantic salmon Salmo salar

Article *in* Journal of Animal Ecology · September 1998

DOI: 10.1046/j.1365-2656.1998.00237.x

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Journal of Animal Ecology 1998,
67, 751–762

The relative role of density-dependent and density-independent survival in the life cycle of Atlantic salmon *Salmo salar*

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Summary

1. Density-dependent factors appeared important for the survival of juvenile Atlantic salmon in the River Imsa whilst density-independent factors were more important for the older fish at sea. In fresh water, density dependence was indicated by a stock – recruitment relationship with increasing loss-rates from eggs to smolts and from eggs to adults as egg density increased. 73% of the loss-rates were explained by variation in egg density. At sea, density independence was indicated by the lack of a significant relationship between loss-rates and smolt densities.

2. The relationship between smolt density and initial egg density was best described by an asymptotic ‘Cushing’ type relationship with a plateau at densities higher than approximately 60 000 eggs for the total river areas of 10 000 m^2. The number of smolts developed from the eggs spawned varied between 350 and 2400.

3. The relationship between smolt biomass in wet mass (kg 10 000 m^{-2}) or energy (kJ 10 000 m^{-2}) and the amount of salmon eggs in the River Imsa increased asymptotically. Annual smolt biomass ranged from 13 to 88 kg 10 000 m^{-2}, or 66 000 and 431 000 kJ 10 000 m^{-2}. Variation in egg density accounted for approximately 45% of the variation in smolt biomass (mass or energy).

4. Total wet mass and energy of adults (kg 10 000 m^{-2} and kJ 10 000 m^{-2}) produced in relation to the amount of eggs at the start of the year-class, were not significantly correlated, due to a high variation among years. The biomass of adults ranged from 73 kg 10 000 m^{-2} to 655 kg 10 000 m^{-2} and in energy from 370 000 kJ 10 000 m^{-2} to 3 270 000 kJ 10 000 m^{-2}.

5. Total adult biomass (adults caught at sea and in rivers) and the returning adults to the River Imsa in mass or energy were correlated with the size of the smolt cohort from which they originated. Yearly total adult biomass ranged between 240 and 3711 kg 10 000 m^{-2}, when the number of smolts ranged from 397 to 2751, respectively. The biomass of adults returning to the River Imsa was between 59 and 614 kg, produced from between 672 and 1621 smolts.

Key-words: adults, eggs, energy, key-factor analysis, smolts, stock recruitment, wet mass.

Journal of Animal Ecology (1998) **67,** 751–762

Introduction

The abundance of animal populations varies with time as do the rates of survival and reproductive success of individuals. Density-dependent factors provide a mechanism for population regulation by affecting birth rates, mortality rates and emigration rates (e.g. Elliott 1994). The chief density-dependent factor is often intraspecific competition for resources, which is most effective at high population densities. Other such factors are predation and parasites. At low densities, however, density-independent mortality caused by the abiotic environment is expected to be important for population abundance (Sinclair 1989).

Therefore, Haldane (1956) hypothesized that density-dependent factors are mainly working in benign environments whereas density-independent factors predominate under hostile conditions. This has, for instance, been supported by studies of upland and lowland populations of the spittle bug *Neophilaenus* *lineatus*, sparrow hawk *Accipiter nisus* in stable and declining forests and brown trout *Salmo trutta* in two highly different but neighbouring streams in England (Whittaker 1971; Newton & Marquiss 1986; Elliott 1989).

Fisheries theory assumes that early juvenile mortality is density-dependent while adult mortality is density-independent (Charnov 1986), and the same notion has been developed for coral reef fish and applied to theories of their community structure (Warner & Chesson 1985). In birds, which have lower reproductive rates than fishes, late juvenile prebreeding density-dependent regulation appears more common, whereas large mammals, with their low reproductive rates, are at least partly regulated through changes in fertility (Sinclair 1989).

Several mathematical models describing density-dependent population regulation have been applied with varying degrees of success in aquatic (Rothschild 1986; Shepherd & Cushing 1990; Fogarty, Sissenwine & Cohen 1991; Elliott 1994) as well as non-aquatic literature (Bellows 1981; May 1981). In salmonids, models describing the numerical relationship between recruits and adults have been tested, e.g. for brown trout *Salmo trutta* (Elliott 1994), Pacific salmon *Oncorhynchus* spp.; (e.g. Ricker 1954, 1975, 1989; Rounsefell 1958; Shepard & Withler 1958; Cushing & Harris 1973; Peterman 1980, 1981; Hilborn & Walters 1992) and Atlantic salmon *Salmo salar* L. (e.g. Elson & Tuomi 1975; Buck & Hay 1984; Chadwick 1985; Gardiner & Shackley 1991; Kennedy & Crozier 1993, 1995; Crozier & Kennedy 1995). In the case of brown trout the relationship between different age-groups of recruits and adults has also been tested (Elliott 1994).

Complex life cycles are common among animals with ontogenetic niche shifts and associated metamorphosis (Werner & Gilliam 1984). The holometabolous insects are well-known examples, but this also applies to vertebrates like fishes and amphibians (Jonsson & Jonsson 1993; Moran 1994). For example, Atlantic salmon live in fresh water as territorial parr for 1–8 years before transforming to smolts and migrating to the ocean (Metcalfe & Thorpe 1990). In the ocean, the salmon are free ranging in surface waters for 1–4 years before attaining maturity and returning to fresh water for spawning (Jonsson, Hansen & Jonsson 1991b). There may be very different mechanisms influencing the population size of a species exploiting such different environments due to different carrying capacities and defensibility of the resources of the habitats. In the case of the Atlantic salmon, we hypothesize that density is regulated in fresh water but not in the ocean. The reason is that rivers are very restricted in area and food availability can be defended by the territorial parr, whereas the ocean is vast in size and the abundant pelagic food resources cannot be defended by juveniles and adults (cf. Murray 1982).

In organisms with indeterminate and flexible growth like fishes, size, growth rate, biomass and production are influenced by abundance. At high population densities, food become restricted and the individual growth rate decreases with consequences for production and biomass (Fryer & Iles 1972; Craig 1987). This growth flexibility provides a plastic phenotypic response to changing environments, not necessarily influencing the genetics of the population. Biomass and production can be given in units of wet or dry mass (Chapman 1978; Mann & Penczak 1986), but the unit of energy gives a more objective measure that is easily comparable across cohorts within and among species and locations and which can be also related to the bioenergetics of the individuals (Craig 1980).

Here, we test the roles of density-dependent and density-independent relationships in the life cycle of Atlantic salmon in fresh water and at sea. The relationships are given in terms of number of individuals, wet mass and energy content.

Materials and methods

STUDY AREA

The River Imsa, located near the city of Stavanger, south-western Norway (58°50′N, 6°E), is 1 km long and approximately 10 m wide (total area equals approximately 10 000 m^2), and drains Lake Liavatn into the Høgsfjord (32 ‰ salt). The annual mean water flow in the river is 5·1 m^3 s^{-1}, with the highest discharge during autumn and winter (mean value: 10 m^3 s^{-1}) and the lowest during summer (mean value: 2 m^3 s^{-1}; Jonsson, Jonsson & Ruud-Hansen 1989). The water temperature ranges from about 2 °C in winter to *c.* 20 °C in summer (Jonsson *et al.* 1989). Atlantic salmon and brown trout are the dominant species in the river, but Arctic charr *Salvelinus alpinus*, whitefish *Coregonus lavaretus*, three-spined stickleback *Gasterosteus aculeatus* and European eels *Anguilla anguilla* are also present, as well as rainbow trout *Oncorhynchus mykiss*, escaped from fish farms.

Fish traps are situated about 100 m above the river mouth and separately catch all ascending (box trap) and descending (Wolf trap; Wolf 1951) fish larger than *c.* 10 cm. Throughout the study period 1975–94, the traps were monitored twice a day. Natural tip lengths (cm; Ricker 1979), weights (g) and sexes of all fish were recorded and scale samples of the spawners were taken for age determination, before they were released downstream or upstream of the traps. There is no salmon fishing in the River Imsa.

SMOLTS

All wild Atlantic salmon smolts descending the River Imsa were counted during the period 1975–93. In the period 1975–79 one-third of the smolts descending the river were individually tagged with numbered Carlin tags (Carlin 1955), one-third were adipose fin-clipped and one-third were unmarked. This was to provide estimates of the mortality caused by handling, anaesthesia and tagging. Total return rates of adult Atlantic salmon to the River Imsa of unmarked, fin-clipped and Carlin-tagged smolts were 7·7% of the unmarked, 4·1% of the fin-clipped and 3·1% of the Carlin-tagged fish (Hansen 1988). The mortality of the Carlin-tagged fish relative to the unmarked fish (tagging mortality) was estimated to be $(7{\cdot}7 - 3{\cdot}1)/7{\cdot}7 = 59{\cdot}7\%$. In the present study this figure has been used when adjusting for the tagging mortality. Furthermore, in 1980, the smolts were retained in the traps and not allowed to migrate to sea, but from 1981 and onwards all the Atlantic salmon smolts descending the river were individually tagged with Carlin tags.

From 1983 and onwards every tenth smolt descending into the trap was sampled for age determination by use of scales and otoliths (cf. Jonsson 1976). The smolt age distribution was used to calculate the actual numbers of smolts originating from different brood years and for previous years we assumed that the distribution was the average of that observed from the age determination.

ADULTS

All adult Atlantic salmon ascending the River Imsa were recorded from 1976 to 1994; in the trap at the river mouth, the fish were divided into two groups: One group was taken into the hatchery for stripping whereas the other group was released upstream of the trap for natural reproduction. This latter group was used for estimating the within river stock–recruitment relationship. In the period 1982–90, sea-ranched salmon of the River Imsa stock, that were hatchery-reared until smolting and then released at the river mouth, were also allowed to spawn in the river. From 1991 to 1993 no adults were released upstream of the trap for spawning in the river.

Adult Atlantic salmon from the River Imsa ascend rivers other than the home river. The mean observed straying-rate of wild River Imsa salmon is 9·5% (Jonsson, Jonsson & Hansen 1991a). This figure is based on recaptures of adults that were Carlin-tagged as smolts. The tagged adults were recaptured by anglers and fishermen both at sea and in rivers other than the River Imsa. For these fish, length, total mass and place of recapture and scale samples were registered. However, not all tags were reported, and crude estimates have suggested that about 50% of the tags were unreported (Hansen & Jonsson 1989). When estimating the overall survival from smolts to adults, we used this figure for correction when estimating the marine stock–recruitment relationship.

Relationships between fecundity (F) and total body mass (M) of the River Imsa Atlantic salmon are (Jonsson, Jonsson & Fleming 1996):

$$F = 1{\cdot}22M + 741{\cdot}90,\ \text{d.f.} = 16,\ r^2 = 0{\cdot}79,$$

$P < 0{\cdot}001$ for wild females and

$$F = 1{\cdot}79M + 238{\cdot}18,\ \text{d.f.} = 41,\ r^2 = 0{\cdot}85,$$

$P < 0{\cdot}001$ for sea-ranched females.

From these regressions, we calculated the number of eggs spawned, assuming that there was no mortality after the fish passed the trap, and that the females spawned all their eggs. Experimental tests indicated that this is a reasonable assumption (Fleming, Lamberg & Jonsson 1997).

STOCK–RECRUITMENT ESTIMATE

The relationship between the number of recruits (R) to the population and the parental stock of fish (B) is in fisheries literature called the stock–recruitment curve (Wootton 1990). If number of offspring increases linearly with parental abundance, the relationship is density-independent. This means that there is a constant proportionate survival (p): $R = pB$. If the recruitment rate changes with density, the relationship is density-dependent. To estimate the stock–recruitment relationship we used the equation described by Shepherd (1982):

$$R = aB/[1 + (B/K)^\beta]$$

where K is the threshold biomass. The model parameters were estimated by non-linear least squares. We used this equation because it is rather versatile. Depending on the value of β the equation can display curves similar to (although mathematically different from) those of Ricker (1954), Beverton & Holt (1957) and Cushing (1973). For $\beta > 1$, the curves are dome shaped. The dome, however, has a more pronounced peak than that of the Ricker model (Elliott 1985). For $\beta = 1$, the curve is identical to that of Beverton and Holt. For $\beta < 1$, it mimics the Cushing equation, except that it has a finite slope at the origin. The null hypothesis tested is that there is no dependence of R on B; values of R vary randomly around a constant, estimated by the arithmetic mean value of R. The alternative hypothesis tested is that R is proportional to B, i.e. $R = pB$ (Elliott 1985).

'Key-factor analysis' was used to compare the loss-rates between life stages (Varley, Gradwell & Hassell 1973). In this analysis, the population density was expressed on a logarithmic scale so that the total loss-rate was the sum of loss-rates between successive stages in the life cycle. We measured the loss-rate between the egg and smolt stages, $k_{\text{smolt}} = \log_e$ (number of eggs in each year-class) – $\log_e$ (number of smolts produced from an egg year-class), and between the smolt and adult stages, $k_{\text{adult}} = \log_e$ (number of smolts from an egg year-class) – $\log_e$ (number of adults from the same egg year-class). Total loss-rate (K) is the sum of the loss-rates between the egg and smolt (k_{smolt}) and smolt and adult (k_{adult}) stages. To test for density dependence, mortalities in different life stages (*k* values) were plotted against the initial density of the stage (or its logarithm) and tested for a significant relationship (Dempster 1975).

ENERGY MEASUREMENTS

We measured the energy contents (kJ) of 21 smolts caught in the trap in the River Imsa in May 1995 and of 15 male and 16 female adult salmon caught when ascending the river for spawning in November 1989 and 1995. The fish were sealed in polyethylene bags and frozen shortly after capture. While still partly frozen, the fish were dissected. The energy content of the fish was estimated by adding the energy in protein, lipid and carbohydrate in the tissue (Craig, Kenley & Talling 1978). Total protein content was determined by the analysis of Kjeldahl (Anonymous 1981), the lipid content according to Anonymous (1987), the carbohydrate according to Anonymous (1987) and the glucose according to Anonymous (1978) and Mason (1983) (details in Jonsson, Jonsson & Hansen 1997).

Results

LIFE HISTORY

The River Imsa supports a small population of anadromous Atlantic salmon spawning between Lake Liavatn and the river mouth. The juveniles (parr) use the river as a nursery, and smolt mainly as 2-year-olds. Based on samples of every tenth smolt during 1983–93, the mean age distribution of 1-, 2-, and $\geqslant$3-year-old smolts was 14, 78 and 8%, respectively. Smolts older than 3 years were rare, and only one 4-year-old smolt was caught during the sampling period (in 1983). In eight of the 11 years of sampling, more than 80% of the smolts were 2 years old. Furthermore, the proportion of 3-year-old smolts was high in 1983 (33%) whereas the proportion of 1-year-old smolts was high in 1990 and 1991 (32%). The mean body mass of 1-, 2- and 3-year-old smolts was 24, 38 and 52 g, respectively, with mean energy contents of 111, 187 and 262 kJ, respectively.

The River Imsa salmon mature sexually mainly as one-sea-winter fish. During 1976–94, 82% of the adults ascending the river were one-sea-winter fish and 18% were multi-sea-winter. The mean weight of one-sea-winter salmon was approximately 2 kg and that of multi-sea-winter salmon was 5·5 kg. The respective energy contents were approximately 8800 kJ and 32 000 kJ. The sex ratio (M/F) of the adults in the river during 1976–94 was 1·17. The majority of one-sea-winter salmon were males (60%), while the majority of two-sea-winter fish were females (76%).

STOCK RECRUITMENT

The relationship between the number of smolts (*S*) and the number of eggs spawned (*E*) in the River Imsa was asymptotic (Fig. 1a). The increase in abundance of smolts was highest at low egg densities and started to level off at densities above *c.* 60 000 eggs per 10 000 m^2 river area. However, the curve did not reach a maximum within the egg densities investigated (maximum 600 000 eggs per 10 000 m^2). The number of smolts in each cohort varied between 337 and 2357 with the highest value in the 1978 cohort. When this year-class was omitted, the coefficient of determination increased from 0·49 to 0·75: $S = (0{\cdot}09E)/[1 + (E/7320)^{0{\cdot}77}]$, $r^2 = 0{\cdot}75$, d.f. = 11, $P < 0{\cdot}01$. Furthermore, the relationship between the number of returning adults to the River Imsa and the initial number of eggs at the start of each year-class is described by an asymptotic curve (Fig. 1b). However, the relationship between the eggs produced by these adults and the number of eggs at the start of each year-class was not significant ($r^2 = 0{\cdot}22$, $P > 0{\cdot}05$), because of the large variability among year-classes (Fig. 1c).

The number of adults surviving in the ocean increased linearly with the number of descending smolts from which these adults were produced; this applies both to the total number of adults caught in rivers and at sea ($r^2 = 0{\cdot}73$, d.f. = 16, $P < 0{\cdot}01$; Fig. 1d) and to the number of adults returning to the River Imsa only ($r^2 = 0{\cdot}38$, d.f. = 17, $P < 0{\cdot}01$; Fig. 1e). Furthermore, there was a significant relationship between numbers of multi-sea-winter salmon and one-sea-winter salmon caught in rivers and at sea from the same smolt cohorts (Fig. 2). In the earliest years of the study, from 1976 to 1981, the numbers of one- and multi-sea-winter fish were higher than in the later years, indicating a much larger population size at the beginning of the sampling period.

In the River Imsa, the loss-rates from eggs to smolts and from eggs to adults appeared to be density dependent, as the loss-rates increased with increasing egg density (Fig. 3a & b). Furthermore, 73% of the variation in loss-rates was explained by the variation in egg density. At sea, however, the loss-rate was density independent as there was no significant difference in loss-rate with changing density (Fig. 3c: $r^2 = 0{\cdot}15$, $P > 0{\cdot}05$ and Fig. 3d: $r^2 = 0{\cdot}12$, $P > 0{\cdot}05$). A graphic illustration of the key factors from eggs to smolts and from smolts to adults as well as the total *K*, shows that the pattern in total loss-rates resembled that from eggs to smolts more than that of smolts to adults (Fig. 4). This indicates that freshwater survival was the main factor influencing the abundance of returning adults.

BIOMASS AND ENERGY

The relationships between smolt biomass (P_{mass}, kg 10 000 m^{-2}) and number of eggs spawned (*E*) in the River Imsa is asymptotic (Fig. 5a). The relationship between energy of smolts (P_{energy}, kJ 10 000 m^{-2}) and number of eggs can be given by a similar model:

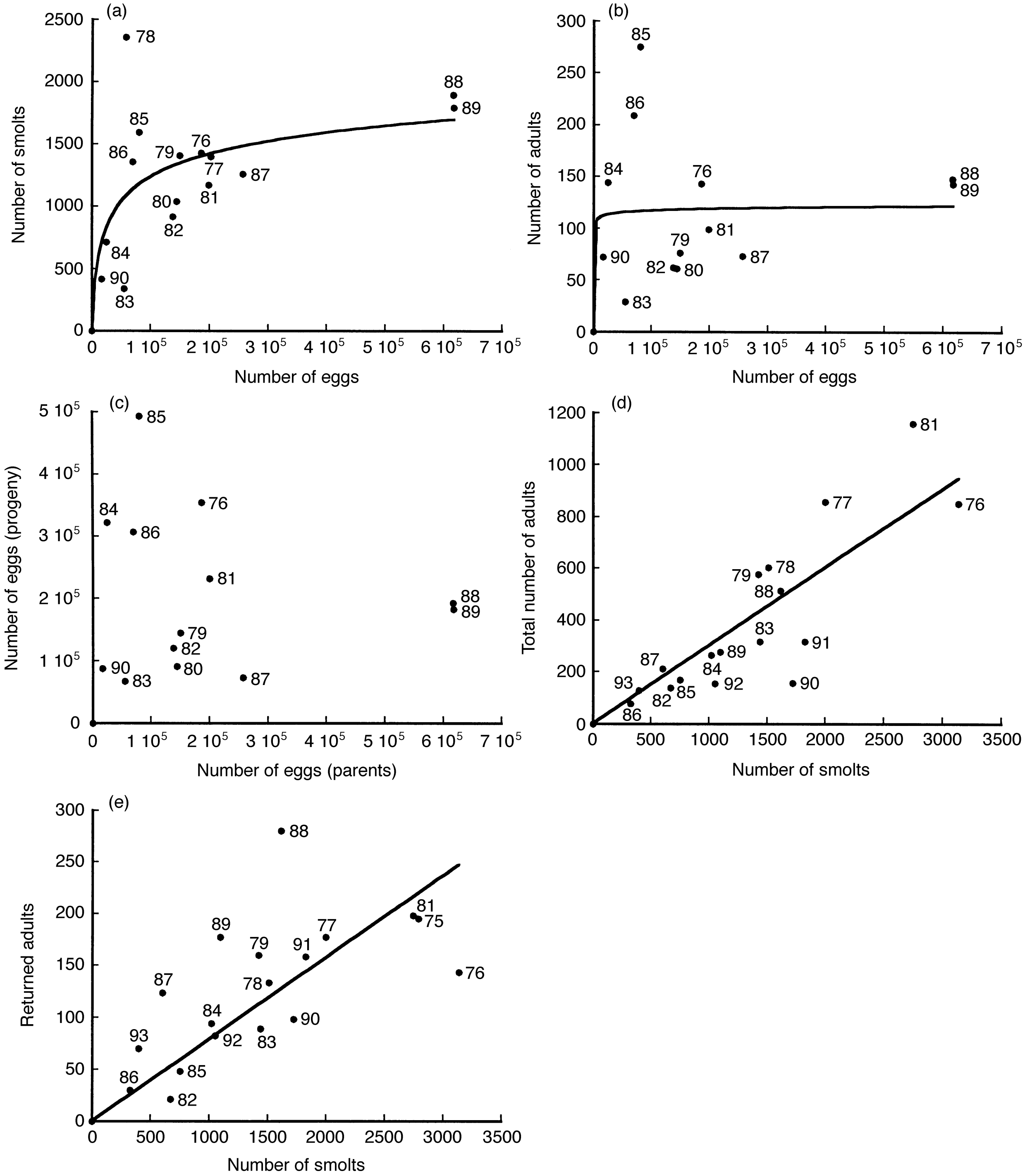

Fig. 1. Relationship, with ± standard error of the parameters in parentheses, between (a) number of eggs in the river (E) and number of smolts produced (S) ($S = 0{\cdot}11\ (\pm 0{\cdot}28)E/\{1 + [E/9129{\cdot}3\ (\pm 35\,131)]^{0{\cdot}88(\pm 0{\cdot}24)}\}$, d.f. = 12, $r^2 = 0{\cdot}49$, $P < 0{\cdot}01$), (b) number of eggs in the river (E) and number of adults produced (A) ($A = 0{\cdot}62\ (\pm 71{\cdot}7)\ E/\{1 + [E/166{\cdot}38\ (\pm 20\,036)]^{0{\cdot}98(\pm 0{\cdot}33)}\}$, d.f. = 10, $r^2 = 0{\cdot}30$, $P = 0{\cdot}05$), (c) number of eggs from the parent stock and number of eggs produced by the offspring year class, (d) number of smolts leaving the river (S) and estimated number of adults (A_1) caught at sea and in freshwater ($A_1 = 0{\cdot}30\ (\pm 0{\cdot}03)S$, d.f. = 16, $r^2 = 0{\cdot}73$, $P < 0{\cdot}01$), (e) number of smolts leaving the river (S) and estimated number of returning adults (A_2) to the River Imsa ($A_2 = 0{\cdot}078\ (\pm 0{\cdot}008)S$, d.f. = 17, $r^2 = 0{\cdot}38$, $P < 0{\cdot}01$). The numbers on parts (a), (b) and (c) refer to the year the eggs were spawned and on (d) and (e) to the year of smolt descent.

$P_{\text{energy}} = 12{\cdot}94\ (\pm 17{\cdot}20)\ E/\{1 + [E/22\,351\ (\pm 41\,638)]^{1{\cdot}01(\pm 0{\cdot}26)}\}$, d.f. = 12, $r^2 = 0{\cdot}45$, $P < 0{\cdot}01$). The ± standard error of the parameters is given in parentheses. The increase in mass was highest at low egg densities and started to level off at densities of *c.* 60 000 eggs per 10 000 m² river area. The yearly biomass of smolts in the River Imsa ranged in mass from 13 kg per 10 000 m² (in 1983) to 88 kg per 10 000 m² (in 1978), in energy equivalent to between 66 000 kJ per 10 000 m² and 431 000 kJ per 10 000 m².

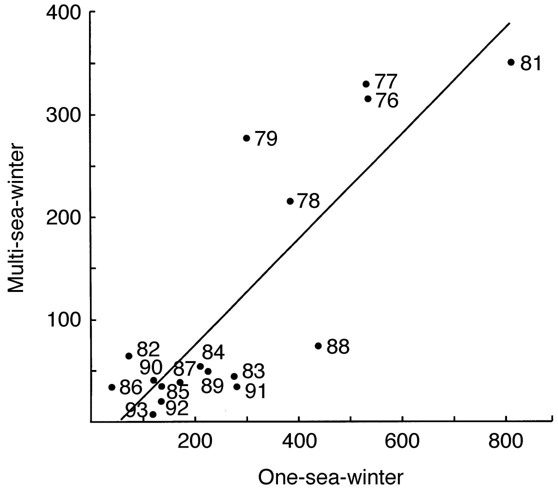

Fig. 2. Relationship between number of one- (G) and multi-sea-winter (M) salmon caught at sea and in freshwater, produced from the same smolt-year-class ($M = 0{\cdot}52G - 27{\cdot}59$, d.f. = 15, $r^2 = 0{\cdot}71$, $P < 0{\cdot}01$). The figure given at each point is the year of the smolt migration to sea.

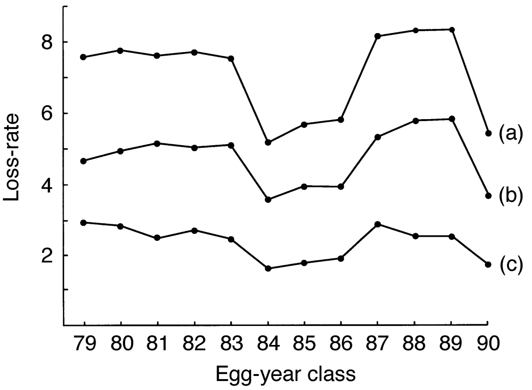

Fig. 4. (a) Total loss-rate, (b) loss-rate in freshwater and (c) loss-rate at sea in each egg year class of Atlantic salmon.

The number of eggs produced in 1978 gave higher biomass estimates than all other egg year-classes studied. When the 1978 figure was omitted from the regression, the coefficient of determination increased and the correlations were: $P_{\mathrm{mass}} = (0{\cdot}00192\ E)/[1 +$

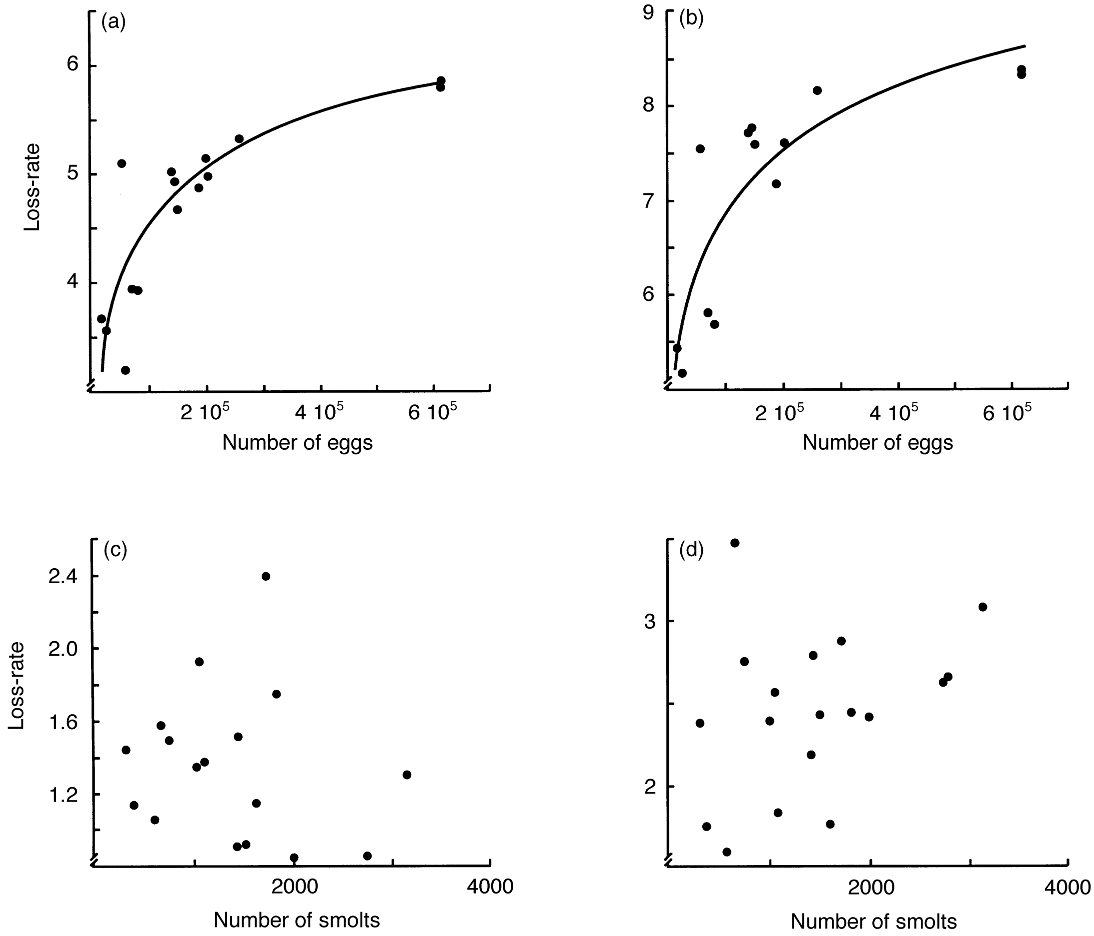

Fig. 3. Relationship between (a) number of eggs (S_e) and loss-rate of the smolts ($\ln S_e/\ln R_s$), where R_s is number of smolts; $\ln (S_e/R_s) = 0{\cdot}67 \ln S_e - 3{\cdot}22$, d.f. = 13, $r^2 = 0{\cdot}73$, $P < 0{\cdot}001$, (b) number of eggs (S_e) and the loss-rate of the parent stock [$\ln S_e/\ln R_a$ (adults)]; $\ln (S_e/R_a) = 0{\cdot}92 \ln S_e - 3{\cdot}62$, d.f. = 11, $r^2 = 0{\cdot}73$, $P < 0{\cdot}001$, (c) number of smolts and loss-rate of Atlantic salmon until fished in the ocean and (d) number of smolts and loss-rate of the Atlantic salmon until ascending the River Imsa as adults (fishing including).

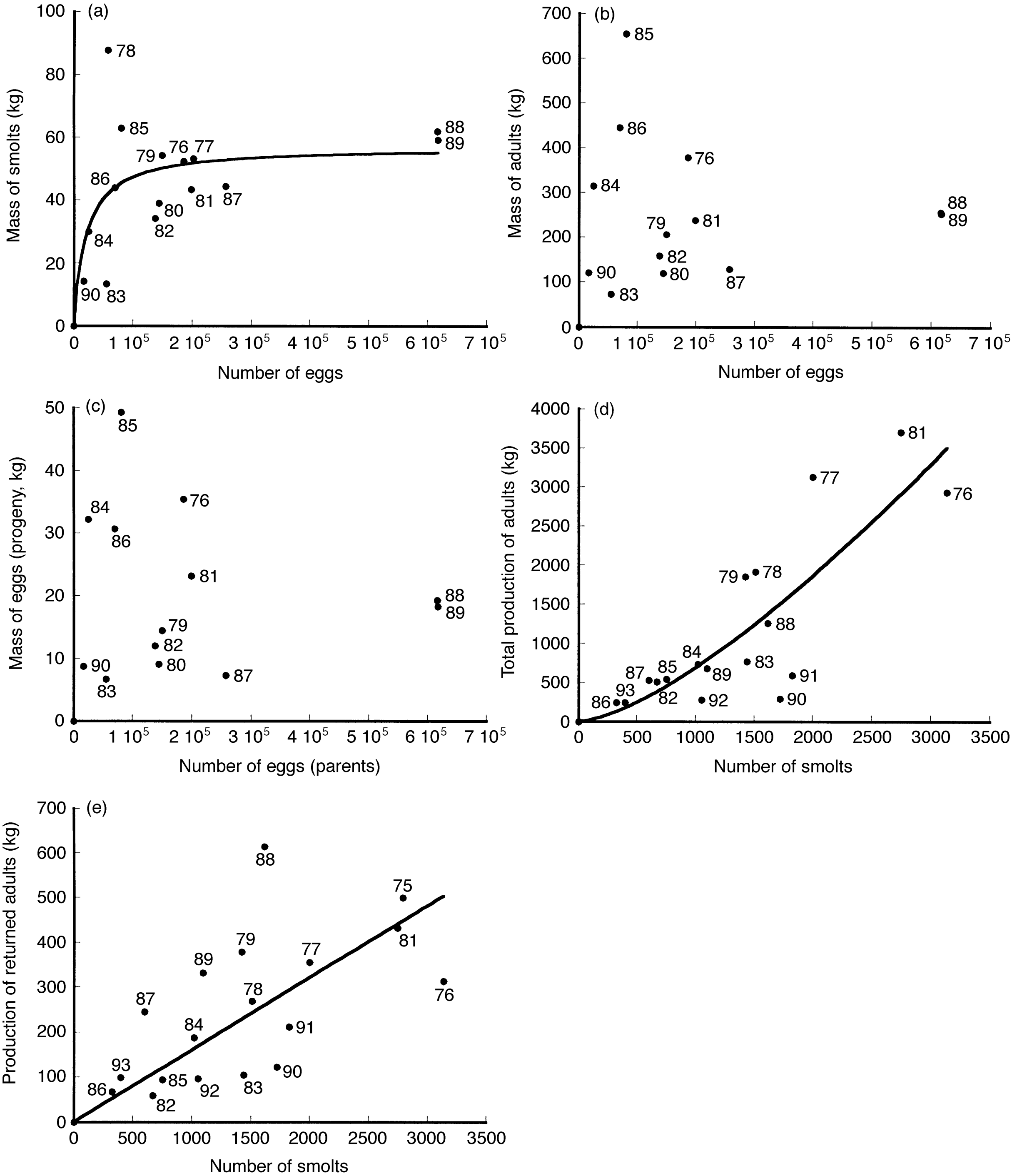

Fig. 5. Relationship, with ± standard error of the parameters in parentheses, between number of eggs (E) in the river and (a) wet mass of smolts produced (P_{mass}, kg 10 000 m^{-2}) ($P_{\mathrm{mass}} = 0{\cdot}003\ (\pm 0{\cdot}004)\ E/\{1 + [E/21\,385\ (\pm 43\,977)]^{1{\cdot}00(\pm 0{\cdot}25)}\}$, d.f. = 12, $r^2 = 0{\cdot}46$ $P < 0{\cdot}01$), (b) mass of the adults produced, (c) mass of the eggs produced, (d) relationship between number of smolts (S) and estimated total wet mass of adults ($A_{1\,\mathrm{mass}}$) ($A_{1\,\mathrm{mass}} = 3{\cdot}30\ (\pm 46{\cdot}39)\ S/\{1 + [S/9960\ (\pm 419\,620)]^{-0{\cdot}59(\pm 3{\cdot}14)}\}$, d.f. = 14, $r^2 = 0{\cdot}69$, $P < 0{\cdot}01$), and (e) between number of smolts (S) and estimated wet mass of returned adults ($A_{2\,\mathrm{mass}}$) to the River Imsa ($A_{2\,\mathrm{mass}} = 0{\cdot}16\ (\pm 0{\cdot}02)\ S$, d.f. = 17, $r^2 = 0{\cdot}43$, $P < 0{\cdot}01$). The numbers on parts (a), (b) and (c) refer to the year the eggs were spawned, and on (d) and (e) to the year of smolt descent.

$(E/24\,533)^{0{\cdot}91}]$, $r^2 = 0{\cdot}71$, d.f. = 11, $P < 0{\cdot}01$ and $P_{\mathrm{energy}} = (9{\cdot}38\ \ E)/[1 + (E/25\,153)^{0{\cdot}92}]$, $r^2 = 0{\cdot}69$, d.f. = 11, $P < 0{\cdot}05$. The variation in egg density accounted for almost 70% of the variation in smolt production in mass and energy, when the 1978 egg year-class was omitted, or *c.* 25% higher than when the 1978 egg-year class was included.

The relationship between wet mass (kg per 10 000 m^2; Fig. 5b) and energy of adults (kJ per 10 000 m^2) and the amount of eggs at the start of the year-class was not significantly correlated due to high variability among years. A dome-shaped model gave the best fit with a coefficient of determination as low as 0·23 for both wet mass and energy, but none of the relationships were significant ($P > 0{\cdot}05$). The biomass of adults ranged from 73 kg per 10 000 m^2 (in 1983) to 655 kg per 10 000 m^2 (in 1985) and in energy from 370 000 kJ per 10 000 m^2 to 3270 000 kJ per 10 000 m^2. Furthermore, the relationship between the number of eggs of the parental population and the eggs of the offspring was not significantly correlated (Fig. 5c). The same holds true if the eggs were measured in energy (kJ). In both cases a non-significant dome-shaped curve gave the best fit ($r^2 = 0{\cdot}22$, $P > 0{\cdot}05$). The egg production in mass ranged between 6·7 kg in 1983 and 49 kg in 1985. The corresponding values in energy were 50 647 kJ and 373 885 kJ.

The total adult biomass (adults caught at sea, in the River Imsa and in other rivers) and the returning adults to the River Imsa were significantly correlated with the size of the smolt cohort from which they originated (Fig. 5d & e). When the biomass was measured in energy (kJ) the relationship between total adult mass ($A_{1\,\text{energy}}$) and number smolt (S) was: $A_{1\,\text{energy}} = 63{\cdot}96\ (\pm 154{\cdot}36)S^{1{\cdot}56(\pm 0{\cdot}31)}$, d.f. = 15, $r^2 = 0{\cdot}70$, $P < 0{\cdot}01$. The energy of the adults returning to the River Imsa ($A_{2\,\text{energy}}$) in relation to number of smolts was: $A_{2\,\text{energy}} = 819{\cdot}61\ (\pm 92{\cdot}59)S$, d.f. = 17, $r^2 = 0{\cdot}46$, $P < 0{\cdot}01$. Standard errors are given in parentheses. The yearly total biomass ranged from 240 kg per 10 000 m^2 in 1993 to 3711 kg per 10 000 m^2 in 1981, when the number of smolts ranged from 397 to 2751. In energy the total production ranged from 1133 961 kJ in 1993 to 19 711492 kJ in 1981. The corresponding values for the adult biomass 10 000 m^{-2} in the River Imsa were 59 kg in 1982 and 614 kg in 1988, with smolt numbers between 672 and 1621. The relationship between adult biomass and smolt density seemed to be density-independent.

Discussion

Density-dependent survival appears to determine the number and biomass of Atlantic salmon smolts leaving the River Imsa. Thus, there seems to be a carrying capacity limiting the population size in fresh water. In the North Atlantic Ocean, on the other hand, density-independent factors were important. In this habitat, the population size of salmon is small compared with many other pelagic fish species. Therefore, freshwater survival influenced the number of adult Atlantic salmon returning to the river. This can be seen both from the shape of the stock–recruitment curve and the fact that the loss rate in fresh water, but not at sea, increased with egg density. The stock–recruitment curve of juveniles in fresh water, which was close to that described by Cushing (1973), increased rapidly at low egg densities and started to level off at egg densities of approximately 60 000 eggs 10 000 m^{-2}, or 6 eggs m^{-2}. This shape of the recruitment curve is similar to those found for Atlantic salmon by Buck & Hay (1984) in Scotland, Chadwick (1985) in Newfoundland and Kennedy & Crozier (1993, 1995) from Ireland. The recruitment curve of North Sea herring is also similar (Rothschild 1986). In brown trout (Elliott 1994) and several Pacific salmon (Ricker 1954, 1975, 1989), the alternative density-dependent 'Ricker-curve' describes the population survival better. Gardiner & Shackley (1991) also fitted a dome-shaped stock–recruitment relationship for juvenile Atlantic salmon in a Scottish stream, and a similar relationship was found by Gee, Milner & Hemsworth (1978) for Atlantic salmon in the River Wye. The reason why the stock–recruitment relationship in these cases differed from the present relationship and those fitted by Buck & Hay (1984), Chadwick (1985) and Kennedy & Crozier (1993, 1995) is unknown, but may indicate that the factors regulating population size vary among systems.

The asymptotic stock–recruitment relationship is the appropriate curve if there is a maximum abundance of the population imposed by food availability or space, or if a predator can adjust its predatory activity immediately to changes in its prey abundance (Ricker 1975). The dome-shaped stock–recruitment relationship is the proper model when the cause of the density dependence is cannibalism of the young by adults, or an increase in the time it takes for the young to grow through a vulnerable size range, or when there is a time-lag in the response of a predator or parasite to the abundance of the fish being attacked. In the River Imsa, the smolt abundance is probably imposed by food availability and space limitations.

The exact optimal egg deposition in the River Imsa could not be determined from the Shepherd (1982) model used (although it did not increase much between 6 and 60 eggs m^{-2} river area); it appears to be higher than that in other Atlantic salmon streams investigated. In tributaries to the Miramichi River and the Pollett River in Canada, Elson (1975) estimated that an egg deposition of 2·4 eggs m^{-2} gave optimal smolt production in suitable rearing habitats. Chadwick (1985), on the other hand, maintained that this value was too low, in spite of its wide use in eastern Canada. In the small Girnock Burn, Scotland, Buck & Hay (1984) counted the number of upstream migrating spawners and downstream descending smolts. Their estimate of optimal egg density was 3·4 eggs m^{-2}. As in the present case, both Chadwick (1985) and Buck & Hay (1984) gave asymptotic stock–recruitment relationships. Optimal egg deposition of anadromous brown trout in the Black Browse Beck in the Lake District, England, on the other hand, is one order of magnitude higher than that of Atlantic salmon in the River Imsa. The reason for this difference is unknown.

In general, causes of density-dependent mortality in fish populations are generally unknown, except that this is an interaction between food supplies and predators affecting early larval stages. High densities lead to competition and lack of food, which causes slow growth and mortality, either directly or indirectly through increased vulnerability to predators (Sinclair 1989). Cushing (1981) found that the degree of regulation acting on a population was directly related to the fecundity of marine fishes. Species with high fecundity experienced strong regulation and species with low fecundity showed weak regulation. Salmonids, with their large eggs, should consequently show weak regulation. Nevertheless, Elliott's (1994) study of young brown trout in the Black Browse Beck is one of the best examples of population regulation in vertebrate species.

In the ocean, density-independent factors seemed important for the survival of the fish, in accordance with our hypothesis presented in the introduction. The number of adults increased recti-linearly with annual smolt number: the higher the average smolt output, the higher the average number of returning spawners. This indicates that the population density is far below the carrying capacity for Atlantic salmon in the North Atlantic. This agrees with the observation that individual growth rate and asymptotic size are much higher at sea than in fresh water. When the population size approaches the carrying capacity of the habitat, individual growth rate and asymptotic fish size should be small due to the plastic growth performance of fishes (Wootton 1990).

A low salmon abundance, relative to the carrying capacity for the species, might be judged as if the ocean is a hostile habitat for salmon (Haldane 1956). The high marine mortality rate gives support to this assumption. However, low recruitment of smolts relative to the carrying capacity of the system would lead to the same results, and the natural smolt output today is certainly much lower than it used to be in historical times. We feel that a combination of the two is limiting the abundance of adults returning to rivers for spawning, and that the abundance of adults would increase given a higher recruitment rate. In the present study, the high tag mortality (60%) and the killing of 10% of the smolts for age determination are important mortality factors. These mortalities were, however, controlled for in the present estimates. General reasons restricting the recruitment of smolts to the North Atlantic are river regulation and dam building that constrain salmon migration into rivers, pollution and acidification of rivers and fish diseases like gyrodactylosis reducing the survival of salmon and young fish in particular (Johnsen & Jensen 1986; Hesthagen & Hansen 1991). A factor working in the opposite direction is the constant output of farmed salmon into the ocean (Hansen, Jacobsen & Lund 1993; Hansen, Reddin & Lund 1997). At present between 25 and 40% of the Atlantic salmon in the oceanic feeding areas in the north Norwegian Sea are of farmed origin. Whether or not the escape of farmed salmon in the long run will increase salmon abundance in the ocean is, however, highly questionable (Hindar, Ryman & Utter 1991; Jonsson 1997).

Density-independent factors are widely believed to be important contributors to variations in population abundance of marine fishes and Peterman (1981), for example, gave evidence for density-independent marine survival in the Oregon coho salmon *Oncorhynchus kisutch*, and ocean climate may be the prime determinant of the annual changes in mortality (cf. Frank & Leggett 1994). The productivity of sockeye salmon *Oncorhynchus nerka* in Bristol Bay appears to be strongly related to fluctuations in climate (Adkinson *et al.* 1996), and the major change in climate over the Pacific Ocean in the winter of 1976–77 seemed to result in a change in productivity of the Fraser River sockeye salmon (Hare & Francis 1995; Beamish, Neville & Cass 1997). For Atlantic salmon, surface water temperature in the North Atlantic Ocean appears to be a promising candidate for the explanation of year class variation in abundance of adult Atlantic salmon in Europe (Friedland, Hansen & Dunkley 1998).

In contrast to large mammals, adult density dependence in population regulation occurs seldom in fishes (Cushing 1988, 1996; Sinclair 1989; Shepherd & Cushing 1990; Bradford 1992; Elliott 1994), and there was no evidence for it in the present investigation. If such a situation occurs, there should be resource limitation for adults but not for younger individuals (Shuter 1990). This may occur in densely populated stocks because the food requirements differ between adults and juveniles (larger particle sizes and amounts), due to their larger body size as found for Arctic charr by Forseth, Ugedal & Jonsson (1994). Hamrin & Persson (1986) proposed that this asymmetric competition was the prime mechanism behind fluctuations in year class abundance of vendace *Coregonus albula* in Scandinavian lakes (but see Sandlund *et al.* 1991 for an alternative explanation). In resident brown trout, Elliott & Hurley (1998) found that the number of spawning females produced in each year class was strongly density-dependent on the initial number of females that laid eggs at the start of the year class, and is probably the first clear evidence for fish population regulation in the adult, rather than the juvenile stage. Obviously, there is no similar situation in Atlantic salmon feeding in the North Atlantic Ocean.

One might have expected positive density dependence of salmon at sea. Sea survival might increase with number of recruits due to improved navigational accuracy (Quinn & Fresh 1984) or increased survival due to a possible functional response of the potential predators (Wood & Hand 1985). There is, however, no support for such effects within the present variation in population sizes tested. On the other hand, there was a slight positive increase in total biomass of adults in years with high rather than low smolt numbers indicating a positive effect of smolt abundance on fish size in the ocean. A similar effect was not found for the fish returning to the River Imsa (Fig. 5).

The fit of a stock–recruitment model to the data provides strong evidence for density-dependent regulation at some stage in the life cycle, but does not indicate at which stage or stages, and it provides no explanation of the mechanism by which density-dependent survival occurs. Here population regulation occurred in fresh water but not in the ocean.

Acknowledgements

We are indebted to the staff of the Norwegian Institute for Nature Research, Research Station at Ims for emptying the fish traps every day all year round in all kinds of weather, to Torbjørn Forseth for helping with the statistical treatment of the data and to J. Malcolm Elliott and Neil Metcalfe for excellent criticism of an earlier draft of the manuscript.

References

Adkinson, M.D., Peterman, R.M., Lapointe, M.F., Gillis, D.M. & Korman, J. (1996) Alternative models of climatic effects on sockeye salmon, *Oncorhynchus nerka*, productivity in Bristol Bay, Alaska, and the Fraser River, British Columbia. *Fisheries Oceanography*, **5**, 137–152.

Anonymous (1978.) *Nordic Committee on Food Analysis (NMKL)*, **93**, 1–3.

Anonymous (1981) Determination of Kjeldahl nitrogen content with Kjeltec auto systems 1, 2, 3 and 4. *Tecator Application Note AN*, **30**, 1–4.

Anonymous (1987) Determination of total crude fat in plant materials and feeding stuffs with the Soxtec Hydrolyzing system. *Tecator Application Note AN*, **92**, 1–6.

Beamish, R.J., Neville, C.-E.M. & Cass, A.J. (1997) Production of Fraser River sockeye salmon (*Oncorhynchus nerka*) in relation to decadal-scale changes in the climate and the ocean. *Canadian Journal of Fisheries and Aquatic Sciences*, **54**, 543–554.

Bellows, T.S. (1981) The descriptive properties of some models for density dependence. *Journal of Animal Ecology*, **50**, 139–156.

Beverton, R.J.H. & Holt, S.J. (1957) On the dynamics of exploited fish populations. *Fishery Investigations, London Series 2*, **19**, 1–533.

Bradford, M.J. (1992) Precision of recruitment predictions from early life stages of marine fishes. *Fishery Bulletin, U.S.*, **90**, 439–453.

Buck, R.J.G. & Hay, D.W. (1984) The relation between stock size and progeny of Atlantic salmon, *Salmo salar* L., in a Scottish stream. *Journal of Fish Biology*, **24**, 1–11.

Carlin, B. (1955) Tagging of salmon smolts in the river Lagan. *Institute of Freshwater Research Drottningholm, Report*, **36**, 57–74.

Chadwick, E.M.P. (1985) The influence of spawning stock on production and yield of Atlantic salmon, *Salmo salar* L., in Canadian rivers. *Aquaculture and Fisheries Management*, **1**, 111–119.

Chapman, D.W. (1978) Production in fish populations. *Ecology of Freshwater Fish Production* (ed. S.D. Gerking), pp. 5–25. Blackwell Scientific Publications, Oxford.

Charnov, E.L. (1986) Life history evolution in a 'recruitment population': why are adult mortality rate constant? *Oikos*, **47**, 129–134.

Craig, J.F. (1980) Growth and production of the 1955–72 cohorts of perch, *Perca fluviatilis* L., in Windermere. *Journal of Animal Ecology*, **49**, 291–315.

Craig, J.F. (1987) *The Biology of the Perch and Related Fish*. Broom Helm, London.

Craig, J.F., Kenley, M.J. & Talling, J.F. (1978) Comparative estimations of the energy content of fish tissue from bomb calorimetry, wet oxidation and proximate analysis. *Freshwater Biology*, **8**, 585–590.

Crozier, W.W. & Kennedy, G.J.A. (1995) The relationship between a summer fry (0+) abundance index, derived from semi-quantitative electrofishing, and egg deposition of Atlantic salmon, in the River Bush, Northern Ireland. *Journal of Fish Biology*, **47**, 1055–1062.

Cushing, D.H. (1973) The dependence of recruitment on parent stock. *Journal of the Fisheries Research Board of Canada*, **30**, 1965–1976.

Cushing, D.H. (1981) *Fisheries Biology. A Study in Population Dynamics*. University of Wisconsin Press, Madison.

Cushing, D.H. (1988) The study of stock and recruitment. *Fish Population Dynamics*, 2nd edn (ed. J.A. Gulland), pp. 105–128. John Wiley and Sons, Chichester.

Cushing, D.H. (1996) *Towards a Science of Recruitment in Fish Populations*. Ecology Institute, Oldendorf.

Cushing, D.H. & Harris, J.G.K. (1973) Stock and recruitment and the problem of density dependence. *Rapports et procès-verbaux des réunions. Conseil international poor l'exploration de la mer*, **164**, 142–155.

Dempster, J.P. (1975) *Animal Population Ecology*. Academic Press, London.

Elliott, J.M. (1985) The choice of a stock–recruitment model for migratory trout, *Salmo trutta*, in an English Lake District stream. *Archiv für Hydrobiologie*, **104**, 145–168.

Elliott, J.M. (1989) The natural regulation of numbers and growth in contrasting populations of brown trout, *Salmo trutta*, in two Lake District streams. *Freshwater Biology*, **21**, 7–19.

Elliott, J.M. (1994) *Quantitative Ecology and the Brown Trout*. Oxford University Press, Oxford.

Elliott, J.M. & Hurley, M.A. (1998) Population regulation in adult, but not juvenile, resident trout (*Salmo trutta*) in a Lake District stream. *Journal of Animal Ecology*, **67**, 280–286.

Elson, P.F. (1975) Atlantic salmon rivers, smolt production and optimal spawning: an overview of natural production. *International Atlantic Salmon Foundation Special Publication Series*, **6**, 96–119.

Elson, P.F. & Tuomi, A.L.W. (1975) *The Foyle Fisheries: New Bases for Rational Management*. Special Report to the Foyle Fisheries Commission, Londonderry, Northern Ireland.

Fleming, I.A., Lamberg, A. & Jonsson, B. (1997) Effects of early experience on the reproductive performance of Atlantic salmon. *Behavioral Ecology*, **8**, 470–480.

Fogarty, M.J., Sissenwine, M.P. & Cohen, E.B. (1991) Recruitment variability and the dynamics of exploited marine populations. *Trends in Ecology and Evolution*, **6**, 241–246.

Forseth, T., Ugedal, O. & Jonsson, B. (1994) The energy budget, niche shift, reproduction and growth in a population of Arctic charr, *Salvelinus alpinus*. *Journal of Animal Ecology*, **63**, 116–126.

Frank, K.T. & Leggett, W.C. (1994) Fisheries ecology in the context of ecological and evolutionary theory. *Annual Review of Ecology and Systematics*, **25**, 401–22.

Friedland, K.D., Hansen, L.P. & Dunkley, D.A. (1998) Marine temperatures experienced by postsmolts and the survival of Atlantic salmon, *Salmo salar* L., in the North Sea area. *Fisheries Oceanography*, **7**, 22–34.

Fryer, G. & Iles, T.D. (1972) *The Cichlid Fishes of the Great Lakes of Africa*. Oliver & Boyd, Edinburgh.

Gardiner, R. & Shackley, P. (1991) Stock and recruitment and inversely density-dependent growth of salmon, *Salmo salar* L., in a Scottish stream. *Journal of Fish Biology*, **38**, 691–696.

Gee, A.S., Milner, N.J. & Hemsworth, R.J. (1978) The production of juvenile Atlantic salmon, *Salmo salar,* in the Upper Wye, Wales. *Journal of Fish Biology*, **13**, 439–451.

Haldane, J.B.S. (1956) The relation between density regulation and natural selection. *Proceedings of The Royal Society B*, **145,** 306–308.

Hamrin, S. & Persson, L. (1986) Asymmetrical competition between age classes as a factor causing population oscillations in an obligate planktivorous fish species. *Oikos*, **47,** 223–232.

Hansen, L.P. (1988) Effect of Carlin tagging and fin clipping on survival of Atlantic salmon (*Salmo salar* L.) released as smolts. *Aquaculture*, **70,** 391–394.

Hansen, L.P., Jacobsen, J.A. & Lund, R.A. (1993) High numbers of farmed Atlantic salmon, *Salmo salar* L., observed in oceanic waters north of the Faroe Islands. *Aquaculture and Fisheries Management*, **24,** 777–781.

Hansen, L.P. & Jonsson, B. (1989) Salmon ranching experiments in the River Imsa: returns of different stocks to the fishery and to River Imsa. *Aquaculture – A Biotechnology in Progress* (eds N. De Pauw, E. Jaspers, H. Ackefors, N. Wilkins), pp. 445–452. European Aquaculture Society, Bredene, Belgium.

Hansen, L.P., Reddin, D.G. & Lund, R.A. (1997) The incidence of reared Atlantic salmon (*Salmo salar* L.) of fish farm origin at West Greenland. *ICES Journal of Marine Science*, **54,** 152–155.

Hare, S.R. & Francis, R.C. (1995) Climate change and salmon production in the northeast Pacific Ocean. *Climate Change and Northern Fish Populations* (ed. R.J. Beamish), pp. 357–372. Canadian Special Publication of Fisheries and Aquatic Sciences 121, National Research Council of Canada, Ottawa.

Hesthagen, T. & Hansen, L.P. (1991) Estimates of the annual loss of Atlantic salmon, *Salmo salar* L., in Norway due to acidification. *Aquaculture and Fisheries Management*, **22,** 85–91.

Hilborn, R. & Walters, C.J. (1992) *Quantitative Fisheries Stock Assessment*. Chapman & Hall, New York.

Hindar, K., Ryman, N. & Utter, F. (1991) Genetic effects of cultured fish on natural fish populations. *Canadian Journal of Fisheries and Aquatic Sciences*, **48,** 945–957.

Johnsen, B.O. & Jensen, A.J. (1986) Infestations of Atlantic salmon, *Salmo salar*, by *Gyrodactylus salaris* in Norwegian rivers. *Journal of Fish Biology*, **29,** 233–241.

Jonsson, B. (1976) Comparison of scales and otoliths for age determination in brown trout, *Salmo trutta* L. *Norwegian Journal of Zoology*, **24,** 295–301.

Jonsson, B. (1997) A review of ecological and behavioural interactions between cultured and wild Atlantic salmon. *ICES Journal of Marine Science*, **54,** 1031–1039.

Jonsson, B. & Jonsson, N. (1993) Partial migration: niche shift versus sexual maturation in fishes. *Reviews in Fish Biology and Fisheries*, **3,** 348–365.

Jonsson, B., Jonsson, N. & Hansen, L.P. (1991a) Differences in life history and migratory behaviour between wild and hatchery-reared Atlantic salmon in nature. *Aquaculture*, **98,** 69–78.

Jonsson, B., Jonsson, N. & Hansen, L.P. (1991b) Variation in age, size and repeat spawning of adult Atlantic salmon in relation to river discharge. *Journal of Animal Ecology*, **60,** 937–947.

Jonsson, B., Jonsson, N. & Ruud-Hansen, J. (1989) Downstream displacement and life history variables of Arctic charr (*Salvelinus alpinus*) in a Norwegian river. *Physiology and Ecology Japan, Special Volume*, **1,** 93–105.

Jonsson, N., Jonsson, B. & Fleming, I.A. (1996) Does early growth cause a phenotypically plastic response in egg production of Atlantic salmon? *Functional Ecology*, **10,** 89–96.

Jonsson, N., Jonsson, B. & Hansen, L.P. (1997) Changes in proximate composition and estimates of energetic costs during upstream migration and spawning in Atlantic salmon *Salmo salar*. *Journal of Animal Ecology*, **66,** 425–436.

Kennedy, G.J.A. & Crozier, W.W. (1993) Juvenile Atlantic salmon (*Salmo salar*) – production and prediction. *Canadian Special Publication of Fisheries and Aquatic Sciences*, **118,** 179–187.

Kennedy, G.J.A. & Crozier, W.W. (1995) Factors affecting recruitment success in salmonids. *The Ecological Basis for River Management* (eds D.M Harper & A.J.D. Ferguson), pp. 349–362. John Wiley, Chichester.

Mann, R.H.K. & Penczak, T. (1986) Fish production in rivers: a review. *Polskie Archivum Hydrobiologii*, **33,** 233–247.

Mason, M.J. (1983) Determination of glucose, sucrose, lactose and ethanol in foods and beverages, using immobilized enzyme electrodes. *Journal of the Association of Official Analytical Chemists*, **66,** 981–984.

May, R.M. (1981) Models for single populations. *Theoretical Ecology: Principles and Applications* (ed. R.M. May), pp. 5–29. Blackwell Scientific Publications, Oxford.

Metcalfe, N.B. & Thorpe, J.E. (1990) Determinants of geographical variation in the age of seaward-migrating salmon, *Salmo salar*. *Journal of Animal Ecology*, **59,** 135–145.

Moran, N.A. (1994) Adaptation and constraint in the complex life cycles of animals. *Annual Reviews of Ecology and Systematics*, **25,** 573–600.

Murray, B.G. (1982) On the meaning of density dependence. *Oecologia*, **53,** 370–373.

Newton, I. & Marquiss, M. (1986) Population regulation in sparrowhawks. *Journal of Animal Ecology*, **55,** 463–480.

Peterman, R.M. (1980) Dynamics of native Indian food fisheries on salmon in British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences*, **37,** 561–566.

Peterman, R.M. (1981) Form of random variation in salmon smolt-to-adult relations and its influence on production estimates. *Canadian Journal of Fisheries and Aquatic Sciences*, **38,** 1113–1119.

Quinn, T.P. & Fresh, K. (1984) Homing and straying in chinook salmon (*Oncorhynchus tshawytscha*) from Cowlitz River Hatchery, Washington. *Canadian Journal of Fisheries and Aquatic Sciences*, **41,** 1078–1082.

Ricker, W.E. (1954) Stock and recruitment. *Journal of the Fisheries Research Board of Canada*, **11,** 559–623.

Ricker, W.E. (1975) Computation and interpretation of biological statistics of fish populations. *Bulletin of the Fisheries Research Board of Canada*, **191,** 1–382.

Ricker, W.E. (1979) Growth rates and models. *Fish Physiology*, Vol. 8 (eds W.S. Hoar, D.J. Randall & J.R. Brett), pp. 677–743. Academic Press, New York.

Ricker, W.E. (1989) History and present state of the odd-year pink salmon runs of the Fraser River region. *Canadian Technical Report, Fisheries and Aquatic Sciences*, **1702,** 1–37.

Rothschild, B.J. (1986) *Dynamics of Marine Fish Populations*. Harvard University Press, Cambridge, Mass.

Rounsefell, G.A. (1958) Factors causing decline in sockeye salmon of Karluk River, Alaska. *U.S. Fish and Wildlife Service, Fishery Bulletin*, **58,** 83–169.

Sandlund, O.T., Jonsson, B., Næsje, T.F. & Aass, P. (1991) Year-class fluctuations in vendace, *Coregonus albula* (Linnaeus): Who's got the upper hand in intraspecific competition? *Journal of Fish Biology*, **38,** 873–885.

Shepard, M.P. & Withler, F.C. (1958) Spawning stock size and resultant production for Skeena sockeye. *Journal of Fisheries Research Board of Canada*, **15,** 1007–1025.

Shepherd, J.G. (1982) A versatile new stock–recruitment relationship for fisheries, and the construction of sustainable yield curves. *Journal de Conseil International Exploration de la Mer*, **40,** 67–75.

Shepherd, J.G. & Cushing, D.H. (1990) Regulation in fish populations: myth or mirage? *Philosophical Transactions of the Royal Society London, Series B*, **330,** 151–164.

Shuter, B.J. (1990) Population-level indicators of stress. *Biological Indicators of Stress in Fish* (ed. S.M. Adams), pp. 145–166. American Fisheries Society, Bethesda.

Sinclair, A.E.G. (1989) The regulation of animal populations. *Ecological Concepts* (ed. M. Cherrett), pp. 197–241. *British Ecological Society Symposium*. Blackwell Scientific Publications, Oxford.

Varley, G.C., Gradwell, G.R. & Hassell, M.P. (1973) *Insect Population Ecology*. Blackwell Scientific Publications, Oxford.

Warner, R.R. & Chesson, P.L. (1985) Coexistence mediated by recruitment fluctuations: a field guide to the storage effect. *American Naturalist*, **125,** 769–787.

Werner, E.E. & Gilliam, J.F. (1984) The ontogenetic niche and species interactions in size-structured populations. *Annual Reviews of Ecology and Systematics*, **15,** 393–425.

Whittaker, J.B. (1971) Population changes in *Neophilaenus lineatus* (L.) (Homoptera: Cercopidae) in different parts of its range. *Journal of Animal Ecology*, **40,** 425–443.

Wolf, P. (1951) A trap for the capture of fish and other organisms moving downstream. *Transaction of the American Fisheries Society*, **80,** 41–45.

Wood, C.C. & Hand, C.M. (1985) Food-searching behaviour of the common merganser (*Mergus merganser*): Functional response to prey and predator density. *Canadian Journal of Zoology*, **63,** 1260–1270.

Wootton, R.J. (1990) *Ecology of Teleost Fishes*. Chapman & Hall, London.

Received 18 September 1997; revision received 22 December 1997

Brief History of Salmon Angling on the Miramichi

By Morris Green former Minister of the New Brunswick Department of Natural Resources

Millennia before Europeans arrived, Indigenous food fishers established permanent communities near the head of tide on the Southwest and Northwest Miramichi Rivers. Furthermore, their birch bark canoes took them to the river’s farthest branches where they found the best and most productive holding pools; so that they knew when and where to fish. The upper Main Southwest Miramichi was an area shared by the Maliseets of the Saint John River Valley and the Mi’kmaq of the lower Miramichi. Indigenous ancestral knowledge of fish and game cycles was essential to their survival.

The establishment of a few French fur traders at the forks of the Southwest and Northwest Miramichi rivers in the early 1600’s little impacted the indigenous food fishery, however, when English settlement followed the French defeat in the War of 1756-1763, the situation changed dramatically.

With their 100,000-acre government grant that included fishing and timber rights, William Davidson and his partner John Cort, soon threatened the traditional food fishery of indigenous people on the river. Focused on export of fish to British, Mediterranean, and Caribbean markets, the partners quickly diminished the spawning numbers returning to the two main Miramichi rivers and their tributaries.

The conflict between indigenous and non-indigenous fishers continues to the present day. Another point of conflict over the resource developed when the upper branches of the Miramichi rivers were settled by dispossessed Americans in the aftermath of the American Revolution in the late eighteenth century.

These pioneering, non-indigenous settlers soon established the same cyclical reliance on the Atlantic salmon as the First Nations. Only two years after their arrival, Ephraim Betts, unofficial leader of the Loyalist refugees who settled the upper Southwest Miramichi in 1795, petitioned the governor to exclude outsiders from exploiting their salmon. Pressure from Betts and others forced the colonial government to take some steps to protect the salmon population from overfishing.

One part of the government's approach to regulate the fishery was limiting seasons and issuing netting licences for salmon and gaspereau to upriver settlers. The 1861 New Brunswick census lists licence holders for the two species. After confederation, the federal government's Marine and Fisheries department assumed the role of issuing fishing licences. The Department of Fisheries and Oceans (DFO) continued issuing a few legacy gaspereau licences on the upper Southwest Miramichi until a few years ago. The tidal water salmon licences continued to be issued until 1972 when the federal Minister of DFO, Jack Davis, bought-out the commercial fishermen in Miramichi Bay and the river's tidal waters for a period of five years that was later made permanent.

The indigenous, commercial, and pioneer salmon food fishers were soon joined another group pursuing Atlantic salmon as a sport, putting more pressure on fish numbers.

By the **1800's**, British Army officers posted to New Brunswick to ward off potential attacks from Americans, having abundant time of their hands, looked for ways to amuse themselves. The officers were sons of wealthy landowners in England, Scotland, Wales, or Ireland who had grown up fishing and hunting on family estates. Landing in New Brunswick's virtually untouched wilderness was like a "return to Eden" for them. Their guides on these adventures were the experienced Indigenous hunters and fishers who knew these lands as well as their own villages. Gabe Aquin of St. Mary's First nation was one of their guides, but so too were members of the Indigenous community at Perth-Andover and the Miramichi.

The **1834** New Brunswick and Nova Scotia Land Company's purchase of **589 000 acres**, the northern tip of York County, that included the upper Main Southwest Miramichi River was a critical event in the evolution of the province's fly-fishing history. After the company realized its dream of establishing a pioneer farming community along a placid scow filled river was doomed because of the rough, infertile land and rapid filled streams, it focused on selling its timber as a revenue source.

Tough, skilled loggers cut the company trees in the fall, sledded them to the river in winter and stream-drove them down raging, ice-filled floods in the spring. This training ground produced a group of men who knew the woods and waters and had no fear of anything nature had to offer. They later became the local guides for the fly fishers who came to the river looking for salmon. Working for cash-paying sports was a novelty for woodsmen typically paid by store credits by lumber contractors or mill owners.

It is not known exactly when this sport began on the Miramichi, although, written records show it was commonplace by the 1840's. An **1854** painting of Burnt Hill by Lt. William Wolfe shows a well-used campsite with tents pitched and anglers and guides present.

Local interest in fly fishing as a sport was prompted by the actions of three successive New Brunswick Lieutenant Governors after 1848. Sir Edmund Head, Hon. John Henry Thomas, and Hon. Arthur Hamilton-Gordon explored New Brunswick's wilderness, engaging in both fishing and hunting. Vivid newspaper accounts of the day that recounted their wilderness adventures encouraged other colonial sportsmen to embrace their native colony's natural sporting resources.

Joining those local anglers were Americans who came to New Brunswick following settlement of the long-standing border dispute with Maine by the Webster-Ashburton Treaty of **1842**. The best known of the early Americans was Robert Barnwell Roosevelt, uncle of future President, Teddy Roosevelt, whose 1862 book ***Game Fish of the Northern States of America and British Provinces*** gives an account of his trip to the Main Southwest Miramichi where he was outfitted by Bloomfield Ridge's William Wilson, relative of the Wilson outfitters of McNamee.

As Canada became a nation in 1867, the trickle of anglers turned into a flood on the upper Main Southwest Miramichi. Soon, conflict arose between the provincial and federal governments over control of inland fisheries while different groups of anglers tried to claim exclusive fishing rights to angle certain prized productive pools.

By the **1870's**, amidst Canada's jurisdictional constitutional debate, prominent American and Canadians formed fishing clubs that claimed control of particular pools and parts of rivers. The first was the Burnt Hill Club, based on its 1000 acre property on the Main Southwest Miramichi formed in **1874.**

To assert its authority the federal government took two actions. Firstly, it required all anglers to purchase a fishing permit. Secondly, federal minister of

Marine and Fisheries, Peter Mitchell of Newcastle, issued an **1873** federal angling lease to the Burnt Hill Club for the Main Southwest Miramichi from Price's Bend to the source. Assaults, charges, counter charges, and lawsuits, quickly followed as wardens and competing anglers tried to establish their rights. It was from the battle between the two levels of government and various fishing groups over fishing rights to certain pools and court findings that led to the **1882**, decision by the Privy Council of the British House of Lords (Canada's highest court at the time), which ruled that riparian rights for angling belonged to landowners adjacent to the river, which included the provincial government on ungranted lands, not the federal government. This meant that the New Brunswick and Nova Scotia Land Company in particular, was now legal owner of some of the world's best salmon producing waters. Local landowners whose properties abutted salmon rivers in the province also gained control of fishing rights on their river fronts because of this decision. The positive economic result of that decision has been part of the local economy ever since.

The other part of this decision verified that province had control of the inland fisheries waters, but the federal government had responsibility for their management.

Recognizing the economic benefit to the province, New Brunswick immediately enacted laws and regulations to organize their new responsibilities. In 1885, New Brunswick sold fishing permits to residents. Government further capitalized on their riparian rights in 1889 by leasing 396 miles of its salmon producing rivers and some lakes to the highest bidders. Lessees were required to hire wardens to enforce provincial laws and regulations. Revenue gained from the leases helped fund the province's first fish and game enforcement branch consisting of a fishery commissioner and wardens.

The riparian rights decision had and still has greatly enriched provincial coffers and spawned generations of guides, outfitters, cooks, fly tiers, canoe builders, while helping support community businesses that catered to the sporting trade. It became part of their way of life.

Often forgotten in the battle over angling between elites of the day, is the loss of netting rights by local settlers who, for the decades since their arrival had been granted food fishery licences by the colonial government. New laws now forbade netting, spearing, or any other non-angling means of gathering salmon from the river. Those who pursued a food fishery were called poachers and those caught by newly appointed provincial or private wardens were given large fines. So, it was a changing world for all involved.

Two big changes took place in the early 1900’s: big game hunting was in decline and a railway was built giving easier access to the upper parts of the river. As a result, some local big game guides started offering river fishing trips to interested anglers and salmon outfitting began. The Griffins of Cross Creek, the Allens and Wades of Penniac, were among the pioneer outfitters on the Miramichi. Over the next thirty years, more outfitting camps were built along the rivers to accommodate guests. And some of the guests became land and camp owners hiring local men and women to host them during their fishing trips. Many of those camps or their successors still survive. Interest in outfitting spread to the lower parts of the river as time went on. Out of this pursuit of salmon grew a major local industry where outfitters, guides, cooks, fly tiers, boat makers, camp builders, merchants, and others benefitted from the dollars the sport brought to river.

As time progressed, so too did local interest in angling. With their intense focus on the river, many local men and women became the most efficient fishers the river has ever seen. Among the crowds gathered on shore with the coming of the “new runs” were Americans, Canadians, and a sprinkling of Europeans from all walks of life, all focused on the excitement of catching a bright straight-from-the-sea Atlantic salmon. Those were heady times. Those days are gone, but current estimates show that the angling business of the Miramichi is still worth thirty million dollars and directly employs six hundred people. That is a big business in rural Miramichi.

Threatening this vital economic generator along the rivers was the continued netting in tidal waters and in Miramichi Bay. As well, fishers from Newfoundland operated shore netting and offshore fishing for Atlantic salmon as part of their livelihoods. The result was diminishing numbers of salmon reaching their mother rivers in New Brunswick and elsewhere.

In 1948, a Montreal based group formed the Atlantic Salmon Association, a conservation group of wealthy and powerful anglers, dedicated to addressing the decline in returning salmon numbers. This group later developed into the Atlantic

Salmon Federation which promoted conservation measures in all salmon producing rivers in the northeastern United States and Canada.

Locally, to respond to those threats to the species, the Miramichi Salmon Association was formed in 1953 by a group of like-minded Americans and Canadians who tackled the problem head-on by lobbying federal and provincial governments to limit the catch limits by net fisheries along the river and beyond.

Later years saw the formation of a trans-Atlantic

Despite some successes over the years, the battles continue as dedicated anglers fight to save this iconic fish. Discovery of the Atlantic salmon feeding ground off Greenland in 1962 by the American nuclear submarine “Nautilus” alerted deep sea fishermen to their location and prompted an all-out assault on those grounds by various nations including Denmark.

The intense war of words beginning in the 1970’s between salmon conservation groups and deep-sea commercial fishers, especially from Denmark about saving the species from extinction has sometimes subsided, but never disappeared.

Prominent American anglers like Ted Williams, Bing Crosby, joined Canadian conservation voices to try to internationally embarrass Denmark into limiting its exploitation of salmon stocks off Greenland. The American government joined forces with Canada to try to limit the damage to the salmon stocks. For its part in 1972, President Richard Nixon signed a bilateral agreement with Denmark allowing it to harvest 100,000 salmon annually. Unfortunately, this unilateral action by Nixon lessened the ability of other countries including Canada to affect change regarding their rivers.

To try to show international leadership, Canada’s DFO minister Jack Davis ended salmon netting in Miramichi tidal waters and salmon fishing areas off Newfoundland. Compensation was paid to commercial salmon fishers. The initial term was for five years but that was extended to ten and then made permanent.

At that time the federal government was engaged in managing Atlantic salmon. Since then, other than gradually removing any angling retention, severely reducing federal warden protection, and closing cold water pools in times of high temperatures, successive federal governments have detached themselves from active management of this significant species.

Unfortunately for the salmon numbers, lately, they have focused their efforts in successfully rebuilding striped bass numbers on the Miramichi—a resurgence that threatens the very existence of its Atlantic salmon. The situation is desperate and getting worse with striped bass being found throughout the watershed. Unlike salmon, striped bass continue feeding in fresh water. Salmon smolts, trout, and any species that swims is food for them. Unchecked, they will destroy the rivers primes species, one that has become part of the culture of the river people.

To add to the problem, small mouth bass have escaped from Miramichi Lake into the main river. Illegally introduced some twenty years ago by camp owners wanting to fish a new species, their whim has become a river nightmare as growing small mouth bass combined with the large striped bass populations threaten the very eco-system of the Miramichi.

Out of desperation, a new alliance of conservationists and academics have joined forces with DFO to establish a new organization called CAST—Collaboration for Atlantic Salmon Tomorrow (2016-2020).

CAST's purpose was to "research and assess factors in the decline of at-risk Atlantic salmon populations." Its purpose was "to implement a comprehensive recovery strategy to reverse the decline of wild Atlantic salmon before it is too late."

Their four areas of focus were: ARIS sonar population tracking, thermal and LIDAR imaging to map habitats accurately, population modelling and a common home for all salmon data. And an adult salmon release program. I short, the group wanted to track salmon on their journey from their home rivers to their feeding grounds and back again, determine available suitable river habitats, establish a common data base for all salmon information, and finally to grow salmon to spawning size for release into the Northwest and Little Southwest Miramichi Rivers to replenish disappearing salmon numbers. The eggs for that experiment were sourced from salmon native to those rivers.

Partners in this long and expensive initiative were devastated by DFO's decision. In response to a letter of complaint about DFO's actions, the minister of the day, Bernadette Jordan, cited scientific studies that suggested such an action could negatively impact the rivers' native salmon stocks.

In summary, the first three objectives were successful as was the third until DFO refused to allow the rivers' restocking.

DFO claims that it has a five-year plan (2018-2023) to address the scientific questions related to Atlantic salmon, some of which were concluded by members of the CAST initiative. However, after gathering all this information about salmon, the time has come when the federal government must act, and act now.

As a reflection of the importance of salmon culture and celebrating the history of salmon fishing, the Atlantic Salmon Museum in Doaktown, was established in 1983. Contained within its walls are artefacts and Hall of Fame that tell the tale of salmon fishing from its Indigenous beginning to present day. Among its outreach programs is a conservation program for young children who will become tomorrow's new anglers and conservationists. Wouldn't it be a sad commentary if the museum is the only vestige of salmon left on the river?

Without DFO's immediate intervention to rebalance this catastrophe, Atlantic salmon will become extinct in the Miramichi, and that would be a national and international crime.

Year	River	Median	Group	species
2003	NW smolt	0.607	Pre-recovery	S. salar
2004	NW smolt	0.635	Pre-recovery	S. salar
2005	NW smolt	0.752	Pre-recovery	S. salar
2006	NW smolt	0.656	Pre-recovery	S. salar
2007	NW smolt	0.686	Pre-recovery	S. salar
2008	NW smolt	0.492	Pre-recovery	S. salar
2009	NW smolt		Pre-recovery	S. salar
2010	NW smolt		Pre-recovery	S. salar
2011	NW smolt		Post-recovery	S. salar
2012	NW smolt		Post-recovery	S. salar
2013	NW smolt	0.169	Post-recovery	S. salar
2014	NW smolt	0.228	Post-recovery	S. salar
2015	NW smolt	0.331	Post-recovery	S. salar
2016	NW smolt	0.359	Post-recovery	S. salar
2017	NW smolt	0.077	Post-recovery	S. salar
2018	NW smolt	0.158	Post-recovery	S. salar
2019	NW smolt	0.2	Post-recovery	S. salar
2020	NW smolt	0.309	Post-recovery	S. salar
2021	NW smolt	0.106	Post-recovery	S. salar
2022	NW smolt	0.038	Post-recovery	S. salar
2023	NW smolt	0.069	Post-recovery	S. salar
2003	Striped bass	20.61	Pre-recovery	M. saxatilis
2004	Striped bass	14.54	Pre-recovery	M. saxatilis
2005	Striped bass	18.3	Pre-recovery	M. saxatilis
2006	Striped bass	22.25	Pre-recovery	M. saxatilis
2007	Striped bass	49.59	Pre-recovery	M. saxatilis
2008	Striped bass	92.27	Pre-recovery	M. saxatilis
2009	Striped bass	48.39	Pre-recovery	M. saxatilis
2010	Striped bass	61.06	Pre-recovery	M. saxatilis
2011	Striped bass	202.9	Post-recovery	M. saxatilis
2012	Striped bass		Post-recovery	M. saxatilis
2013	Striped bass	254.8	Post-recovery	M. saxatilis
2014	Striped bass	138	Post-recovery	M. saxatilis
2015	Striped bass	297.5	Post-recovery	M. saxatilis
2016	Striped bass	314.5	Post-recovery	M. saxatilis
2017	Striped bass	998.1	Post-recovery	M. saxatilis
2018	Striped bass	335.6	Post-recovery	M. saxatilis
2019	Striped bass	313.8	Post-recovery	M. saxatilis
2020	Striped bass		Post-recovery	M. saxatilis
2021	Striped bass	260.7	Post-recovery	M. saxatilis
2022	Striped bass	471.8	Post-recovery	M. saxatilis
2003	SW smolt	0.436	Pre-recovery	S. salar
2004	SW smolt	0.462	Pre-recovery	S. salar

2005	SW smolt	0.617	Pre-recovery	S. salar
2006	SW smolt	0.675	Pre-recovery	S. salar
2007	SW smolt	0.674	Pre-recovery	S. salar
2008	SW smolt	0.569	Pre-recovery	S. salar
2009	SW smolt	0.561	Pre-recovery	S. salar
2010	SW smolt	0.564	Pre-recovery	S. salar
2011	SW smolt	0.6	Post-recovery	S. salar
2012	SW smolt	0.564	Post-recovery	S. salar
2013	SW smolt	0.39	Post-recovery	S. salar
2014	SW smolt	0.396	Post-recovery	S. salar
2015	SW smolt	0.347	Post-recovery	S. salar
2016	SW smolt	0.373	Post-recovery	S. salar
2017	SW smolt	0.196	Post-recovery	S. salar
2018	SW smolt	0.363	Post-recovery	S. salar
2019	SW smolt	0.336	Post-recovery	S. salar
2020	SW smolt	0.167	Post-recovery	S. salar
2021	SW smolt	0.355	Post-recovery	S. salar
2022	SW smolt	0.269	Post-recovery	S. salar
2023	SW smolt	0.209	Post-recovery	S. salar

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Research Document 2006/014 | **Document de recherche 2006/014**

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Aspects of the Life History, Biology, and Population Dynamics of Atlantic Salmon (*Salmo salar* L.) in Eastern Canada

Aspects du cycle, de la biologie et de la dynamique des populations de saumon atlantique (*Salmo salar* L.) dans l'Est du Canada

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* This series documents the scientific basis for the evaluation of fisheries resources in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

* La présente série documente les bases scientifiques des évaluations des ressources halieutiques du Canada. Elle traite des problèmes courants selon les échéanciers dictés. Les documents qu'elle contient ne doivent pas être considérés comme des énoncés définitifs sur les sujets traités, mais plutôt comme des rapports d'étape sur les études en cours.

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ISSN 1499-3848 (Printed / Imprimé)

Canada

Foreword

This document is a product from a workshop that was not conducted under the Department of Fisheries Oceans (DFO) Science Advisory Process coordinated by the Canadian Science Advisory Secretariat (CSAS). However, it is being documented in the CSAS Research Document series as it presents some key scientific information related to the advisory process. It is one of a number of contributions first tabled at a DFO-SARCEP (Species at Risk Committee / *Comité sur les espèces en péril*) sponsored workshop in Moncton (February 2006) to begin the development of a 'Conservation Status Report' (CSR) for Atlantic salmon. When completed in 2007, the CSR could form the basis for a Committee on the Status of Endangered Wildlife in Canada (COSEWIC) status report, recovery potential assessment and recovery strategy, and most importantly, enable DFO to implement pre-emptive management measures prior to engagement in any listing process.

Avant-propos

Le présent document est issu d'un atelier qui ne faisait pas partie du processus consultatif scientifique du ministère des Pêches et des Océans, coordonné par le Secrétariat canadien de consultation scientifique (SCCS). Cependant, il est intégré à la collection de documents de recherche du SCCS car il présente certains renseignements scientifiques clés, liés au processus consultatif. Il fait partie des nombreuses contributions présentées au départ lors d'un atelier parrainé par le MPO-SARCEP (*Species at Risk Committee* / Comité sur les espèces en péril) à Moncton (février 2006) en vue de commencer l'élaboration d'un rapport sur la situation de la conservation du saumon atlantique. Lorsqu'il sera terminé, en 2007, ce rapport pourrait servir de base à un rapport de situation du Comité sur la situation des espèces en péril au Canada (COSEPAC), à une évaluation du potentiel de rétablissement et à un programme de rétablissement mais, avant tout, il permettra au MPO de mettre en œuvre des mesures de gestion anticipées avant même de s'engager dans un processus d'inscription.

Abstract

A review of selected aspects of the life history, biology, and population dynamics of the Atlantic salmon of eastern Canada is presented. Information is provided on the following: life cycle and variability and plasticity in life history characteristics, size and growth in fresh water, biological characteristics of smolts, size and growth in marine water, survival in fresh water and at sea, and fecundity. A comprehensive list of references including those for major reviews of various topics is provided. The review is intended to provide pertinent resource material for the purpose outlined in the Foreword section above and as such is narrowed in scope accordingly.

Résumé

Le présent document passe en revue différents aspects du cycle, de la biologie et de la dynamique des populations de saumon atlantique de l’est du Canada. On y trouve des données sur les éléments suivants : cycle biologique et variations, ainsi que plasticité des caractéristiques du cycle biologique, taille et croissance en eau douce, caractéristiques biologiques des saumoneaux, taille et croissance en eau de mer, survie en eau douce et en mer et fécondité. Le document contient également une liste exhaustive de documents de référence, notamment des examens importants de divers sujets. Le présent rapport vise à fournir des données pertinentes aux fins mentionnées dans l’avant-propos qui précède, et sa portée a été limitée en conséquence.

Introduction

This paper presents a review of aspects of the life history, biology, and population dynamics of the Atlantic salmon of eastern Canada. Though the literature on these topics for Atlantic salmon in general is quite voluminous and comprehensive, information provided in this report is selective and tailored mainly to requirements specified in the Foreword section. Pertaining to the process of developing a Conservation Status Report for Atlantic salmon, those who wish to pursue various topics in greater detail than dealt with here can avail of the major reviews and works cited throughout the document.

Reference is made in this document to rivers within the provinces of eastern Canada, Salmon Fishing Areas (SFAs), and fishing zones (Qs) for the Province of Quebéc (Fig. 1).

Life History

Anadromous Populations

Anadromous Atlantic salmon display considerable phenotypic plasticity and variability in life history characters (Riddell and Leggett 1981; Saunders and Schom 1985; Thorpe 1986, 1989, 1994, 1998; Thorpe et al. 1998; Fleming 1996; Hutchings and Jones 1998; Klemetsen et al. 2003). It is an iteroparous species, one that can spawn repeatedly, as opposed to most species of Pacific salmon (*Oncorhynchus*), which are semelparous and die after a single spawning (Schaffer 1974; Fleming and Reynolds 2004). A defining characteristic of anadromous Atlantic salmon is the ability to return with a high degree of fidelity to the natal river or tributary for spawning (homing) (see reviews by Hasler and Scholz 1983; Stabell 1982, 1984; Hansen and Quinn 1998). Such precision in homing has led to the formation and maintenance of river-specific or local adaptations, resulting in much of the variability in genetic, life-history, behavioral, and other traits observed throughout the range of the species (Saunders and Schom 1985; Taylor 1991).

Figure 2 is a generalized depiction of the salient features of the life cycle of Atlantic salmon. This illustration will be the focus of an up-front broad-scale description of life history, to be followed by more detailed treatments of various aspects in the ensuing sections.

Depending on the stock, spawners returning to rivers are comprised of varying proportions maiden fish (those spawning for the first time) and repeat spawners. Most maiden salmon in turn consist of varying proportions of smaller fish that return to spawn after one winter at sea (1-sea-winter or 1SW) also known as grilse and larger fish that return after two or more winters at sea (2, 3-sea-winter, also designated as multi-sea-winter or MSW). Some rivers possess a component that returns to spawn after only a few months at sea (0-sea-winter or 0SW).

Collectively over its entire range in North America, adult Atlantic salmon return to rivers from feeding and staging areas in the sea mainly between May and November, but some runs can begin as early as March and April. In general, run timing varies by river, sea age, year, and hydrographic conditions. Run timing metrics pertain to where salmon are counted within a river system, whether it be an enumeration facility in the lower section of the river close to the estuary/ocean, or at a fishway located well upstream from the river mouth. As such, direct comparisons among rivers can be problematic although variability within a stock over time (years) is consistent for each individual system. Occasionally there can be some element of predictability in run timing of individual stocks; however, variation among years within rivers can still be substantive with median dates differing by five to six weeks or more in some populations. Large salmon enter earlier, on average, than small salmon in some rivers (e.g. LaHave River, Humber River, Western Arm Brook, Causapscal River). In other stocks the reverse is often true (e.g. Margaree River, Nashwaak River, Campbellton River, Highlands River, Mistassinin River) or there is little or no apparent difference (e.g. South River, Big Salmon River, Middle Brook, Terra Nova River, Bec-Scie River).

Run timing in Maritime rivers is frequently later, with return migrations extending over a greater interval of time than those of Quebec, Newfoundland, and Labrador salmon rivers. In the Miramichi, there are two distinct runs of salmon, an early run with peak migrations in early July and a late run with a peak in late Sept. to early Oct. (Chaput et al. 2001). Conne River and Humber River, Newfoundland, are characterized by median run timing dates of June 28, and are among the earliest in all of Atlantic Canada. In contrast, Nepisiguit, Big Salmon, and Margaree rivers have late migrations with median run dates occurring in late August and early September. Run timing is believed to be a heritable trait (Hansen and Jonsson 1991a) that is also influenced by environmental conditions. An example of two adjacent stocks influenced by similar environmental conditions but with very different run timing characteristics are Northeast Brook, Trepassey, and Biscay Bay River, Newfoundland. Here, median dates differ by more than three weeks between the two stocks with the former more similar to run timing of salmon returning to rivers in northern Labrador.

Spawning usually occurs in October and November in gravel-bottomed riffle areas of streams. Fertilization of eggs can involve both adult males and precocious male parr. Spawned-out or spent fish (kelts) either return to sea immediately after spawning or remain in fresh water until the following spring. Eggs incubate in the spawning nests or redds over the winter months and hatching usually begins in April. The hatchlings or alevins remain in the gravel for several weeks living off large yolk sacs. Upon emergence from the gravel in late May – early June, the yolk sac is absorbed and the free-swimming young fish, now referred to as fry or under-yearling parr, begin active feeding. Parr rear in fluvial (riverine) and lacustrine (standing water) habitats for two to eight years (Klemetsen et al. 2003) after which time they enter the smolt stage and migrate to sea.

Relative proportions of the various maiden sea-age and repeat spawning age groups and associated biological characteristics vary widely among stocks and with geographic location in North America (Porter et al. 1986). For example, stock composition can vary from being comprised of three sea ages, such as those on the Gaspé Peninsula, to consisting of a single sea age, which is characteristic of most of Newfoundland (Fig. 3). Even within populations with a simple sea-age structure like many of those of Newfoundland, there can be a multiplicity of individuals with different spawning histories contributing to egg deposition in a given year. In addition to virgin fish corresponding to the various smolt-age groups, which constitute the bulk of spawning escapements, there can be significant numbers of consecutive and alternate spawners present. All sea-age groups possess repeat spawners and this obviously greatly increases the potential for additional spawning types depending on the population. All these adult spawning types plus precociously maturing male parr may occur in the same population, constituting a “bet hedging” strategy (Stearns 1976; Lacey et al. 1983; Orzack and Tuljapurkar 1989; Roff 1992; Ellner and Hairston 1994; Einum and Fleming 2004), maximizing survival and population stability.

In Ungava Bay, at the northern extremity of the North American distribution, a form referred to as “estuarine” salmon occurs in addition to 1SW and MSW salmon (Robitaille et al. 1986). Unlike 1SW and older salmon, which undergo extensive oceanic migrations (Hansen and Quinn 1998), these fish can reach maturity and return to freshwater after only a few months in the estuary and are referred to as 0-sea-winter (0SW) salmon in Fig. 2. This form has also been encountered sporadically in low numbers over the years in some Newfoundland rivers (e.g. Campbellton River), but has become more prevalent in recent years (e.g. Downton et al. 2001). In eastern Hudson Bay (Nastapoka River), there is also a northern form that migrates to the estuary for a brief time, but this one differs from the estuarine salmon of Ungava Bay in that it does not undergo smoltification and growth rates are lower (Morin 1991). There are also populations in the inner Bay of Fundy that do not undertake long, distant migrations but rather tend to stay more localized (Amiro 1998, 2003). Unlike the estuarine or 0SW type however, inner Bay of Fundy populations are characteristically 1SW with a significant repeat spawner component (Amiro 1990).

The sexual maturation of wild anadromous male parr is widespread and highly variable throughout the distribution of the species (Dalley et al. 1983; Myers et al. 1986; Heinimaa and Erkinaro 2004) and they successfully mate with adult females both in the presence and absence of adult males (Myers and Hutchings 1987). Different mechanisms have been proposed to explain the co-existence of precocious and adult males within a single population of Atlantic salmon (see reviews by Fleming (1996) and Fleming and Reynolds (2004)). There is some evidence from breeding studies for a heritable basis for early maturity, or the existence of genetically distinct strategies. Other studies indicate a significant environmental influence, acting on genetically determined thresholds related to growth or physiological (e.g., energetic) state. Coexistence has been explained in terms of game theory wherein the two phenotypes represent alternate reproductive tactics within a single population. In contrast, the incidence of the maturation of wild anadromous female parr in fresh water is rare (Power 1969; Prouzet 1981; Baglinière and Maisse 1985; Hindar and Nordland 1989; Moore and Riley 1992). There have been cases where anadromous females that were stocked into ponds as swim-up fry have reached sexual maturity in fresh water (Harris 1973; Morrison 1983; O’Connell and Gibson 1989). These fish were subject to rapid growth and outlet configurations of the ponds prevented them from going to sea.

Non-anadromous Populations

Non-anadromous or resident salmon, complete their life cycle entirely in freshwater (Power 1958; Berg 1985). Isolation of non-anadromous from anadromous salmon appears to have occurred during isostatic rebound of coastal regions following the last ice age, approximately 10,000 years ago (Power 1958; Sutterlin and MacLean 1984; Berg 1985). Non-anadromous salmon occur not only above impassable physical barriers (e.g. Andrews 1966), but are also found in sympatry with anadromous salmon in river systems that are fully accessible to the sea (Scott and Crossman 1964; Hutchings 1986; Verspoor and Cole 1989). For some sympatric occurrences, there is evidence to suggest the forms are reproductively isolated (Verspoor and Cole 1989; Birt et al. 1991a, 1991b). Hutchings (1985) however, also viewed the occurrence of both forms in sympatry as phenotypic polymorphisms within a single population, as alternative evolutionarily stable strategies in the context of game theory as proposed by Maynard Smith (1982).

Size and Growth in Fresh Water

Atlantic salmon inhabit cool temperate streams of eastern Canada and can tolerate fresh water temperatures ranging from 0 to 28 °C (Elliott 1991). Juvenile Atlantic salmon begin feeding in the spring at water temperatures of 6 to 7 °C, and grow optimally at 16 to 19 °C (Javaid and Anderson 1967; Gibson 1978; Dwyer and Piper 1987; Jensen et al. 1989; Peterson and Martin-Robichaud 1989; Elliott, 1991; Elliott and Hurley 1997). Feeding ceases in larger juveniles at temperatures above 23°C and at water temperatures ranging from 22 to 24 °C, juvenile salmon seek refuge from thermal stress (Cunjak et al. 1993). The factors with the strongest correlation to body size and growth of juvenile salmon in fresh water include temperature, food availability and density (Gibson 1993; Elliott and Hurley 1997; Grant et al. 1998). Variations in growth rate are expected to result in variations in size and age at smoltification (Mangel 1994; Marschall et al. 1998). Several studies have reported on the short growing season of wild Atlantic salmon such that the size at age is reached earlier in the year than would be expected based on temperature alone (Randall and Paim 1982; Metcalfe 1994; Elliott and Hurley 1997; Strothotte et al. 2005).

Variations in Juvenile Salmon Size-At-Age

Growth in length and weight in Atlantic salmon juveniles in fluvial fresh water habitats tends to be initially rapid in June and July, sometimes into August, but limited in September and October (Randall 1982; Randall and Paim 1982; Heggenes and Borgstrom 1991; Juanes et al. 2000; Strothotte et al. 2005). The growth trajectories may also differ among age groups with growth of fry occurring over a longer period of time than that of parr (Randall and Chadwick 1986; Heggenes and Borgstrom 1991; Strothotte et al. 2005). Repeated sampling of juveniles from May to November in three rivers in the southern Gulf of St. Lawrence has provided further evidence of this seasonal growth dynamic (Fig. 4). In samples collected in the Miramichi River over the past 35 years, there is a strong association between size of fry and sampling date but this is not evident for parr of ages 1 and 2, which indicates that most of the parr growth for the year had probably occurred by the time of sampling in July and August (Swansburg et al. 2002).

Within a river, there are important among site differences in fork length of juveniles attained at the end of the growing season. In the Margaree River, end of growing season sizes of fry ranged from 5.0 to 7.0 cm whereas in the Miramichi, length ranged from less than 5.0 to over 6.0 cm (Fig. 4). Age-1 parr end of season sizes ranged from 7.5 to 11.0 cm among four sites in the Margaree River whereas in the Miramichi River, sizes ranged from 7.3 to 9.4 cm (Fig. 4).

Over a larger number of sites sampled in the Miramichi in 2001, age-1 parr mean size attained 10.4 cm and end of season age-1 parr length ranged almost 4 cm among sites whereas fry mean length ranged 1.5 cm. The Miramichi River covers 14,000 km^2 of drainage area and there are cool and warm water tributaries in the Northwest and Southwest Miramichi rivers which result in contrasting rearing environments for juveniles.

Swansburg et al. (2002) reported that in the Miramichi River, the fork length of juvenile Atlantic salmon parr decreased significantly during 1971 to 2000 and the decreased sizes-at-age were associated with increases in spring air and water temperatures. Although there remained a large component of the variance in size-at-age which could not be explained by density, date of sampling, tributary effects, and annual temperature indices, there was a strong correlation in the mean annual size-at-age of juveniles between the Northwest Miramichi and Southwest Miramichi rivers. There was also a positive and significant association between size-at-age of age-1 parr and size of fry in the previous year and size of age-2 parr and age-1 parr the previous year indicating that size advantages can be maintained at later ages.

Juvenile anadromous salmon use lacustrine habitat for rearing purposes to a considerable degree in Newfoundland rivers and over the years there have been varied studies of this aspect of life history (Pepper 1976; Pepper et al. 1984, 1985; Chadwick and Green 1985; Hutchings 1986; Ryan 1986; O'Connell and Ash 1989, 1993; Ryan et al. 1993; Dempson et al. 1996; O'Connell and Dempson 1996; Erkinaro and Gibson 1997a, 1997b; Gibson 2002). Habitats other than fluvial, including lacustrine, have been considered as marginal or secondary for juvenile anadromous salmon, occupied by individuals displaced from preferred stream habitat (Pepper et al. 1985; Gibson 1993), which is consistent with the traditional view that they are stream dwellers (Keenleyside 1962; Gibson 1966, 1988, 1993; Marschall et al. 1998). The widespread use of lacustrine habitat by parr is believed to be due to the relative lack of predators and competitors (Hutchings 1986; Gibson 1993; Gibson et al. 1993).

Growth of parr occupying lakes and ponds has been shown to be higher than for those of fluvial habitat (Pepper et al. 1985; Hutchings 1986; O'Connell and Ash 1993; Dempson et al. 1996). O'Connell and Ash (1993) demonstrated that juvenile growth rate in rivers in Newfoundland dominated by lacustrine habitat was higher than in those comprised mainly of fluvial habitat; Fig. 5a shows results for several rivers combined in each category. The lacustrine systems in this analysis were widespread geographically while the fluvial systems were mainly those of southwestern Newfoundland. Dempson et al. (1996) obtained similar results for lacustrine versus fluvial habitats within a single river system, Conne River (Fig. 5b). In another study for Conne River, Dempson et al. (2004a) reported higher lipid, protein, and energy levels for lacustrine parr than for fluvial parr. Hutchings (1986) considered the use of lacustrine habitat to be non-random and resultant enhanced growth and survival relative to fluvial habitat to be of adaptive significance.

Smolts

Variations in growth rate of juveniles are expected to result in variations in size and age at smoltification (Mangel 1994; Marschall et al. 1998). There is a negative association between an index of growth potential (combination of degree-days and day length) and mean age at smoltification for Atlantic salmon throughout its range (Metcalfe and Thorpe 1990). The age at smoltification has been shown to be in part associated with the growth rate or size achieved at several points in the life cycle and there is an increased probability of smoltification if a threshold size or growth rate is achieved by a critical time (Thorpe et al. 1998). As a consequence, small changes in initial growth rates of a cohort can affect the mean age at smoltification. In the Margaree River population, Strothotte et al. (2005) described a positive association between the age at smoltification (2 or 3 years) and size achieved at the end of the first freshwater year. In Newfoundland stocks, there are general and positive associations between marine survival, smolt size, and smolt condition (Dempson et al. 2003).

Smolt Size

During the last ten years, smolt monitoring programs in the Maritime provinces have provided information on wild Atlantic salmon smolt characteristics to complement the index river times series of the province of Québec. Wild smolts in the Nashwaak River and Big Salmon River in the Bay of Fundy are large relative to other MSW stocks, averaging between 14 and 16 cm fork length (Fig. 6). The lengths of smolts from the Gulf of St. Lawrence rivers extending from the western side of Cape Breton Island (SFA 18) to the north shore of the St. Lawrence (Q7) have ranged between 12 and 14 cm with the smallest overall sizes in the St-Jean River on the Gaspe peninsula (Q2) (Fig. 6). Smolts are also longer at age in the southern stocks of SFA 23, ranging from 14 to 16 cm for 2-year-old smolts and 15.5 to over 18 cm for 3- year-old smolts. The highest mean smolt lengths are recorded from Ungava Bay at 215 mm (Power 1969).

In Newfoundland, length varies from approximately 12 to 15 cm in the fluvial systems on the southwestern portion of the island; data for Highlands River as representative the MSW stocks of this area are shown in Fig. 6. For the remainder of Newfoundland (1SW stocks), mean smolt length ranges from around 15 to 18 cm. Lengths for two systems possessing substantive lacustrine habitat (Campbellton River and Western Arm Brook) exceeded all others presented in Fig. 6. Hutchings and Jones (1998) misquoted mean smolt length for two southeastern Newfoundland systems dominated by lacustrine habitat presented in O'Connell and Ash (1993). Values for the lacustrine systems Beaver River and Northeast River, Placentia should be much higher, namely 16.9 and 17.5 cm. O'Connell and Ash (1993) showed smolt size for lacustrine dominated systems to be higher than for fluvial systems. The importance of lacustrine habitat in determining smolt size is further illustrated in Figure 5c. An analysis of available data for 16 river systems, with watershed areas varying in size from < 100 to 6,400 km^2, shows a significant positive relationship between mean empirical smolt length and the ratio of lacustrine to fluvial (L/F) habitat (expressed as m^2) (Klemetsen et al. 2003).

With the exception of Newfoundland, there is a tendency for smolt length to increase with latitude (data summarized in Hutchings and Jones (1998)). Some rivers referred to above, ranging geographically from the south to the north coasts of Newfoundland, have mean smolt lengths overlapping those reported for Sand Hill River in southern Labrador (generally around 16 cm) by Anderson (1985). The departure from the clinal tendency exhibited by Newfoundland populations is an example of the modifying effects of local environmental conditions on growth, in this case most likely the utilization of lacustrine habitat for rearing.

Weight of salmon smolts is generally well described by length but mean weight at a standardized length can vary annually. Wild smolts at a fork length of 13.0 cm had annual predicted weights ranging from 20.5 to 21.5 for the Margaree River, 20.7 to 22.4 g for the Miramichi River, and 19.4 to 22.4 for the Restigouche River. Fulton's K condition factor, expressed as the ratio of the weight to the length cubed, provides an index of the relative weight at length. In the seven MSW salmon stocks of the mainland of eastern Canada, K was the lowest in the Saint-Jean River and highest in the southern Gulf stocks (Fig. 7). Smolts from the Margaree River were of similar K to de la Trinité stock and the Bay of Fundy rivers. Condition of smolts in the three Newfoundland systems (Campbellton, Highlands, Western Arm) was comparable to that of de la Trinité and St.-Jean. Smolts of higher condition factor have generally been shown to have better sea survival (Dempson et al. 2003).

Smolt Age

There are relatively few smolt monitoring programs in eastern Canada with which to describe smolt age distributions but there is much wider coverage in adult sampling programs. The proportion at smolt age from adult returns has been used to describe the smolt age characteristics of rivers in eastern Canada. While exceptions occur, there is a general tendency for smolt (river) age to increase with latitude in wild North American populations (Power 1981) (Fig. 8). Much of the variability has been explained by opportunities for growth that are influenced by local environmental characteristics (Power 1986). Maritime populations often have mean smolt ages between 2 and 3 years (see summary by Hutchings and Jones 1998); a similar situation applies to populations in southwest Newfoundland. In contrast, the rest of Newfoundland has salmon populations characterized by mean smolt ages between 3 and 4 years, similar to many Quebec stocks. Populations with mean smolt age of 4 or more years occur along the northwest coast of Newfoundland and into southern Labrador while mean smolt in some north Labrador stocks exceeds 5 years. Populations in Ungava Bay, northern Quebec also have mean smolt ages that exceed 5 years (Power 1969).

The fresh water age of returning adults to eastern Canada ranges between 1 and 7 years. Smolts of river age 8 have been reported from the rivers of Ungava Bay (Power 1969; Robitaille et al. 1986). There are few records of one year smolts in the large scale sample collections dating back to the 1970s and these are mostly from the southern portions of the range (the one-year old smolt records from Bonaventure (Q1), Grande rivière (Q2), and de la Trinité (Q7) are suspect). Maximum smolt ages were highest in the northern areas of eastern Canada and smolt ages 6 to 7 years were only observed in salmon from the northeast coast of Newfoundland, eastern north shore of Québec (Q9) and Labrador. Age 3-year-old smolts are found throughout the range of Atlantic salmon but at very low abundance in the most northern populations in Labrador (SFA 1) and Ungava Bay.

After adjusting for the year of smolt migration, two-sea-winter maiden salmon are comprised of higher proportions of younger smolt ages relative to the one-sea-winter maiden salmon, notably in the multi-sea-winter salmon stocks in the southern portion of the range. This is consistent with other observations that larger smolts frequently mature at younger sea ages than smaller smolts. In the two most northern rivers examined (St. Jean, Q2; de la Trinité, Q7), however, the proportion of age 3-year-old smolts in the returns of 1SW and 2SW from the same smolt cohort were not statistically different. The preponderance of 1SW salmon in many Newfoundland rivers could be related to the growth, size, and survival advantages of smolts conferred by lacustrine habitat (O'Connell and Ash 1993; Klemetsen et al. 2003).

Sex Ratios of Smolts

The incidence of male precocity in a population has been shown to be a function of growth rate, i.e. faster growth favors early maturity (Dalley et al. 1983; Myers et al. 1986; Randall et al. 1986; Thorpe 1986, 1987; Hutchings and Myers 1994). Precocious maturation however comes at a cost, with mature parr incurring greater mortality than non-maturing individuals (Saunders et al. 1982; Dalley et al. 1983; Myers 1984) and maturation can also inhibit smolting (Thorpe 1987; Saunders et al. 1994; Whalen et al. 2000; Dustin et al. 2005). Losses of males from both these processes can result in smolt runs dominated by females (Forsythe 1967; Saunders et al. 1982; Dalley et al. 1983) and there can be high correspondence between the proportions of females in smolt runs and subsequent 1SW adults (e.g. Dempson et al. 2004b). O'Connell and Ash (1993) reported female:male ratios for smolts and 1SW adults subject to enhanced growth in the lacustrine systems referred to above to be substantially higher than for counterparts in fluvial systems.

As already alluded to, the sex ratios in the smolt runs of eastern Canada show differences which are consistent with the sea age structure of the returning adults. In the rivers of insular Newfoundland which are comprised primarily of 1SW maiden salmon, the smolt migrations are strongly skewed toward females, generally in excess of 70% and as high as 90% depending on the river and year. In the MSW stocks of the mainland portion of eastern Canada, the percentage female is variable and determines in large part the proportion of the sea-age structure of the returning adults. There are limited data for sex ratios in the southern stocks of the Bay of Fundy and the Atlantic coast of Nova Scotia but Jessop (1975) reported smolt runs comprised of 60 and 66% female in 1971 and 1972 for Big Salmon River. In the southern Gulf of St. Lawrence, the percentage female in the Margaree has been the highest of the southern Gulf of St. Lawrence rivers, ranging between 70 and 79 for the years 2001 to 2005. In the Miramichi and Restigouche rivers, the sex ratios are nearly balanced with ranges of 52-61% and 44-60%, respectively. The percentage female in the smolts of Québec monitored rivers ranges from 56 to 72 for the Saint-Jean River and 53 to 70 for de la Trinité River.

Size and Growth of Non-Anadromous Salmon

In North America, there have been reports of non-anadromous Atlantic salmon measuring up to 56 cm in length for Labrador (Bruce 1974) and weighing as much as 3.9 kg for Newfoundland (Scott and Crossman 1964), 20.4 kg for Lake Ontario (Scott and Crossman 1998), and in excess of 12.2 kg for Maine, USA (Warner and Havey 1985), comparable in size to anadromous fish of various sea ages. At the opposite end of the size spectrum, populations of dwarf non-anadromous salmon occur in both fluvial (Gibson et al. 1996) and lacustrine (Bruce 1976; Barbour et al. 1979; Sutterlin and MacLean 1984) habitats. In the fluvial situation, dwarf mature females ranged in length from 8.4 to 12.3 cm (mean = 10.2) (Gibson et al. 1996) while lacustrine mature females ranged from around 11.0 to 14.9 cm (Bruce 1976). A variation of the above noted earlier is a transplantation experiment where anadromous salmon fry were stocked into a small landlocked urban pond in Newfoundland. The resulting growth rate of young salmon in this pond was two to five times as fast as in other Newfoundland ponds with age 2+ fish approximately 28.0 cm in length (O'Connell and Gibson 1989) illustrating the capacity for growth in certain situations.

Size and Growth in Marine Waters

Atlantic salmon management in eastern Canada is based on two size groups defined by fork length; small salmon < 63 cm in fork length, large salmon are of length ≥ 63 cm. Salmon return to rivers of eastern Canada to spawn predominantly after one, two, and in some areas three years at sea. A handful of scale samples from salmon from rivers in SFA 15 (Restigouche and Nepisiguit) have been interpreted as possessing four maiden years. In most regions, 1SW salmon are the most abundant maiden age group, representing almost 100% of maiden spawners in rivers of insular Newfoundland, with the exception of fish on the southwest coast (SFA 13). Two-sea-winter salmon are abundant in most rivers of the mainland of eastern Canada, especially in the southern Gulf of St. Lawrence, and are found rivers on the southwest coast of Newfoundland (but virtually absent for the remainder of Newfoundland) and in Labrador. Three-sea-winter salmon are infrequent other than in the rivers of the southwestern Gulf of St. Lawrence (Baie des Chaleurs – SFA 15, Q1, Q2), in southwestern Newfoundland (SFA 13) (Blair 1965) and in some rivers of the north shore of the St. Lawrence (Q8); they have occurred historically in the Saint John River (SFA 23) and St. Mary's River (SFA 20). With few exceptions, small salmon are comprised of 1SW maiden fish whereas the large salmon category is comprised of multiple maiden age groups and repeat spawners.

The observed differences in the proportions at age at maturity in the anadromous salmon of eastern Canada suggest that this trait reflects an adaptation of the populations to spatially variable conditions (Hutchings and Jones 1998). The factors which are considered to be important determinants of age at maturity include growth rates (in fresh water and at sea), survival in fresh water and at sea, fecundity, and heritability (Meerburg 1986; Chadwick et al. 1987; Friedland and Hass 1997; Hutchings and Jones 1998). Age at maturity of parents remains an important factor in determining age at maturity of progeny (see papers in Meerburg 1986).

Repeat spawning salmon are present in all rivers of eastern Canada and they contribute both biologically and economically to the value of the salmon resource (Atkinson and Moore 1999). As a result of changes in fisheries, repeat spawner abundances have increased in numerous rivers of eastern Canada, most notably in the MSW stock of the Miramichi River (Fig. 9). Repeat spawners, historically particularly prominent in some of the inner Bay of Fundy rivers in SFA 22 and have now become so in other areas since the reduction of exploitation in fisheries (Ducharme 1969; Moore et al. 1995). The percentage of large salmon comprised of repeat spawners over the past 35 years has increased from < 5 to between 30 and 55 in the last decade (Chaput et al. 2001). In contrast, the repeat spawner proportions have not changed in the Saint John River and the Saint Jean River (Fig. 9).

Small salmon are greater than 90% maiden salmon with repeat spawners in that size group most abundant in the rivers of the south and northeast coasts of Newfoundland. Repeat spawners constitute between 70 and 98% of the large salmon category in the south and northeast coasts of Newfoundland compared to only 5 to 30% in most other rivers of eastern Canada. Dempson et al. (2004c), noted an increase in the contribution of consecutive spawners during the Atlantic salmon commercial fishery moratorium years (post-1991) although in some stocks (e.g. Gander and Conne) the increase was only observed after several years into the moratorium. In MSW stocks, repeat spawners are derived from all age groups of maiden salmon. In the Miramichi River, repeat spawners were predominantly derived from 1SW maiden salmon in the 1970s but 2SW maidens are now proportionally as abundant as 1SW maidens attributable to the reduction on harvest of 2SW salmon resulting from the closure of the Maritime commercial fisheries and the mandatory release of all large salmon in the recreational fisheries since 1984.

Spawning age structure has become more complex in many rivers with salmon in the Miramichi River on their seventh spawning migrations observed almost annually since 1995 (Chaput and Jones 2006). Salmon repeat as either consecutive or alternate spawners and in all combinations of these in subsequent spawning migrations. Switching between consecutive and alternate life histories also occurs frequently. Collectively over the years, with smolt-age combinations included, as many as 20 spawning types have been encountered for Conne River (Dempson et al. 2001) and 26 for Gander River (O'Connell et al. 2001), both of which are 1SW stocks, whereas in the Miramichi River with its significant MSW component, 49 sea age types have been interpreted, excluding the freshwater age combinations. The broad spawning age structure has resulted in 8 or 9 year classes present on the spawning grounds in the recent decade, an increase from the 4 to 5 year classes in the earlier years when there were fewer repeat spawners (Chaput and Jones 2006).

Growth

Growth in the marine environment is rapid relative to that in fresh water. Whereas after two to four years of growth in fresh water, smolts attain average fork lengths of 12 to 18 cm, after one year of growth at sea, length can range from 45 to as much as 65 cm. After two years at sea, salmon measure in the range of 70 to 80 cm. At relatively similar age, there is a large variation in fork length (Cairns 2003). Weights increase about 75-fold between the smolt stage and 1SW salmon stage, and over 200 fold from smolts to 2SW salmon (Cairns 2003). Weight is strongly associated with length and salmon measuring 60 cm weigh about 2 kg, 80 cm salmon weigh about 5 kg, and a 100 cm salmon weighs upwards of 11 kg.

Growth of adults at sea, like juveniles in fresh water, is seasonal and described by the cyclic patterns of circuli spacings on the scales, from observations of non-maturing 1SW salmon at West Greenland, from post-smolt surveys at sea, and from monitoring of wild smolts reared in sea cages (see Cairns 2003; Dempson et al. 1999). Atlantic salmon which are destined to spawn reduce or cease their feeding activities on their return migrations, as much as several months prior to entering the rivers (see summary in Cairns 2003). In the Miramichi River where salmon return to fresh water over a five month period from late May to late October, there is a very modest increase in length during the season of return, by about 3 to 4 cm (+6%) for 1SW salmon and 3 cm (+4%) for 2SW salmon (Moore et al. 1995). Differences in length between male and female salmon are small, with male salmon in the Miramichi River longer by about 1.5 cm in both 1SW and 2SW salmon (Moore et al. 1995).

Size at maiden age varies among rivers. One-sea-winter maiden salmon mean lengths range between 48 and 59 cm. There does not seem to be any consistent pattern regarding size of 1SW salmon and latitude. Indeed larger 1SW salmon, say ≥ 55 cm, can be found among populations from the Maritimes, Québec, and Newfoundland and Labrador with 1SW salmon averaging greater than 60 cm occurring in some Ungava Bay rivers (Power 1969; Hutchings and Jones 1998). Two-sea-winter salmon mean lengths range from just over 60 cm in some rivers of the south coast of Newfoundland to between 75 and 80 cm in most of the other rivers. 3SW salmon mean lengths vary between 75 cm to over 100 cm.

The lengths of salmon at maiden age of return as measured at several river monitoring facilities have increased over time (Fig. 10). Some of the changes correspond to the closures of commercial fisheries instituted over the period 1970 to 2004. Mean sizes of 1SW maiden salmon are greater post-1991 in 11 of 14 rivers, the exceptions being Restigouche River, Saint Jean River and Conne River (Scheffe a posteriori test, $P < 0.05$; Fig. 11). Generally, the mean sizes during 1970 to 1983 and 1984 to 1991 did not change with the exception of the Miramichi River in which size increased over the three management periods. 1SW maiden salmon mean length in the recent management period ranges between 54 and 59 cm in contrast to the 1970-1983 period when mean length ranged from just under 52 to almost 57 cm (Fig. 11). Mean lengths of 2SW maiden salmon have increased in some rivers but are unchanged in others. Some of the changes in mean lengths of 2SW maiden salmon observed in the Miramichi have been attributed to closures of size-selective commercial fisheries (Moore et al. 1995). Mean sizes of 2SW maiden salmon are greater post-1991 in only 2 of 8 rivers, the Miramichi River and de la Trinité River (Scheffe a posteriori test, $P < 0.05$; Fig. 12).

Size of repeat spawners varies in relation to the spawning history of individual fish and whether spawning occurred as consecutive or alternate year events. For example, whereas maiden 1SW salmon at Conne River, Newfoundland, average 51.1 cm in fork length, first-time consecutive spawners averaged 55.9 cm while salmon with two consecutive spawning marks had a mean length of 62.0 cm. In contrast, alternate spawners that typically spend another entire year at sea averaged 69.6 cm in length. In two MSW salmon stocks of the Gulf, first time consecutive 1SW maiden salmon had an average length of 67.2 in the Miramichi compared to 63.2 cm in the Restigouche (Fig. 13). First time 1SW alternate spawners had mean lengths of 83.5 and 81.7 cm. 2SW repeat spawners had mean lengths of 81.9 to 86.8 as first time consecutives, and 91.5 to 98.0 cm as first time alternates. Growth of salmon between spawning events results in repeat spawner lengths overlapping onto and eventually exceeding maiden salmon lengths.

Sex Ratios

There are important regional differences in the proportion female in the 1SW maiden salmon components in eastern Canada. 1SW maiden salmon in the Bay of Fundy and Atlantic coast of Nova Scotia are generally comprised of between 20 and 70% female, the exceptions being some stocks from eastern Cape Breton and the salmon stock in the Saint John River above Mactaquac which are generally ≤ 10%. The stocks from the Gulf of St. Lawrence and Québec

(SFA 15-18, Q1-Q7, Q10) are characterized by a low percentage of female in the 1SW maiden salmon (1 to 20%), with very few females (< 5%) in many rivers of Chaleur Bay (SFA 15, Q1-Q2). In Newfoundland, maiden 1SW salmon are predominantly female, ranging from 45 to almost 100% in some rivers. The Labrador and north shore Quebec stocks (Q8, Q9) are more similar to the Bay of Fundy/Atlantic coast of Nova Scotia rivers, percentages female ranging from 13 to 71% for the seventeen rivers with information.

Large salmon are characterized by percentages female which are generally greater than 50% whereas 2SW maiden salmon have variable but high proportions female which can exceed 90% in some stocks.

For stocks in which large salmon represent greater than 50% of the total returns to the river, the small salmon (1SW maiden) are predominantly males, greater than 80% (Fig. 14). In other stocks where small salmon are greater than 50% of the returns, the proportion female in the small salmon will generally be biased to females, with exceptions to this for some Labrador stocks and a few Gulf of St. Lawrence stocks.

Percentage female within a river varies annually, for some within a fairly narrow range (for example 1SW salmon in Saint-Jean) while in other stocks, the annual variation is quite large (for example 1SW salmon in Middle Brook SFA 5) (Fig. 15). For 1SW maiden salmon, there was no statistically significant (linear regression, P>0.05) change in the proportion female over time in 7 of 13 rivers examined. There were significant increases in the proportion female in 4 of the 13 rivers (Saint John, de la Trinité, Northeast Trepassey, Sand Hill) whereas in the Miramichi River and the Exploits River, there were significant decreases. For 2SW salmon there were statistically significant increases in the proportion female in 4 of 6 rivers whereas there was no change in the Saint John River and the Nashwaak River.

Survival

The anadromous life history of Atlantic salmon provides opportunities to monitor survival in the two distinct environments occupied by salmon at the different stages of its life cycle. In a large number of rivers, total counts (or estimates) of returning adults can be obtained. From these and by accounting for removals in fisheries, egg depositions can be derived. All ages of juvenile salmon can be readily monitored up to the smolt stage at which point the total production of a cohort going to sea can be quantified. From these, fresh water survival from the egg to the smolt stage can be studied. Marine survival or in some cases return rates can also be quantified on those rivers where the total smolt production and subsequent adult returns are monitored.

In Freshwater

The abundance of Atlantic salmon in fresh water is regulated by density dependent and density independent factors (Elliott 2001). In fresh water, survival is at least compensatory such that relative survival decreases as abundance increases, and vice versa. Compensatory survival in fresh water results from competition for limited resources including food and space, with its multidimensional components (Marschall et al. 1998; Cunjak et al. 1998). Self thinning principles have been proposed as the mechanism for regulation of population abundance (Grant et al. 1998). The major density dependent regulatory factors are assumed to occur generally in the early life stages but can also act at later stages as individuals compete for limited resources (Elliott 2001; Jonsson et al. 1998).

Overall survival in freshwater can be approximated from estimates of numbers of eggs deposited and subsequent production of migrating smolts. Indices of inter-stage survivals in freshwater, as for example, fry to age-1 parr, can be obtained from indices of juvenile abundances at age. The translation of these indices into absolute measures of survival are difficult because these life stages were frequently sampled in classic rearing habitat (Elson 1957) which represents different proportions of all habitat within rivers. As a result, the juvenile survival indices derived may differ from those integrated over all habitat types.

Symons (1979) provided an earlier review of interstage survival rates in fresh water and summarized ranges of survival rates for egg to fry of 9 to 20%, annual rates for parr of 28 to 57% and parr to smolt survival rates of 35 to 65%. Locke (1998) summarized a number of interstage survival rates from the literature and illustrates the wide range of estimated survival rates at all stages among the studies. Symons (1979) suggested that based on interstage survival rates, egg-to-smolt survival would be higher for populations with younger smolt ages but the observations from eastern Canada as reported by Chaput et al. (1998) do not support that view; egg-to-smolt survival is actually higher in the more northern rivers with older mean smolt age.

Egg-to-smolt or freshwater survival can vary substantially both among rivers as well as within rivers over time to the extent that the variability often exceeds that observed among estimates of marine survival. Egg-to-smolt survival rates from monitored rivers in eastern Canada range from a low of 0.1% to a high of 6.5% (Fig. 16). Based on the data available, Chaput et al. (1998) reported that egg to smolt survival was higher in Newfoundland rivers where juveniles have the potential to rear in lacustrine habitat, rather than exclusively fluvial habitat. Within Newfoundland, Klemetsen et al. (2003) reported that salmon rearing in lacustrine habitat may have somewhat higher freshwater survival rates than corresponding stocks rearing predominately in fluvial environments. Indeed, egg-to-smolt survival averaged 0.52% (minimum = 0.36; maximum = 1.09%) for Northeast Brook, Trepassey, with an L/F value of 5.2 over 12 year-classes. In contrast, freshwater survival for Conne River (N = 14 year classes), with an L/F value of 24.1, averaged 1.24%, varying from 0.45 to 2.55% among individual year-classes. For Western Arm Brook, freshwater survival averaged 1.39% over 29 year-classes where the L/F value is 69.6. Klemetsen et al. (2003) showed some evidence that smolt-to-adult survival increases with L/F ratio, suggesting that smolt survival increases with smolt size (see also below).

The overall compensatory function of survival is expressed in the decrease in egg-to-smolt survival with increasing egg deposition observed in numerous monitored stocks of eastern Canada (Fig. 17).

At Sea

For salmon, there are numerous estimates of M for the freshwater stages because they can be readily monitored and are not subjected to fisheries. Estimates of M during the marine phase are much more difficult to obtain because the adults enumerated back to the river have until recently been exposed to both natural and fishing mortality factors at sea. Ritter (1989) reviewed the literature on survival of salmon at sea and concluded that sea survivals were stock dependent with average sea survivals (return rates) for North American populations to the first sea winter of 10 to 20% per year. The sea survivals of European stocks are generally higher with return rates to the coast of River Bush smolts in the order of 35% (Crozier and Kennedy 1994) and those to other rivers generally being greater than 10% (Ó Maoiléidigh et al. 2003).

Doubleday et al. (1979) suggested that the greatest mortality occurred in the initial stages at sea when the fish were small compared with later in life (after one year at sea) when the fish were much larger, consistent with the inverse-weight hypothesis. Since smolts are about 1% the weight of salmon after one year at sea (20-40 g versus 2000-4000 g), then variations in integrated mortality would be expected to be defined by smolt size.

Many factors act to influence the survival and production of Atlantic salmon (Saunders 1981; Dempson et al. 1998; McCormick et al. 1998; Parrish et al. 1998; Armstrong et al. 1998). Consequently, survival, and hence adult salmon abundance, is often highly variable, both in Atlantic (Chadwick 1988; Dempson et al. 1998) as well as in Pacific salmon populations (Noakes et al. 1990; Hargreaves 1994). Some factors, such as run timing and smolt size, can have a consistent influence on the subsequent survival to the adult life stage (e.g. Ward and Slaney 1988; Hansen and Jonsson 1989, 1991b; Ritter 1989; McCormick et al. 1998; Salminen et al. 1995; Finstad and Jonsson 2001). In contrast, Hargreaves (1994) stated that many attempts have been made to relate marine survival rates to environmental parameters, and while observed patterns have often assisted in forecasting subsequent salmon abundance, frequently the resulting relationships showed little consistency among stocks, and among years within a stock.

A review of marine mortality of Atlantic salmon and its measurement concluded that contributory factors are complex and attempts to identify a single, dominant factor have been unfounded (Potter et al. 2003). Survival was found to vary substantially both among stocks and regions as well as within a stock over time. In populations for which multiple sea-age classes exist, estimates of return rates are underestimates of survival because some of the fish are destined to remain at sea and either die or return as MSW fish (Hutchings and Jones 1998; Chaput 2003). No attempt has been made to adjust for this in the current synopsis.

Counts of Atlantic salmon smolts and adults enable estimates of marine survival to be derived. Examination of survival trends over time can provide insight into the effects of management measures designed to reduce marine exploitation, or, in the absence of fisheries allow estimates of natural survival to be calculated. Alternate methods including inverse weight and maturity schedules have also been applied to determine estimates of natural survival (see review by Chaput 2003).

Survival of smolts to 1SW, or in the case of Newfoundland salmon < 63 cm in length, is generally low. Since the Newfoundland commercial salmon fishery moratorium began in 1992, and hence the opportunity to obtain 'natural' survival in the absence of directed marine fisheries, survival has exceeded 10% in less than 3% of all individual estimates available (N = 76) (Fig. 18). The majority of survival values for monitored Newfoundland stocks fall within the range of 2 to 7%, averaging around 5% and thus similar to the return rates reported for 1SW Miramichi salmon, while return rates to 1SW salmon in Québec rivers are generally less than 1% (Fig. 18). Indeed, for Newfoundland small salmon there is no difference in the distribution of survival values between the pre-moratorium and moratorium periods (G = 4.090, P = 0.665). In some populations (e.g. St. Jean, de la Trinité, Conne, and Northeast Brook, Trepassey) survival is lower since the Newfoundland commercial salmon fishery closed than it was prior to the moratorium in 1992 even in the absence of adjustments to account for marine exploitation prior to fishery closures. This was largely unexpected given that estimates of the median marine exploitation rates during the period 1984 to 1991 were 45.3% (29.6 – 57.1%) on small salmon and 74.2% (57.7 – 83.7%) on large salmon (Dempson et al. 2001a). Survival to the 2SW stage is also quite low averaging less than 1% in most monitored rivers and again with several stocks experiencing lower survival in recent years by comparison with those years affected by directed commercial fisheries (Fig. 19).

Doubleday et al. (1979) used the inverse weight hypothesis to estimate natural survival of non-maturing 1SW salmon during the second year at sea. Assuming an exponential growth function, they obtained natural mortality rate estimates between Greenland and home waters (approx. 12 months) from 3% to 12%, or less than 1% per month. Based on recent size at age data from several North American rivers and assuming a linear growth function, monthly mortality rates of about 3% were estimated (Chaput 2003).

In addition to marine survival estimates of maiden salmon, survival of repeat spawners can also be determined for some stocks. Dempson et al. (2004c) used scale pattern analysis to identify first time consecutive spawners for six Newfoundland rivers. Survival was estimated by comparing the numbers of salmon returning to spawn a second time in year $i + 1$ with the corresponding number of maiden 1SW fish that spawned previously in year i. Similar to the situation in freshwater and that observed among maiden salmon, survival of repeat spawners also varied considerably among years within stocks. While survival of first time repeat spawners was commonly less than 20%, estimates of over 30% or greater occurred in some rivers in some years. Mean survival of repeat spawners was highest for Terra Nova River, Northeast River, Placentia and Middle Brook (mean > 18%), and lowest for Exploits River, Gander River, and Conne River (mean < 10%). This contrasts with repeat spawner survivals to a second spawning for the Miramchi which were about 5% for 1SW and between 5% and over 30% for 2SW salmon in the late 1980s (Moore et al. 1995). More recent analysis of the Miramichi stock shows an increase in the return rate as consecutive spawners in both the 1SW and 2SW salmon components with the overall return to a second spawning of 2SW salmon at about 20% over the recent five years. 1SW repeat spawner return rates remain at less than 10%.

Fecundity

Fecundity varies considerably both within and among salmon stocks. Egg number and size increase with body size (Thorpe et al. 1984; Jonsson et al. 1996). In a dwarf or stunted freshwater resident population from Newfoundland, mean fecundity was 33.0 eggs (Gibson et al. 1996). In contrast, Randall (1989) reported mean fecundities of 12606 and 16585 eggs for 3SW and previous spawning salmon in Restigouche River. Although absolute fecundity varies greatly among individuals, as expected owing to high variability in adult body size, relative fecundity (eggs per kilogram) as a measure of reproductive effort, varies much less but is inversely related to fish size. For Miramichi River, New Brunswick, relative fecundity ranged from 1331 eggs kg^{-1} in previous spawning salmon (mean length 82.1 cm) to 2035 eggs kg^{-1} in 1SW fish (Randall 1989). Rouleau and Tremblay (1990) reported values of 1628 eggs kg^{-1} for 2SW salmon, 1256 eggs kg^{-1} for 3SW salmon, and 1244 eggs kg^{-1} for repeat spawners. In a survey of 2440 specimens from 10 Newfoundland rivers, mean relative fecundity varied from 1278 to 2500 (O'Connell et al. 1997). The variability in relative fecundity that can occur within a stock is rarely, if ever, taken into consideration when estimates of egg deposition are determined. Consequences related to conservation spawning levels achieved and studies on population dynamics are obvious.

There can be a trade-off between egg size and fecundity (Fleming 1996; Jonsson et al. 1996). Fish either may spawn large and few eggs or small and many eggs. Fast growth of parr in fresh water before smolting has been associated with smaller relative egg size at maturity, a phenotypic response that has been explained as an adaptation to the potential growth opportunities in the nursery river. This assumes that feeding opportunities that parents experience as juveniles in rivers is a good predictor of what their offspring will experience. Thorpe et al. (1984) cited instances of both negative and positive correlations between egg size and egg number and also one study showing no correlation.

References

Amiro, P. G. 1990. Recruitment variation in Atlantic salmon stocks of the inner Bay of Fundy. Canadian Atlantic Fisheries Scientific Advisory Committee Research Document 90/41. 26 pp.

Amiro, P. G. 1998. The mystery of the missing inner Bay of Fundy salmon. Atlantic Salmon Journal 47(3): 50-53.

Amiro, P. G. 2003. Population status of inner Bay of Fundy Atlantic salmon (*Salmo salar*), to 1993. Canadian Technical Report of Fisheries and Aquatic Sciences No. 2488. 44 pp.

Anderson, T. C. 1985. The rivers of Labrador. Canadian Special Publication of Fisheries and Aquatic Sciences 81. 389 pp.

Andrews, C. W. 1966. Landlocked Atlantic salmon (*Salmo salar* L.) in the Terra Nova River system, Newfoundland. The Canadian Field-Naturalist 80: 101-109.

Armstrong, J. D., J. W. A. Grant, H. L. Forsgren, K. D. Fausch, R. M. DeGraaf, I. A. Fleming, T. D. Prowse, and I . J. Schlosser. 1998. The application of science to the management of Atlantic salmon (*Salmo salar*): integration across scales. Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1): 303-311.

Atkinson, G. and D. Moore. 1999. The case for releasing grilse. Atlantic Salmon Journal 48: 68-72.

Baglinière, T. B., and G. Maisse. 1985. Precocious maturation and smoltification in wild Atlantic salmon in the Amorican Massif, France. Aquaculture 45: 249-263.

Barbour, S. E., P. J. Rombough, and J. J. Kerekes. 1979. A life history and ecologic study of an isolated population of "dwarf " ouananiche, *Salmo salar*, from Gros Morne National Park, Newfoundland. Naturaliste canadien 106: 305-311.

Berg, O. K. 1985. The formation of non-anadromous populations of Atlantic salmon, *Salmo salar* L., in Europe. Journal of Fish Biology 27: 805-815.

Birt, T. P., J. M. Green, and W. S. Davidson. 1991a. Mitochondrial DNA variation reveals genetically distinct sympatric populations of anadromous and nonanadromous Atlantic salmon, *Salmo salar*. Canadian Journal of Fisheries and Aquatic Sciences 48: 577-582.

Birt, T. M., J. M. Green, and W. S. Davidson. 1991b. Contrasts in development and smolting of genetically distinct sympatric anadromous and nonanadromous Atlantic salmon, *Salmo salar*. Canadian Journal of Zoology 69: 2075-2084.

Blair, A. A. 1965. Bay of Islands and Humber River Atlantic salmon investigations. Journal of the Fisheries Research Board of Canada 22: 599-620.

Bruce, W. J. 1974. The limnology and fish populations of Jacopie Lake, West Forebay, Smallwood Reservoir, Labrador. Fisheries and Marine Service Technical Report Series No. NEW/T-74-2. 74 pp.

Bruce, W. J. 1976. Age, growth, maturity and food habits of landlocked salmon (*Salmo salar*) in Soldiers Pond, a Newfoundland Lake. Fisheries and Marine Service Technical Report 668. 16 pp.

Cairns, D. K. 2003. Feeding, fasting, and weight-based estimation of natural mortality in marine-phase Atlantic salmon (*Salmo salar* L.). *In* Marine mortality of Atlantic salmon, *Salmo salar* L: methods and measures, pp. 162-197. Potter, E.C.E., N. Ó Maoiléidigh, and G. Chaput [eds.] Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2003/101. 213 pp.

Chadwick, E. M. P. 1988. Relationship between Atlantic salmon smolts and adults in Canadian waters. *In* Atlantic salmon: planning for the future, pp. 301-324. Mills, D., and D. Piggins [eds.] Croom Helm, London.

Chadwick, E. M. P., and J. M. Green. 1985. Atlantic salmon (*Salmo salar*) production in a largely lacustrine Newfoundland watershed. Verhandlungen Internationale Vereinigung für Theoretische und Angewandte Limnologie 22: 2509-2525.

Chadwick, E.M.P., R. R. Claytor, C. E. Léger, and R. L. Saunders. 1987. Inverse correlation between ovarian development of Atlantic salmon (*Salmo salar*) smolts and sea age. Canadian Journal of Fisheries and Aquatic Sciences 44: 1320-1325.

Chaput, G. 2003. Estimation of mortality for Atlantic salmon (*Salmo salar* L.). *In* Marine mortality of Atlantic salmon, *Salmo salar* L: methods and measures, pp. 59-82. Potter, E.C.E., N. Ó Maoiléidigh, and G. Chaput [eds.] Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2003/101. 213 pp.

Chaput G., J. Allard, F. Caron, J. B. Dempson, C. C. Mullins, and M. F. O'Connell. 1998. River-specific target spawning requirements for Atlantic salmon (*Salmo salar*) based on a generalized smolt production model. Canadian Journal of Fisheries and Aquatic Sciences 55: 246-261.

Chaput, G., and R. Jones. 2006. Replacement ratios and rebuilding potential for two multi-sea-winter-salmon stocks of the Maritime provinces. Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2006/027.

Chaput, G., D. Moore, J. Hayward, J. Sheasgreen, and B. Dubee. 2001. Stock Status of Atlantic Salmon (*Salmo salar*) in the Miramichi River, 2000. Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2001/008. 88 pp.

Crozier, W. W., and G.J.A. Kennedy. 1994. Marine exploitation of Atlantic salmon (*Salmo salar* L.) from the River Bush, Northern Ireland. Fisheries Research 19:141-155.

Cunjak, R. A., D. Caissie, N. El-Jabi, P. Hardie, J. H. Conlon, T. L. Pollock, D. J. Giberson, and S. Komadina-Douthwright. 1993. The Catamaran Brook (New Brunswick) habitat research project: biological, physical, and chemical conditions (1990-1992). Canadian Technical Report of Fisheries and Aquatic Sciences 1914. 81 pp.

Cunjak, R., T. D. Prowse, and D. L. Parrish. 1998. Atlantic salmon (*Salmo salar*) in winter: “the season of parr discontent”. Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1): 161-180.

Dalley, E. L., C. W. Andrews, and J. M. Green. 1983. Precocious male Atlantic salmon parr (*Salmo salar*) in insular Newfoundland. Canadian Journal of Fisheries and Aquatic Sciences 40: 647-652.

Dempson, J. B., G. Furey, and M. Bloom. 2001. Assessment of the status of the Atlantic salmon stock of Conne River, SFA 11, Newfoundland, 2000. Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2001/030. 45 pp.

Dempson, J. B., G. Furey, and M. Bloom. 2004b. Status of Atlantic salmon, *Salmo* salar, in Conne River, SFA 11, Newfoundland, 2003. Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2004/057. 31 pp.

Dempson, J. B., C. C. Mullins, C. E. Bourgeois, M. F. O'Connell, and D. G. Reddin. 2003. Perspectives on smolt production and marine survival of Newfoundland Atlantic salmon (*Salmo salar* L.) related to smolt size and run timing. *In*: Marine mortality of Atlantic salmon, *Salmo salar* L: methods and measures, pp. 27-43. E.C.E., Potter, N. Ó Maoiléidigh, and G. Chaput [eds.] Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2003/101. 213 pp.

Dempson, J. B., M. F. O'Connell, and C. J. Schwarz. 2004c. Spatial and temporal trends in abundance of Atlantic salmon, *Salmo salar*, in Newfoundland with emphasis on impacts of the 1992 closure of the commercial fishery. Fisheries Management and Ecology 11: 387-402.

Dempson, J. B., M. F. O'Connell, and M. Shears. 1996. Relative production of Atlantic salmon from fluvial and lacustrine habitats estimated from analyses of scale characteristics. Journal of Fish Biology 48: 329-341.

Dempson, J. B., V. A. Pepper, G. Furey, M. Bloom, T. Nicholls, and G. Hoskins. 1999. Evaluation of an alternative strategy to enhance salmon populations: cage rearing wild smolts from Conne River, Newfoundland. ICES Journal of Marine Science 56: 422-432.

Dempson, J. B., D. G. Reddin, M. F. O'Connell, J. Helbig, C. E. Bourgeois, C. Mullins, T. R. Porter, G. Lilly, J. Carscadden, G. B. Stenson, and D. Kulka. 1998. Spatial and temporal variation in Atlantic salmon abundance in the Newfoundland-Labrador region with emphasis on factors that may have contributed to low returns in1997. Department of Fisheries and Oceans Canadian Stock Assessment Secretariat Research Document 98/114. 161 pp.

Dempson, J. B., C. J. Schwarz, D. G. Reddin, M. F. O'Connell, C. C. Mullins, and C. E. Bourgeois. 2001a. Estimation of marine exploitation rates on Atlantic salmon (*Salmo salar* L.) stocks in Newfoundland, Canada. ICES Journal of Marine Science 58: 331-341.

Dempson, J. B., C. J. Schwarz, M. Shears, and G. Furey. 2004a. Comparative proximate body composition of Atlantic salmon with emphasis on parr from fluvial and lacustrine habitats. Journal of Fish Biology 64: 1257-1271.

Doubleday, W. G., D. R. Rivard, J. A. Ritter, and K. U. Vickers. 1979. Natural mortality rate estimates for North Atlantic salmon in the sea. ICES C.M. 1979/M:26. 15 pp.

Downton, P. R., D. G. Reddin, and R. W. Johnson. 2001. Status of Atlantic salmon (*Salmo salar* L.) in Campbellton River, Notre Dame Bay (SFA 4), Newfoundland in 2000. Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2001/031. 73 pp.

Ducharme, L.J.A. 1969. Atlantic salmon returning for their fifth and sixth consecutive spawning trips. Journal of the Fisheries Research Board of Canada 26: 1661-1664.

Duston, J., T. Astatkie, and P. F. MacIsaac. 2005. Genetic influence of parr versus anadromous sires on the life histories of Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 62: 2067-2075.

Dwyer, W. P., and R. G. Piper. 1987. Atlantic salmon growth efficiency as affected by temperature. Progressive Fish Culturist 49: 57-59.

Einum, S., and I. A. Fleming. 2004. Environmental unpredictability and offspring size: conservative versus diversified bet-hedging. Evolutionary Ecology Research 6: 443-455.

Elliott, J. M. 1991. Tolerance and resistance to thermal stress in juvenile Atlantic salmon, *Salmo salar*. Freshwater Biology 25: 61-70.

Elliott, J. M., and M. A. Hurley. 1997. A functional model for maximum growth of Atlantic salmon parr, *Salmo salar*, from two populations in northwest England. Functional Ecology 11: 592-603.

Elliott, J. M. 2001. The relative role of density in the stock-recruitment relationship of salmonids. *In* Stock, recruitment and reference points : assessment and management of Atlantic salmon, pp. 25-66. Prevost, E., and G. Chaput [eds.] INRA Editions.

Ellner, S., and N. G. Hairston, Jr. 1994. Role of overlapping generations in maintaining genetic variation in a fluctuating environment. American Naturalist 143: 403-417.

Elson, P. F. 1957. Using hatchery reared Atlantic salmon to best advantage. Canadian Fish Culturist 21: 7-17.

Erkinaro, J., and R. J. Gibson. 1997a. Interhabitat migration of juvenile Atlantic salmon in a Newfoundland river system, Canada. Journal of Fish Biology 51: 373-388.

Erkinaro, J., and R. J. Gibson. 1997b. Movements of Atlantic salmon, *Salmo salar* L., parr and brook trout, *Salvelinus fontinalis* (Mitchill), in lakes, and their impact on single-census population estimation. Fisheries Management and Ecology 4: 369-384.

Finstad, B., and N. Jonsson. 2001. Factors influencing the yield of smolt releases in Norway. Nordic Journal of Freshwater Research 75: 37-55.

Fleming, I. A. 1996. Reproductive strategies of Atlantic salmon: ecology and evolution. Reviews in Fish Biology 6: 379-416.

Fleming, I. A., and J. D. Reynolds. 2004. Salmonid breeding systems. *In* Evolution illuminated: salmon and their relatives, pp. 264-294. Henry, A. P., and S. C. Stearns [eds.] Oxford University Press, Oxford, New York.

Forsythe, M. G. 1967. Analysis of the 1965 smolt run in the Northwest Miramichi River, New Brunswick. Fish. Res. Oard Can. Tech Rep. 4. 73 pp.

Friedland, K. D., and R. E. Haas. 1996. Marine post-smolt growth and age at maturity of Atlantic salmon. Journal of Fish Biology 48: 1-15.

Gibson, R. J. 1966. Some factors influencing the distributions of brook trout and young Atlantic salmon. Journal of the Fisheries Research Board of Canada 23: 1977-1980.

Gibson, R. J. 1978. The behaviour of juvenile Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) with regard to temperature and to water velocity. Transactions of the American Fisheries Society 107, 703-712.

Gibson, R. J. 1988. Mechanisms regulating species composition, population structure, and production of stream salmonids; a review. Poliskie Archiwum Hydrobiologii 35: 469-495.

Gibson, R. J. 1993. The Atlantic salmon in freshwater: spawning, rearing and production. Reviews in Fish Biology and Fisheries 3: 39-73.

Gibson, R. J. 2002. The effects of fluvial processes and habitat heterogeneity on distribution, growth and densities of juvenile Atlantic salmon (*Salmo salar* L.), with consequences on the abundance of the adult fish. Ecology of Freshwater Fish 11: 207-222.

Gibson, R. J., D. E. Stansbury, R. R. Whalen, and K. G. Hillier. 1993. Relative habitat use, and inter-specific and intra-specific competition of brook trout (*Salvelinus fontinalis*) and juvenile Atlantic salmon (*Salmo salar*) in some Newfoundland rivers. *In* Production of juvenile Atlantic salmon, *Salmo salar*, in natural waters, pp. 53-69. Gibson, R. J., and R. E. Cutting [eds.] Canadian Special Publication of Fisheries and Aquatic Sciences 118. 262 pp.

Gibson, R. J., D. D. Williams, C. McGowan, and W. S. Davidson, W. S. 1996. The ecology of dwarf fluvial Atlantic salmon, *Salmo salar* L., cohabiting with brook trout, *Salvelinus fontinalis* (Mitchill), in southeastern Newfoundland, Canada. Polskie Archiwum Hydrobiologii 43: 145-166.

Grant, J.W.A., S.Ó. Steingrímsson, E.R. Keeley, and R.A. Cunjak. 1998. Implications of territory size for the measurement and prediction of salmonid abundance in streams. Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1): 181-190.

Hansen, L. P., and B. Jonsson. 1989. Salmon ranching experiments in the River Imsa: effect of timing of Atlantic salmon (*Salmo salar*) smolt migration on survival to adults. Aquaculture 82: 367-373.

Hansen, L. P. and B. Jonsson. 1991a. Evidence of a genetic component in the seasonal return pattern of Atlantic salmon, *Salmo salar* L. Journal of Fish Biology 38: 251-258.

Hansen, L. P., and B. Jonsson. 1991b. The effect of timing of Atlantic salmon smolt and post-smolt release on the distribution of adult return. Aquaculture 98: 61-67.

Hansen, L. P., and T. P. Quinn. 1998. The marine phase of the Atlantic salmon (*Salmo salar*) life cycle, with comparisons to Pacific salmon. Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1): 104-118.

Hargreaves, N. B. 1994. Processes controlling behaviour and mortality of salmonids during the early sea life period in the ocean. Nordic Journal of Freshwater Research 69: 100.

Harris, G. S. 1973. Rearing smolts in mountain lakes to supplement salmon stocks. International Atlantic Salmon Foundation Special Publication Series 4: 237-252.

Hasler, A. D., and A. T. Scholz. 1983. Olfactory imprinting and homing in salmon. Springer-Verlag, Berlin. 134 pp.

Heggenes, J., and R. Borgstrøm. 1991. Effect of habitat types on survival, spatial distribution and production of an allopatric cohort of Atlantic salmon, *Salmo salar* L., under conditions of low competition. Journal of Fish Biology 38: 267-280.

Heinimaa, S., and J. Erkinaro. 2004. Characteristics of mature male parr in the northernmost Atlantic salmon populations. Journal of Fish Biology 64: 219-226.

Hindar, K., and J. Nordland. 1989. A female salmon, *Salmo salar* L., maturing sexually in the parr stage. Journal of Fish Biology 35: 461-463.

Hutchings, J. A. 1985. The adaptive significance of lakeward migrations by juvenile Atlantic salmon, *Salmo salar* L. M.Sc. thesis. St. John's, Newfoundland: Memorial University of Newfoundland. 132 pp.

Hutchings, J. A. 1986. Lakeward migrations by juvenile Atlantic salmon, *Salmo salar*. Canadian Journal of Fisheries and Aquatic Sciences 43: 732-741.

Hutchings, J. A. and M. E. B. Jones. 1998. Life history variation and growth rate thresholds for maturity in Atlantic salmon, *Salmo salar*. Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1): 22-47.

Hutchings, J. A., and R. A. Myers. 1994. The evolution of alternative mating strategies in variable environments. Evolutionary Ecology 8: 256-268.

Javaid, M. Y., and J. M. Anderson. 1967. Thermal acclimation and temperature selection in Atlantic salmon, *Salmo salar*, and rainbow trout, *S. gairdneri*. Journal of the Fisheries Research Board of Canada 24: 1507-1513.

Jensen, A. J., B. O. Johnsen, and L. Saksgard. 1989. Temperature requirements in Atlantic salmon (*Salmo salar*), brown trout (*Salmo trutta*), and Arctic Char (*Salvelinus alpinus*) from hatching to initial feeding compared with geographic distribution. Canadian Journal of Fisheries and Aquatic Sciences 46: 786-789.

Jessop, B. M. 1975. Investigation of the salmon (*Salmo salar*) smolt migration of the Big Salmon River, New Brunswick, 1966-72. Environment Canada Fisheries and Marine Service, Technical Report Series No. MAR/T-75-1.

Jonsson, N., B. Jonsson, and I. A. Fleming. 1996. Does early growth cause a phenotypically plastic response in egg production of Atlantic salmon? Functional Ecology 10: 89-96.

Jonsson, N., B. Jonsson, and L.P. Hansen. 1998. Density-dependent and density-independent relationships in the life-cycle of Atlantic salmon, *Salmo salar*. J. Animal Ecology 67: 751-762.

Juanes, F., B. H. Letcher, and G. Gries. 2000. Ecology of stream fish: insights gained from an individual-based approach to juvenile Atlantic salmon. Ecology of Freshwater Fish 9: 65-73.

Keenleyside, M. H. A. 1962. Skin-diving observations of Atlantic salmon and brook trout in the Miramichi River, New Brunswick. Journal of the Fisheries Research Board of Canada 19: 625-634.

Klemetsen, A., P.-A. Amundsen, J. B. Dempson, B. Jonsson, M. F. O'Connell, and E. Mortensen. 2003. Atlantic salmon *Salmo salar* L., brown trout *Salmo trutta* L. and Arctic charr *Salvelinus alpinus* (L.): a review of aspects of their life histories. Ecology of Freshwater Fish 12: 1-59.

Lacey, E. P., L. R. Real, J. Antonovics, and D. G. Heckel. 1983. Variance models in the study of life histories. American Naturalist 122: 114-131.

Locke, A. 1998. Modeling the effects of poststocking survival rates on the success of stocking hatchery Atlantic salmon in a New Brunswick river. North American Journal of Fisheries Management 18: 547-560.

Mangel, M. 1994. Climate change and salmonid life history variation. Deep Sea Research II 41: 75-106.

Marschall, E. A., T. P. Quinn, D. A. Roff, J. A. Hutchings, N. B. Metcalfe, T. A. Bakke, R. L. Saunders, and N. L. Poff. 1998. A framework for understanding Atlantic salmon (*Salmo salar*) life history. Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1): 48-58.

Maynard Smith, J. 1982. Evolution and the theory of games. Cambridge: Cambridge University Press. 224 pp.

McCormick, S. D., L. P. Hansen, T. P. Quinn, and R. L. Saunders. 1998. Movement, migration, and smolting of Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1): 77-92.

Meerburg, D. J. [ed.]. 1986. Salmonid age at maturity. Canadian Special Publication of Fisheries and Aquatic Sciences 89. 118 pp.

Metcalfe, N. B. 1994. The role of behaviour in determining salmon growth and development. Aquaculture and Fisheries Management 25: 67-76.

Metcalfe, N. B., and J. E. Thorpe. 1990. Determinants of geographic variation in the age of seaward-migrating salmon, *Salmo salar*. Journal of Animal Ecology 59: 135-145.

Moore, D. S., G. Chaput, and R. Pickard. 1995. The effect of fisheries on the biological characteristics and survival of mature Atlantic salmon (*Salmo salar*) from the Miramichi River. *In* Water, science, and the public: the Miramichi ecosystem, pp. 229-247. E.M.P. Chadwick [ed.] Canadian Special Publication of Fisheries and Aquatic Sciences 123. 300 pp.

Moore, A., and W. D. Riley. 1992. A sexually mature female Atlantic salmon, *Salmo salar* L., smolt. Aquaculture and Fisheries Management 23: 273-274.

Morin, R. 1991. Atlantic salmon (*Salmo salar*) in the lower Nastapoka River, Quebec: distribution and origins of salmon in eastern Hudson Bay. Canadian Journal of Zoology 69: 1674-1681.

Morrison, B.R.S. 1983. Observations on the food of juvenile Atlantic salmon, *Salmo salar* L., reared in a Scottish hill loch. Journal of Fish Biology 23: 305-313.

Myers, R. A. 1984. Demographic consequences of precocious maturation of Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 41: 1349-1353.

Myers, R. A., J. A. Hutchings, and R. J. Gibson. 1986. Variation in male parr maturation within and among populations of Atlantic salmon. Canadian Journal of Fisheries and Aquatic Sciences 43: 1242-1248.

Myers, R. A., and J. A. Hutchings. 1987. Mating of anadromous Atlantic salmon parr, *Salmo salar* L., with mature male parr. Journal of Fish Biology 31: 143-146.

Noakes, D. J., D. W. Welch, M. Henderson, and E. Mansfield. 1990. A comparison of pre-season forecasting methods for returns of two British Columbia sockeye salmon stocks. North American Journal of Fisheries Management 10: 46-57.

O'Connell, M. F., and E.G.M. Ash. 1989. Atlantic salmon (*Salmo salar*) smolt production in a Newfoundland river system characterized by lacustrine habitat. Internationale Revue der Gesamten Hydrobiologie 74: 73-82.

O'Connell, M. F., and E.G.M. Ash. 1993. Smolt size in relation to age at first maturity of Atlantic salmon (*Salmo salar*): the role of lacustrine habitat. Journal of Fish Biology 42: 551-569.

O'Connell, M. F., and J. B. Dempson. 1996. Spatial and temporal distributions of salmonids in two ponds in Newfoundland, Canada. Journal of Fish Biology 48: 738-757.

O'Connell, M. F., J. B. Dempson, and D. G. Reddin. 1997. Inter-annual and inter-river variability in fecundity in Atlantic salmon (*Salmo salar* L.) in Newfoundland region rivers. Department of Fisheries and Oceans Canadian Stock Assessment Secretariat Research Document 97/94. 33 pp.

O'Connell, M. F., and R. J. Gibson. 1989. The maturation of anadromous female Atlantic salmon, *Salmo salar* L., stocked in a small pond in urban St. John's, Newfoundland, Canada. Journal of Fish Biology 34: 937-946.

O'Connell, M. F., A. Walsh, and N. M. Cochrane. 2001. Status of Atlantic salmon (*Salmo salar* L.) in Gander River, Newfoundland, 2000. Department of Fisheries and Oceans Canadian Stock Assessment Secretariat Research Document 2001/041. 63 pp.

Ó Maoiléidigh, N., E.C.E. Potter, P. McGinnity, W. W. Crozier, L. P. Hansen, G. Gudbergsson, E. Prévost, L. Karlsson, and J. MacLean. 2003. Examination of marine survival datasets for Atlantic salmon (*Salmo salar* L.) in the north east Atlantic area. *In* Marine mortality of Atlantic salmon, *Salmo salar* L: methods and measures, pp. 10-26. Potter, E.C.E., N. Ó Maoiléidigh, and G. Chaput [eds.] Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2003/101. 213 pp.

Orzack, S. H., and S. Tuljapurkar. 1989. Population dynamics in variable environments. VII. The demography and evolution of iteroparity. American Naturalist 133: 901-923.

Parrish, D. L., R. J. Behnke, S. R. Gephard, S. D. McCormick, and G. H. Reeves. 1998. Why aren't there more Atlantic salmon (*Salmo salar*)? Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1): 281-287.

Pepper, V. A. 1976. Lacustrine nursery areas for Atlantic salmon in insular Newfoundland. Fisheries and Marine Service Technical Report 671. xiii + 61 pp.

Pepper, V. A., N. P. Oliver, and R. Blunden. 1984. Lake surveys and biological potential for natural lacustrine rearing of juvenile Atlantic salmon (*Salmo salar*) in Newfoundland. Canadian Technical Report of Fisheries and Aquatic Sciences 1295. iv + 72 pp.

Pepper, V. A., N. P. Oliver, and R. Blunden. 1985. Juvenile anadromous Atlantic salmon of three lakes of Newfoundland. Internationale Revue der Gesamten Hydrobiologie 70: 733-753.

Peterson, R. H., and D. J. Martin Robichaud. 1989. First feeding of Atlantic salmon fry as influenced by temperature regime. Aquaculture 78: 35-53.

Porter, T. R., M. C. Healey, M. F. O'Connell, E. T. Baum, A. T. Bielak, and Y. Côté. 1986. Implications of varying sea age at maturity of Atlantic salmon (*Salmo salar*) on yield to the fisheries. *In* Salmonid age at maturity, pp. 110-117. D. J. Meerburg [ed.] Canadian Special Publication of Fisheries and Aquatic Sciences 89. 118 pp.

Potter, E. C. E., N. Ó Maoiléidigh, and G. Chaput [eds.]. 2003. Marine mortality of Atlantic salmon, *Salmo salar* L: methods and measures. Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2003/101. 213 pp.

Power, G. 1958. The evolution of the freshwater races of the Atlantic salmon (*Salmo salar* L.) in eastern North America. Arctic 11: 86-92.

Power, G. 1969. The salmon of Ungava Bay. Arctic Institute of North America. Technical Paper No. 22. 72 pp.

Power, G. 1981. Stock characteristics and catches of Atlantic salmon (*Salmo salar*) in Quebec, and Newfoundland and Labrador in relation to environmental variables. Canadian Journal of Fisheries and Aquatic Sciences 38: 1601-1611.

Power, G. 1986. Physical influences on age at maturity of Atlantic salmon (*Salmo salar*): a synthesis of ideas and questions. *In* Salmonid age at maturity, pp. 97-101. D. J. Meerburg [ed.] Canadian Special Publication of Fisheries and Aquatic Sciences 89. 118 pp.

Prouzet, P. 1981. Observations d'une femelle de tacon de saumon atlantique (*Salmo salar* L.) parvenu à maturité sexuelle en rivière. Bulletin Francais de la Pisciculture 282: 16-19.

Randall, R. G. 1982. Emergence, population densities, and growth of salmon and trout fry in two New Brunswick streams. Canadian Journal of Zoology 60: 2239-2244.

Randall, R. G. and U. Paim. 1982. Growth, biomass and production of juvenile Atlantic salmon (*Salmo salar*) in two Miramichi River, New Brunswick, tributary streams. Canadian Journal of Zoology 60: 1647-1659

Randall, R.G., and Chadwick, E.M.P. 1986. Density as a factor affecting the production of juvenile Atlantic salmon (*Salmo salar*) in the Miramichi and Restigouche rivers, New Brunswick. Polskie Archiwum Hydrobiologii 33: 391-409.

Randall, R. G., Thorpe, J. E., Gibson, R. J. & Reddin, D. G. 1986. Biological factors affecting age at maturity in Atlantic salmon (*Salmo salar*). *In* Salmonid age at maturity, pp. 90-96. D. J. Meerburg [ed.] Canadian Special Publication of Fisheries and Aquatic Sciences 89. 118 pp.

Randall, R. G. 1989. Effect of sea-age on the reproductive potential of Atlantic salmon (*Salmo salar*) in eastern Canada. Canadian Journal of Fisheries and Aquatic Sciences 46: 2210-2218.

Riddell, B. E., and W. C. Leggett. 1981. Evidence of an adaptive basis for geographic variation in body morphology and time of downstream migration of juvenile Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 38: 308-320.

Ritter, J. A. 1989. Marine migration and natural mortality of North American Atlantic salmon (*Salmo salar* L.). Canadian Manuscript Report of Fisheries and Aquatic Sciences No. 2041. 136 pp.

Robitaille, J. A., Y. Côté, G. Shooner, and G. Hayeur. 1986. Growth and maturation patterns of Atlantic salmon of Atlantic salmon, *Salmo salar*, in Koksoak River, Ungava, Quebec. *In*

Salmonid age at maturity, pp. 62-69. D. J. Meerburg [ed.] Canadian Special Publication of Fisheries and Aquatic Sciences 89. 118 p.

Roff, D. A. 1992. The evolution of life histories. New York, London: Chapman & Hall. xii + 535 pp.

Rouleau, A., et G. Temblay. 1990. Détermination du nombre d'ovules par femelle chez le saumon Atlantique anadrome du Québec. *In* Compte rendu de l'atelier sur le nombre de reproducteurs requis dans les rivières à saumon, île aux Coudres, février 1988, pp. 154-167. Samson, N., et J. P. le Bel [éds.] Ministère du Loisir, de la Chasse et de la Pêche du Québec, Direction de la gestion des espèces et des habitats. 329 pp.

Ryan, P. M. 1986. Lake use by wild anadromous Atlantic salmon, *Salmo salar*, as an index of subsequent adult abundance. Canadian Journal of Fisheries and Aquatic Sciences 43: 2-11.

Ryan, P. M., M. F. O'Connell, and V. A. Pepper [eds.]. 1993. Report from the workshop on lake use by Atlantic salmon in Newfoundland, Canada. Canadian Manuscript Report of Fisheries and Aquatic Sciences 2222. iv + 54 pp.

Salminen, M., S. Kuikka, and E. Erkamo. 1995. Annual variability in survival of sea-ranched Baltic salmon, *Salmo salar* L.: significance of smolt size and marine conditions. Fisheries Management and Ecology 2: 171-184.

Saunders, R. L. 1981. Atlantic salmon (*Salmo salar*) stocks and management implications in the Canadian Atlantic Provinces and New England, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 38: 1612-1625.

Saunders, R. L., P. R. Harmon, and D. E. Knox. 1994. Smolt development and subsequent sexual maturation in previously mature male Atlantic salmon (*Salmo salar*). Aquaculture 121: 79-93.

Saunders, R. L., E. B. Henderson, and B. D. Glebe. 1982. Precocious sexual maturation and smoltification in male Atlantic salmon (*Salmo salar*). Aquaculture 28: 211-229.

Saunders, R. L., and C. B. Schom. 1985. Importance of the variation in life history parameters of Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 42: 615-618.

Schaffer, W. M. 1974. Selection for optimal life histories: the effects of age structure. Ecology 55: 291-303.

Scott, W. B., and E. J. Crossman. 1964. Fishes occurring in the fresh waters of insular Newfoundland. Ottawa: Department of Fisheries. iv + 124 pp.

Scott, W. B., and E. J. Crossman. 1998. Freshwater fishes of Canada. Oakville, Ontario: Galt House Publications Ltd. xx + 966 pp.

Stabell, O. B. 1982. Homing of Atlantic salmon in relation to olfaction and genetics. *In* Proceedings of the salmon and trout migratory behavior symposium, pp. 71-78. Brannon, E. L., and E. O. Salo [eds.] University of Washington, Seattle. 309 pp.

Stabell, O. B. 1984. Homing and olfaction in salmonids: a critical review with special reference to the Atlantic salmon. Biological Reviews 59: 333-388.

Stearns, S. C. 1976. Life-history tactics: a review of the ideas. Quarterly Review of Biology 51: 3-47.

Strothotte, E., G. J. Chaput and H. Rosenthalt. 2005. Seasonal growth of wild Atlantic salmon (*Salmo salar* L.) juveniles and implications on age at smoltification. Journal of Fish Biology 67: 1585-1602.

Sutterlin, A. M., and D. MacLean. 1984. Age at first maturity and early expression of oocyte recruitment processes in two forms of Atlantic salmon (*Salmo salar*) and their hybrids. Canadian Journal of Fisheries and Aquatic Sciences 41: 1139-1149.

Symons, P. 1979. Estimated escapement of Atlantic salmon (*Salmo salar*) for maximum smolt production in rivers of different productivity. Journal of the Fisheries Research Board of Canada 36:132-140.

Swansburg, E., G. Chaput, D. Moore, D. Caissie, and N. El-Jabi. 2002. Size variability of juvenile Atlantic salmon: links to environmental conditions. Journal of Fish Biology 61: 661-683.

Taylor, E. B. 1991. A review of local adaptation in Salmonidae, with particular reference to Pacific and Atlantic salmon. Aquaculture 98: 185-207.

Thorpe, J. E. 1986. Age at first maturity in Atlantic salmon, *Salmo salar*: freshwater period influences and conflicts with smolting. *In* Salmonid age at maturity, pp. 7-14. D. J. Meerburg [ed.] Canadian Special Publication of Fisheries and Aquatic Sciences 89. 118 pp.

Thorpe, J. E. 1987. Smolting versus residency: developmental conflicts in salmonids. American Fisheries Society Symposium 1: 244-262.

Thorpe, J. E. 1989. Developmental variation in salmonid populations. Journal of Fish Biology 35(Suppl. A): 295-303.

Thorpe, J. E. 1994. Reproductive strategies in Atlantic salmon, *Salmo salar*. Aquaculture and Fisheries Management 25: 77-87.

Thorpe, J. E. 1998. Salmonid life-history evolution as a constraint on marine stock enhancement. Bulletin of Marine Science 62: 465-475.

Thorpe, J. E., M. Mangel, N. B. Metcalfe, and F. A. Huntingford. 1998. Modelling the proximate basis of salmonid life-history variation, with application to Atlantic salmon, *Salmo salar* L. Evolutionary Ecology 12: 581-599.

Thorpe, J. E., M. S. Miles, and D. S. Keay. 1984. Developmental rate, fecundity and egg size in Atlantic salmon, *Salmo salar* L. Aquaculture 43: 289-305.

Verspoor, E., and L. J. Cole. 1989. Genetically distinct sympatric populations of resident and anadromous Atlantic salmon, *Salmo salar*. Canadian Journal of Zoology 67: 1453-1461.

Ward, B. R., and P. A. Slaney. 1988. Life history and smolt-to-adult survival of Keogh River steelhead trout (*Salmo gairdneri*) and the relationship to smolt size. Canadian Journal of Fisheries and Aquatic Sciences 45: 1110-1122.

Warner, K., and K. A. Havey. 1985. Life history, ecology and management of Maine landlocked salmon. Maine Department of Inland Fisheries and Wildlife. vii + 127 pp.

Whalen, K. G., D. L. Parrish, M. E. Mather, and J. R. McMenemy. 2000. Cross-tributary analysis of parr to smolt recruitment of Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 57: 1607-1616.

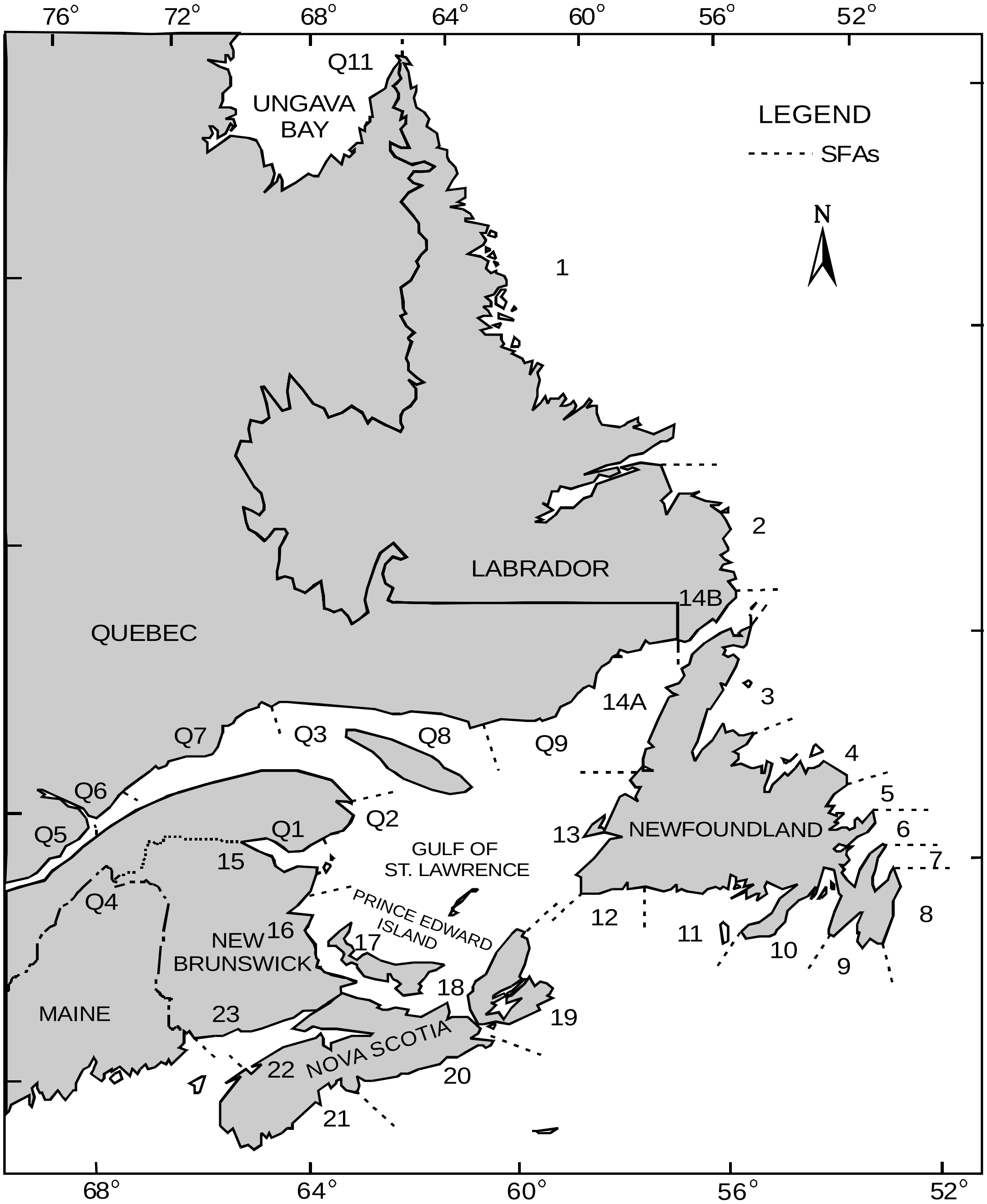


Figure 1. Map showing the Salmon Fishing Areas (SFAs) of Newfoundland and Labrador, Nova Scotia, New Brunswick, and Prince Edward Island, and Management Zones (Qs) of Quebec.

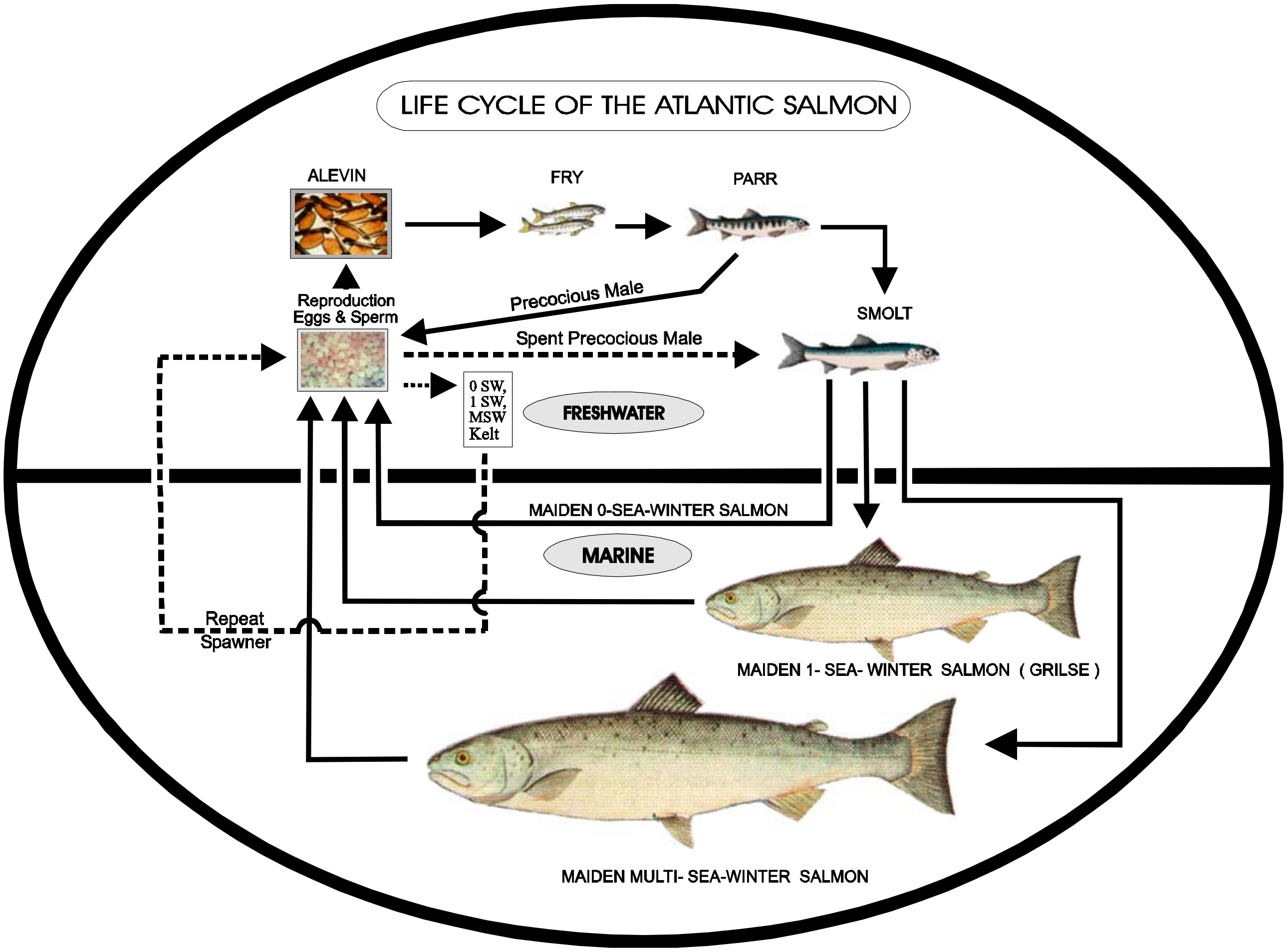
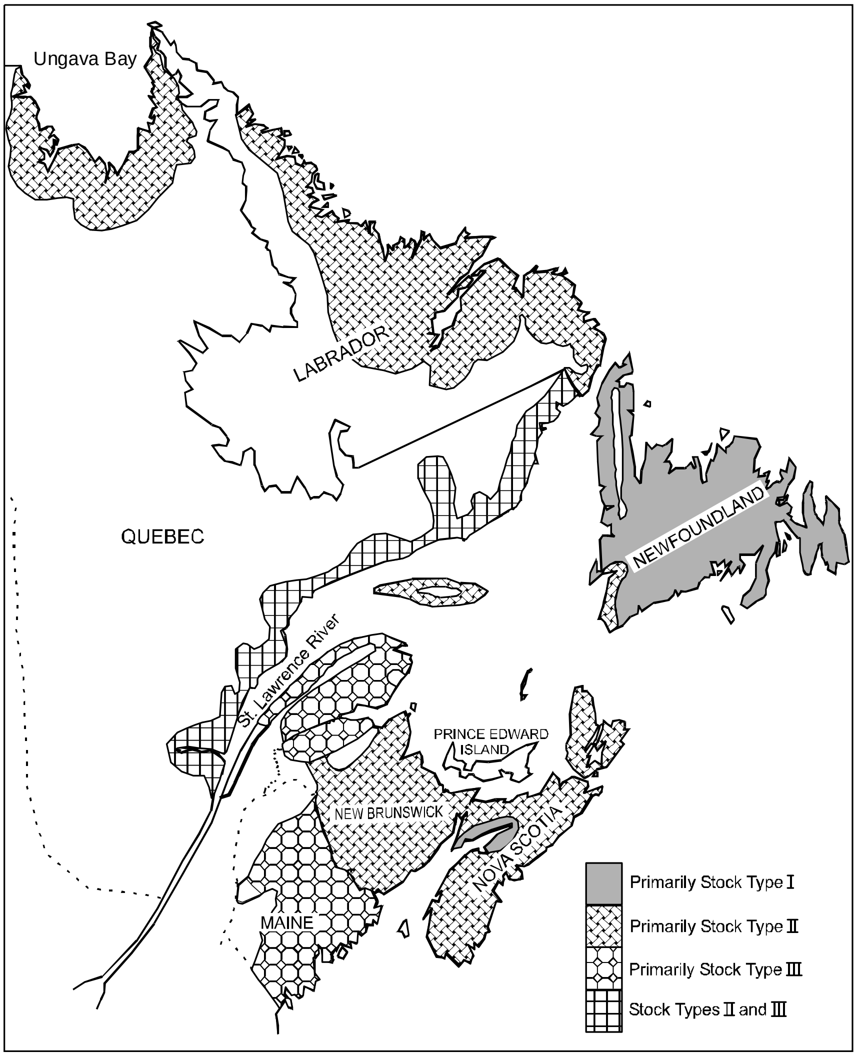


Figure 2. Generalized life cycle of the Atlantic salmon.



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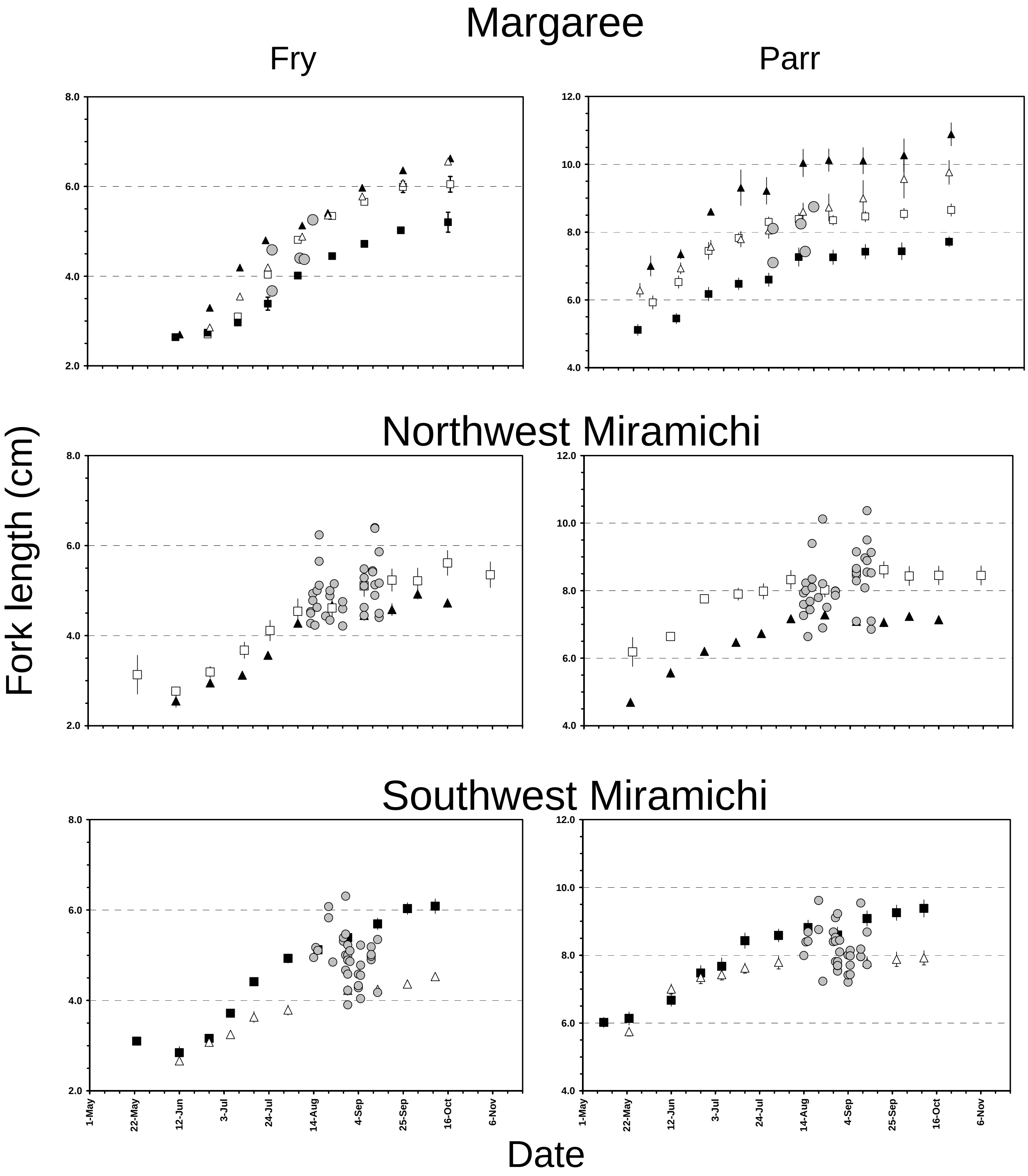

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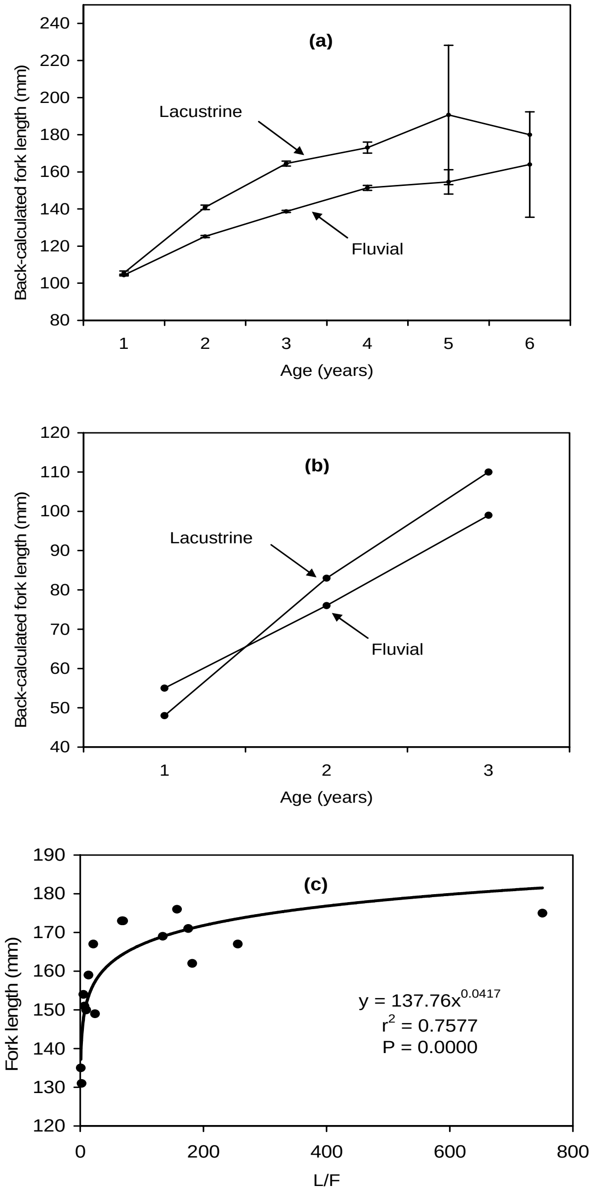

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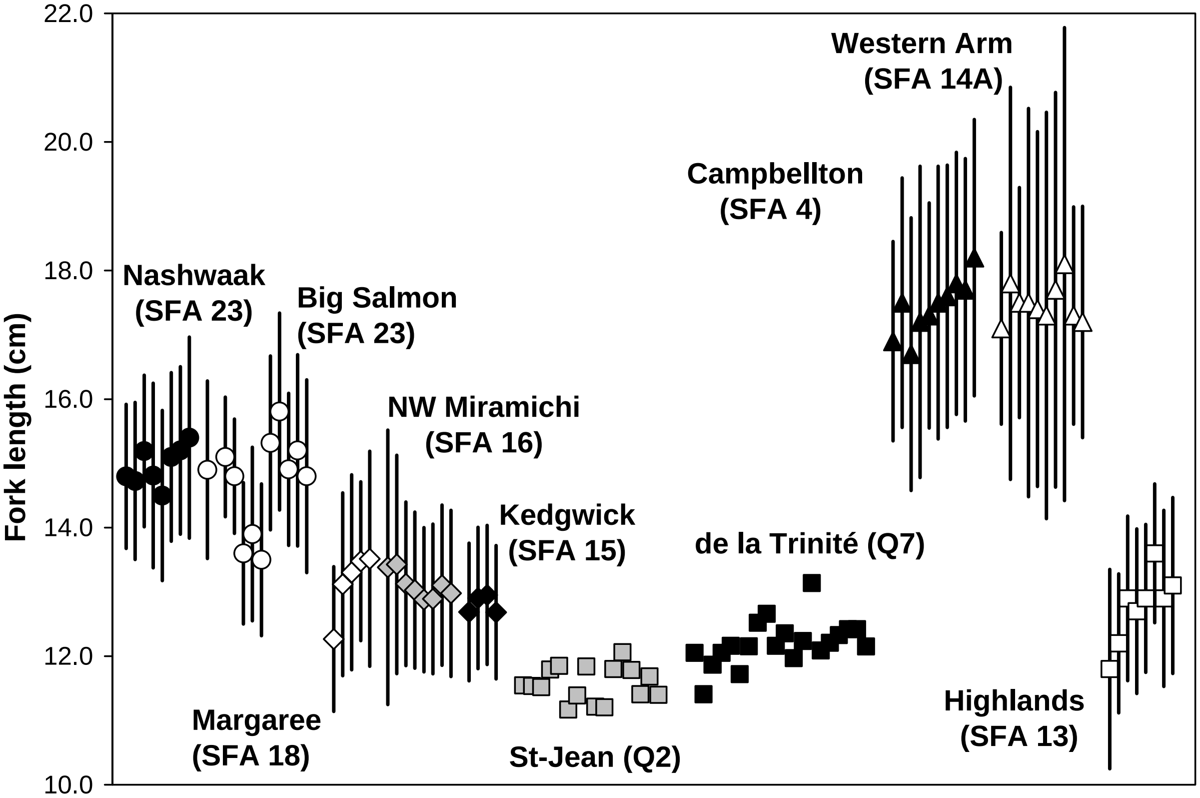

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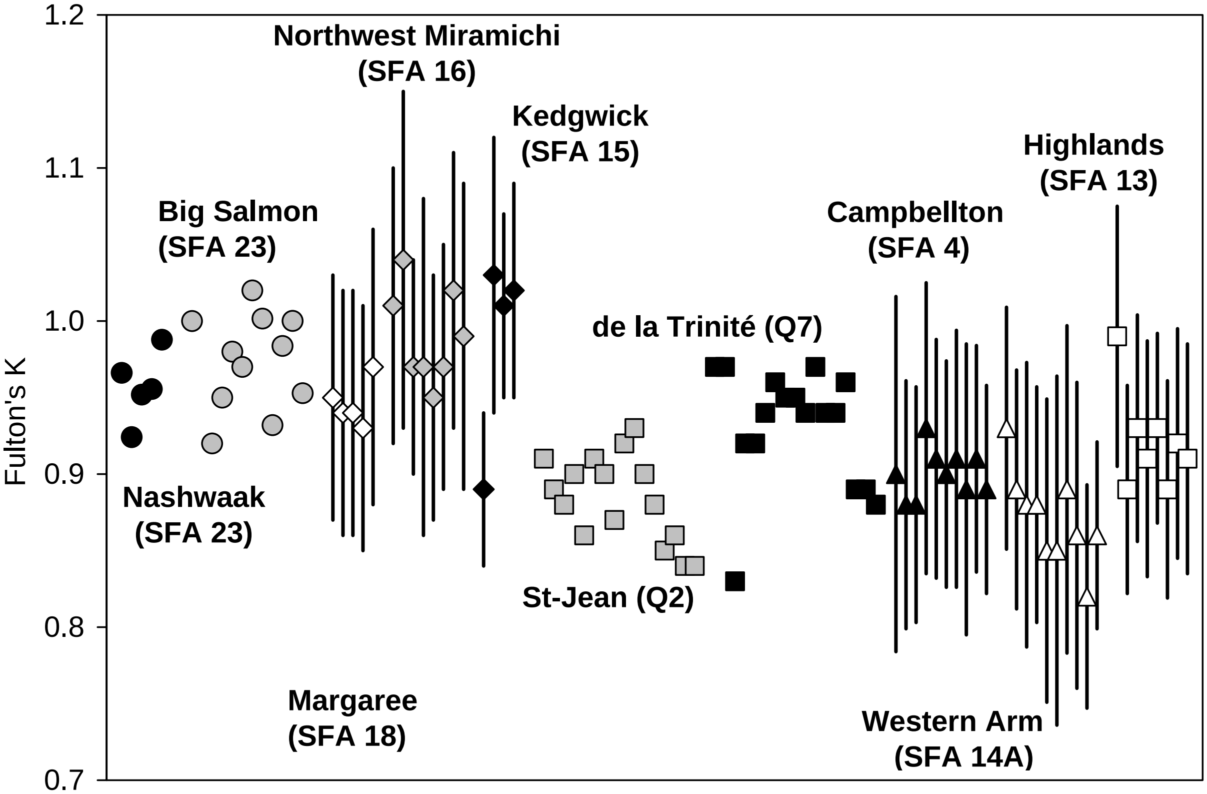
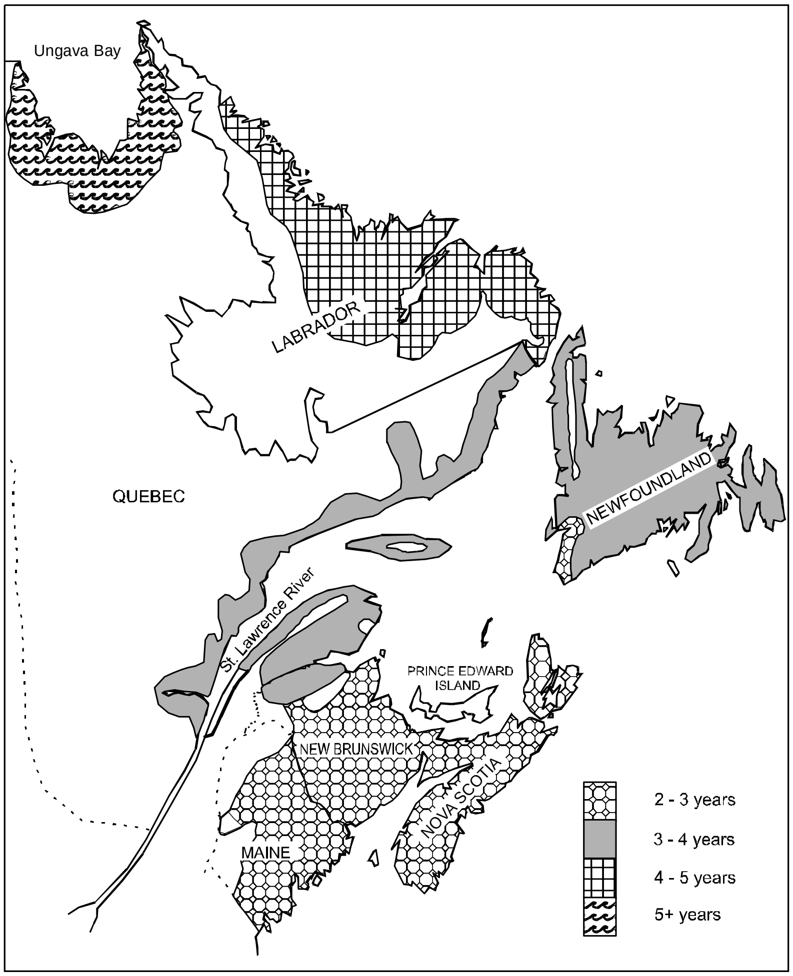


Figure 7. Condition factor (mean, ± 1 std. dev.) of wild Atlantic salmon smolts, expressed as Fulton's K in seven MSW salmon stocks of the mainland of eastern Canada and two 1SW (Campbellton and Western Arm) stocks and one MSW (Highlands) stock in Newfoundland (see caption for Figure 6 for further details on geographic location).



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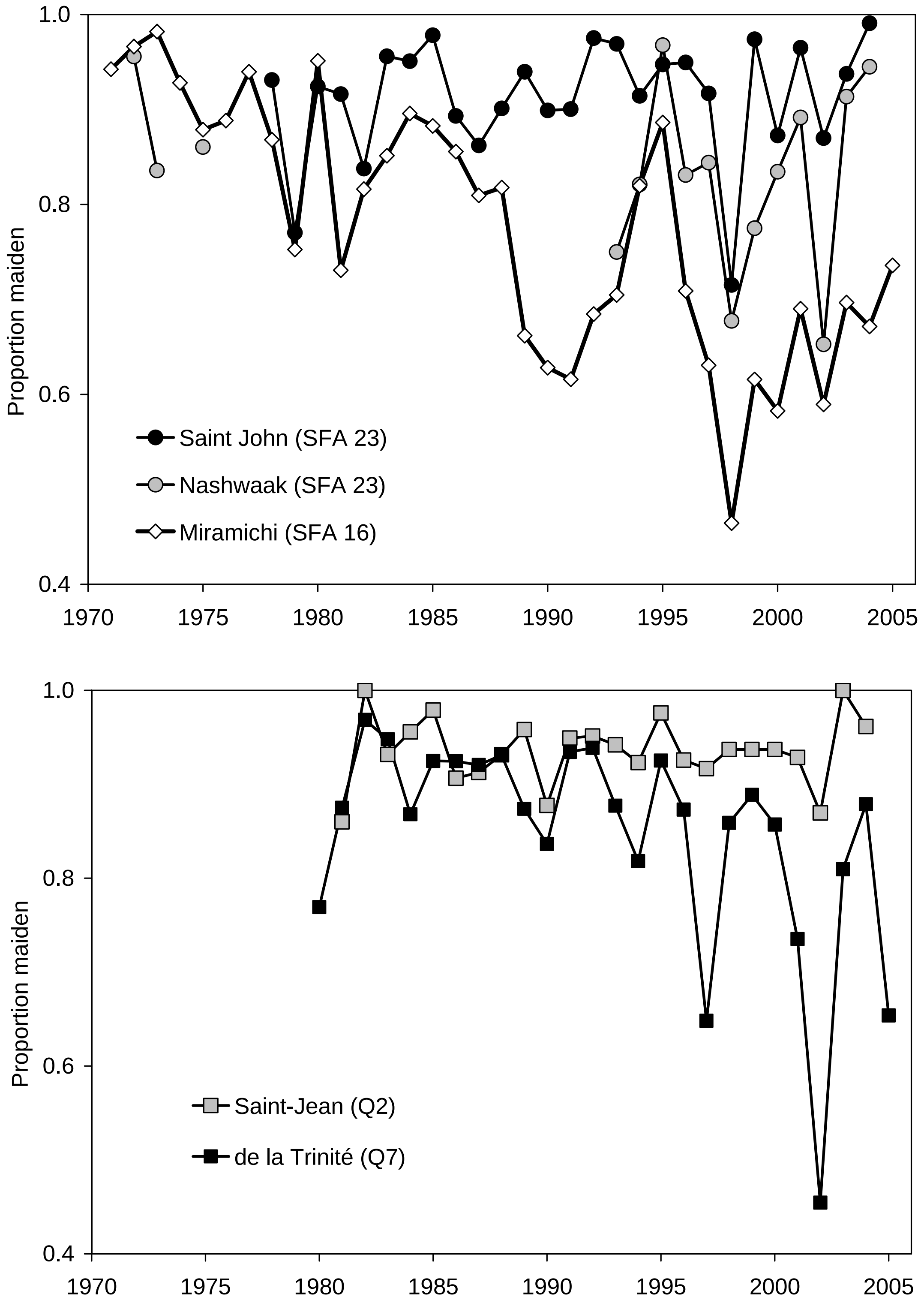

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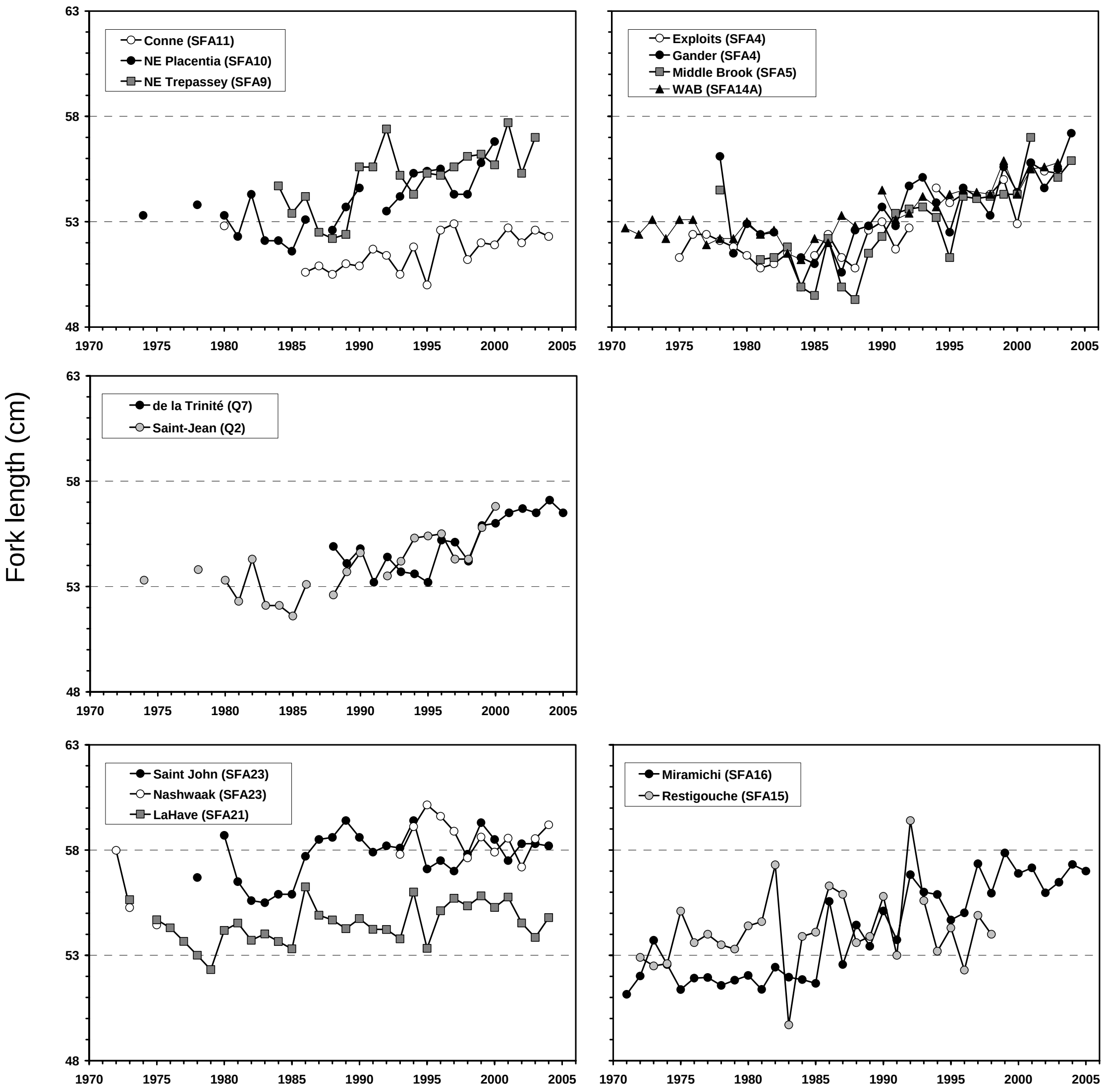

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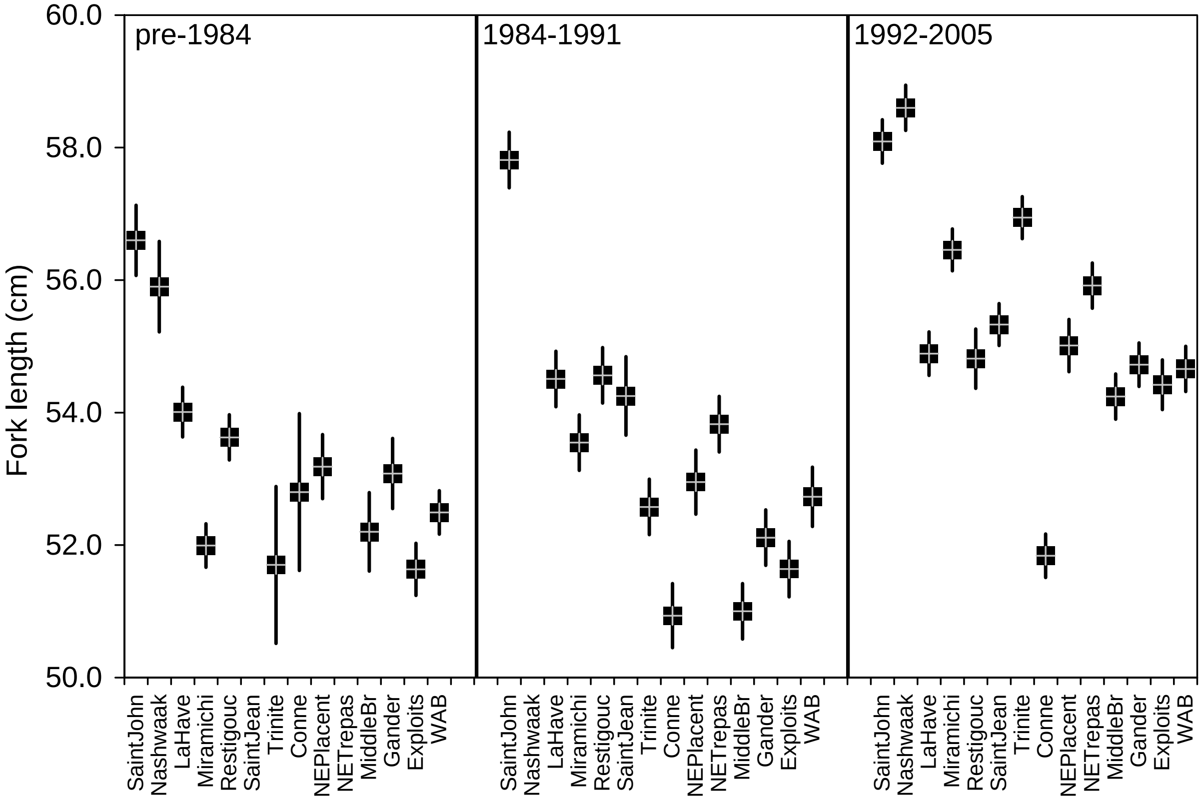
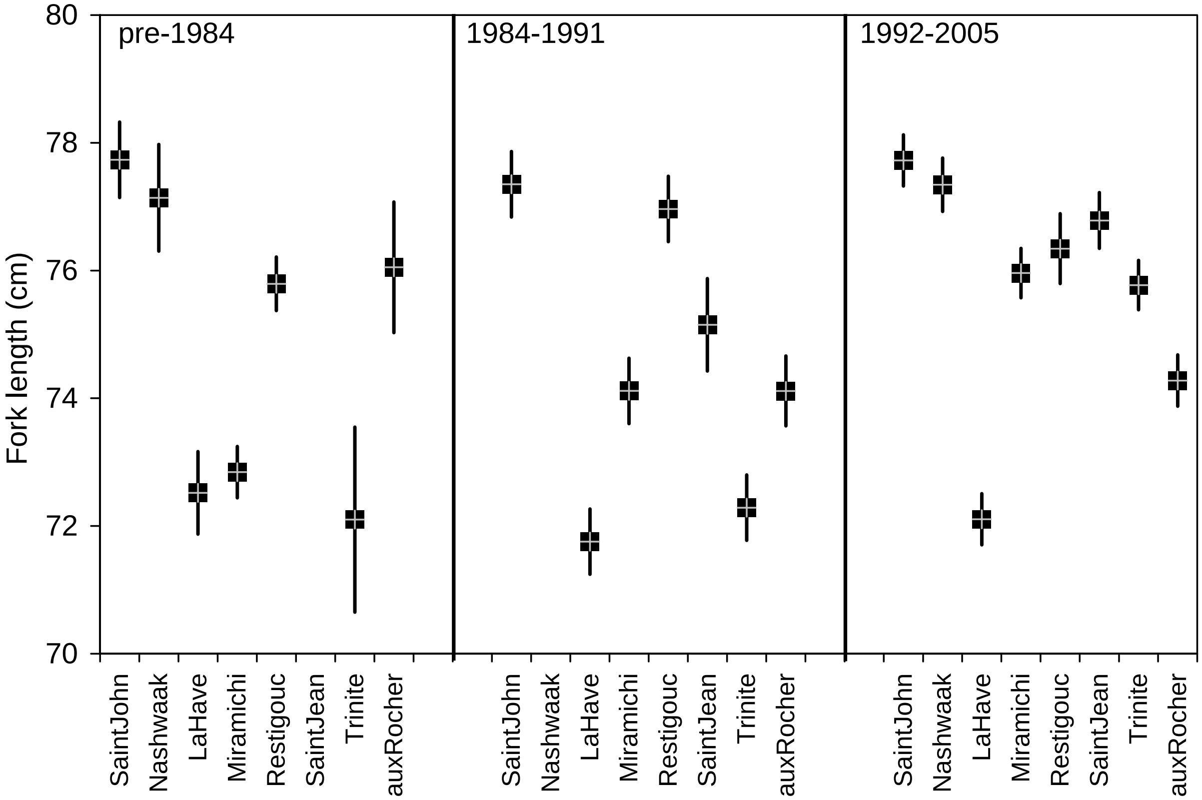


Figure 11. Mean fork length of 1SW maiden salmon from 14 selected rivers for three management periods. The management periods are: pre-1984 represents 1970 to 1983 when the commercial fisheries of the Maritime provinces and portions of Québec were closed or under quota management, 1984 to 1991 when the commercial fisheries of the Maritimes and portions of Québec were closed and Newfoundland commercial fisheries quotas were introduced, and post 1991 when the moratoria on salmon and cod commercial fisheries were introduced. Mean (± std. err.) of the annual values by river are shown.



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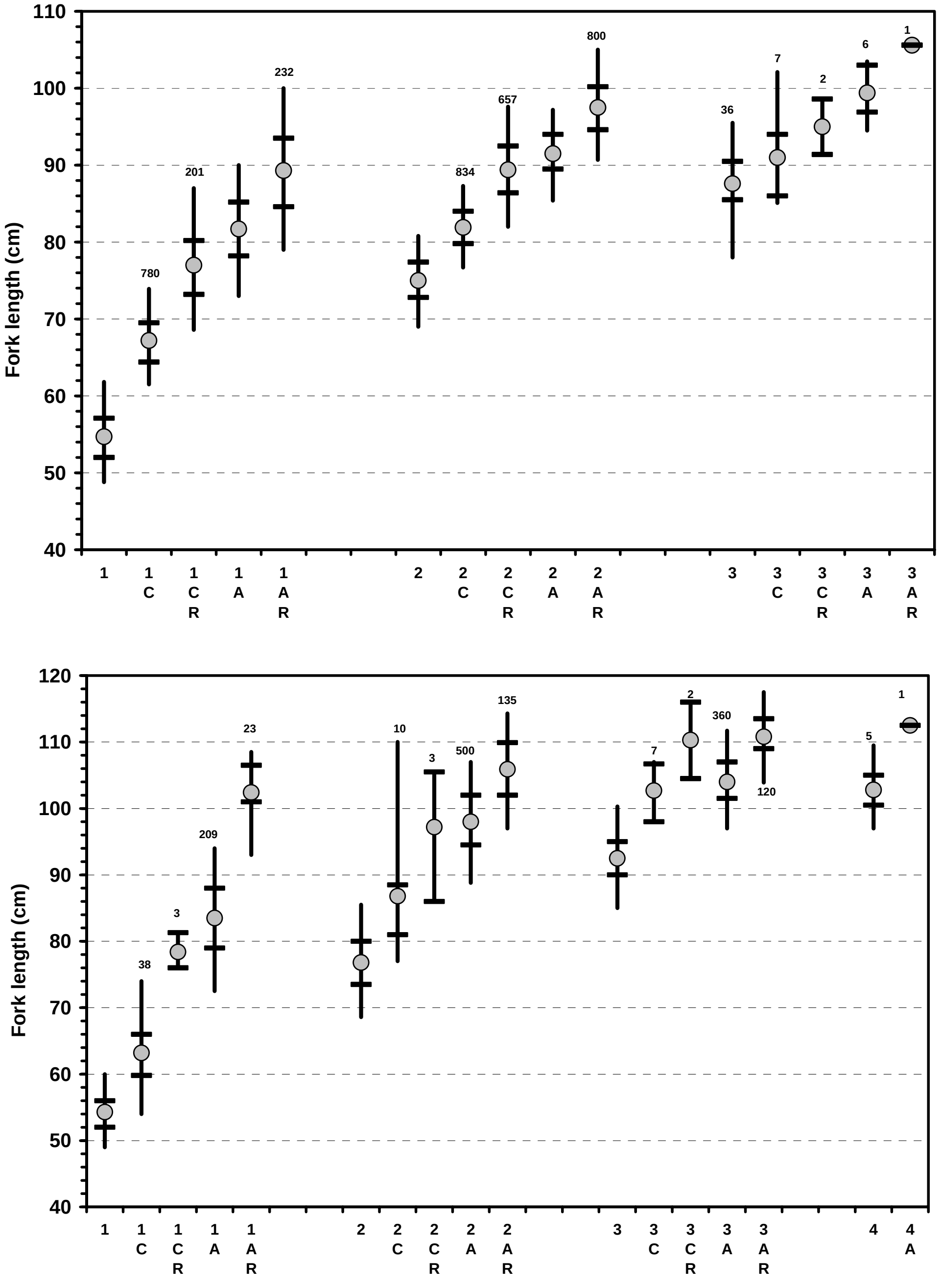

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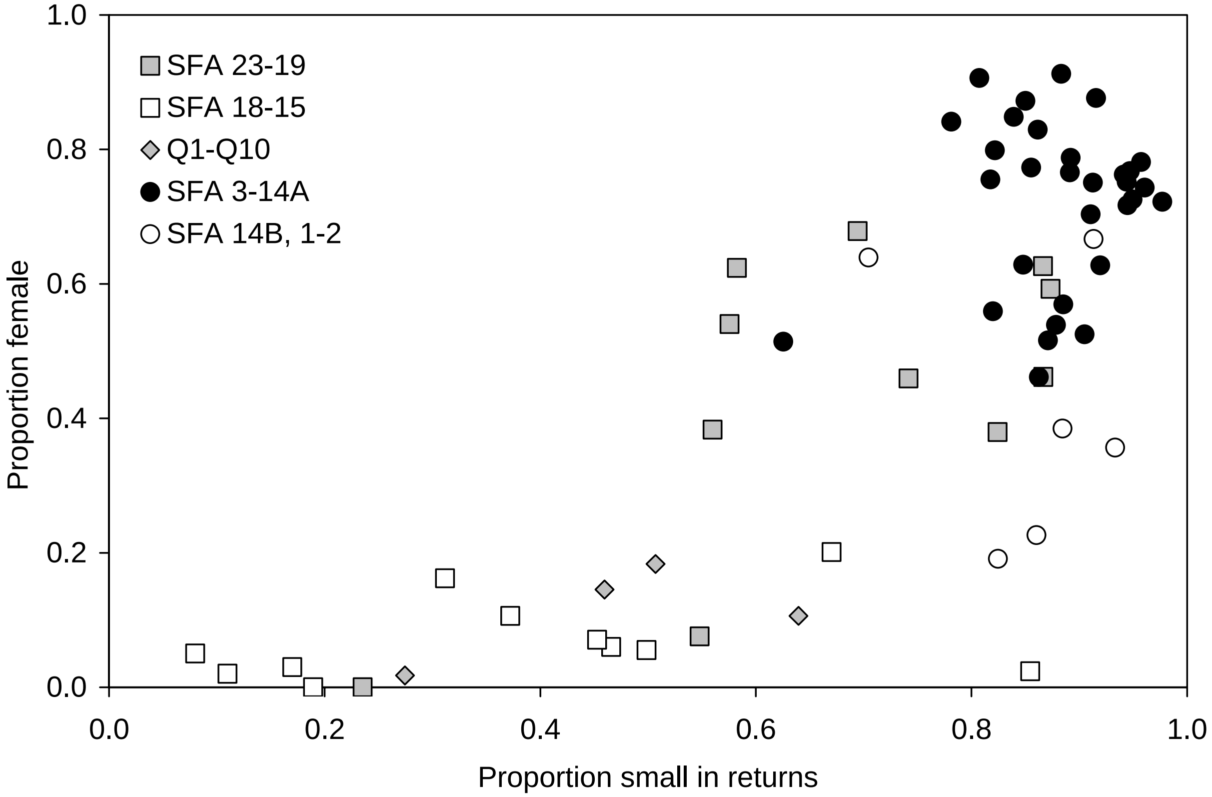

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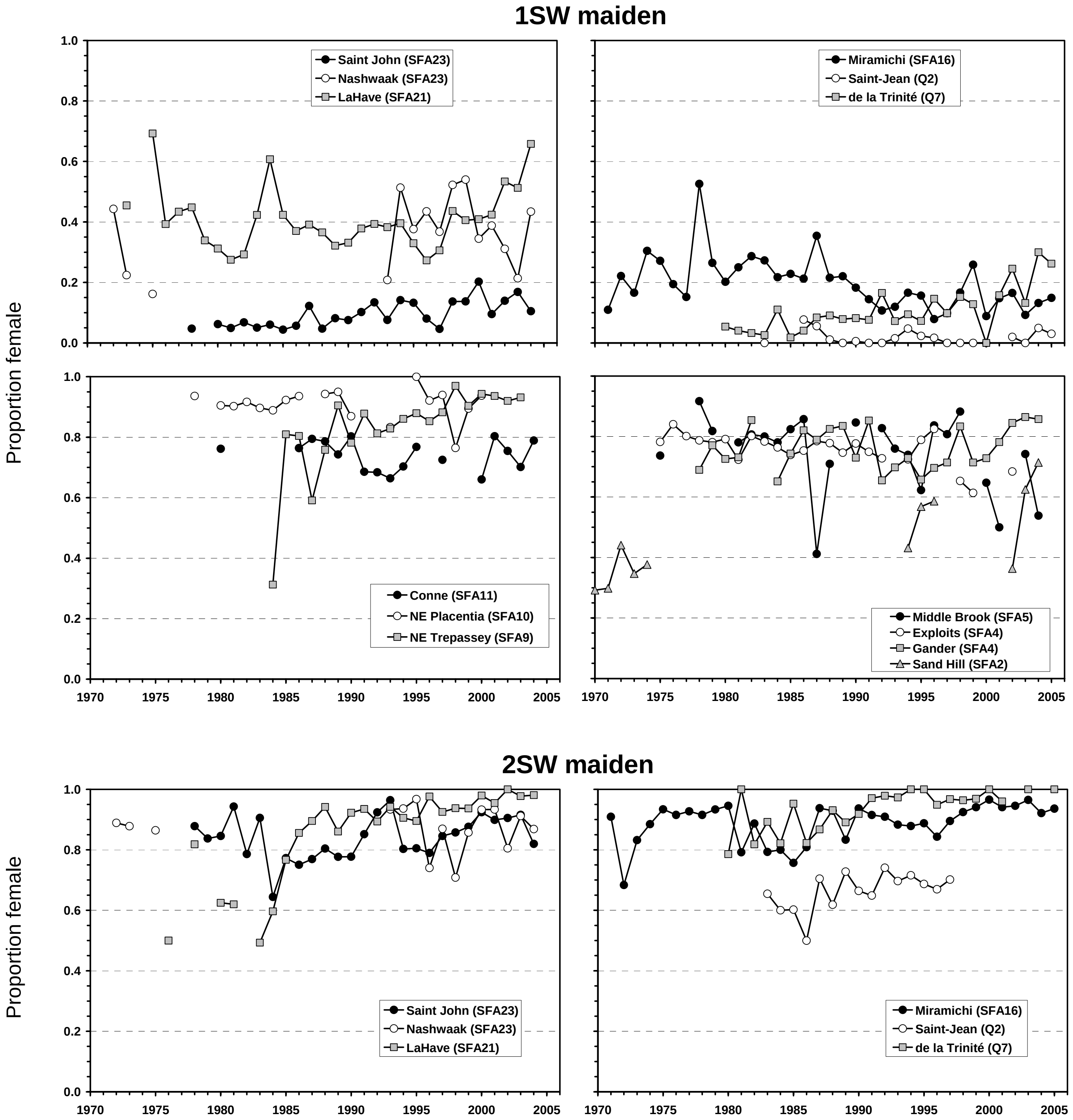

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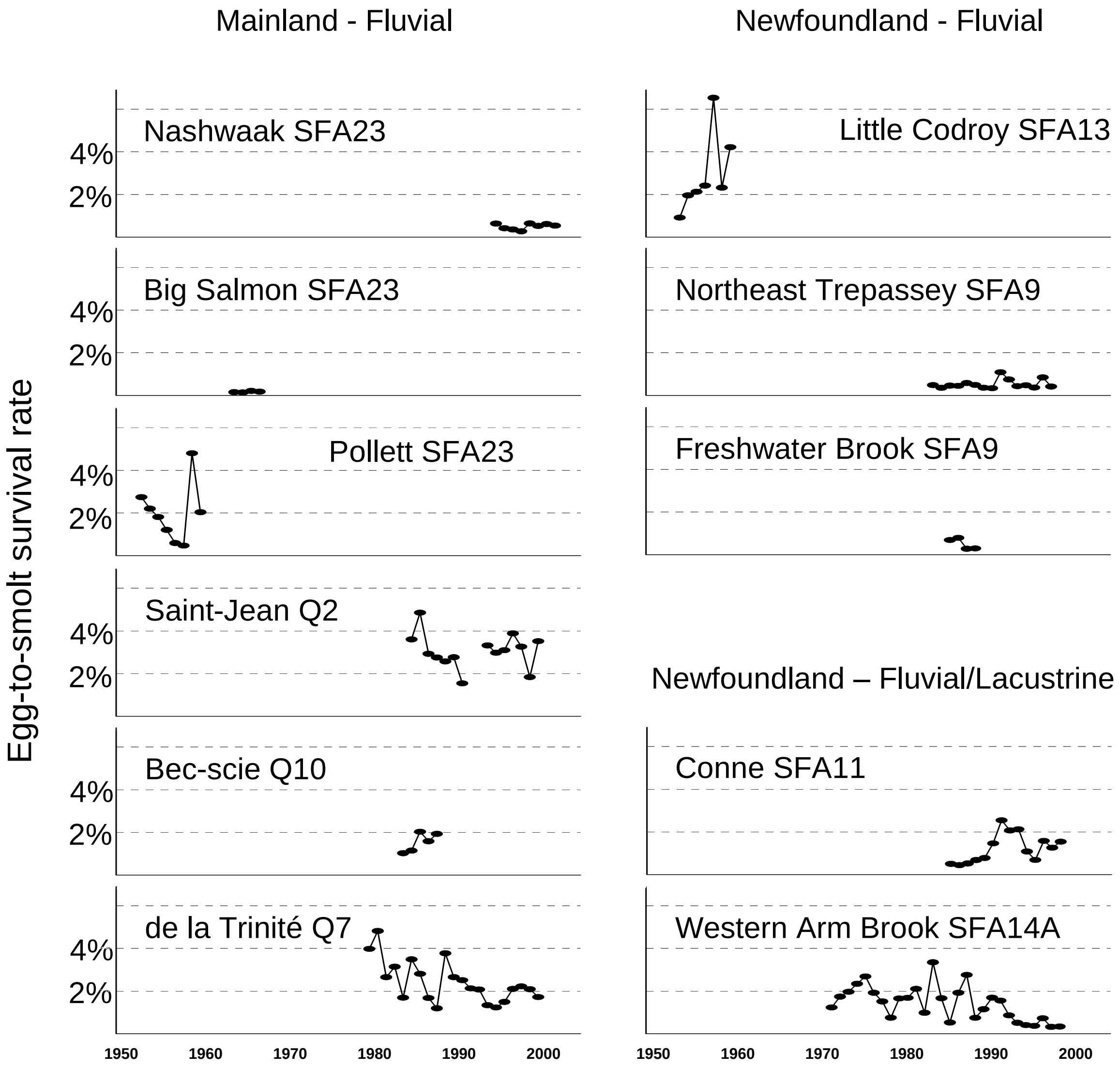

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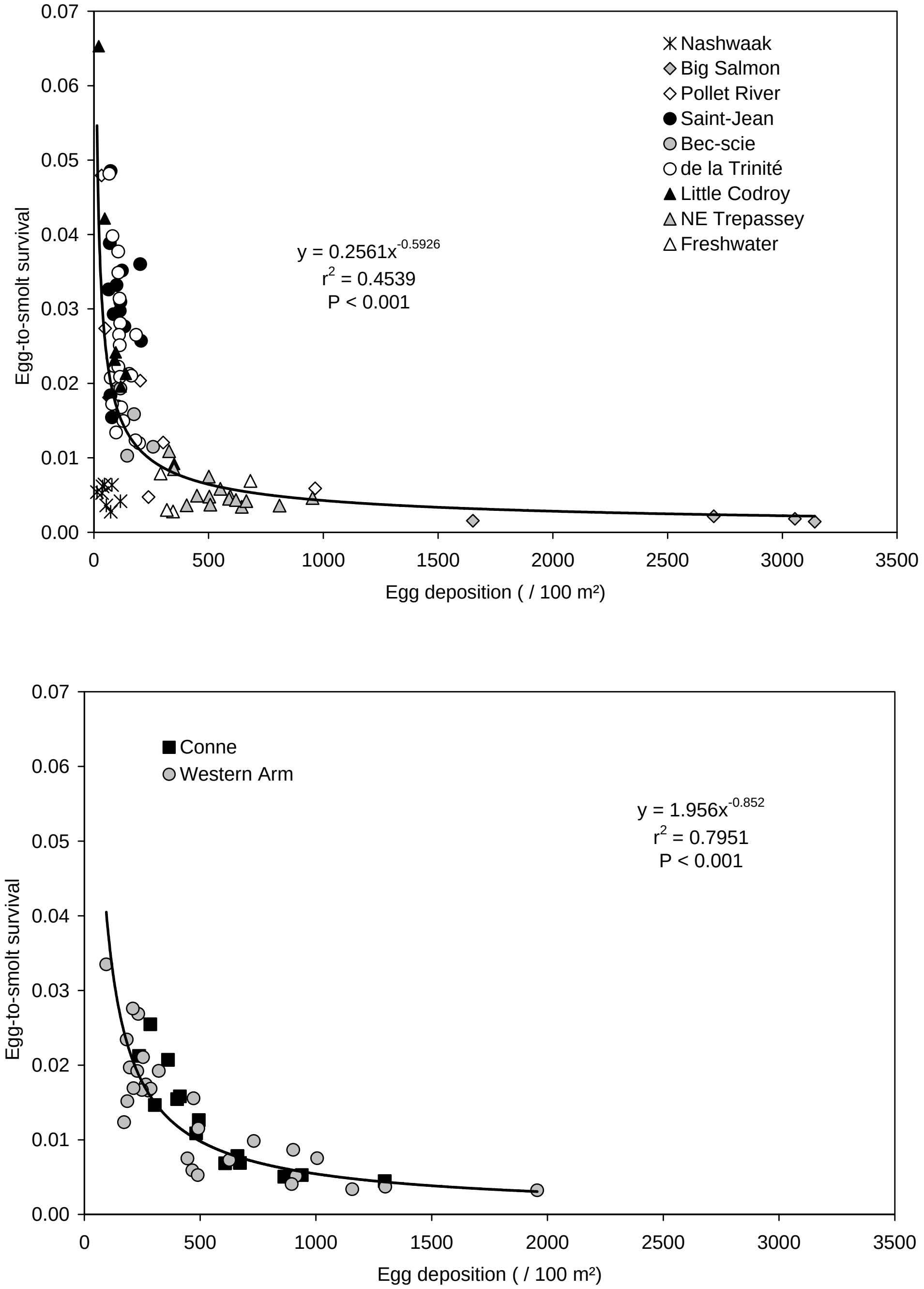

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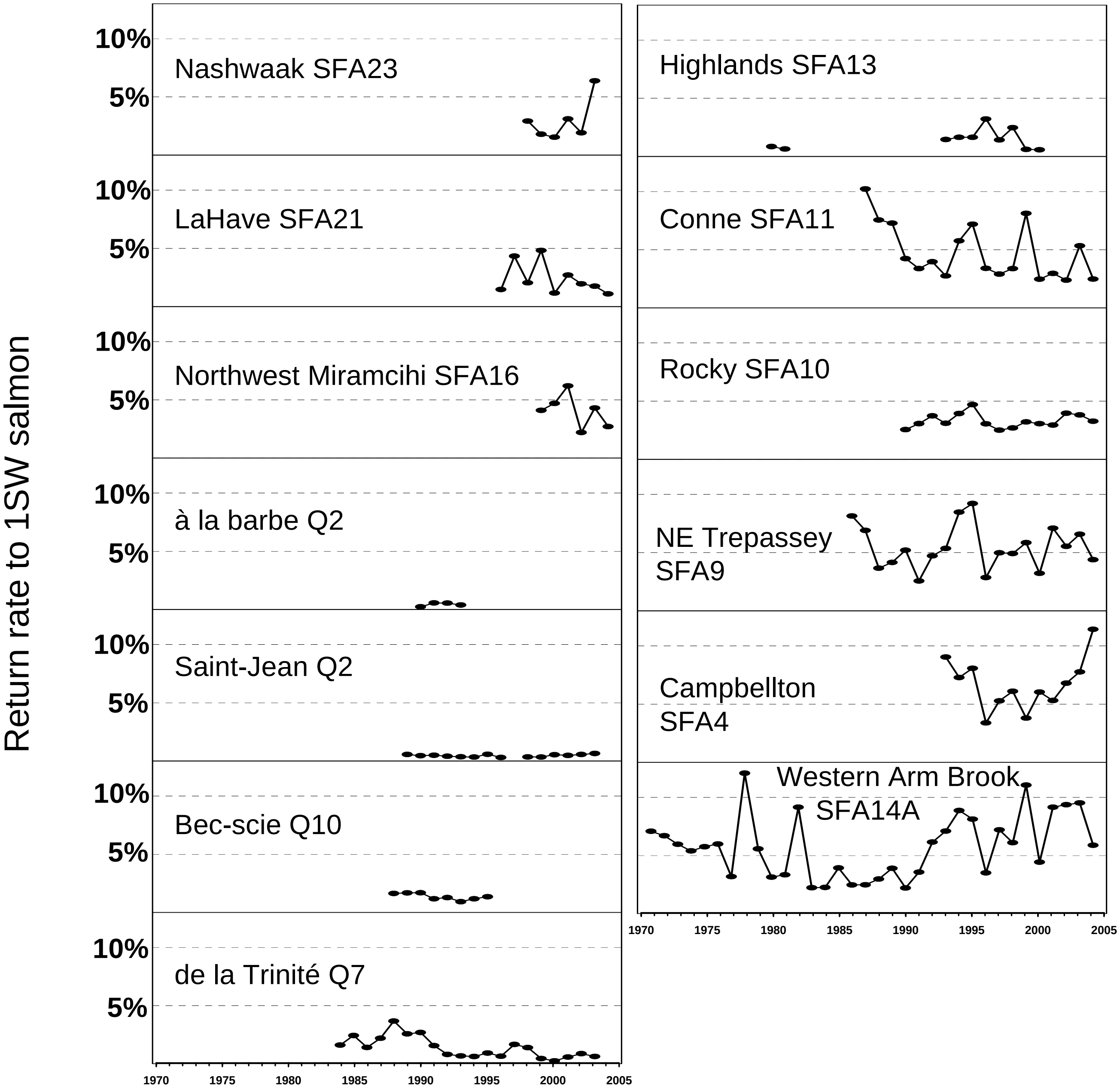

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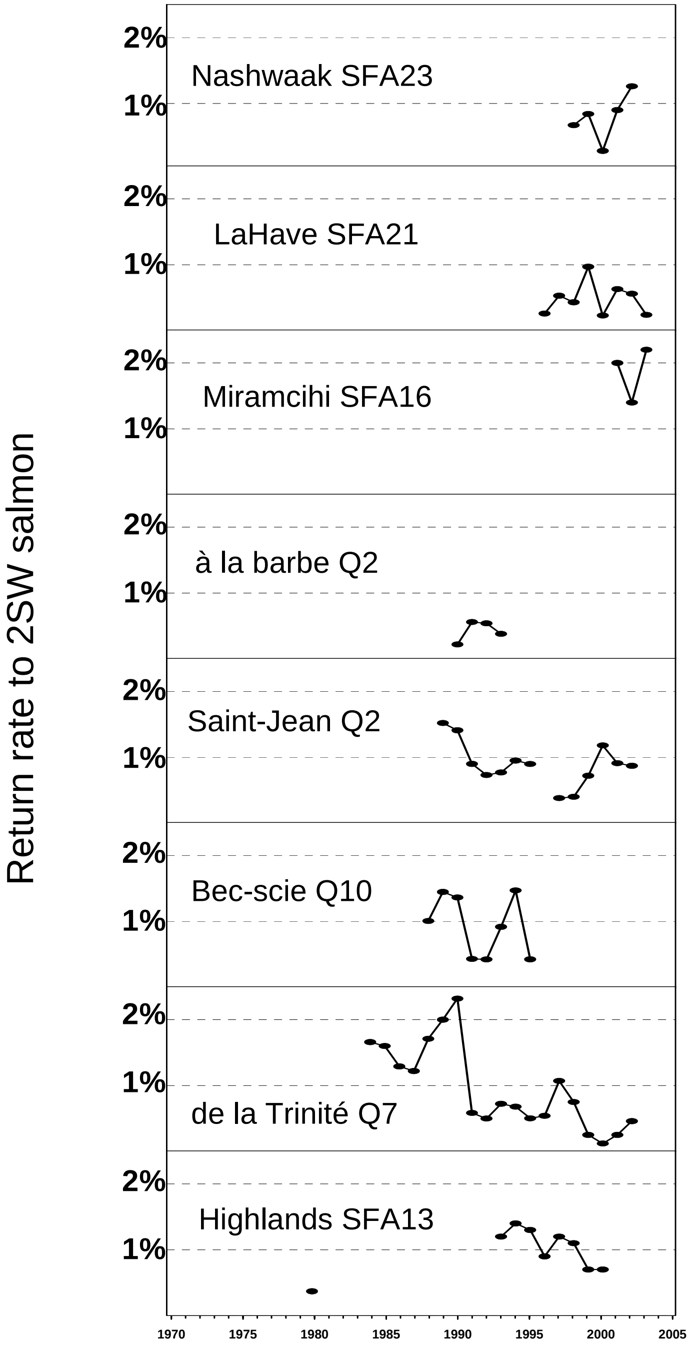

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Research Document 2016/029

Gulf Region

Biological Characteristics and Population Dynamics of Atlantic Salmon (*Salmo salar*) from the Miramichi River, New Brunswick, Canada

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Gulf Region
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May 2016

Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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Published by:

Fisheries and Oceans Canada
Canadian Science Advisory Secretariat
200 Kent Street
Ottawa ON K1A 0E6

http://www.dfo-mpo.gc.ca/csas-sccs/
csas-sccs@dfo-mpo.gc.ca

ISSN 1919-5044

Correct citation for this publication:

Chaput, G., Douglas, S.G., and Hayward, J. 2016. Biological Characteristics and Population Dynamics of Atlantic Salmon (*Salmo salar*) from the Miramichi River, New Brunswick, Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2016/029. v + 53 p.

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ABSTRACT

This report summarizes the information on biological characteristics of Atlantic salmon (*Salmo salar*) from the Miramichi River (New Brunswick, Canada) over the period 1971 to 2014. Emphasis is placed on characteristics of salmon from the Northwest Miramichi in support of a review of the risks to long-term fitness of wild Atlantic salmon of a proposed supplementation program consisting of the captive-rearing in freshwater of wild salmon smolts to the adult stage and release back to the river to spawn. Freshwater dynamics of Atlantic salmon are examined to assist in the evaluation of the risks and /or benefits of the proposed supplementation activity in achieving one of the stated goals which is to increase abundance of adult Atlantic salmon to the river. The Atlantic salmon population from the Miramichi is characterized by complex phenotypic diversity. In any given year, there are typically 6 year classes of immature fish and as many as 9 year classes of mature fish in the combined freshwater and marine ecosystems. There are defined run timing components of the population to the river and headwater areas at higher elevations are primarily utilized by early-run fish. There is an important sex ratio bias between the maiden sea age groups of fish. Increased return rate to a second spawning is an example of phenotypic plasticity in this population responding to changes in the ecosystem. Indices of juvenile abundance for the past four decades indicate that freshwater production of Atlantic salmon increased from low values in the 1970s and peaked in the late 1990s and early 2000s. Abundance indices of fry and small parr are currently well above those of the 1970s to mid-1980s. Preliminary stock and recruitment analyses indicate that the most important density dependent control occurs between the egg and fry stage with modelled theoretical carrying capacity of fry almost realized in the late 1990s. If the egg to fry recruitment dynamic is as severely compensatory as suggested by these analyses, there may be very little gain to be realized in increased smolt production and subsequent adult returns by supplementing the current spawning escapement levels with large numbers of captive-reared adult spawners. Density dependent factors will rapidly adjust the abundances of juveniles to levels which can be sustained by the productive capacity of the freshwater habitat.

Caractéristiques biologiques et dynamique des populations de saumon de l’Atlantique (*Salmo salar*) de la rivière Miramichi, au Nouveau-Brunswick (Canada)

RÉSUMÉ

Ce rapport résume les renseignements sur les caractéristiques biologiques du saumon de l’Atlantique (*Salmo salar*) de la rivière Miramichi (Nouveau-Brunswick, Canada) au cours de la période de 1971 à 2014. On a placé l'accent sur les caractéristiques du saumon dans la rivière Miramichi Nord-Ouest pour appuyer un examen des risques pour la valeur adaptative à long terme du saumon sauvage de l'Atlantique d'un programme d'ensemencement proposé consistant à élever en captivité des saumoneaux sauvages en eau douce jusqu'à l'âge adulte et à les remettre en liberté dans la rivière pour qu'ils aillent frayer. La dynamique du saumon de l’Atlantique en eau douce est examinée afin de faciliter l'évaluation des risques et des avantages de l'activité d'ensemencement proposée pour atteindre l'un des objectifs énoncés, qui est d'accroître l'abondance des saumons de l'Atlantique adultes dans la rivière. La population de saumon de l'Atlantique de la rivière Miramichi est caractérisée par une diversité phénotypique complexe. Pendant une année donnée, on trouve habituellement six classes d'âge de poissons immatures et neuf classes d'âge de poissons matures dans les écosystèmes marins et d'eau douce combinés. La population de la rivière compte des stocks de montaison définis, et les eaux d'amont situées à des élévations supérieures sont principalement utilisées par les stocks de montaison hâtive. Il existe un important biais sex-ratio entre les groupes d'âge en mer de poissons vierges. L'augmentation du taux de montaison au cours d'une deuxième période de frai est un exemple de plasticité phénotypique dans cette population en réponse aux changements dans l'écosystème. Les indices d'abondance des juvéniles pour les quatre dernières décennies indiquent que la production de saumon de l’Atlantique en eau douce a augmenté, passant de valeurs faibles dans les années 1970 pour culminer à la fin des années 1990 et au début des années 2000. Les indices d'abondance des alevins et des petits tacons se situent actuellement bien au-dessus de ceux enregistrés entre les années 1970 et le milieu des années 1980. Les analyses préliminaires des stocks et du recrutement indiquent que le plus important contrôle de la densité a lieu entre l'étape de l'œuf et celle de l'alevin et que la capacité de charge théorique de l'alevin a presque été atteinte à la fin des années 1990. La dynamique de recrutement œufs-alevins est fortement compensatoire, comme le laissaient croire ces analyses. Il se peut que le fait de compléter les niveaux actuels d'échappée de géniteurs par de grands nombres de reproducteurs adultes élevés en captivité ne procure que très peu de gains pour ce qui est d'accroître la production de saumoneaux et les montaisons d'adultes subséquentes. Les facteurs dépendants de la densité ajusteront rapidement l'abondance des juvéniles à des niveaux qui peuvent être maintenus par la capacité de production de l'habitat d'eau douce.

INTRODUCTION

The Miramichi River, at a maximum axial length of 250 km and draining an area of about 14,000 km², had until recently, the largest Atlantic salmon run of eastern North America. There are two major branches: the Northwest Branch covers about 3,900 km² and the Southwest Branch about 7,700 km² of drainage area (Randall et al. 1989). The two branches drain into a common estuary and subsequently drain into the Gulf of St. Lawrence at latitude 47°N (Fig. 1).

The Atlantic salmon (*Salmo salar*) population from the Miramichi River has been one of the most intensively monitored stocks of Atlantic salmon in eastern Canada. Two size groups of Atlantic salmon return to the river to spawn. The small salmon category consists of fish less than 63 cm fork length, which are generally referred to as grilse. These fish have usually spent only one full year at sea (one-sea-winter; 1SW) prior to returning to the river but the size group may also contain some previously spawned salmon. The large salmon category consists of fish greater than or equal to 63 cm fork length. This size group is generally referred to as multi-sea-winter (MSW) or just salmon and contains varying proportions of 1SW, 2SW (two-sea-winter) and 3SW (three-sea-winter) maiden (first time) spawners as well as previous spawners (Moore et al. 1995). Salmon which have spawned and have not returned to sea in the spring of the year are referred to as kelts or black salmon in contrast to bright salmon which are mature adult salmon moving into freshwater from the ocean.

In addition to the different runs and size groups, the Miramichi River is also considered to contain several stocks of Atlantic salmon (Saunders 1981). Separate branch assessments were introduced to account for some of this diversity and for the differences in exploitation between the Northwest and Southwest branches. Aboriginal fisheries were historically conducted almost exclusively in the Northwest Miramichi (exploitation also occurs in the estuarial waters of the Miramichi River, downstream of the confluence of the two branches) and recreational fisheries exploitation also differs between the Northwest and Southwest branches.

Annual assessments of the Atlantic salmon stock of the Miramichi River have been prepared since 1982 (Randall and Chadwick 1983) and the most recent assessment document is to the 2014 return year (DFO 2015a; Douglas et al. 2015). Status of the river in terms of adult returns, estimated egg depositions, and juvenile indices of abundance to 2014 are summarized in DFO (2015a).

This report summarizes the information on biological characteristics (size and age structure, sex ratios, fecundities, run-timing, and migrations at sea) over the period 1971 to 2014. Emphasis is placed on characteristics of salmon from the Northwest Miramichi in support of a review of the risks to long-term fitness of wild Atlantic salmon of a proposed supplementation program consisting of the captive-rearing in freshwater of wild salmon smolts to the adult stage and release back to the river to spawn (DFO 2016). Freshwater dynamics of Atlantic salmon are examined to assist in the evaluation of the risks and /or benefits of the proposed supplementation activity in achieving one of the stated goals which is to increase abundance of adult Atlantic salmon to the river.

PHYSICAL AND ECOSYSTEM CHARACTERISTICS

Detailed geophysical information of the Miramichi River is provided by Blousfield (1955). The two main branches of the Miramichi River are each comprised of two main rivers which join near the head of tide; the Renous and Southwest Miramichi rivers join at the head of tide in the Southwest Branch of the Miramichi, the Little Southwest and Northwest Miramichi rivers join at the head of tide in the Northwest branch of the Miramichi (Fig. 1). The two branches of the

Northwest Miramichi and the Renous River within the Southwest branch are characterized by a steep profile, achieving maximum elevations of almost 450 m within the first 100 km of river length (Randall et al. 1989; Fig. 1). The Southwest Miramichi within the Southwest branch has a more shallow profile, attaining a maximum elevation of just under 400 m at 200 km upstream of the head of the tide.

Wetted area for juvenile salmon production in the Miramichi River was calculated by Amiro (1983). The estimates are total wetted area, unweighted by habitat type, gradient or other variable of productive capacity. Total wetted area in the Northwest Miramichi is estimated at 1,679 ha of habitat, with more than 50% of the estimated area in stream order 4 or less (Table 1). Within the Northwest Miramichi, the habitat areas are approximately equivalent between the Little Southwest Miramichi River and the Northwest Miramichi River.

Eleven diadromous and 19 freshwater fish have been reported from the Miramichi River (Randall et al. 1989). As for non-native species, brown trout, rainbow trout, tiger trout, chain pickerel, and smallmouth bass have been recorded in the Miramichi. The single known occurrence of chain pickerel was eradicated shortly after discovery from Depres Lake in 2001, a headwater lake of the Cains River in the Southwest Miramichi (Connell et al. 2002). Smallmouth bass were first reported from Miramichi Lake in 2008, a headwater tributary of the Southwest Miramichi River, and efforts have been ongoing since to control the spread, and reduce abundance with the objective of eradicating the species from the watershed (DFO 2013; Biron et al. 2014).

BIOLOGICAL CHARACTERISTICS

This section on biological characteristics includes information on river ages, juvenile salmon characteristics, smolt characteristics, and adult salmon characteristics. The adult salmon characteristics include information on lengths and weights by spawning history groups and within season of return, fecundity, sex ratio by sea age group, run timing and marine migrations.

Returning adult salmon have been systematically captured and sampled in the Miramichi River since 1971. Salmon are captured in tidal trapnets which are fished daily over the entire migration period from the middle of May to late October. Fork length, origin (hatchery released fish are identified based on the absence of the adipose fin which was clipped from appropriate life stages prior to release), sex by external characteristics, and a scale sample is collected from up to 30 small salmon per day and generally all large salmon (>= 63 cm fork length). Small salmon and large salmon are tagged and released with external individually numbered tags prior to release. Data on other fish captured are also recorded. Hayward (2001) provides an overview of all counting facilities in the Miramichi River and more details of sampling operations are in Hayward et al. (2014).

RIVER AGE DISTRIBUTIONS

Juvenile salmon from the Miramichi River spend between two and five years in freshwater before going to sea. For the Northwest Miramichi system, based on sampling and run size estimates for the smolt migration years 1999 to 2006, the percentage of a yearclass going to sea after two years in freshwater varied from 29% to 61% whereas river age 4 smolts were never more than 2% of a yearclass (Table 2).

Based on characteristics of returning adult Atlantic salmon and weighted by estimates of returns, the majority (> 95%) of a yearclass of salmon from the Northwest Miramichi spent 2 or 3 years in rivers with on average 47% of all maiden-aged returning salmon having a river age of 2 years. A similar percentage (average 47%: range 11% to 85%) of the 1SW maiden salmon were of river age 2. For the 2SW maiden salmon, a slightly higher percentage was of river age 2 (average 53%; range 24% to 82%) (Fig. 2). There is a large amount of variation in the percentages at river age in the returns as adults from a yearclass. This is due to the annual variations in sea survival to which a yearclass is exposed. In the case of the Northwest Miramichi, a yearclass of salmon is at sea over four consecutive years of maiden returns (see text table for yearclass 2000 below).

Years at sea for the example yearclass = 2000			
Maiden sea age	river age 2	river age 3	river age 4
1SW	2003/2004	2004/2005	2005/2006
2SW	2003/2004 2004/2005	2004/2005 2005/2006	2005/2006 2006/2007

JUVENILE SALMON CHARACTERISTICS

Growth rates of salmon juveniles are highly variable among sites and years. Average size of age-0 parr (fry) is annually variable, mean fork lengths ranging between 4.0 and 5.5 cm (Fig. 3; Swansburg et al. 2002). Age-1 parr also show important variations in mean size among sites and among years, ranging between 7.5 cm to just over 9.0 cm in the Northwest Miramichi (Fig. 3). Age-2+ parr, those juveniles not leaving the river as 2-year old smolts range in mean size between 10.5 and 12.4 cm fork length (Fig. 3). Precocious male maturation is common in juveniles in the Miramichi (Cunjak and Therrien 1998; Brodeur 2006).

Smolt characteristics

Information on size, age, sex ratio and timing of Atlantic salmon smolts was obtained during monitoring and assessment programs in the Northwest Miramichi system from tidal waters and from the Little Southwest Miramichi River using a rotary screw trap (see description in Chaput et al. 2002).

Atlantic salmon smolts migrate from the Northwest Miramichi primarily from mid-May to early June. Date of peak catches at the estuary trapnet ranged from 16 May to 8 June over sampling years 1999 to 2011(Table 2). The date of the 5th percentile of catches, as an indicator of the initiation of the smolt migration ranged from 13 May to 24 May for the same years sampled (Table 2). Peak catches occurred in most years when water temperatures attained / exceeded 15ºC (Figs. 4a and 4b). Run duration is generally short, occurring over a period of about three weeks.

Smolts from the Northwest Miramichi are of relatively consistent size distribution annually, ranging between 10.5 to 18.0 cm with mean lengths of 13 cm (Fig. 5; Table 2). Mean weight of smolts ranged between 18 and 22 g annually (Table 2).

There are usually more females than males in the smolt run, the percentage female ranging between 42% and 63% with greater than 50% female in most years (Table 2).

ADULT SALMON CHARACTERISTICS

Atlantic salmon returning to the Miramichi are assessed on the basis of abundance of two size groups: small salmon (less than 63 cm fork length) and large salmon (>= 63 cm fork length).

Over the period 1992 to 2014, small salmon have comprised more than 50% of the returns to the Northwest Miramichi, with the maximum percentage of 90% in 2002 and the lowest percentage (50%) in 2012 and 2014 (Fig. 6). On average, the returns of small salmon have a lower percentage in the Southwest Miramichi (58%) than in the Northwest Miramichi (68%) although the annual variation in the percentages of small salmon is just as important in the Southwest Miramichi system (range: 34% to 83%) (Fig. 6). The proportion of the total annual returns made up of small salmon has declined over the period 1992 to 2014, especially so in recent years. The same general pattern is noted for 1SW maiden salmon as a proportion of the annual returns of maiden 1SW and 2SW salmon, 3SW salmon are very rare in the Miramichi (Fig. 6).

Sea age composition

Fish in the small salmon category are predominantly (>95%) maiden 1SW salmon, the other age groups include 2SW salmon and repeat spawning 1SW salmon as consecutives. The large salmon category is comprised of a more diverse life history including 1SW maiden salmon, 2SW maiden, 3SW maiden, and a large number of categories of repeat spawning salmon. Repeat spawning salmon can be short duration migrants (consecutive) which spend a few months at sea to recondition and return to rivers to spawn in consecutive years, or long duration migrants (alternates) that spend more than one year at sea post-spawning to recondition before returning to rivers to spawn. A total of 52 unique spawning histories have been interpreted from scales of salmon in the Miramichi and repeat spawners up to a seventh spawning migration have been sampled since the mid-1990s (Table 3).

When considered by smolt class, the majority of returning maiden salmon are 1SW maiden sea age (Fig. 6). Notable exceptions to this pattern were the returns from the 2008 and 2011 smolt migration years when 40% or more of the smolt class returned to the Northwest Miramichi at 2SW maiden sea age (Fig. 6). For the 2011 smolt class of the Southwest Miramichi, almost 60% of the smolts returned at 2SW maiden sea age when usually the percentage of 2SW in the return of the smolt class is less than 30% (Fig. 6). When considered by yearclass, taking into account the variable river ages and the different years when the salmon are at sea, the percentage of all maiden sea age returns comprised of 1SW salmon has oscillated at around 80% for the Northwest Miramichi and about 75% for the Southwest Miramichi (Fig. 6). There is less variability in the proportion of the maiden salmon that are 1SW and there is no apparent change in the proportion over time in the Northwest Miramichi but there is a perceptible decline in the Southwest Miramichi (Fig. 6).

Lengths and weights

Fork lengths of adult Atlantic salmon increase with the number of years at sea (Fig. 7). Maiden salmon at 1SW age have a median fork length of about 58 cm, 2SW salmon have a median fork length of about 75 cm, and 3SW salmon although rare in the Miramichi have a median fork length of about 84 cm (Fig. 7). Post spawners can return to sea to feed and grow and return to spawn repeatedly. As mentioned previously, Atlantic salmon on a seventh spawning migration have been observed in the Miramichi (Table 3). Consecutive spawners put on less length at each return migration than alternate spawners. First time repeat 1SW salmon that return as consecutives (1SWC) are intermediate in length between 1SW and 2SW maiden salmon whereas first time repeat alternate 1SW spawners (1SWA) are intermediate in length between 2SW and 3SW salmon (Fig. 7). The longest salmon recorded in the Miramichi have been 2SW repeat alternate spawners (Fig. 7). Sizes at age are similar between the Northwest Miramichi and Southwest Miramichi systems (Fig.7).

Adult salmon return to the Miramichi River over an extended period from late May to late October. Fork lengths of 1SW maiden and 1SW consecutive salmon increase over the season of return; median lengths of 1SW maiden salmon increase from 56 cm to about 59 cm over the June to October migration period whereas 1SW consecutive spawners increase in length from 62 cm in June to about 68 cm in October (Fig. 8). There is no discernible change in fork lengths of 2SW maiden and repeat spawning salmon during the seasonal migration, with perhaps the exception of the June returning 2SW salmon which were slightly shorter than 2SW salmon in the other months (Fig. 8).

Fork length distributions over years 1992 to 2013 show important (> 5 cm at median values) annual variations in size particularly for 1SW maiden salmon returns in October (Fig. 9). Although there are important variations in size distribution of 1SW consecutive and 1SW alternate spawners, the sample sizes are much smaller for these size groups. Annual variations in fork lengths of 2SW maiden and repeat salmon are also noted (Fig. 10).

Weights are not collected systematically at the monitoring trapnets of the Miramichi. Systematic sacrificing of salmon from the index trapnet prior to 1992 was conducted and these samples combined with opportunistic sampling in recent years of incidental mortalities were used to describe the weights at sea age. Maiden 1SW salmon have a median weight of 1.57 kg, 2SW maiden have a median weight of 4.50 kg. First time alternate repeat spawning 1SW salmon have a median weight of 5.51 kg whereas first time alternate repeat spawning 2SW salmon have a median weight of 9.00 kg (Fig. 11).

Sex ratio, fecundity, egg size

There are important differences in the proportion of females among the size categories and sea age histories. Small salmon, comprising the majority of 1SW salmon, are the majority male whereas large salmon, comprised by a majority of 2SW salmon, are the majority female (Fig. 12). Small salmon in the Northwest Miramichi River have a higher proportion of females than in the Southwest Miramichi. The proportion of small salmon from the early run (prior to September 1) averages 0.38 female whereas small salmon from the late run averages 0.18 female salmon (Fig. 12). The same proportions are noted from sampling sites in the Little Southwest Miramichi. There is much less difference in the proportion of females of large salmon, averaging about 0.80 female in both the early and late runs in the Northwest Miramichi and the Southwest Miramichi (Fig. 12). There are noted annual variations in the proportions female (Fig. 13).

Fecundity to body size relationships for salmon from the Miramichi River were reported by Randall (1989). These relationships, one for small salmon and the other for large salmon, combined with the average fork lengths and the proportions female in the runs of each size group, are used to estimate the number of eggs per fish for the annual stock assessments (Fig. 13; Douglas et al. 2015).

Data from spawning of females in the hatchery at South Esk were collected during 1991 to 1995 and are presented in Figure 14 (J. Hayward, DFO, unpublished data). Reid and Chaput (2012) also estimated eggs per female, by size and spawning history (Fig. 15).

Fecundity (number of eggs) increases with both length and with weight (Figs. 14, 15). The fecundity length relationships from Reid and Chaput (2012) are essentially identical to those from the hatchery data of 1991 to 1995 (Figs. 14, 15). These two relationships differ from those derived by Randall (1989) in having a lower number of eggs for small salmon and differing slopes for the large salmon.

Female 1SW maiden salmon, of median fork length 58 cm, have a predicted fecundity of about 2,900 eggs. Maiden 2SW salmon of median fork length 75 cm have a predicted fecundity of 5,900 eggs whereas salmon measuring 84 cm fork length (median length of 3SW salmon) have a predicted fecundity of 8,000 eggs. The highest measured fecundity from the hatchery data was 14,600 eggs from a female salmon measuring 97 cm fork length. Reid and Chaput (2012) measured a maximum fecundity of 15,500 eggs from a female salmon measuring 104 cm fork length.

Egg size (diameter, mm) is smallest for 1SW maiden salmon but otherwise unrelated to body size in other spawning age groups (Fig. 16). The largest eggs are produced by 2SW maiden and alternate repeat spawning salmon (Fig. 16). The egg sizes of consecutive repeat spawners are intermediate in size from those of 1SW maiden salmon and 2SW maiden and repeat alternate spawners (Fig. 16).

Reid and Chaput (2012) measured egg survival rate from female salmon to placement of eggs into incubation boxes in the hatchery (Fig. 17). Egg survival was very high, ranging from a low of 67.0% for a 2SW maiden female to a maximum of 99.4%. The majority (95%) of the measured survival rates exceeded 81%, and half of the measured survival rates exceeded 95% (Fig. 17). Spawning history was a statistically significant explanatory variable, although accounting for a very small proportion of the variance. The survival rate of eggs of consecutive repeat spawners was significantly lower than for 2SW maiden and alternate repeat spawners (Fig. 17).

Run timing

Temporal stock distinctiveness has also been highlighted as an important component of the Atlantic salmon resource (Saunders 1967). The run timing of Atlantic salmon to the Miramichi River has been previously characterized as bimodal, with the first mode occurring in the summer (prior to August 31) and the second in the fall (after August 31) (Fig. 18). Early runs and late runs have different composition in terms of small and large salmon proportions and sex ratios. The early runs in both branches are also exploited more heavily than the late runs.

Early and late runs of salmon to the Miramichi were obvious from DFO index trapnet catches in the early and mid-1990s but appears to have changed over time to a dominant summer mode (Douglas et al. 2015). These changes in run timing have been consistent for both large and small salmon and on both major branches of the Miramichi River (Fig. 19). The proportion of salmon captured at DFO index trapnets by August 31 has increased on the SW Miramichi River since 1994, attaining levels of 75-90% in recent years. A similar pattern was observed for salmon on the NW Miramichi River but the trend was less pronounced (Fig. 19). Decreases in the late run component have generally corresponded with increases in single-day peak catches in the early run portion (Douglas et al. 2015). The reduced late run of salmon to the Miramichi River is not believed to be related to fish abundance but rather to a shift in behavior where the fish enter the river during the summer and no longer stage in Miramichi Bay until autumn.

Movements of salmon at monitoring facilities in freshwater are characterized by a gradation of summer (prior to September 1) to fall (after August 31) movement dominance (Fig. 20). There was a dominant summer movement of salmon at the Northwest Miramichi and the Dungarvon River barriers; on average, 74% of the fish movements at the Northwest Miramichi barrier and 68% of fish movements at the Dungarvon River barrier occurred prior to September 1 (Fig. 20). The movement of fish was predominantly in the fall at the two mid-location facilities in the Southwest Miramichi; 82% fall movements at Clearwater Brook and 81% fall movements at Burnthill Brook (Fig. 20). Almost exclusively fall movements were recorded at Catamaran Brook, a small tributary in the lower portion of the Little Southwest Miramichi (Fig. 20).

There is a positive association between the proportion of the run which ascends to the counting facility early and the elevation of the facility, with fewer fish ascending the higher elevation sites in the fall.

Although early running salmon at the upstream facilities must correspond to early running salmon in the estuary, the fish which migrate to the upstream facilities in the fall are a mixture of early and late run salmon at the estuary. The fish tagged early in the estuary represent a higher proportion of all tag recoveries at the higher elevation sites and there is a strong linear positive association with elevation but not with distance of the facility (Fig. 21). Not all early tagged salmon ascend rapidly to the headwater sites. There was no statistically significant association between the proportion of the early tagged salmon which ascended to the facilities prior to September 1 and neither the elevation nor the distance of the facility from the head of tide (Fig. 21). However, there was a positive and significant association between the proportion of the tagged salmon recovered at the facilities which had been tagged from the early run in the tidal trapnets and the elevation of the facility, although not the distance of the facilities from the head of tide (Fig. 22). This suggests that salmon at the higher elevation areas of the Miramichi are mostly from the early run component; early and late run components return to intermediate and lower elevation sites.

An unpublished study by Dodson and Colombani (1997. The genetic identity of the Clearwater Brook population of Atlantic salmon (*Salmo salar*) ; a temporal and spatial study of Atlantic salmon population genetic structure in the Miramichi, St. John and Margaree, Atlantic Salmon Federation, Final Report) concluded that despite differences in the timing of runs, early and late-run fish mingle on the spawning grounds and that the timing of the spawning run is not genetically determined, but that development at sea and/or differences in marine migratory routes may determine run timing in these stocks of Atlantic salmon.

That being said, there is an important heritable component to run-timing. Salmon from Rocky Brook, a tributary of the Southwest Miramichi, are known to have an important proportion of fish which return to the Miramichi early and consequently have been extensively used to stock salmon in a number of rivers including the Margaree River (NS) and the Morell River (PEI) with the expressed objective of increasing early run returns to these rivers (Claytor et al. 1987; Cairns et al. 1996).

MARINE MIGRATIONS

Atlantic salmon from the Miramichi River undergo long oceanic migrations (Reddin 1985; Reddin and Lear 1990) and were historically harvested in a number of marine commercial fisheries including those of the Gulf of St. Lawrence, Newfoundland, Labrador, St. Pierre & Miquelon, and Greenland (see Saunders 1969; Kerswill 1971; Paloheimo and Elson 1974). Tags applied to smolts from the Miramichi River continue to be captured at West Greenland as non-maturing 1SW salmon in their second summer feeding at sea and recaptures of previously spawned adult salmon from the Miramichi River have also been consistently returned from the West Greenland fishery. Four Atlantic salmon tagged in their second winter at sea north of the Faroes Islands were recovered in southern Gulf of St. Lawrence rivers in the summer following tagging, three of these from the Miramichi River (Hansen and Jacobsen 2000).

The migrations of 1SW salmon at sea are less well known. They were captured as post-smolts (first year at sea) in a number of fisheries in eastern Canada (Ritter 1989) but are not available for capture (due to size, timing) in the Greenland fisheries.

GENETICS AND POPULATION STRUCTURING IN ATLANTIC SALMON FROM MIRAMICHI

Despite the long history of monitoring the Atlantic salmon population of the Miramichi, there is a paucity of information on their genetics and extent of local adaptation.

Møller (1970) indicated that based on transferrin polymorphisms, that there were genetically distinct populations of salmon between the Northwest Miramichi and the Southwest Miramichi systems and based on more homozygotes than expected in samples from the Miramichi River, he concluded that the samples were likely to have been mixtures of different populations. In a follow-up paper, Møller (2005) reported on differences in frequencies among samples collected during the smolt run and of adults returning to the Northwest Miramichi and concluded that these differences reflected population structuring within the river.

Riddell et al. (1981) reported on heritable quantitative genetic differences in body morphology of juvenile salmon from two tributaries of the Southwest Miramichi River, and suggested that these differences were adaptive and associated with differences in geomorphology of the streams.

Recently, salmon from the Miramichi River were sampled from four freshwater locations and two estuary points and genotyped at 15 microsatellite markers and 5568 loci using single nucleotide polymorphisms (SNPs) (Moore et al. 2014). The samples were collected from:

- Northwest Miramichi at barrier (headwaters)
- Little Southwest Miramichi seining samples (headwaters)
- Northwest Miramichi system sampled at tidal trapnet near head of tide
- Southwest Miramichi system sampled at tidal trapnet near head of tide
- Southwest Miramichi at Clearwater Brook (headwaters)
- Southwest Miramichi at Dungarvon Barrier (headwaters)

This work was part of a continent initiative to characterize the population structure of Atlantic salmon in eastern North America. Moore et al. (2014) confirmed the earlier work of Bradbury et al. (2014) that found 12 regional groups of salmon populations with the Miramichi River samples clustering with other populations of the southern Gulf of St. Lawrence (NB, NS, PEI) and the Saint John River system of NB into one group.

HATCHERY SUPPLEMENTATION

One of the earliest supplementation activities for Atlantic salmon in eastern Canada began in 1873 with the establishment of the Atlantic salmon hatchery on the Miramichi River at South Esk. The South Esk hatchery has been in continuous operation since 1873 stocking a variety of juvenile Atlantic salmon life stages annually into the Miramichi watershed (Table 4). Between 1959 and 1970 experimental plantings of Restigouche origin stock (Salmon Fishing Area 15) were distributed to the Southwest Miramichi, Northwest Miramichi, and Little Southwest Miramichi rivers as well as to Rocky Brook (upper tributary of the Southwest Miramichi River). Within the past 30 years, Miramichi origin stock were distributed in the Tabusintac and Buctouche rivers and Rocky Brook (Southwest Miramichi) stock was outplanted to rivers in Prince Edward Island as well as the Margaree River in Nova Scotia. All recent enhancement activities have involved targeted supplementation activities placing juvenile progeny back to tributaries from which the parents were collected.

The Miramichi River remains reliant on natural production with on average, 99% of returning adults coming from wild production (Chaput et al. 2001).

POPULATION DYNAMICS

The benefits to long-term stock abundance of supplementation activities must take into account the factors that regulate population abundances at different stages. Density dependent compensatory mortality in the freshwater phase is well established for Atlantic salmon (Elliott 2001; Jonsson et al. 1998; Gibson 2006). On the other hand, survival at sea is considered to be density independent (Hansen and Quinn 1998; Gibson 2006).

Cunjak and Therrien (1998) provided estimates of egg to 0+ parr and 0+parr to 1+ parr survival rates from Catamaran Brook, a small tributary of the Little Southwest Miramichi. Unadjusted for abundance of the preceding age group, egg to 0+ survival varied between 9% to 61% while survival rates from 0+ to 1+ parr ranged from 14% to 75%. The annual variations were explained in terms of both density dependence and environmental variability (density independent) effects. Interstage survival rates can therefore only be discussed relative to the abundance of the preceding age group. As there are multiple yearclasses of juveniles in the river at the same time, the density dependent mortality resulting from competition for limited resources (food, refuge) occurs within and among yearclasses.

POPULATION DYNAMICS IN FRESHWATER

Juvenile freshwater stages

Data and Methods

Since 1993, the number of sites sampled for juvenile abundance in the Miramichi has varied from a low of three sites in 1991 to a high of 77 sites in 1994 (Moore and Chaput 2007). Different numbers of sites were annually sampled in each river, and the number of sites was not proportional to river size.

Juvenile abundance at a site is expressed in terms of density (fish per 100 m²), for fry, small parr, and large parr size groups. The method for converting single sweep catch per unit effort data to density is described by Chaput et al. (2005). Annual calibration data, or combinations of several years, were used to derive the regression relationships.

Biomass of Atlantic salmon juveniles is derived from the mean density at size and the mean weight at size for each site. Percent Habitat Saturation (PHS), proposed as a relative measure of habitat use and potential interaction between juveniles within the stream, was calculated for each site (Grant and Kramer 1990). It considers both the densities of fish and body lengths. A PHS value of 28 is used as a reference value; it represents the value at which density dependent effects have a 50% probability of being expressed (Grant and Kramer 1990).

Douglas et al. (2015) provided indices of abundance for the four main rivers of the Miramichi: the Northwest Miramichi and the Little Southwest Miramichi for the Northwest Miramichi system, the Southwest Miramichi and the Renous River for the Southwest Miramichi system. Spawning escapements are estimated for the Northwest Miramichi system and the Southwest Miramichi system since 1992. To derive an index of juvenile abundance for each system corresponding to the years with corresponding egg deposition estimates, the river specific mean juvenile indices from Douglas et al. (2015) were weighted by the respective habitat areas in each river (8.21 million m² for the Little Southwest Miramichi and 8.38 million m² for the Northwest Miramichi River; 5.83 million m² for the Renous River and 29.54 million m² for the Southwest Miramichi River). For the years 1992 to 2014, there were a total of 22 stock and recruitment observations for the Northwest Miramichi River and 21 for the Southwest Miramichi River.

Relationships between fry and small parr were examined over the longer time series of observations, from 1971 to 2014. Annual indices were calculated when there were at least four sites in a main river sampled. Over the period 1971 to 2014, there were 37 observations for the Northwest Miramichi system and 36 for the Southwest Miramichi system.

Two stock recruitment models were examined for the relationships between eggs and fry, and for fry to small parr. All data were expressed in units of abundance per 100 m² of habitat. A linear proportional model was used as the default model against which the compensatory Beverton-Holt model was compared.

The linear proportional model was of the form:

$$R_{t+1} = \alpha \, S_t \, e^{\varepsilon_t} \;\; with \, \varepsilon \sim N(0, \sigma^2) \tag{1}$$

with R_{t+1} the abundance of fry, or small parr,

S_t the abundance of eggs, or fry

α the mean survival rate, distributed a priori [0,1], and

e^{ε} the residual error with lognormal distribution.

The Beverton-Holt model was of the form:

$$R_{t+1} = \frac{\alpha \, S_t}{\left(1+\frac{\alpha}{Rmax} S_t\right)} \; e^{\varepsilon_t} \;\; with \, \varepsilon \sim N(0, \sigma^2) \tag{2}$$

with R_{t+1}, S_t, α, and e^{ε} as above and

Rmax the maximum abundance of the recruiting life stage (carrying capacity).

For the Beverton-Holt model, the mean egg deposition (eggs per 100 m²) that results in 50% of carrying capacity (half saturation) is calculated directly as Rmax/α. This value has been proposed as a potential limit reference point for conservation of fish populations (DFO 2015b).

The models were fitted in a Bayesian framework with uninformative priors for α, Rmax, and σ^2:

- $\alpha \sim$ Beta(1,1)
- Rmax ~ Uniform(0, 500)
- $\sigma^2 \sim$ Inv-gamma(0.01, 0.01)

The parameters were estimated using Monte Carlo Markov Chain (MCMC) in Gibbs sampling with OpenBUGS (Lunn et al. 2013). Convergence was interpreted based on visualization of MCMC draw sequences and of quantile distributions of variance. Posterior distributions of the parameters were summarized from 10,000 MCMC draws after burn-in of 50,000 draws. Model fits were assessed using the DIC values in OpenBUGS, a synonymous index to AIC of model sufficiency.

The consequence of inter-yearclass competition on survival rates was examined by plotting the residuals of the abundances of the recruiting stages against the annual mean biomass of the juvenile size groups which potentially would have competed with the recruitment stage. For the fry recruitment stage, small and large parr present during the year of sampling at the fry stage would have competed for resources with fry; for the small parr recruitment stage, it would be fry and large parr of the same year that would have competed. Biomass (g per 100 m²) rather than numerical abundance of the competing life stages was used, as resource requirements for the life stages were assumed to be more closely related to mass than number of animals.

Results

Trends in egg depositions (eggs per 100 m²) and juvenile abundance by age/size groups (fry or age 0+ parr, small parr or age 1, large parr or age 2+) are shown in Figure 23. Over the period of assessment 1971 to 2014 for the Miramichi, the egg depositions were estimated to have exceeded the conservation egg deposition rate of 240 eggs per 100 m² in the mid-1970s and again from 1986 to 1997 (Fig. 23). Estimated egg depositions in both the Northwest and Southwest Miramichi systems were highest at the beginning of the assessment series in 1992 and declined and remained at lower levels into the 2000s. Egg depositions in the Southwest Miramichi exceeded the conservation requirement in most years and have been higher than in the Northwest Miramichi. Estimated egg depositions were high in both systems in 2011 (Fig. 23).

Juvenile densities over the longer time series from 1971 to 2014 showed relatively low levels until the late 1980s when indices of abundance increased, in response to increased egg depositions estimated for the Miramichi River overall (Douglas et al. 2015). Indices of fry abundance peaked in the late 1990s and have generally declined since, with the exception of the indices of abundance noted in 2012 following on the high egg depositions of 2011 (Fig. 23). The trends in abundance indices of small parr follow those of fry, with abundances peaking in the late 1990s to early 2000s and declining since (Fig. 23). Only the Northwest Miramichi River and the Renous River were sampled at a sufficient number of sites (> 4 per river) in 2013 but at least in the Northwest Miramichi, the small parr abundance index was high, following on the high fry index of the previous year (Fig. 23). Trends in large parr abundance indices differ somewhat from trends of fry and small parr; large parr indices have generally been increasing over the period 1971 to 2014 in both the Northwest and Southwest systems with large parr abundances being higher in the Northwest versus the Southwest (Fig. 23).

Trends in indices of standing stock biomass of salmon mirror the trends in abundances at age; biomass values have been highest in the late 1990s and 2000s, declining recently from peak values of over 400 g of salmon biomass per 100 m² in both systems (Fig. 24). Percent habitat saturation also increased from low values in the 1970s and 1980s to high mean values that exceeded the reference value of 28 in the late 1990s and 2000s, but has declined in recent years (Fig. 24).

Of interest in the context of the supplementation question is how much additional juvenile production, and ultimately adult returns can be expected from increasing egg depositions. Abundance indices, as number of fish per size group, standing stock biomass of all salmon, and percent habitat saturation were at maximum values in the late 1990s and have declined in recent years. The indices remain above the lower values estimated in the 1970s and 1980s.

The associations between the estimated egg deposition rates (total eggs divided by total freshwater rearing area) and indices of fry abundance, small parr abundance lagged to the year of egg deposition are shown in Figure 25. The maximum mean fry density in any of the four rivers over the time series has been 143 fry per 100 m² in the Southwest Miramichi River in 1999 (Fig. 25). The maximum mean annual value for the Northwest or Southwest system, or the Miramichi River has been 134 fry per 100 m².

The estimated egg depositions for the Miramichi during the 1970s are not consistent with juvenile indices with mean fry indices being substantially too low compared to contemporary values since 1992 (Figs. 23, 25). This suggests that the egg depositions for that time period were overestimated.

The fry to small parr associations and the small parr to large parr associations show cohort consistency through the time series (Fig. 25).

Egg to fry dynamic

The subsequent stock and recruitment dynamic was modelled using the Northwest and Southwest system time series which extends from the 1998 to 2013 egg deposition years. The estimated egg deposition data from 1992 to 2013 and corresponding fry abundance indices from 1993 to 2014 suggest a strong compensatory relationship (Figs. 26, 27). There was stronger statistical support for the Beverton-Holt (BH) SR relationship than the linear proportional relationship for the Southwest Miramichi system (Fig. 26) and the Northwest Miramichi system (Fig. 27) and the individual main rivers in each of the systems (Figs. 26, 27). The median value of α, the slope at the origin, of the BH relationship is about 0.69 for the Southwest Miramichi and 0.93 for the Northwest Miramichi. This indicates that egg to fry survival is very high at very low egg densities however the values should be interpreted with caution:

- The α parameter was constrained in the model to be between 0 and 1 which is biologically realistic however the posterior distribution of α is highly uncertain (95% Bayesian Credibility Interval 0.39 – 0.98 for the Southwest system, 0.70-1.00 for the Northwest system).
- The total egg deposition is divided by total rearing area but the fry indices are derived from sampling at a limited number of sites which are not randomly or proportionally distributed among habitat types, tributaries, and stream orders of the rivers (Moore and Chaput 2007). As such, the fry indices from these selected sites are likely higher than the average density of fry over all habitats in the river. As a result the α parameter would be biased upward.
- There is no accounting of the uncertainties in either the egg deposition estimates nor of the indices of fry. If these uncertainties were included, the average relationships would be different and the parameter estimates even more uncertain.

The maximum carrying capacities of fry, as derived using the monitored sites in the river, are greater than 158 fry per 100 m² (95% BCI 114-409 fry per 100 m²) for the Southwest Miramichi system and 105 fry per 100 m² (95% BCI 84 – 144 fry per 100 m²) for the Northwest Miramichi system (Figs. 26, 27). Within each of the systems, the Southwest Miramichi River (exclusive of the Renous River) has a carrying capacity for fry of 173 fish per 100 m² and the Northwest Miramichi River is at 148 fry per 100 m² (Figs. 26, 27), values higher in each case than the system values because of the lower abundances of fry sampled from the Renous River in the Southwest system and in the Little Southwest in the Northwest system (Fig. 23).

The same caveats regarding the interpretation of the carrying capacity values as absolute levels discussed for the α parameter apply here as well.

In terms of fits, there is a pattern of temporal blocking of the residuals in the Southwest Miramichi data, with positive residuals for approximately the first half of the analysed time series (1993 to 2001 year classes) followed by negative residuals in the second half (Fig. 26). The consequence of inter yearclass competition on the residuals was not consistent with the hypothesis of fry survival being reduced when there is a high abundance of parr; the opposite dynamic was suggested in the Southwest with the abundance of fry higher in years when parr biomass was also high (Fig. 26).

In contrast, there was no pattern of temporal blocking of residuals for the Northwest Miramichi data series. There was no evident effect of biomass of parr size groups on predicted fry abundance (Fig. 27).

The predicted egg density that results in 50% of carrying capacity (half saturation) is estimated to be 221 eggs per 100 m² (median; 95% BCI 122 – 1,033 eggs per 100 m²) for the Southwest Miramichi system. For the Northwest Miramichi system, the egg density for half saturation is estimated at 113 eggs per 100 m² (95% BCI 87 – 191 eggs per 100 m²). For the Northwest

Miramichi River (excluding the Little Southwest Miramichi), the half saturation value is 159 eggs per 100 m² (95% BCI 118-278 eggs per 100 m²).

Fry to small parr dynamic

The fry to small parr dynamics did not suggest a strong compensatory relationship; the abundance of small parr is a linearly proportional function of fry the previous year in the Southwest Miramichi but there is a weak compensatory relationship for the Northwest Miramichi (Fig. 28). The proportionality parameter for the Southwest Miramichi was estimated at 0.29 (95% BCI 0.25 – 0.37) and a very high carrying capacity (median = 227 small parr per 100 m²; 95% BCI 60-483). For the Northwest Miramichi, the proportionality value was estimated at 0.81 (95% BCI 0.56-0.99) and a carrying capacity value of 51 small parr per 100 m² (95% BCI 35 – 148) (Fig. 28). There is no pattern of temporal blocking of residuals for either river system and the biomass of competing yearclasses (estimated as the average of the biomass of combined small parr and large parr in the fry year and the combined biomass of fry and large parr in the small parr year) was not associated with variations in small parr abundance corrected for fry abundance (Fig. 28).

Smolt production

The contemporary estimates of annual smolt abundance from the Northwest Miramichi and the Southwest Miramichi are highly variable and generally low relative to values expected from rivers in this area (Elson 1975; Symons 1979). Smolt abundance estimates from the Northwest Miramichi system varied from 1.0 to 4.6 smolts per 100 m² of total riverine habitat with more than half the estimates being less than 2 smolts per 100 m² (Table 2). Estimated abundances of migrating smolts from the Little Southwest Miramichi have consistently been less than 1.6 smolts per 100 m² (Table 2). Smolt production from the Southwest Miramichi system has generally been much higher than the Northwest Miramichi, ranging from 1.0 to 6.1 smolts per 100 m², with annual estimates greater than 2.5 smolts per 100 m² since 2004 (Table 2).

Compared to the parr abundances at the monitored sites, that ranged from greater than 30 per 100 m² small parr and large parr combined, it seems that there is a low smoltification rate (product of survival rate and smolt emigration probability) for juveniles in both branches of the Miramichi.

MARINE RETURN RATES

Smolt return rates

Estimated return rates of Northwest Miramichi system smolts to maiden spawners (sum of 1SW and 2SW returns from a smolt class) were estimated to be as low as 0.6% to as high of 7.6% for the smolt migration years 1999 to 2006 and 2011 (Table 2). Estimated return rates for the Southwest Miramichi system were also variable, ranging from 1.7% to 11.9%, the high values for each branch being estimated from returns of the 2001 smolt class (Table 2).

These return rate values are in the range of values estimated from monitored multi-sea-winter salmon populations of the Maritime provinces and Quebec (ICES 2015).

Repeat spawner return rates

The Atlantic salmon population of the Miramichi River is characterized by an expanding spawning history structure (Table 3). Between 1971 and 1986, there were few repeat spawners in the river with at most two previous spawning migrations. Since 1992 and 1995, adult salmon on their sixth and seventh spawning migrations, respectively, have been sampled in the catches at the estuary trapnets and repeat spawning salmon have comprised 6% to 21% of the total returns of all age groups (Chaput and Jones 2006). Post-spawned salmon (kelt) over-winter in the Miramichi River and return to the ocean early in the spring, immediately following ice-out of the river. The area occupied at sea by consecutive spawners from the Miramichi River during reconditioning is not known, though it is likely limited to the Gulf of St. Lawrence given the timing of their return back to the Miramichi. Alternate spawners from the Miramichi River undertake long oceanic migrations, as far as West Greenland, as evidenced from recaptures in marine fisheries.

The proportion of maiden salmon in the total returns to the Miramichi has declined from over 95% prior to 1986 to about 85% since 1996 while the relative abundance of salmon on a second spawning migration has increased from less than 5% of total returns prior to 1995 to over 10% in most years since (Fig. 29). Repeat spawners have become most important in the large salmon category as these fish grow when they return to the sea post-spawning (Fig. 29).

Estimated return rates to a second spawning of both 1SW and 2SW salmon increased between1972 and 2010 (Chaput and Benoît 2012). Since the late 1990s, return rates to a second spawning have ranged from 8% to 25% for 1SW salmon, and 10% to 40% for 2SW salmon. Increased return rates to consecutive spawning have contributed the most to the increased return rates for both the 1SW and 2SW maiden life histories (Fig. 30). A higher proportion of the returns to a second spawning were of the alternate spawning history in both 1SW and 2SW salmon prior to the 1990s but since then, the proportions of the second consecutive spawning returns have exceeded those of the alternate spawning history in both 1SW and 2SW salmon (Fig. 30). Chaput and Benoît (2012) reported on a positive association between the variations in the return rates of repeat spawners and the variations in a small fish biomass index from the southern Gulf of St. Lawrence, an area which could be used by kelts early in the reconditioning year at sea, which provides evidence that abundant food supplies at sea may be beneficial for the survival of Atlantic salmon to a second consecutive spawning. This contrasted with the absence of an association between prey availability and return rates of alternate repeat spawners, suggesting that return rates of the alternate strategy are conditioned by high seas factors.

CONCLUSIONS

The Atlantic salmon population from the Miramichi is characterized by complex phenotypic diversity that is moderately variable over the medium term (about two generations or 10 years). Juveniles rear in freshwater for two to five years, with most migrating to sea after two and three years of freshwater residency and return to rivers as 1SW maiden and 2SW maiden salmon. In any given year, there are six year classes of immature fish in the combined freshwater and marine ecosystem (four years in river including eggs, two years at sea). Chaput and Jones (2006) estimated that for the Miramichi River, the number of year classes present in the annual spawning migration has increased from four to five in the 1970s to as many as nine year classes in the returns of the 1990s. This large number of immature age groups and the increased number of contributing year classes provides population resilience to stochastic and demographic perturbations.

Although there is a paucity of empirical evidence to inform on sub-basin population structuring in the Miramichi, a number of phenotype characteristics are consistent with such structure. Salmon in headwater areas of the river at higher elevations are predominantly fish which returned to tidal waters prior to September 1 and the proportions of late run salmon increase in lower elevation areas in the river. There is an important sex ratio bias between the maiden sea age groups with males being more abundant in the 1SW salmon than in 2SW salmon. This is consistent with life history theory: fitness of males can be optimized at smaller body sizes (i.e. precocious male parr, small salmon) whereas reproductive fitness of female salmon is enhanced by increased body size which is associated with increased egg size, increased egg number per female, and more diverse spawning habitat resulting in increased survival of offspring (Fleming 1996).

Although presently subjected to seemingly lower return rates from sea for maiden age groups, the contemporary return rates of maiden salmon estimated for the Miramichi are in the same range of return rates for other stocks in the Maritime provinces and Quebec. Empirical evidence from monitored stocks indicates that sea survival of maiden salmon was substantially higher historically than at present and that it is low marine survival that is constraining abundance of Atlantic salmon throughout the North Atlantic (ICES 2015).

Increased return rates to a second spawning, particularly for the consecutive life history component, is interpreted to be a response to improved feeding opportunities in the southern Gulf of St. Lawrence, an example of phenotypic plasticity in this population in response to changes in the ecosystem.

Separate branch estimates of returning adult salmon were initiated in 1992. Although the methods used to estimate the returns of 1992 to 1997 warrant revisiting, the model used to estimate the returns for the period 1998 to 2014 has been peer reviewed and is considered to provide accurate estimates of branch specific returns.

For the Miramichi River and its branches, the question remains whether freshwater productivity is at a level expected from a “normal” multi-sea-winter Atlantic salmon population of the Maritime provinces. Indices of juvenile abundance, dating more than four decades, obtained using consistent procedures and using models to estimate site specific abundances, indicate that freshwater production of Atlantic salmon increased to peak values in the late 1990s and early 2000s. Although abundance indices of fry and small parr have declined somewhat since, the abundance indices are at levels well above those of the 1970s to mid-1980s. Increased abundances of large parr over the four decades of monitoring points to changes in the freshwater dynamics which may be contributing to increased density dependent regulation in freshwater (through increased competition for resources) and reduced smolt production. Smolt production from the Northwest Miramichi was at low to medium rates (less than three smolts per 100 m^2) despite the high indices of abundance of small parr (potential contributors of 2-year old smolts) and large parr (potential contributors of 3-year old and older smolts).

Preliminary stock and recruitment analyses presented here indicate that the most important density dependent control occurs between the egg and fry stage with modelled theoretical carrying capacity of fry almost realized in the late 1990s. Fry densities, although highly variable annually, remain at moderate levels in all rivers despite the decline in egg depositions to values below the conservation requirement (management reference point synonymous with a limit reference point) in the past decade.

If the egg to fry recruitment dynamic is as severely compensatory as suggested by these analyses, there may be very little gain to be realized in smolt production and subsequent adult returns by supplementing the spawning escapement with large numbers of captive-reared adult spawners. Density dependent factors will rapidly adjust the abundances of juveniles to levels which can be sustained by the productive capacity of the freshwater habitat, carrying capacity constraints which are inferred to be at play in the Miramichi at the present time.

Due to the strong density dependent survival that is realized in freshwater, the addition of a large number of captive-reared adult progeny to the river will result in increased density-dependent mortality of natural/wild progeny. While an immediate increase in the number of juveniles may appear to be beneficial overall, any reduced fitness of the captive-reared progeny with phenotypic differences (body size, growth rates, maturation rates) will result in reduced abundance of wild Atlantic salmon (DFO 2016; Fraser 2016).

REFERENCES CITED

Amiro, P.G. 1983. Aerial photographic measurement of Atlantic salmon habitat of the Miramichi River, New Brunswick. CAFSAC Res. Doc. 83/74.

Biron, M., Clément, M., Moore, D., and Chaput, G. 2014. Results of a Multi-year Control and Eradication Program for Smallmouth Bass (*Micropterus dolomieu*) in Miramichi Lake, New Brunswick, 2011-2012. DFO Can. Sci. Advis. Sec. Res. Doc. 2014/073. v + 58 p.

Blousfield, E.L. 1955. Some physical features of the Miramichi estuary. J. Fish. Res. Board Canada 12: 342-361.

Bradbury, I., Hamilton, L., Rafferty, S., Meerburg, D., Poole, R., Dempson, J.B., Robertson, M., Reddin, D., Bourret, V., Dionne, M., Chaput, G., Sheehan, T., King, T. and Bernatchez, L. 2014. Genetic evidence of local exploitation of Atlantic salmon in a coastal subsistence fishery in the Northwest Atlantic. *Canadian Journal of Fisheries and Aquatic Sciences* 72: 83–95.

Brodeur, N. 2006. Dispersion patterns of kin in young-of-the-year Atlantic salmon (*Salmo salar*) in Catamaran Brook, New Brunswick. MSc Thesis, Concordia University, Montreal, Quebec.

Cairns, D., Angus, R., Murray, M., and Davidson, K. 1996. Status of Atlantic salmon in the Morell, Mill, Dunk, West and Valleyfield rivers, Prince Edward Island, in 1995. DFO Atl, Fish. Res. Doc. 1996/120. iii + 33 p.

Chaput, G., and Benoît, H.P. 2012. Evidence for bottom–up trophic effects on return rates to a second spawning for Atlantic salmon (*Salmo salar*) from the Miramichi River, Canada. – ICES J. Mar. Sci. 69: 1656–1667.

Chaput, G., and Jones, R. 2006. Reproductive rates and rebuilding potential for two multi-sea-winter Atlantic salmon (*Salmo salar* L.) stocks of the Maritime provinces. DFO Can. Sci. Adv. Secr. Res. Doc. 2006/027.

Chaput, G., Moore, D., Hayward, J., Sheasgreen, J., and Dubee, B. 2001. Stock status of Atlantic salmon (*Salmo salar*) in the Miramichi River, 2000. DFO Can. Sci. Adv. Secr. Res. Doc. 2001/008. 89 p.

Chaput, G., Hardie, P., Hayward, J., Moore, D., Shaesgreen, J., and NSPA. 2002. Migrations and biological characteristics of Atlantic salmon (*Salmo Salar*) smolts from the Northwest Miramichi River, 1998 to 2000. Can. Tech. Rep. Fish. Aquat. Sci. No. 2415. 70p.

Chaput, G., Moore, D., and Peterson, D. 2005. Predicting Atlantic salmon (*Salmo salar*) juvenile densities using catch per unit effort open site electrofishing. Can. Tech. Rep. Fish. Aquat. Sci. No. 2600. v + 25 p.

Chaput, G., Moore, D., Hardie, P., and P. Mallet. 2010. Information on Atlantic salmon (*Salmo salar*) from Salmon Fishing Area 16 (Gulf New Brunswick) of relevance to the development of a COSEWIC status report. DFO Can. Sci. Advis. Sec. Res. Doc. 2010/064. iv + 50 p.

Claytor, R.R., Chaput, G.J., and Lutzac, T.G. 1987. Assessment of Atlantic salmon, (*Salmo salar*), in the Margaree River, 1987. CAFSAC Res. Doc. 87/105. 36 p.

Connell, C.B., Dubee, B.L., and Cronin, P.J. 2002. Using rotenone to eradicate chain pickerel, *Esox niger*, from Despres Lake, New Brunswick, Canada. NB DNRE Fisheries Management Report 2002-01-E.

Cunjak, R.A., and Therrien, J. 1998. Inter-stage survival of wild juvenile Atlantic salmon, *Salmo salar* L. Fish. Manag. Ecol. 5: 209-223.

DFO. 2013. Review of control and eradication activities in 2010 to 2012 targeting Smallmouth Bass in Miramichi Lake, New Brunswick. DFO Can. Sci. Advis. Sec. Sci. Resp. 2013/012.

DFO. 2015a. Update of stock status of Atlantic Salmon (*Salmo salar*) in DFO Gulf Region (New Brunswick Salmon Fishing Areas 15 and 16) for 2014. DFO Can. Sci. Advis. Sec. Sci. Resp. 2015/008.

DFO. 2015b. Development of reference points for Atlantic salmon (*Salmo salar*) that conform to the Precautionary Approach. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2015/058.

DFO. 2016. Review and risk assessment of proposed adult captive-reared supplementation program on fitness of wild Atlantic salmon in the Miramichi River, New Brunswick, Canada. DFO Can. Sci. Adv. Secr. Adv. Rep. 2016/017.

Douglas, S.G., Chaput, G., Hayward, J., and Sheasgreen, J. 2015. Assessment of Atlantic Salmon (*Salmo salar*) in Salmon Fishing Area 16 of the southern Gulf of St. Lawrence to 2013. DFO Can. Sci. Advis. Sec. Res. Doc. 2015/049. v + 36 p.

El-Jabi, N., Swansburg, E., Chaput, G., and Caissie, D. 2004. Climate Change Impacts on Run-Timing of Atlantic Salmon in Eastern Canada and Adaptation of Inseason Models and Management to Improve Resource Access Opportunities. Climate Change Impacts and Adaptation Program Project A505.

Elliott, J.M. 2001. The relative role of density in stock-recruitment relationships of salmonids. Pages 25 – 66. In E. Prevost and G. Chaput (eds). Stock, recruitment and reference points, assessment and management of Atlantic salmon. INRA editions.

Elson, P.F. 1975. Atlantic salmon rivers smolt production and optimal spawning: an overview of natural production. International Atlantic Salmon Foundation Special Publication Ser. 6: 96-119.

Fleming, I.A. 1996. Reproductive strategies of Atlantic salmon: ecology and evolution. Reviews in Fish Biology and Fisheries 6: 379–416.

Fraser, D.J. 2016. Risks and benefits of mitigating low marine survival in wild salmon using smolt-to-adult captive-reared supplementation. DFO Can. Sci. Advis. Sec. Res. Doc. 2016/030. v + 31 p.

Gibson, A.J.F. 2006. Population Regulation in Eastern Canadian Atlantic salmon (*Salmo salar*) populations. DFO Can. Sci.Adv. Secr. Res. Doc 2006/016.

Grant, J.W.A., and Kramer, D.L. 1990. Territory size as a predictor of the upper limit to population density of juvenile salmonids in streams. Can. J. Fish. Aquat. Sci. 47: 1724-1737.

Hansen, L.P., and Jacobsen, J.A. 2000. Distribution and Migration of Atlantic Salmon, *Salmo salar* L., in the Sea. p. 75-87. In D. Mills [ed.] The Ocean Life of Atlantic Salmon: Environmental and Biological Factors Influencing Survival. Fishing News Books, Blackwell Science.

Hansen, L.P., and Quinn, T.P. 1998. The marine phase of the Atlantic salmon (*Salmo salar*) life cycle, with comparisons to Pacific salmon. Can. J. Fish. Aquat. Sci. 55 (Suppl. 1): 104-118.

Hayward, J. 2001. Weekly fish counts from in-river traps and barrier pools in the Miramichi River, New Brunswick, 1994 to 1999. Can. Data Rep. of Fish. Aquat. Sci. 1080: 104 p.

Hayward, J., Sheasgreen, J., Douglas, S. and Reid, J. 2014. Diadromous fish monitoring programs in the Miramichi River system in 2011. Can. Manuscr. Rep. Fish. Aquat. Sci. No. 3042: xiii + 102 p.

ICES. 2015. Report of the Working Group on North Atlantic Salmon (WGNAS), 17–26 March, Moncton, Canada. ICES CM 2015/ACOM:09. 332 pp.

Jonsson, N., Jonsson, B., and Hansen, L.P. 1998. The relative role of density-dependent and density-independent survival in the life cycle of Atlantic salmon *Salmo salar*. J. Anim. Ecol. 67:751-762.

Kerswill, C.J. 1971. Relative rates of utilization by commercial and sport fisheries of Atlantic salmon (*Salmo salar*) from the Miramichi River, New Brunswick. J. Fish.Res Bd Canada 28: 351-363.

Lunn, D., Jackson, C., Best, N., Thomas, A., and Spiegelhalter, D. 2013. The BUGS Book: a practical introduction to Bayesian Analysis. CRC Press, Chapman & Hall Book.

Møller, D. 1970. Transferrin polymorphism in Atlantic salmon (*Salmo salar*). J. Fish. Res. Board Can. 27: 1617–1625.

Møller, D. 2005. Genetic studies on serum transferrins in Atlantic salmon. J. Fish Biol. 67 (Suppl. A): 55–67.

Moore, D., and Chaput, G. 2007. Juvenile Atlantic salmon (*Salmo salar*) surveys in the Miramichi River Watershed from 1970-2004. Can. Data. Rep. Fish. Aquat. Sci. No. 1188. 117p.

Moore, D.S., Chaput, G.J., and Pickard, P.R. 1995. The effect of fisheries on the biological characteristics and survival of mature Atlantic salmon (*Salmo salar*) from the Miramichi River, p. 229-247. In E.M.P. Chadwick [editor]. Water, science, and the public: the Miramichi ecosystem. Can. Spec. Publ. Fish. Aquat. Sci. 123.

Moore, J.S., Bourret, V., Dionne, M., Bradbury, I., O'Reilly, P., Kent, M., Chaput, G., and Bernatchez L. 2014. Conservation genomics of anadromous Atlantic salmon across its North American range: outlier loci identify the same patterns of population structure as neutral loci. Mol. Ecol. 23: 5680-5697.

Paloheimo, J.E., and Elson, P.F. 1974. Reduction of Atlantic Salmon (*Salmo salar*) Catches in Canada Attributed to the Greenland Fishery. J. Fish. Res. Board Can. 31: 1467-1480.

Randall, R.G. 1989. Effect of sea age on the reproductive potential of Atlantic salmon (*Salmo salar*) in eastern Canada. Can. J. Fish. Aquat. Sci. 46: 2210-2218.

Randall, R.G., and Chadwick, E.M.P. 1983. Assessment of the Miramichi River salmon stock in 1982. CAFSAC Res. Doc. 83/21. 24p.

Randall, R.G., O'Connell, M.F., and Chadwick, E.M.P. 1989. Fish production in two large Atlantic coast rivers: Miramichi and Exploits, p. 92-308. *In* D.P. Dodge [ed.] Proceedings of the International Large River Symposium. Can. Spec. Publ. Fish. Aquat. Sci. 106.

Reddin, D.G. 1985. Atlantic salmon (*Salmo salar*) on and east of the Grand Bank. J. Northwest. Atl. Sci. 6:157-164.

Reddin, D.G., and Lear, W.H. 1990. Summary of marine tagging studies of Atlantic salmon (*Salmo salar* L.) in the Northwest Atlantic area. Can. Tech. Rep. Fish. Aquat. Sci. 1737. iv + 114 p.

Reid, J.E., and Chaput, G. 2012. Spawning history influence on fecundity, egg size, and egg survival of Atlantic salmon (*Salmo salar*) from the Miramichi River, New Brunswick, Canada. – ICES Journal of Marine Science, 69: 1678–1685.

Riddell, B.E., Leggett, W.C., and Saunders, R.L. 1981. Evidence of adaptive polygenic variation between two populations of Atlantic salmon (*Salmo salar*) native to tributaries of the S.W. Miramichi River, N.B. Can. J. Fish. Aquat. Sci. 38: 321-331.

Ritter, J.A. 1989. Marine migration and natural mortality of North American Atlantic salmon (*Salmo salar* L.). Can. MS Rep. Fish. Aquat. Sci. No. 2041. 136 p.

Saunders, R.L. 1967. Seasonal pattern of return of Atlantic salmon in the Northwest Miramichi River, New Brunswick. J. Fish. Res. Bd. Canada 24:21-32.

Saunders, R.L. 1969. Contributions of salmon from the Northwest Miramichi River, New Brunswick, to various fisheries. J. Fish. Res. Bd. Canada 26: 269-278.

Saunders, R.L. 1981. Atlantic salmon (*Salmo salar*) stocks and management implications in the Canadian Atlantic Provinces and New England, USA. Can. J. Fish. Aquat. Sci. 38: 1612-1625.

Swansburg, E., Chaput, G., Moore, D., Caissie, D., and El-Jabi, N. 2002. Size variability of juvenile Atlantic salmon: links to environmental conditions. J. Fish Biol. 61: 661-683.

Symons, P. 1979. Estimated escapement of Atlantic salmon (*Salmo salar*) for maximum smolt production in rivers of different productivity. J. Fish. Res. Board Can. 36:132-140.

TABLES

Table 1. Physical and environmental characteristics of the Miramichi River and the two main branches, the Southwest Miramichi and the Northwest Miramichi (Figure 1).

Characteristics	Miramichi	Southwest Miramichi	Northwest Miramichi
Latitude (range)		46º10’ to 47º20’	
General direction of flow		west to east into Gulf of St. Lawrence	
Max. elevation		470 m	
Longest distance from the sea		250 km	
Number of days with ice cover		December to April; 100 to 170 days (1961 – 2002)	
Mean air temperature		-11ºC in January; +19ºC in July	
River order	7	7	6
Basin area (million ha)	1.36	0.77	0.39
Wetted rearing area (ha) for salmon production by stream order			
Stream Order			
1	15	6	8
2	222	125	97
3	666	338	314
4	1,493	885	507
5	1,250	617	633
6	1,478	1,358	120
7	336	336	0
Total	5,461	3,666	1,679

Table 2. Summary of smolt migration characteristics from monitoring programs in the Northwest Miramichi system, the Little Southwest Miramichi River, and the Southwest Miramichi system, 1998 to 2011. Values in grey shading are uncertain and considered to be underestimates.

River	Smolt year	Run size estimate			Smolts per 100 m²		Size (mean)		Prop. female	Prop. at freshwater age			Run timing		Return rates at maiden age		
		Median	95% confidence interval		Median	95% C.I.	mm	g		2	3	4	Peak	5th perc.	1SW	2SW	Combined
Northwest Miramichi	1998	na	na	na	na	na	129	21.8	0.49	0.28	0.71	0.01	16-May	15-May	na	na	na
	1999	390,500	315,500	506,000	2.3	1.9 - 3.0	132	22.4	0.63	0.36	0.62	0.02	19-May	15-May	3.1%	1.3%	4.3%
	2000	162,000	118,000	256,000	1.0	0.7 - 1.5	131	21.2	0.58	0.34	0.63	0.03	02-Jun	18-May	5.2%	0.5%	5.7%
	2001	220,000	169,000	310,000	1.3	1.0 - 1.8	130	21.1	0.53	0.38	0.60	0.01	29-May	21-May	6.8%	0.8%	7.6%
	2002	241,000	198,000	306,000	1.4	1.2 - 1.8	128	20.7	0.57	0.52	0.48	0.00	02-Jun	24-May	2.5%	0.8%	3.3%
	2003	286,000	224,500	388,000	1.7	1.3 - 2.3	128	21.2	0.53	0.50	0.49	0.01	28-May	24-May	4.2%	1.0%	5.1%
	2004	368,000	290,000	496,000	2.2	1.7 - 3.0	131	22.1	0.57	0.41	0.58	0.01	19-May	16-May	2.6%	0.5%	3.1%
	2005	*151,200*	*86,000*	*216,000*	0.9	na	130	21.4	0.52	0.40	0.60	0.01	08-Jun	19-May	na	na	na
	2006	*435,000*	*255,000*	*1,230,000*	2.6	na	130	23.3	0.56	0.44	0.56	0.01	16-May	13-May	na	na	na
	2011	768,000	576,000	1,137,000	4.6	3.4 - 6.8	133	18.1	0.42	0.61	0.38	0.00	21-May	21-May	0.3%	0.2%	0.6%
Little Southwest Miramichi	2005	*46,330*	*32,710*	*68,050*	na	na	130	na	0.58	0.22	0.76	0.02	14-May	13-May	na	na	na
	2006	87,520	41,760	665,300	1.0	0.5 - 7.6	130	na	0.51	0.51	0.49	0.00	18-May	10-May	na	na	na
	2007	138,200	106,000	185,500	1.6	1.2 - 2.1	125	na	0.57	0.34	0.66	0.00	22-May	12-May	na	na	na
	2008	124,100	96,320	164,900	1.4	1.1 - 1.9	130	21.6	0.50	0.38	0.61	0.01	21-May	16-May	na	na	na
	2009	85,000	66,000	112,000	1.0	0.8 - 1.3	129	na	0.52	0.38	0.62	0.00	18-May	13-May	na	na	na
	2010	46,500	28,500	82,500	0.5	0.3 - 0.9	140	na	na	0.35	0.64	0.01	12-May	07-May	na	na	na
	2011	67,900	49,900	104,500	0.7	na	131	22.8	0.47	0.44	0.56	0.00	26-May	21-May	na	na	na
Southwest Miramichi	2001	306,300	290,000	464,000	1.0	0.8 - 1.3	127	19.2	0.47	0.64	0.35	0.00	31-May	22-May	8.6%	3.3%	11.9%
	2002	711,400	498,000	798,000	1.7	1.4 - 2.3	126	18.8	0.54	0.55	0.44	0.01	01-Jun	19-May	3.1%	1.4%	4.5%
	2003	485,000	393,000	615,000	1.3	1.1 - 1.7	128	19.6	0.58	0.59	0.41	0.00	22-May	22-May	6.8%	2.0%	8.8%
	2004	1,167,000	969,000	1,470,000	3.2	2.6 - 3.5	130	21.1	0.54	0.60	0.40	0.00	17-May	16-May	1.8%	0.8%	2.5%
	2006	1,332,000	983,000	1,809,000	3.8	2.8 - 5.1	131	23.1	0.55	0.54	0.46	0.00	17-May	09-May	1.5%	0.5%	2.0%
	2007	1,344,000	1,120,000	1,668,000	3.8	3.2 - 4.7	132	20.7	0.49	0.59	0.41	0.00	27-May	21-May	1.6%	0.8%	2.4%
	2008	901,500	698,000	1,262,000	2.5	2.0 - 3.6	126	19.7	0.60	0.67	0.33	0.00	28-May	22-May	1.0%	0.7%	1.7%
	2009	1,035,000	807,000	1,441,000	2.9	2.3 - 4.1	128	22.1	0.53	0.69	0.31	0.00	18-May	15-May	3.3%	2.2%	5.5%
	2010	2,165,000	1,745,000	2,725,000	6.1	4.9 - 7.7	137	23.9	0.51	0.57	0.43	0.00	21-May	07-May	1.5%	0.4%	1.8%

Table 3. Number of samples by spawning histories of Atlantic salmon aged from the Southwest Miramichi system and the Northwest Miramichi system, 1992 to 2013. Spawning histories are interpreted as: XSW is the maiden sea winter age at first spawning, the sequence of C (consecutive) and A (alternate) represent the at sea reconditioning history for each successive spawning event. The maximum total sea age of salmon interpreted to date is nine years (2SWAAAC, 2SWACCCCC).

Spawning History	Southwest Miramichi	Northwest Miramichi
1SW	17,792	9,791
1SWA	631	331
1SWAA	26	16
1SWAAA	2	2
1SWAAAC	1	na
1SWAAC	6	6
1SWAACC	2	na
1SWAC	66	46
1SWACA	1	na
1SWACC	22	24
1SWACCC	11	2
1SWACCCC	1	na
1SWACCCCC	na	1
1SWC	869	393
1SWCA	10	na
1SWCAC	1	na
1SWCC	151	63
1SWCCA	1	na
1SWCCC	38	10
1SWCCCC	8	4
1SWCCCCC	3	1
1SWCCCCCC	3	na
2SW	9,043	4,479
2SWA	705	366
2SWAA	89	48
2SWAAA	8	4
2SWAAAC	1	na
2SWAAC	18	7
2SWAACC	3	na
2SWAACCC	1	na
2SWAC	314	139
2SWACA	3	na
2SWACC	121	63
2SWACCC	23	10
2SWACCCC	5	1
2SWACCCCC	1	na
2SWC	910	431
2SWCA	12	5
2SWCAC	7	1
2SWCACC	1	na
2SWCC	334	145
2SWCCA	1	1
2SWCCC	174	69
2SWCCCC	65	25
2SWCCCCA	na	1
2SWCCCCC	17	7
2SWCCCCCC	3	1
3SW	14	7
3SWA	1	2
3SWAC	na	1
3SWC	4	1
3SWCC	1	1

Table 4. Enhancement activities conducted in the Miramichi River, 1978 to 2008 (Chaput et al. 2010). Stage of stocking is represented by: UF = unfed fry, FF = feeding fry, FG = fall fingerlings, P = 1+ parr, Sm = smolts.

River	Longitude (degree decimal W)	Latitude (degree decimal N)	Origin of fish stocked	Life stages of fish stocked	Range in annual numbers of fish stocked	Range of years when stocking occurred
Northwest Miramichi	-65.8333	46.9500	NW Miramichi	F, FG, P, Sm	13,000 - 133,000	1978 - 2007
Little Southwest Miramichi	-65.8333	46.9500	LSW Miramichi	F, FG, Sm	800 - 106,400	1978 - 2008
Renous and Tributaries	-65.7833	46.8167	SW Mir., Dungarvon	F, FG, P, Sm	2,200 - 118,000	1987 - 2007
Southwest Miramichi	-65.5833	46.9667	Tributary specific	F, FG, P, Sm	9,000 - 469,400	1978 - 2008

FIGURES

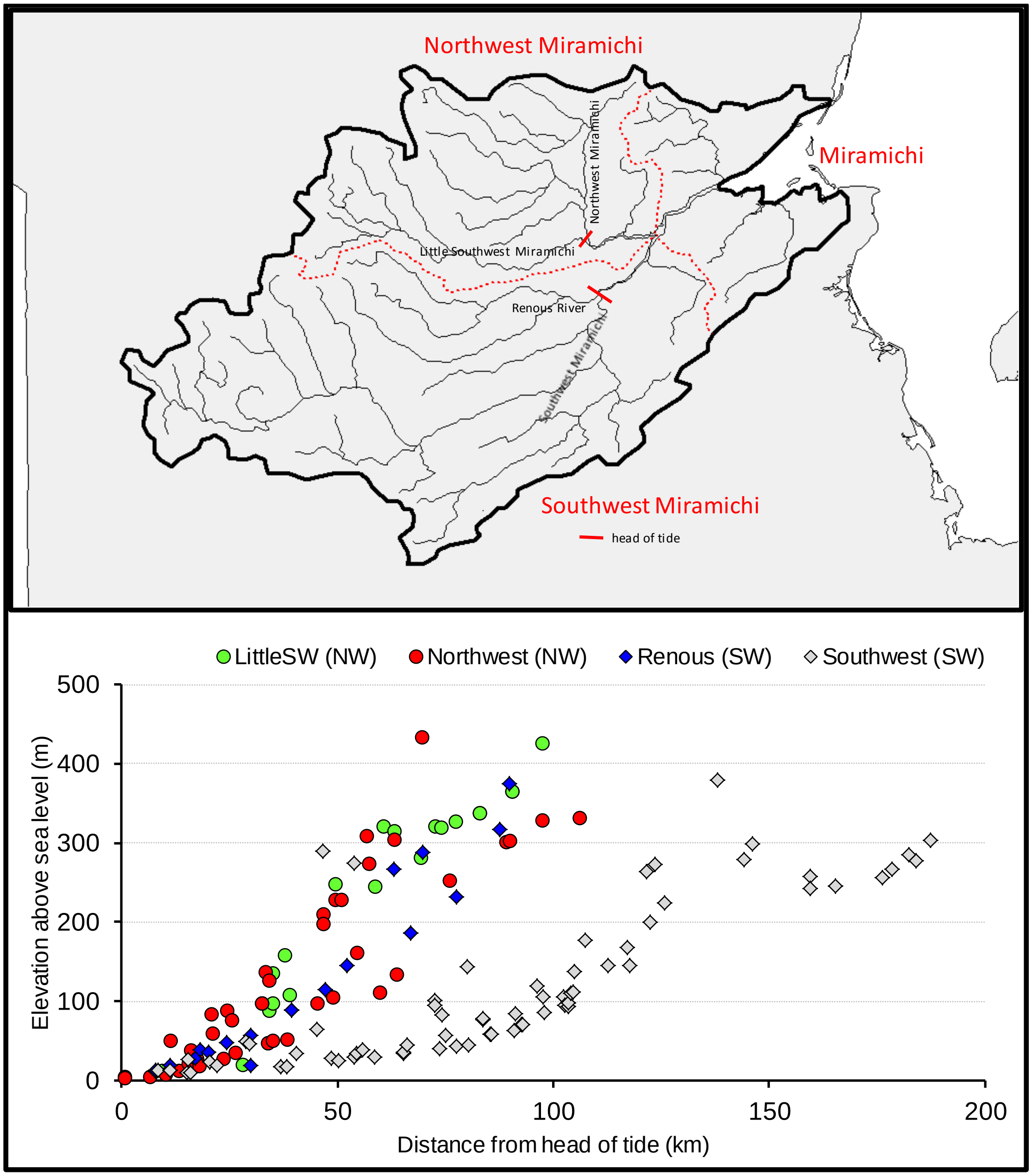

Figure 1. The Miramichi River drainage area and location of main rivers within the Miramichi referred to in text (upper panel) and approximate relief profile (elevation in m above sea level versus distance in km from the head of tide) of the four main rivers based on locations of electrofishing sites within the river (lower panel).

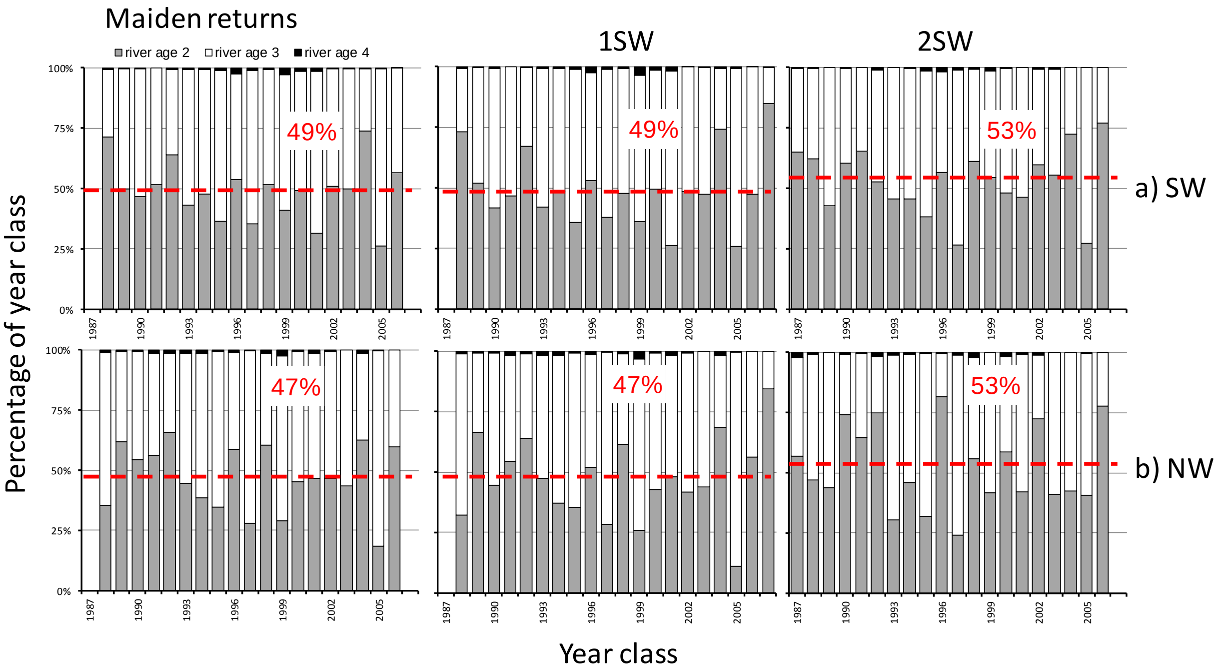

Figure 2. Percent at river age (2, 3, 4 years) by year class for the sum of maiden returns (1SW, 2SW) (left panel), for 1SW maiden (middle panels), and 2SW maiden (right panels) for the Southwest Miramichi River (a; top row) and the Northwest Miramichi River (b; lower row). The values in each cell are the average percentage by year class which were river age 2. The year class values are derived based on scale sampling of adult salmon returning to the estuary trapnets in each branch weighted by the returns of salmon to each branch (median value) for 1998 to 2013. Year class refers to the year of spawning (fall) and is calculated as the year of return minus maiden sea age (1 or 2), minus river age (2, 3, 4) minus one.

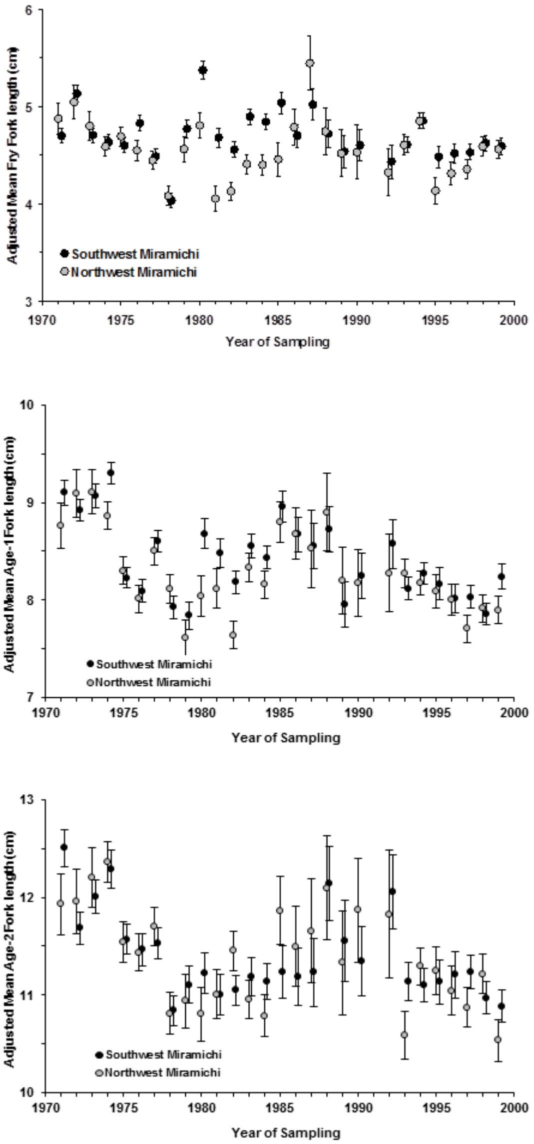

Figure 3. Annual adjusted mean fork length of fry (top), age-1 parr (middle), and age-2 parr (lower) from the Northwest Miramichi and Southwest Miramichi rivers, 1971 to 1999 (Swansburg et al. 2002). Mean fork lengths (cm +/- 1 standard error) were adjusted for date of sampling, density of the age group, tributary, and year separately for each branch. Data from 1991 are omitted because an inadequate number of sites were sampled.

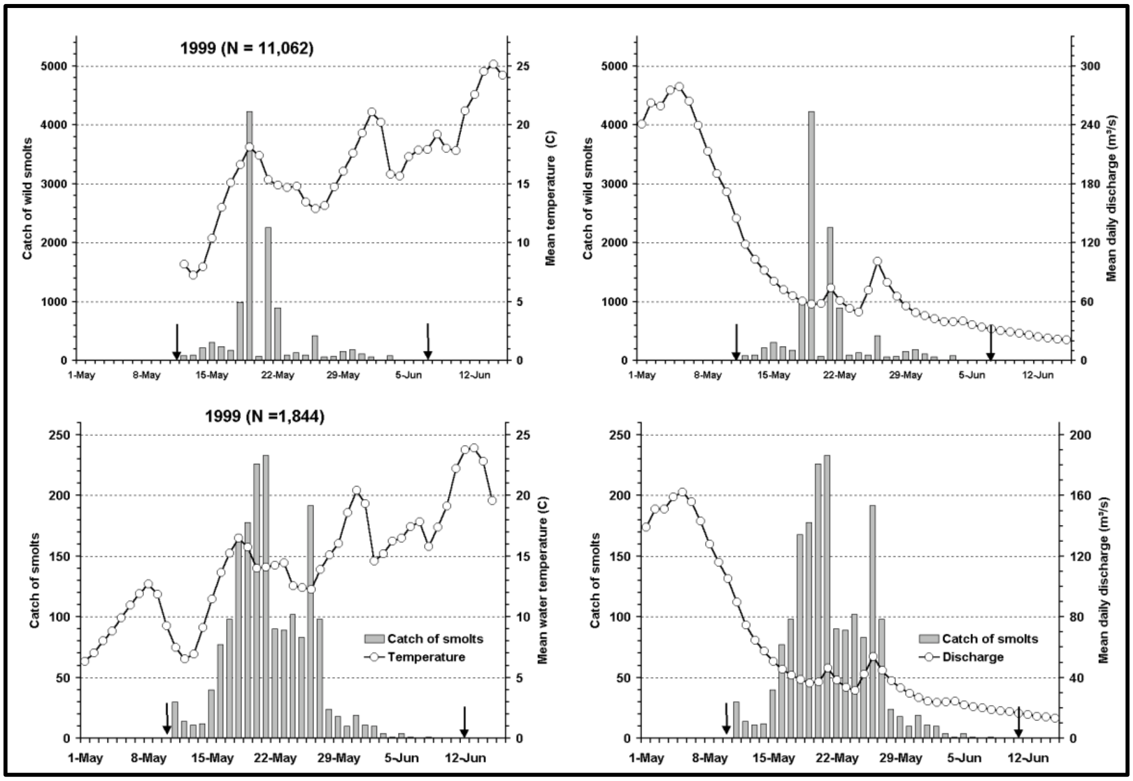

Figure 4a. Timing of catches of wild smolts in 1999 from the estuary tidal trapnet of the Northwest Miramichi system (top row) and at the rotary screw trap in the Little Southwest Miramichi (near Catamaran Brook, about 29 km above the head of tide; bottom row) relative to the mean daily water temperature (left column) and mean daily discharge (right column) (from Chaput et al. 2002). Arrows represent the total smolt migration period.

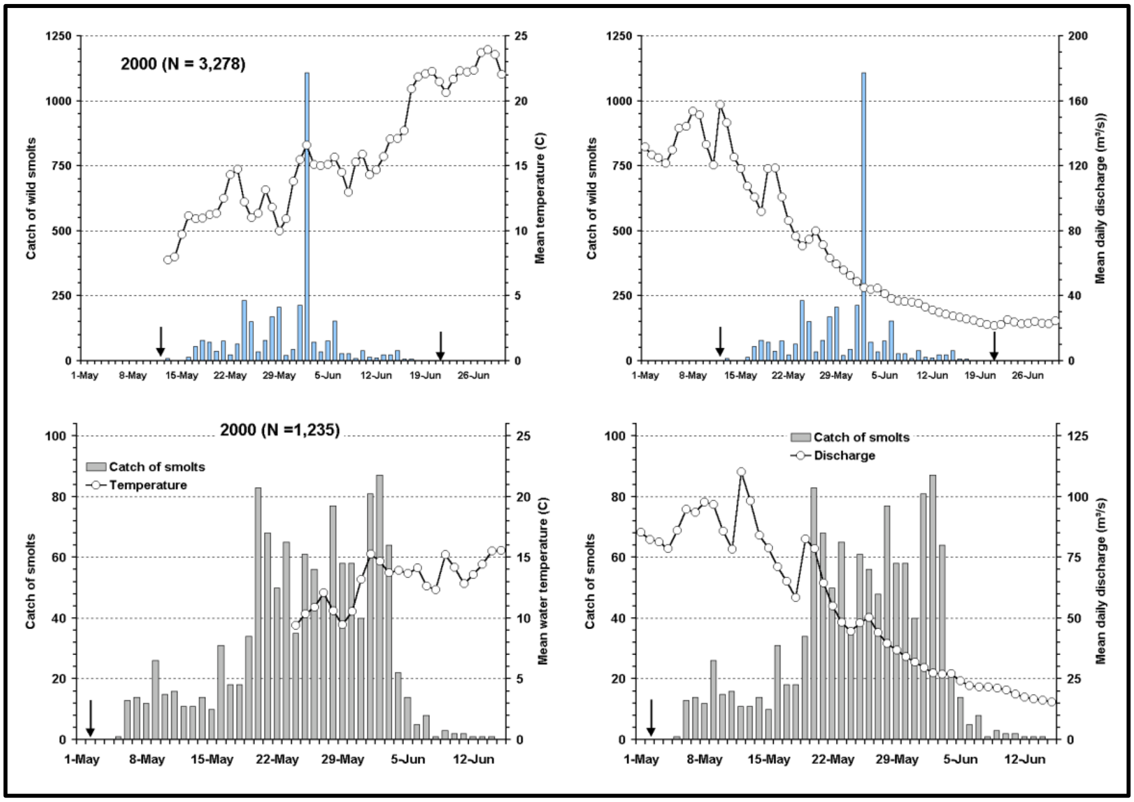

Figure 4b. Timing of catches of wild smolts in 2000 from the estuary tidal trapnet of the Northwest Miramichi system (top row) and at the rotary screw trap in the Little Southwest Miramichi (near Catamaran Brook, about 29 km above the head of tide; bottom row) relative to mean daily water temperature (left column) and mean daily discharge (right column) (from Chaput et al. 2002). Arrows represent the total smolt migration period.

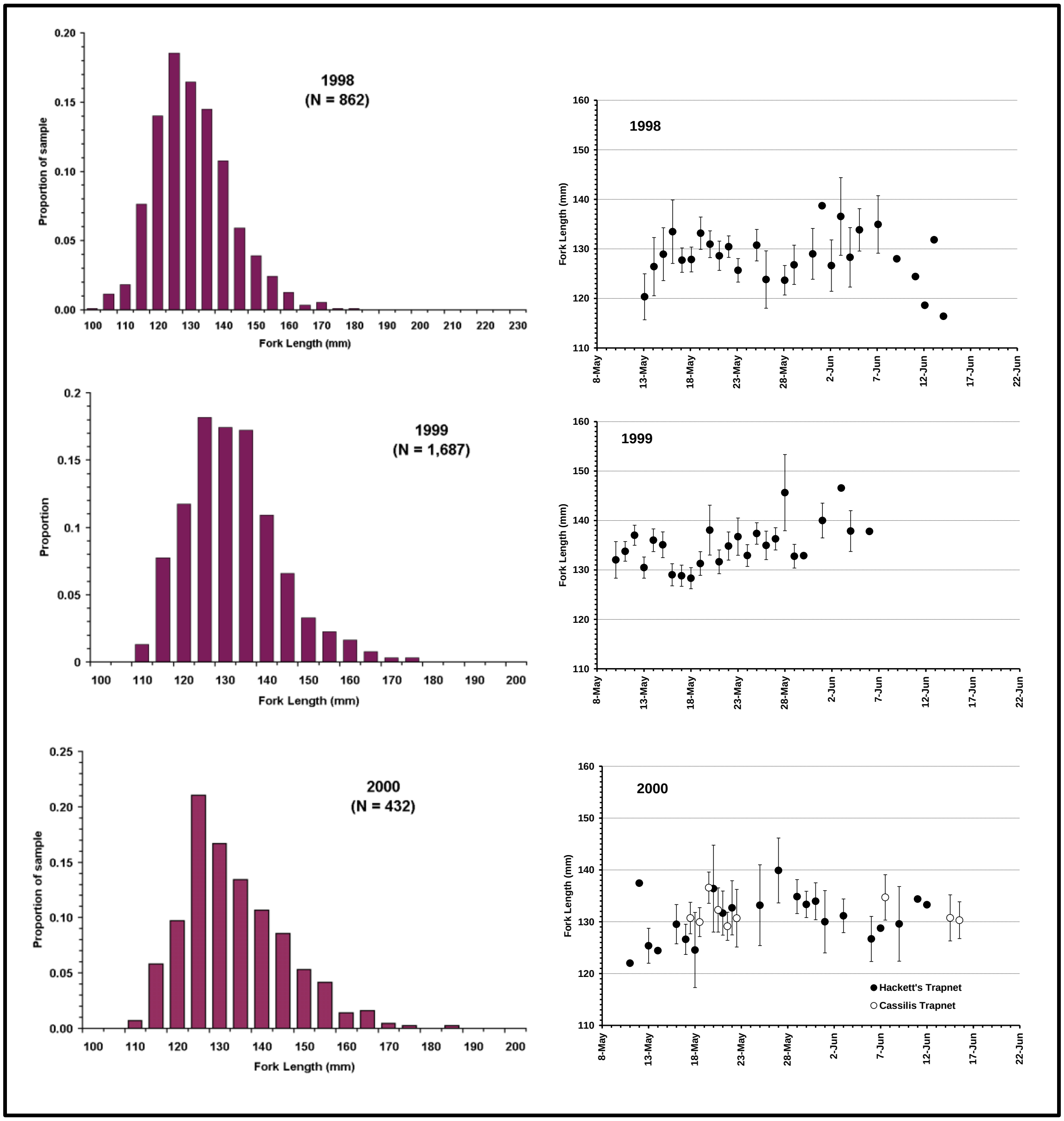

Figure 5. Fork length (mm) distribution of wild smolts (daily samples weighted by daily total catch) (left panel) and mean size (with 2 standard error bars) by date within year from the Northwest Miramichi from 1998 to 2000 (from Chaput et al. 2002).

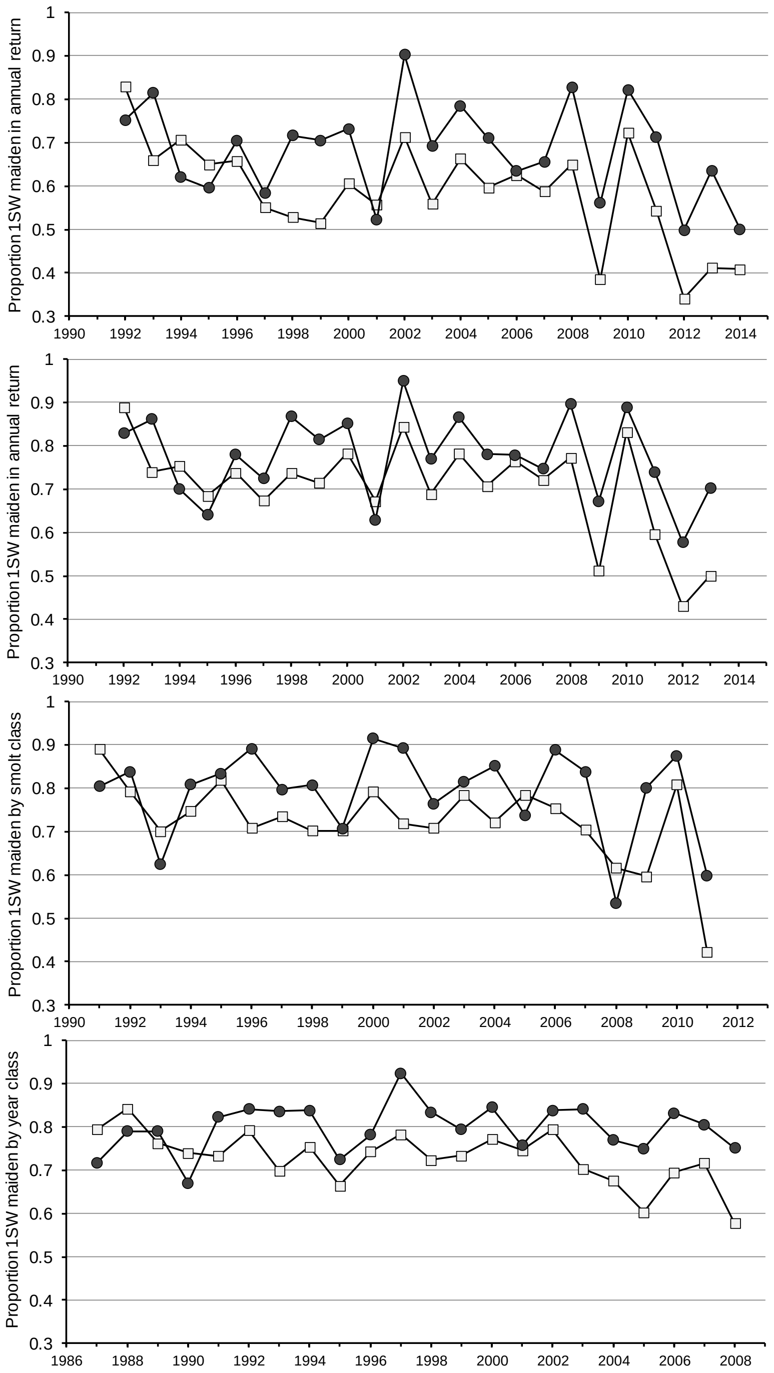

Figure 6. Proportion small salmon in the annual return of all salmon (upper panel), proportion 1SW maiden in the annual return of maiden sea age salmon (second row), proportion 1SW maiden in the returns of maiden sea age salmon by smolt class (year of smolt migration, third row), and proportion 1SW maiden of total maiden return by yearclass (bottom panel), for the Northwest Miramichi (black symbols) and the Southwest Miramichi (open square symbols) systems (Douglas et al. 2015; DFO 2015a).

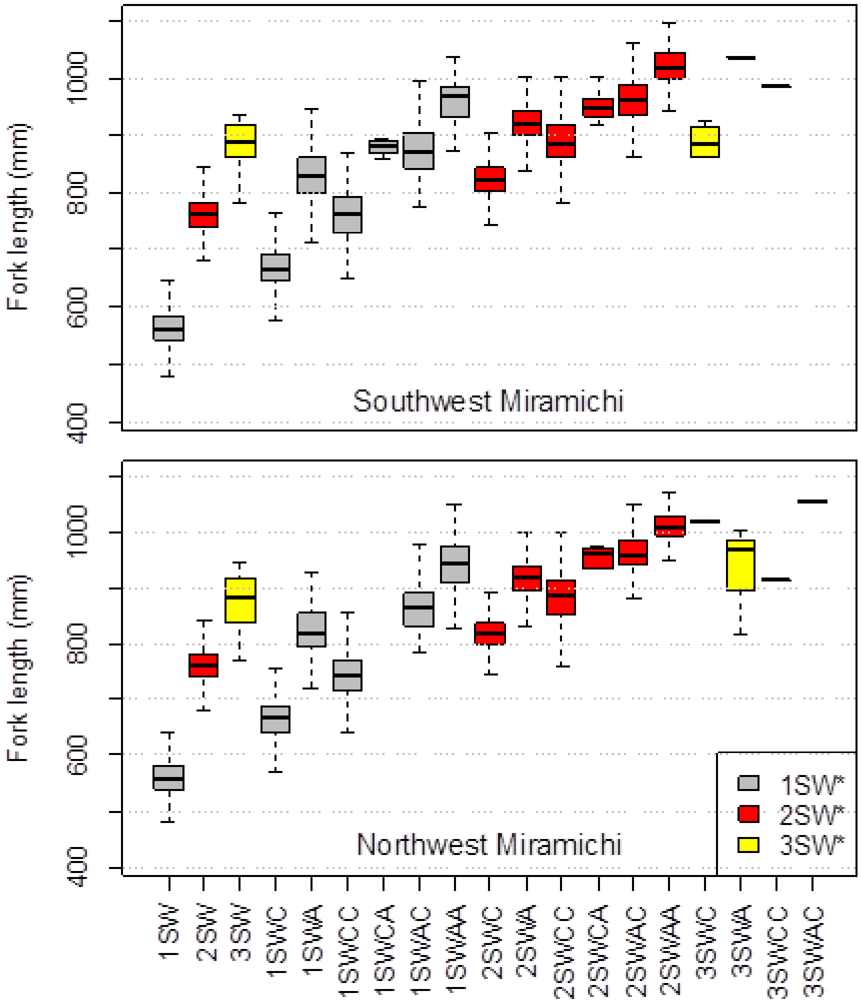

Figure 7. Boxplots of fork length (mm) distributions of wild Atlantic salmon from the Southwest Miramichi system (top panel) and the Northwest Miramichi system (bottom panel) by spawning history type from 1992 to 2013. The 1SW, 2SW and 3SW labels are maiden first time spawners. The other categories are repeat spawners according to sea age at first spawning followed by a sequence of repeat spawner types, with C representing consecutive spawning life history and A representing alternate spawning life history. Single letters (C, A) are categories of fish on a second spawning. CC, CA, AC, and AA represent categories of fish with three or more spawning events with the first two repeat spawning histories indicated by the letter codes.

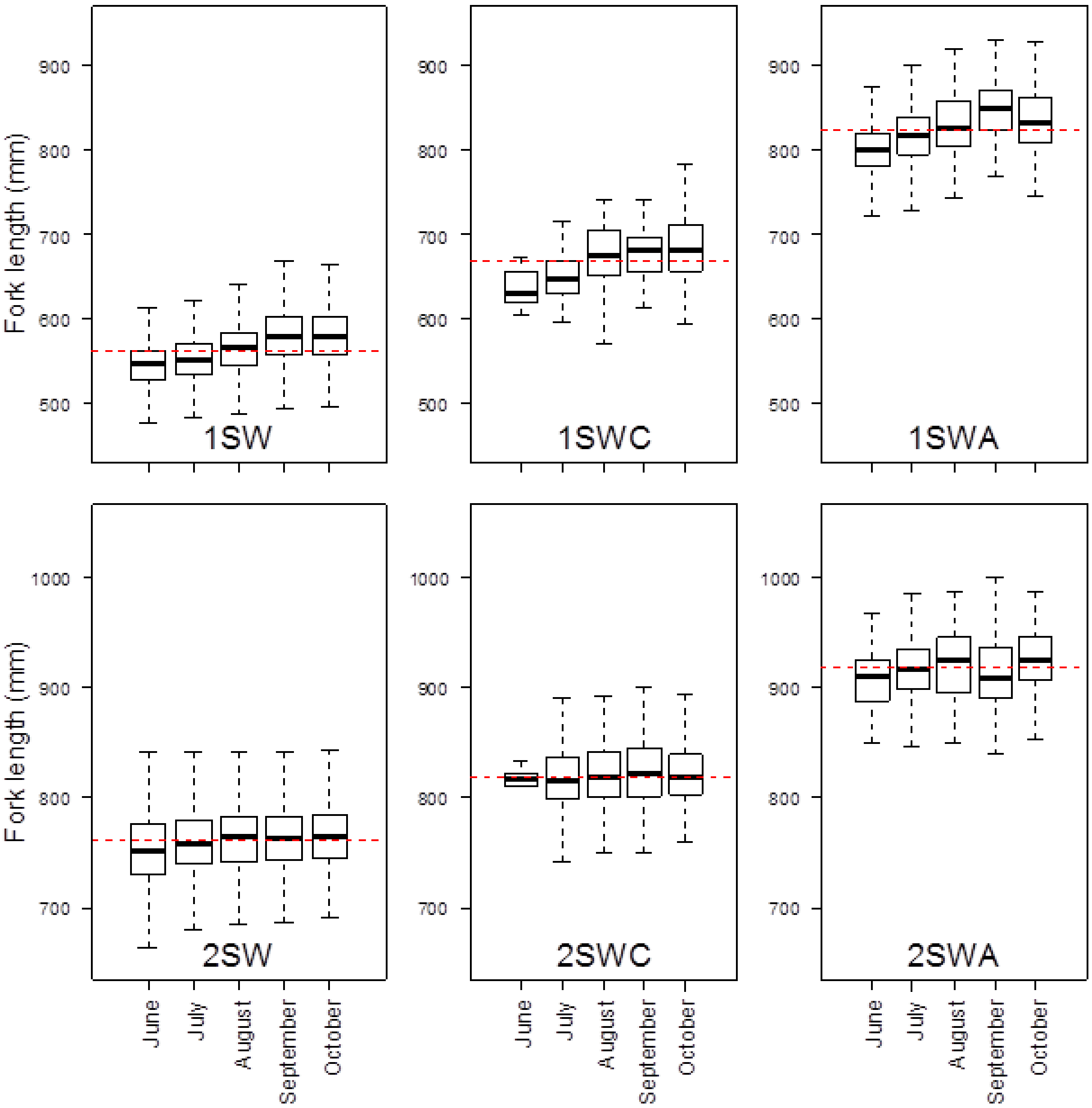

Figure 8. Fork length (mm) distributions by month (June to October) of 1SW (top row) and 2SW (bottom row) life histories as maiden return (left column), consecutive first time repeat spawners (middle column) and alternate first time repeat spawners (right column) based on catches at tidal trapnets in the Northwest Miramichi, 1992 to 2013. The horizontal dashed line in each plot is the mean over all years for the corresponding age group.

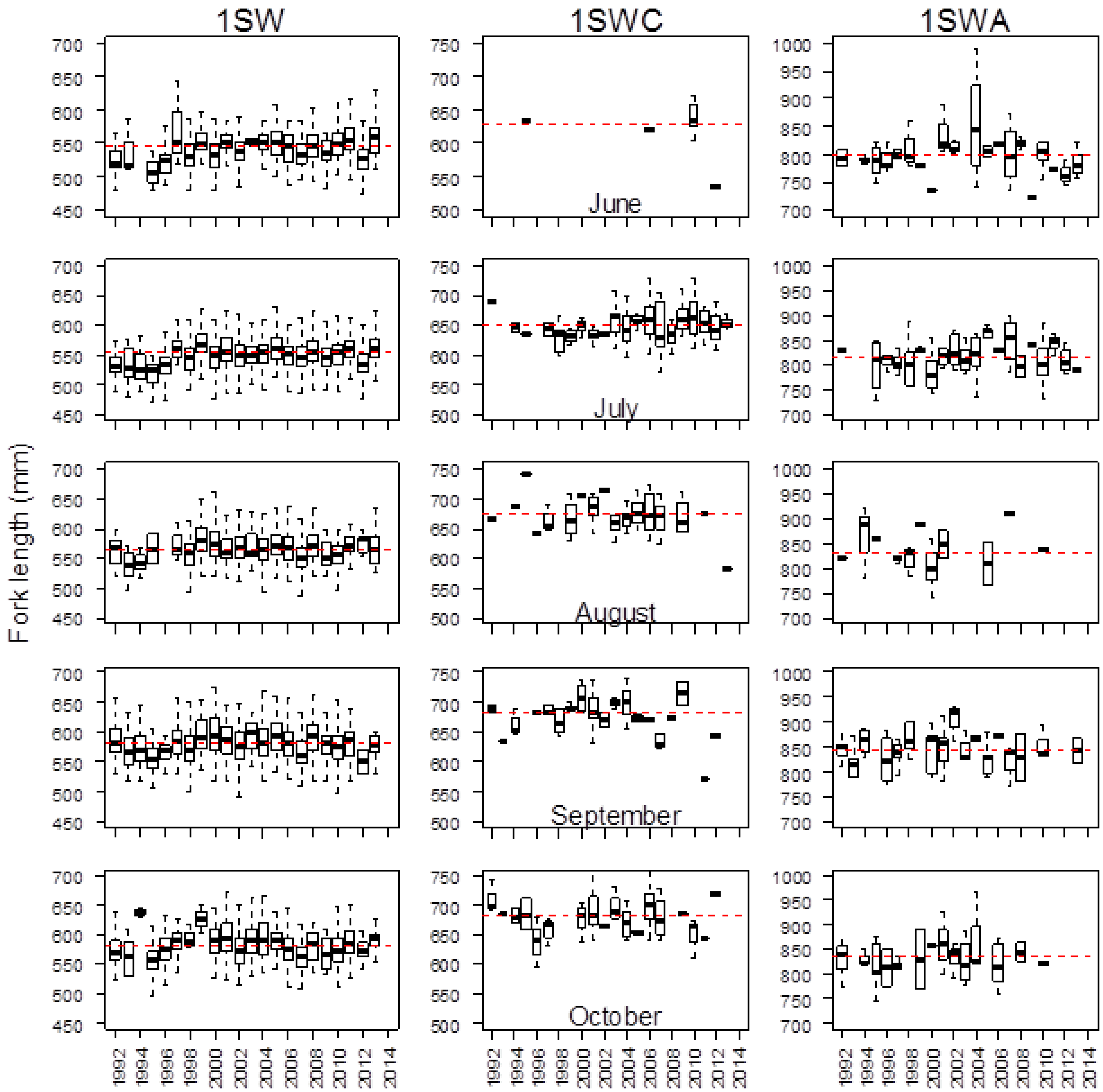

Figure 9. Fork length (mm) distributions by month (June to October) of 1SW maiden (left column), 1SW consecutive first time repeat spawner (middle column) and 1SW alternate first time repeat spawner (right column) based on catches at tidal trapnets in the Northwest Miramichi, 1992 to 2013. Horizontal dashed line in each plot is the mean over all years for the corresponding age group and month.

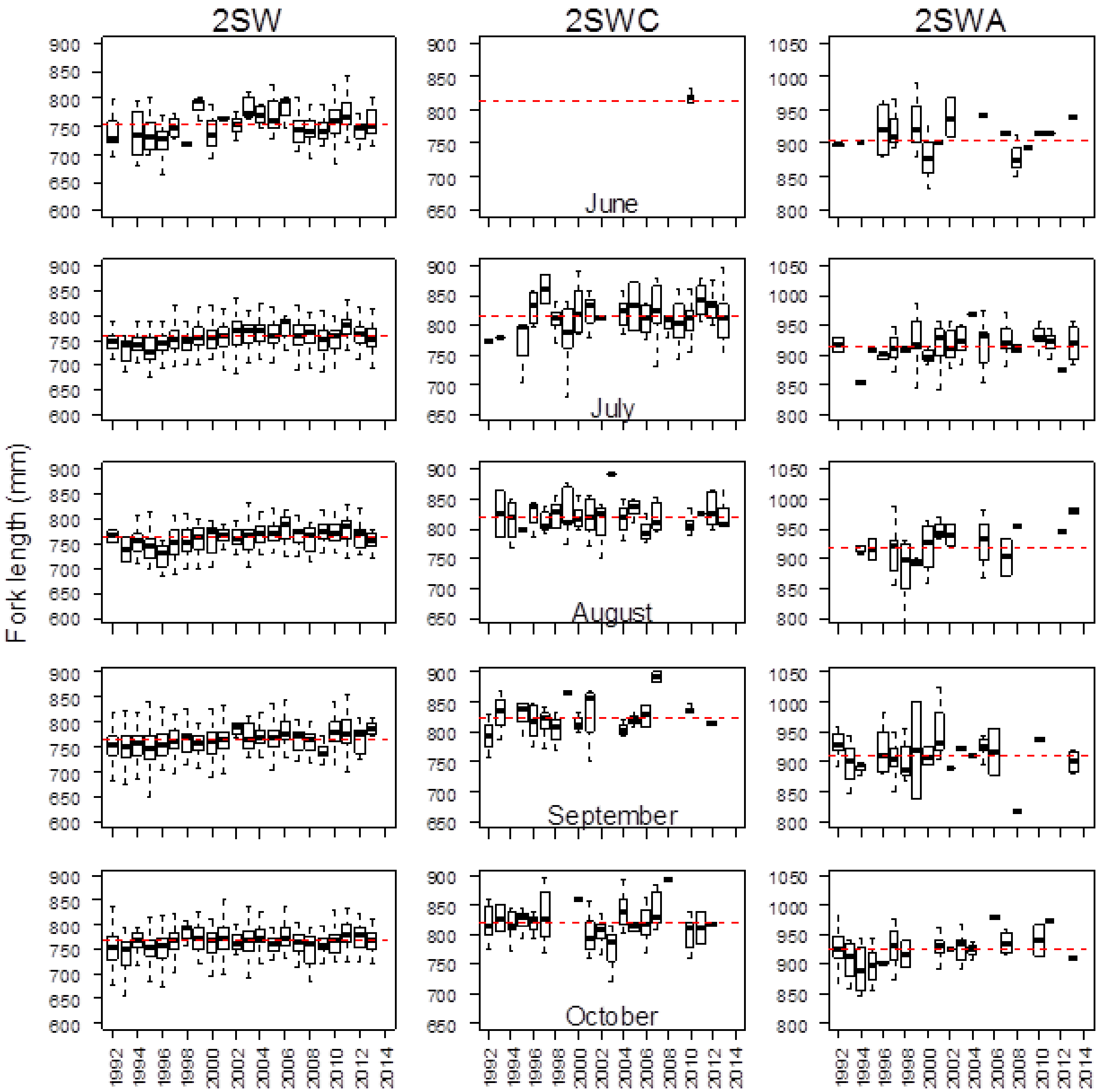

Figure 10. Fork length (mm) distributions by month (June to October) of 2SW maiden (left column), 2SW consecutive first time repeat spawner (middle column) and 2SW alternate first time repeat spawner (right column) based on catches at tidal trapnets in the Northwest Miramichi, 1992 to 2013. Horizontal dashed line in each plot is the mean over all years for the corresponding age group and month.

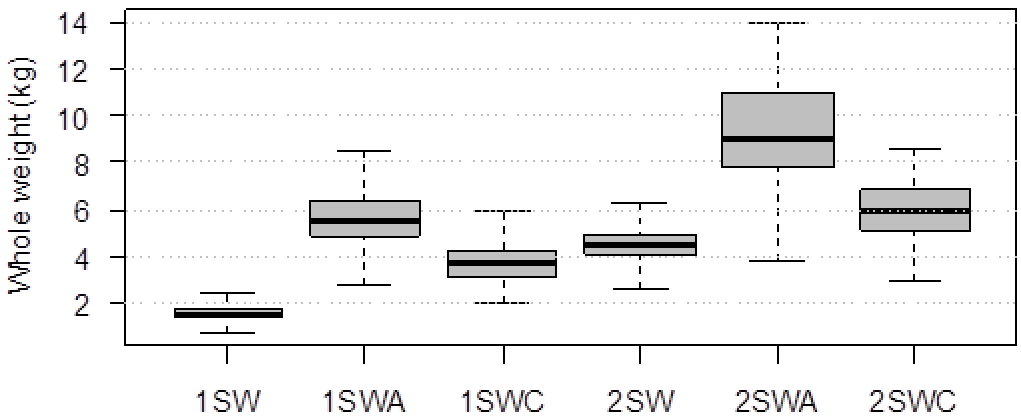

Figure 11. Whole weight (kg) of Atlantic salmon from the Miramichi River by spawning history type. Spawning history types are limited to maiden sea ages (1SW, 2SW), first time consecutive repeat spawners (1SWC, 2SWC), and first time alternate repeat spawners (1SWA, 2SWA), over all years (1971 to 2013) and months.

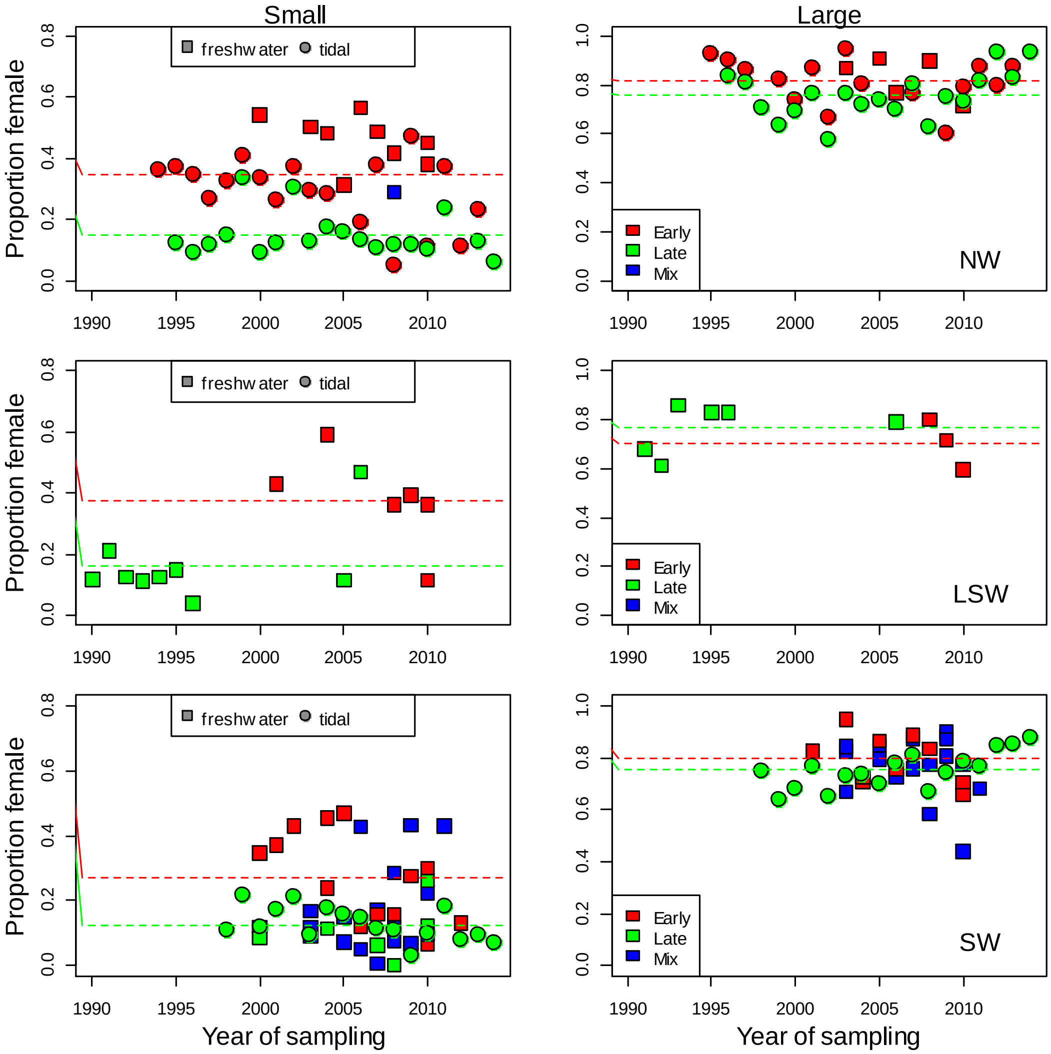

Figure 12. Proportion of females in small salmon (left column) and large salmon (right column) by season of return group (early, late, mixed) from samples in the Northwest Miramichi system (top row), the Little Southwest Miramichi River (middle row), and the Southwest Miramichi system (bottom row). Only samples for which sex was determined for 30 or more fish are shown. Square symbols represent samples obtained at freshwater locations (counting fences, seining) whereas circles are samples from estuary trapnets. Horizontal dashed lines and corresponding colours are the means of the samples in each panel. The early run trapnet samples from the Northwest are from the FSC trapnet catches of June and July. The late run trapnet samples from the Northwest are from the Cassilis trapnet for the months of September and October whereas for the Southwest Miramichi the samples are from the Millerton trapnet. Early run freshwater samples are from the Northwest Barrier for the Northwest Miramichi or from the Dungarvon Barrier and Rocky Brook for the Southwest Miramichi. Samples from the Little Southwest are either from the Catamaran Brook counting fence for the late run or broodstock sampling at Moose Landing and Smiths Forks considered to be from the early run component. Mixed run timing samples from the Northwest are from the Sevogle River. Mixed run timing samples from the Southwest Miramichi include Clearwater, Burnthill, Juniper, and Big Hole samples.

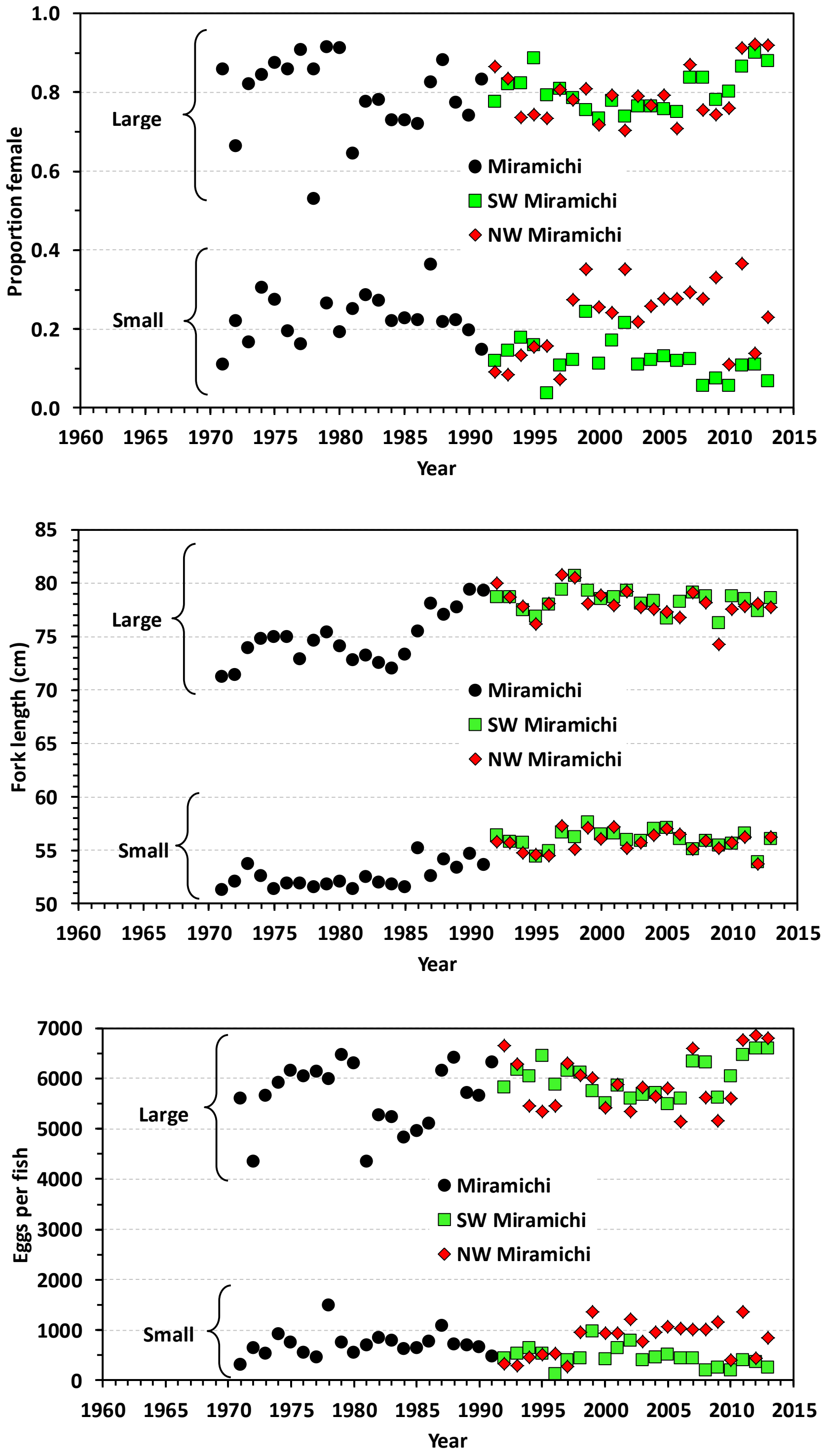

Figure 13. Plots of proportion of females (upper panel), mean length (cm, middle panel), and eggs per fish (lower panel) of wild Atlantic salmon by size group (small salmon, large salmon) from the Miramichi River overall (1971 to 1991) and in the Northwest Miramichi and the Southwest Miramichi branches, 1992 to 2013 (Douglas et al. 2015).

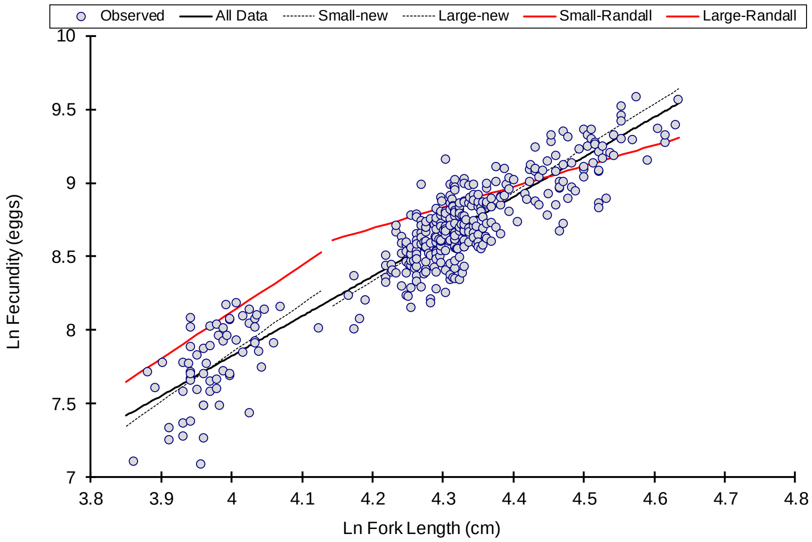

Figure 14. Length (cm) to fecundity relationships for Atlantic salmon from the Miramichi River. The data are from egg estimations in the hatchery collected during 1991 to 1995 (J. Hayward, DFO, unpublished data). The eggs per fish were estimated by volume displacement. The red lines are the relationships from Randall (1989) based on immature eggs from salmon sampled on entry to the river. The Randall (1989) relationships by size group are the ones used to estimate eggs in estimated returns and spawners to the Miramichi River. The parameters of the natural log of the regression of fecundity on length are: slope = 2.7075, intercept = -3.0065.

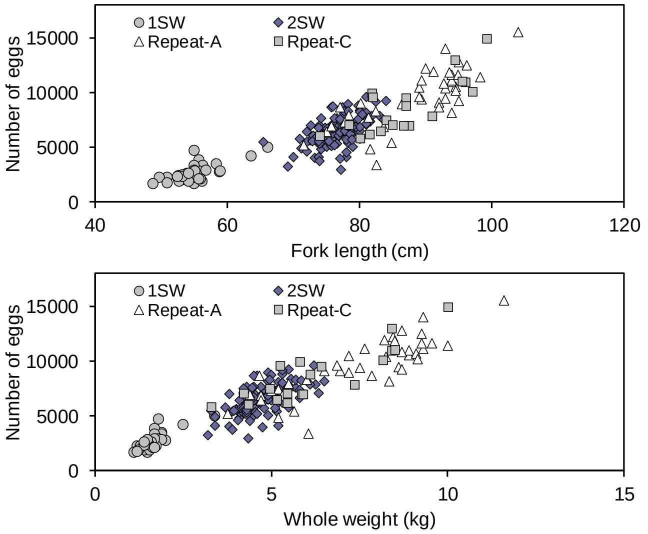

Figure 15. Fecundity (number of eggs) at fork length (cm) (top panel), and at whole weight (kg) (bottom panel) for Atlantic salmon from the Miramichi River. Data and analyses are from Reid and Chaput (2012). The parameters of the natural log of the regression of fecundity on length are very similar to those from J. Hayward (DFO, unpublished data) (slope = 2.7005, intercept = -2.9768).

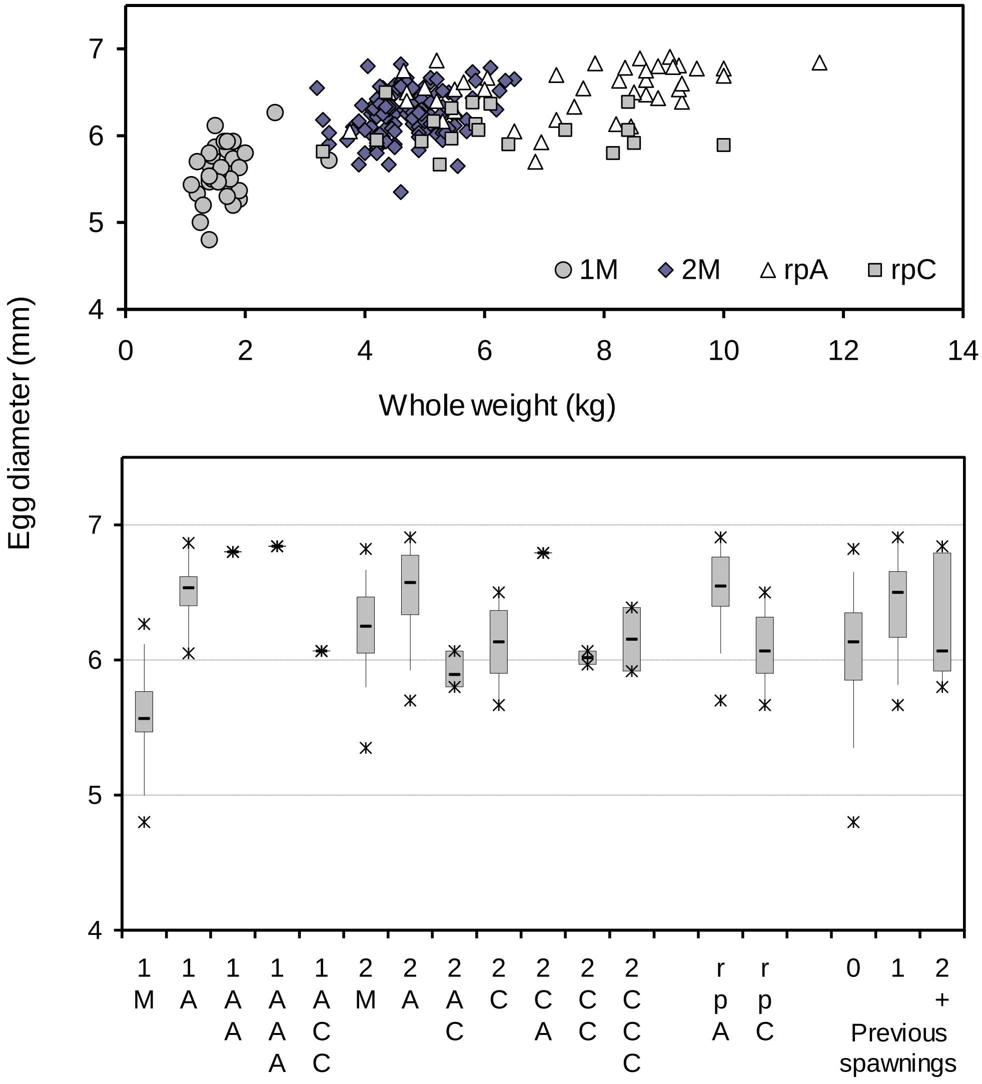

Figure 16. Estimated diameter (mm) of Atlantic salmon eggs from the Miramichi River relative to the spawning history of the female salmon (Reid and Chaput 2012). Maiden spawners are 1M and 2M. The number in each spawning history group refers to the maiden sea age. First time repeat consecutive spawners are 1C and 2C whereas first time repeat alternate spawners are 1A and 2A and so on. rpA represents all salmon that returned to second spawning as alternate, regardless of maiden sea age or the number of spawning events. Similarly, rpC represents all salmon that returned to a second spawning as consecutive regardless of maiden sea age or number of spawning events.

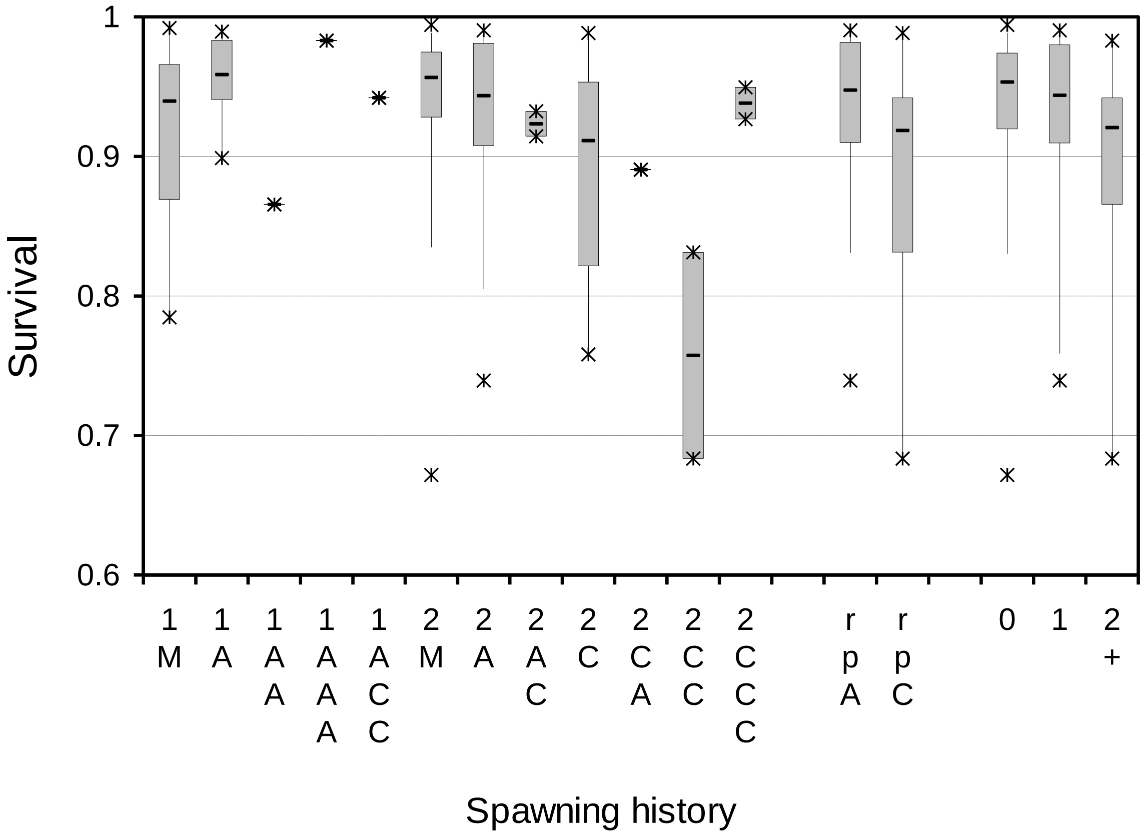

Figure 17. Estimated survival (proportion) to placement in incubation boxes in the hatchery of Atlantic salmon eggs from the Miramichi River relative to the spawning history of the female salmon (from Reid and Chaput 2012). Spawning history refers to those in Figure 15.

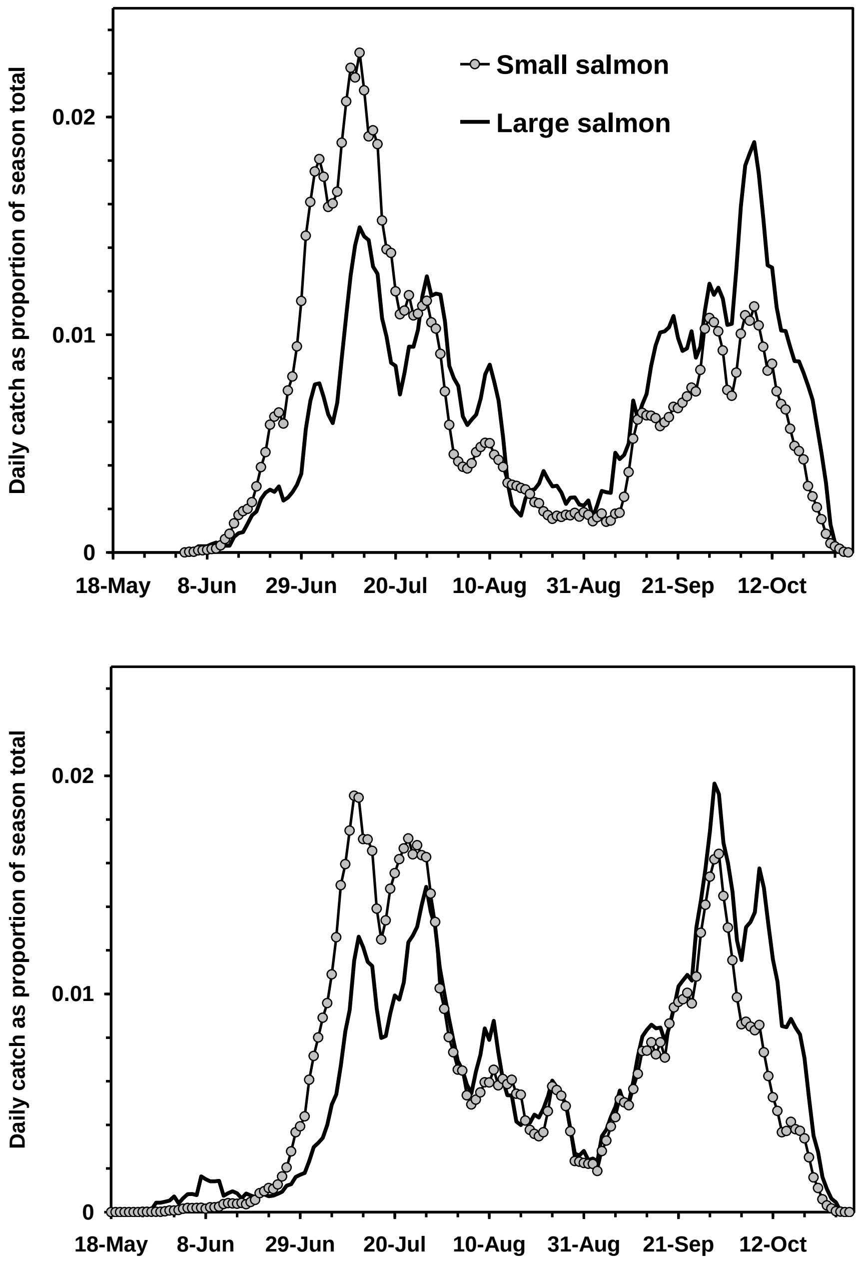

Figure 18. Timing of catches at estuarine trapnets in the Northwest Miramichi (upper) and the Southwest Miramichi (lower). Plots are mean proportions of the total annual catch for the years 1998 to 2004 (from Chaput et al. 2010).

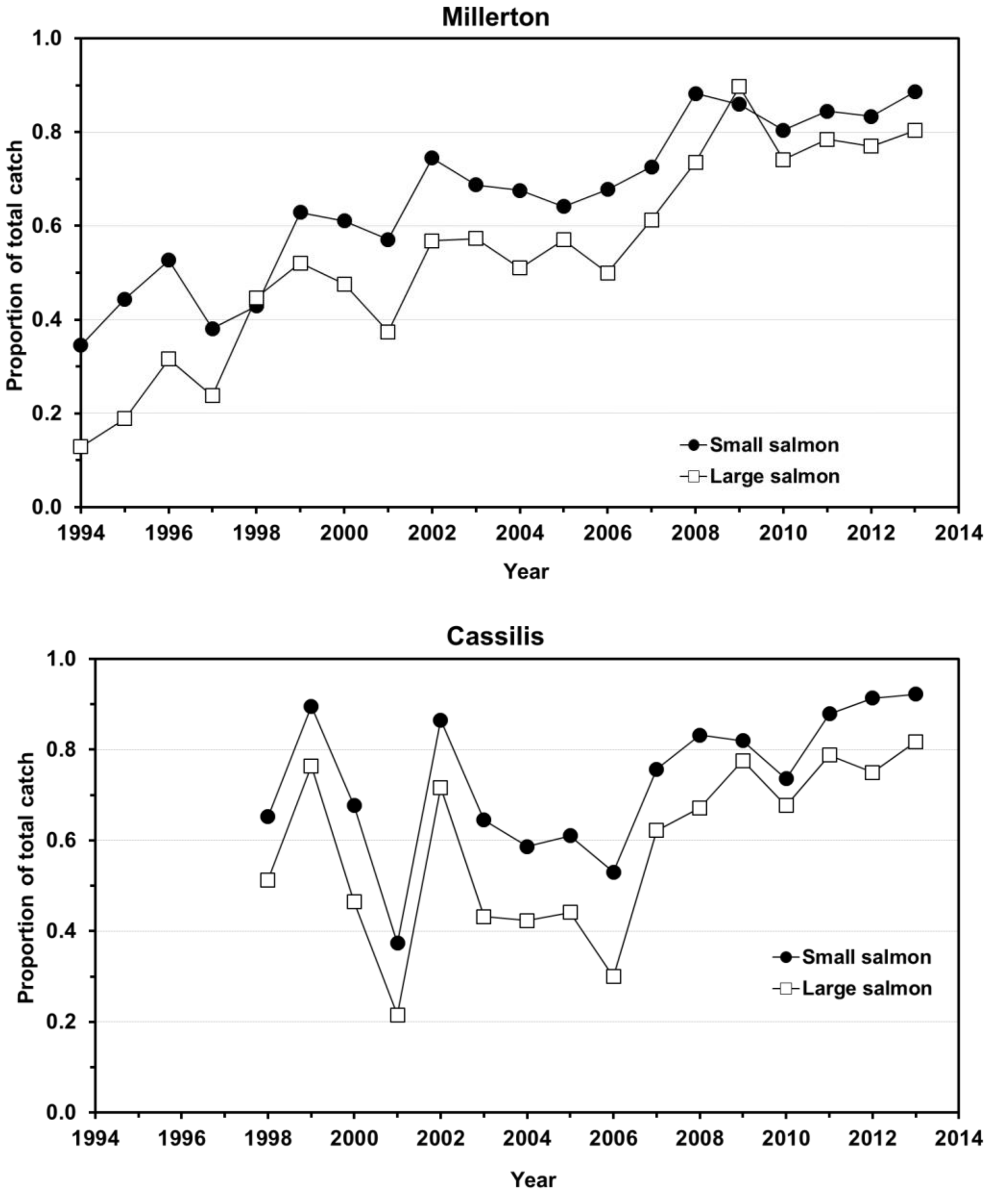

Figure 19. Proportion of annual catches of large salmon and small salmon which were captured by August 31 at DFO Index trapnets at Millerton on the Southwest Miramichi River (upper panel) and at Cassilis on the Northwest Miramichi River (lower panel) for 1998 to 2013. (from Douglas et al. 2015).

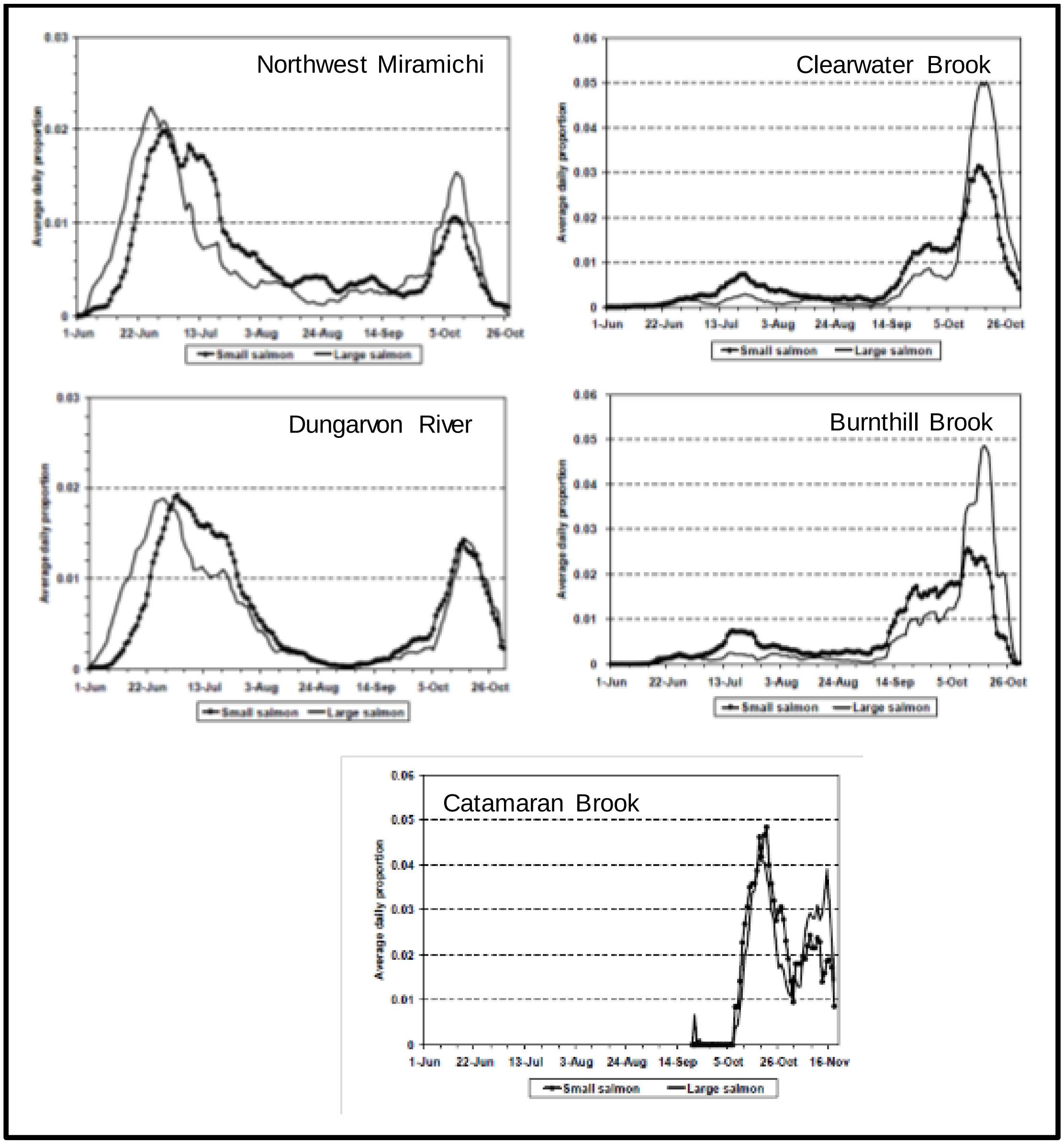

Figure 20. Run-timing of small salmon and large salmon at the Northwest Miramichi barrier (left upper panel), the Dungarvon River barrier (Southwest Miramichi; left lower panel), the Clearwater Brook counting fence (Southwest Miramichi; right upper panel) and the Burnthill Brook counting fence (Southwest Miramichi; right lower panel), and the Catamaran Brook counting fence (Little Southwest Miramichi; bottom row) (from El-Jabi et al. 2004). Daily proportions are the average of 1995-2003 for Northwest Miramichi, of 1995-2002 for Dungarvon River, of 1999-2003 for Clearwater Brook, of 2000 - 2003 for Burnthill Brook, and 1999-2002 for Catamaran Brook.

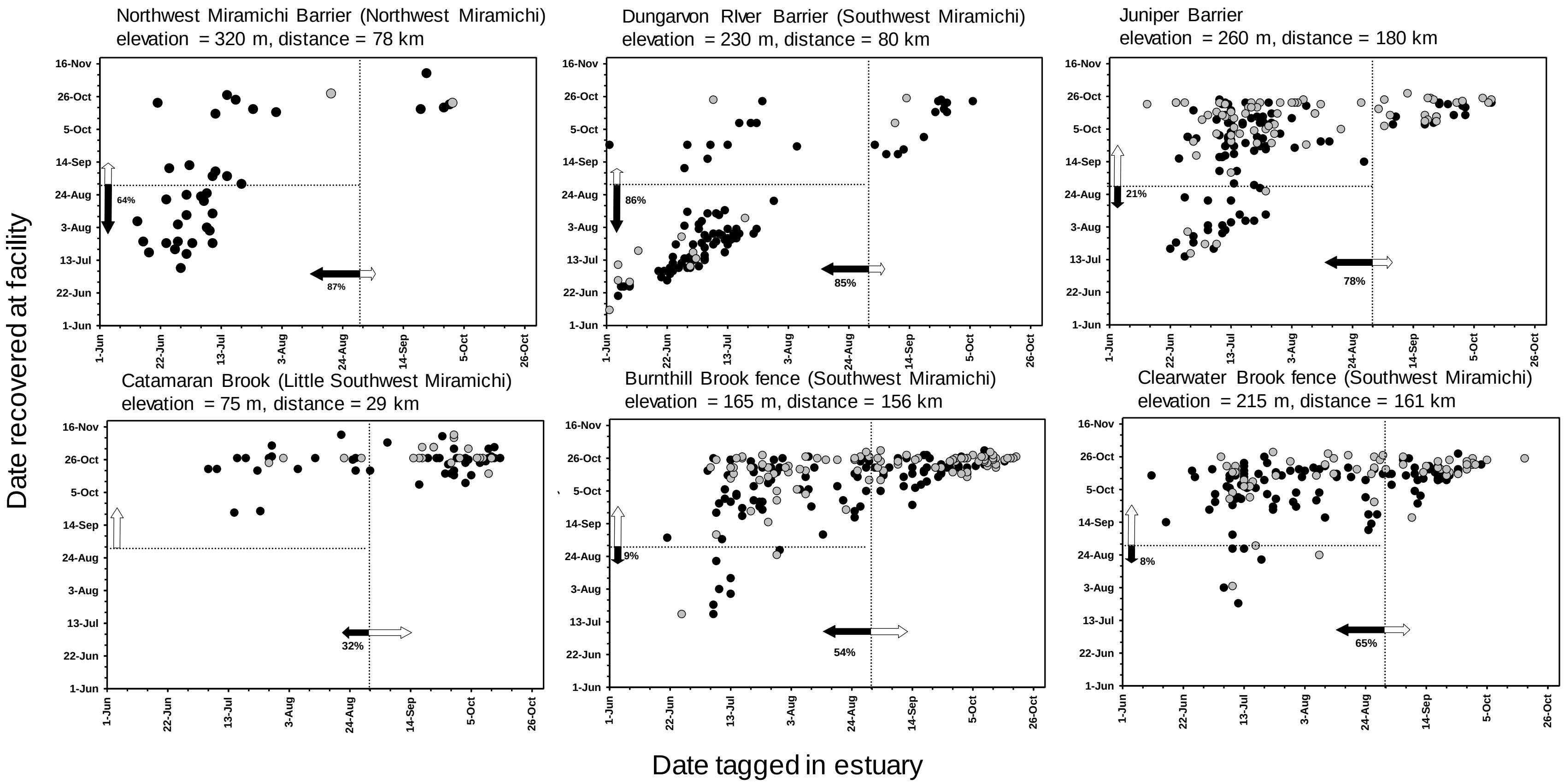

Figure 21. Date of recovery of salmon at upriver counting fences relative to the date when fish were originally tagged in the estuary for two facilities in the Northwest Miramichi system (left column) and four counting facilities in the Southwest Miramichi system (middle and right columns). Grey symbols are large salmon (fork length >= 63 cm) and black symbols are small salmon (fork length < 63 cm). The distance from head of tide and the elevation (m) of each facility are indicated above each panel. The vertical arrow with black shading in each panel and the corresponding value indicates the percentage of the tagged fish which were recovered at the facility in the early portion (prior to September 1) of the season. The horizontal arrow in black shading and the corresponding value in each panel is the percentage of all tags recovered at the facility which had been placed on salmon captured in the estuary prior to September 1 (i.e. early run salmon) (from El-Jabi et al. 2004).

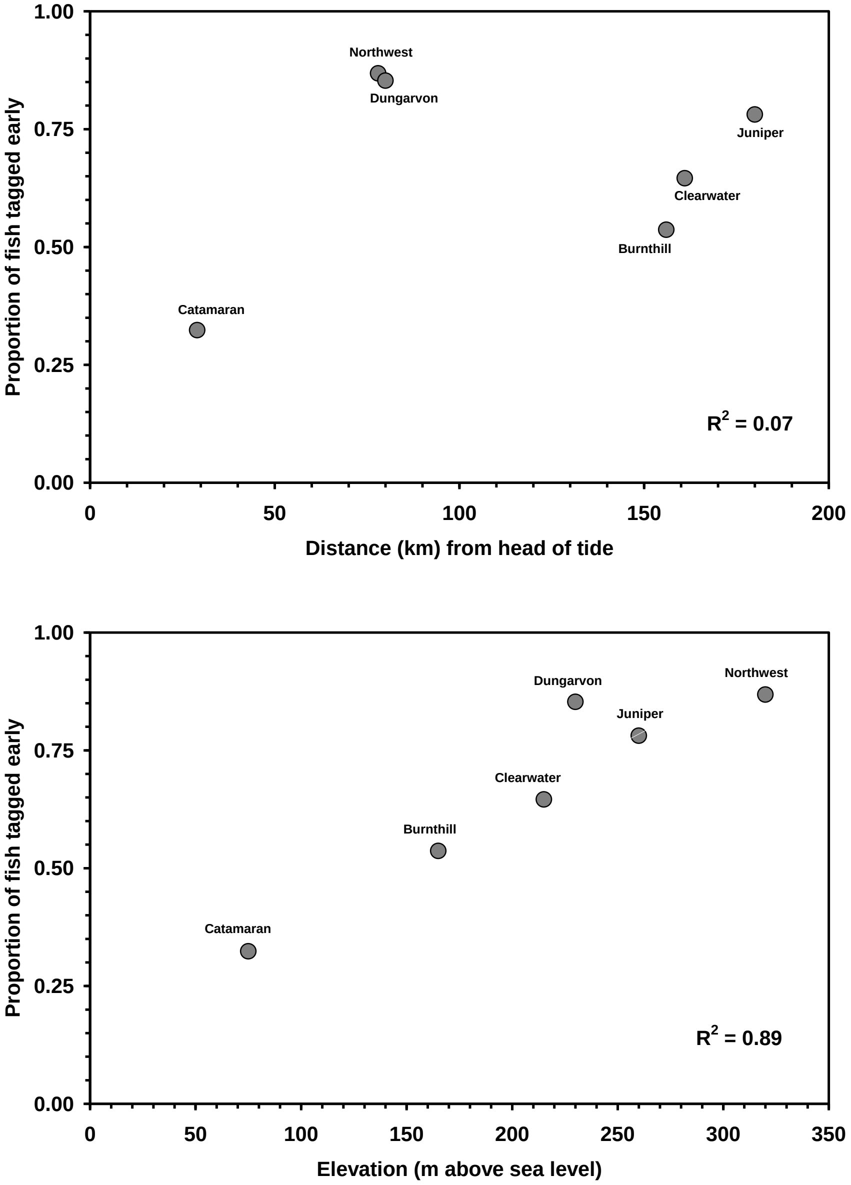

Figure 22. Association between the proportion of all tagged fish recovered at the counting facilities which had been tagged before September 1 relative to the distance (km) from the head of tide (upper panel) or the elevation (m above sea level) (lower panel) of the counting facility (from El-Jabi et al. 2004).

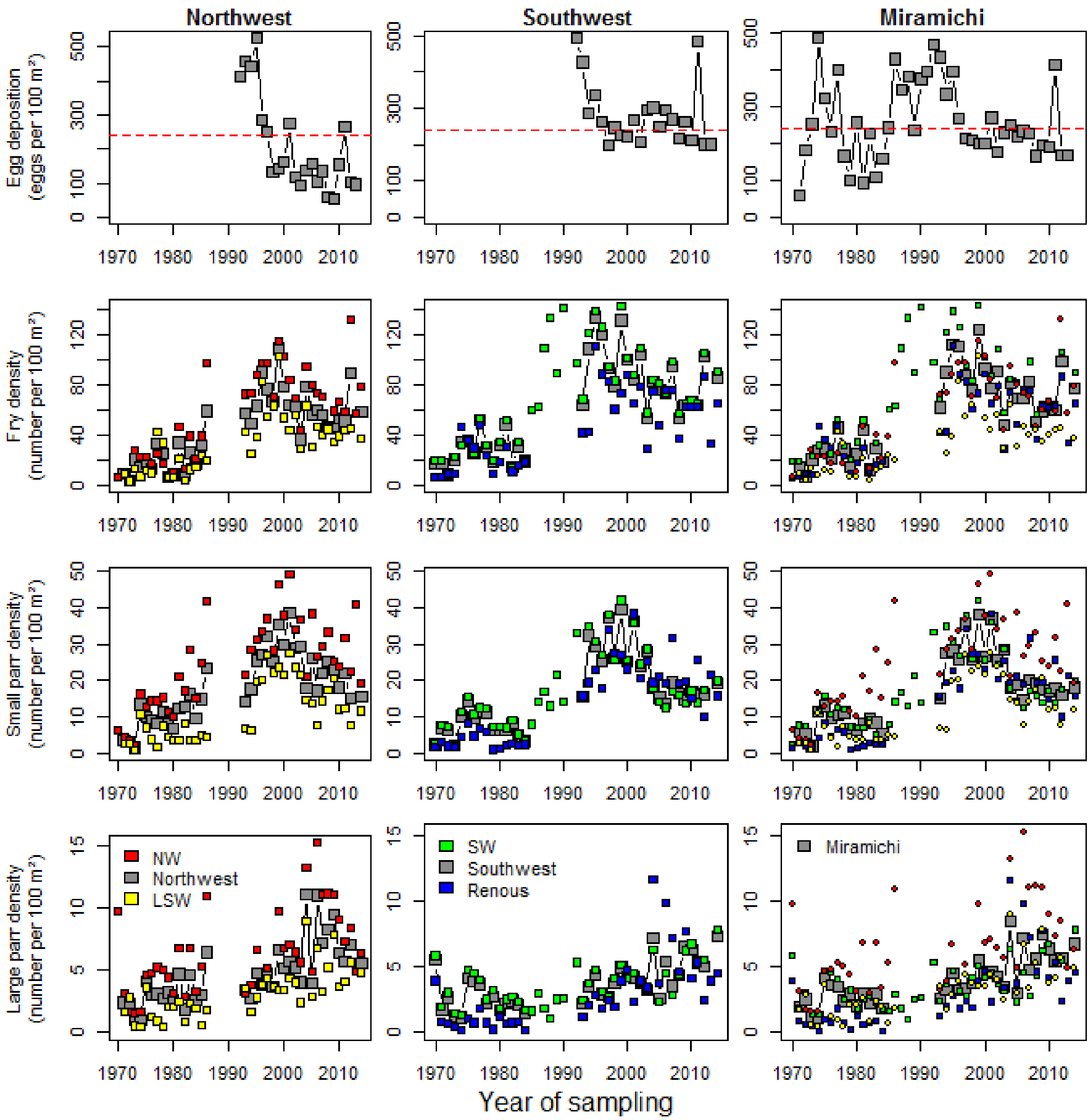

Figure 23. Annual estimates of egg depositions (eggs per 100 m²; upper row), age 0+ parr densities (fish per 100 m²; second row), small parr densities (third row), and large parr densities (bottom row), from the Northwest Miramichi system (left column) and the Southwest Miramichi system (middle column), and the Miramichi River (right column), 1971 to 2014. Average values are shown for years where four or more sites were sampled in each river. The horizontal dashed red line in the upper panel is the conservation egg deposition rate of 240 eggs per 100 m². The symbols in grey for juvenile indices are the weighted (by habitat area) average values of the river indices.

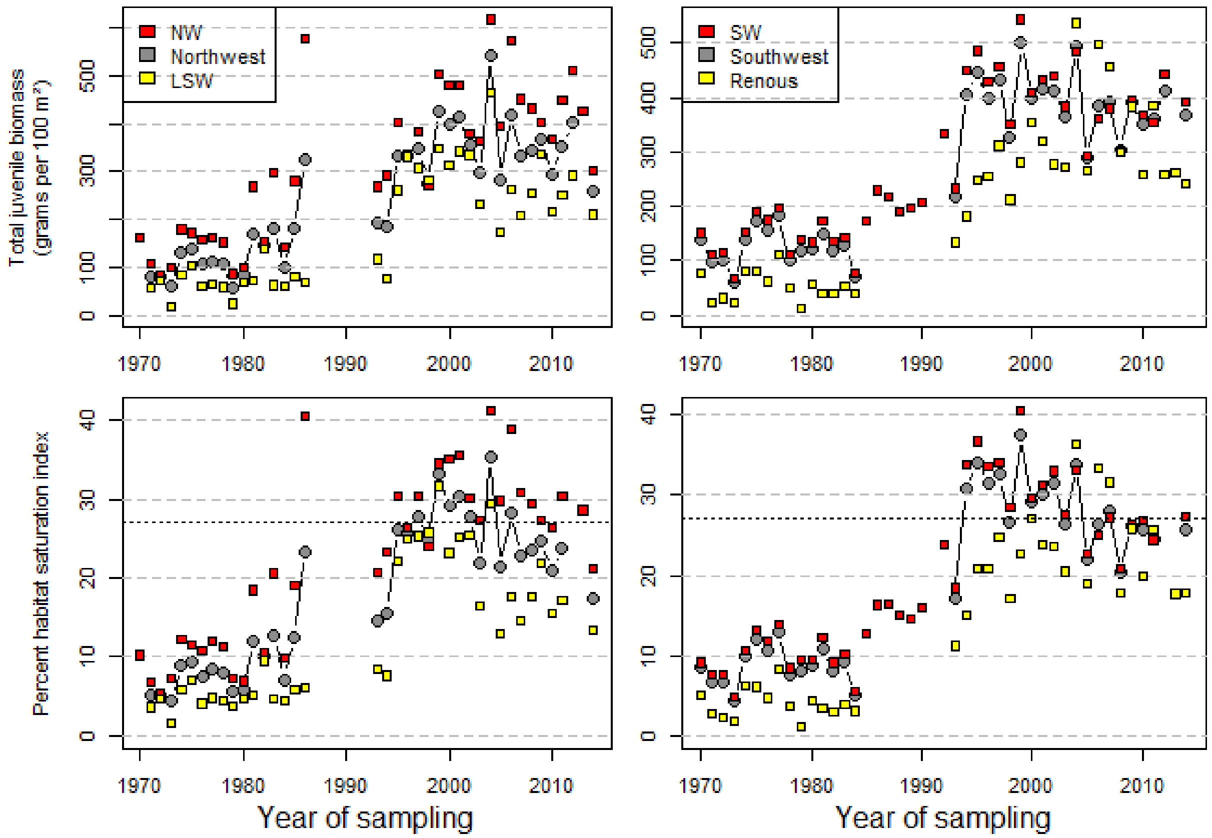

Figure 24. Annual estimates of total salmon juvenile biomass (grams per 100 m²; upper row) and Percent Habitat Saturation Index (PHS, Grant and Kramer 1990) (bottom row) for the Northwest Miramichi system (left column) and the Southwest Miramichi system (right column) 1971 to 2014. Average values are shown for years where four or more sites were sampled in each river. The dotted black line in the panels of the lower row refers to the PHS reference value of 28. The symbols in grey for juvenile indices are the weighted (by habitat area) average values of the river indices.

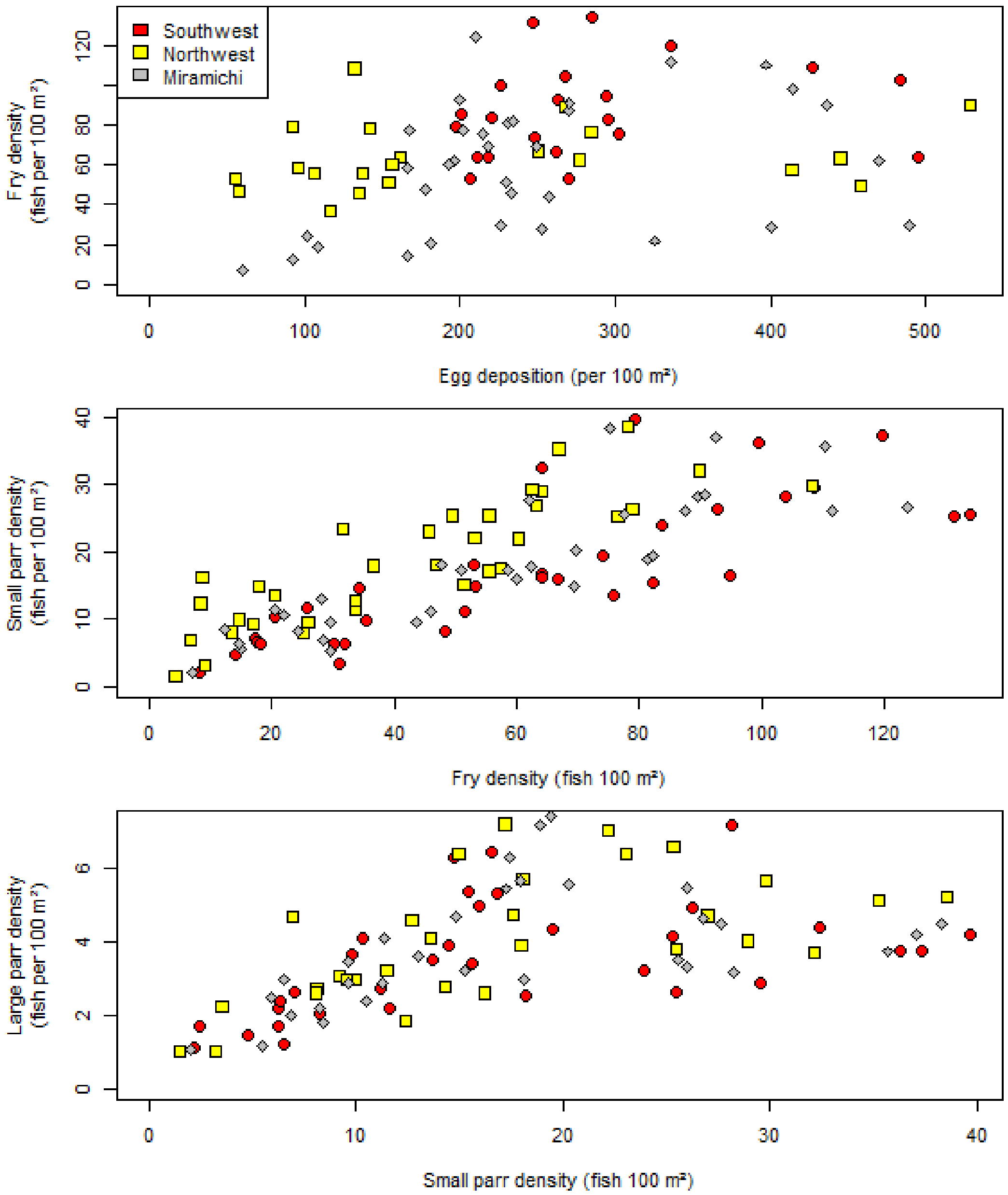

Figure 25. Stock and recruitment associations between estimated egg deposition density in year i and subsequent indices of fry densities in year i+1 (upper row), fry densities in year i and subsequent indices of small parr in year i+1 (middle row), and small parr densities in year i and subsequent large parr densities in year i+1 (bottom row) for the Northwest Miramichi system, the Southwest Miramichi system, and the Miramichi River overall.

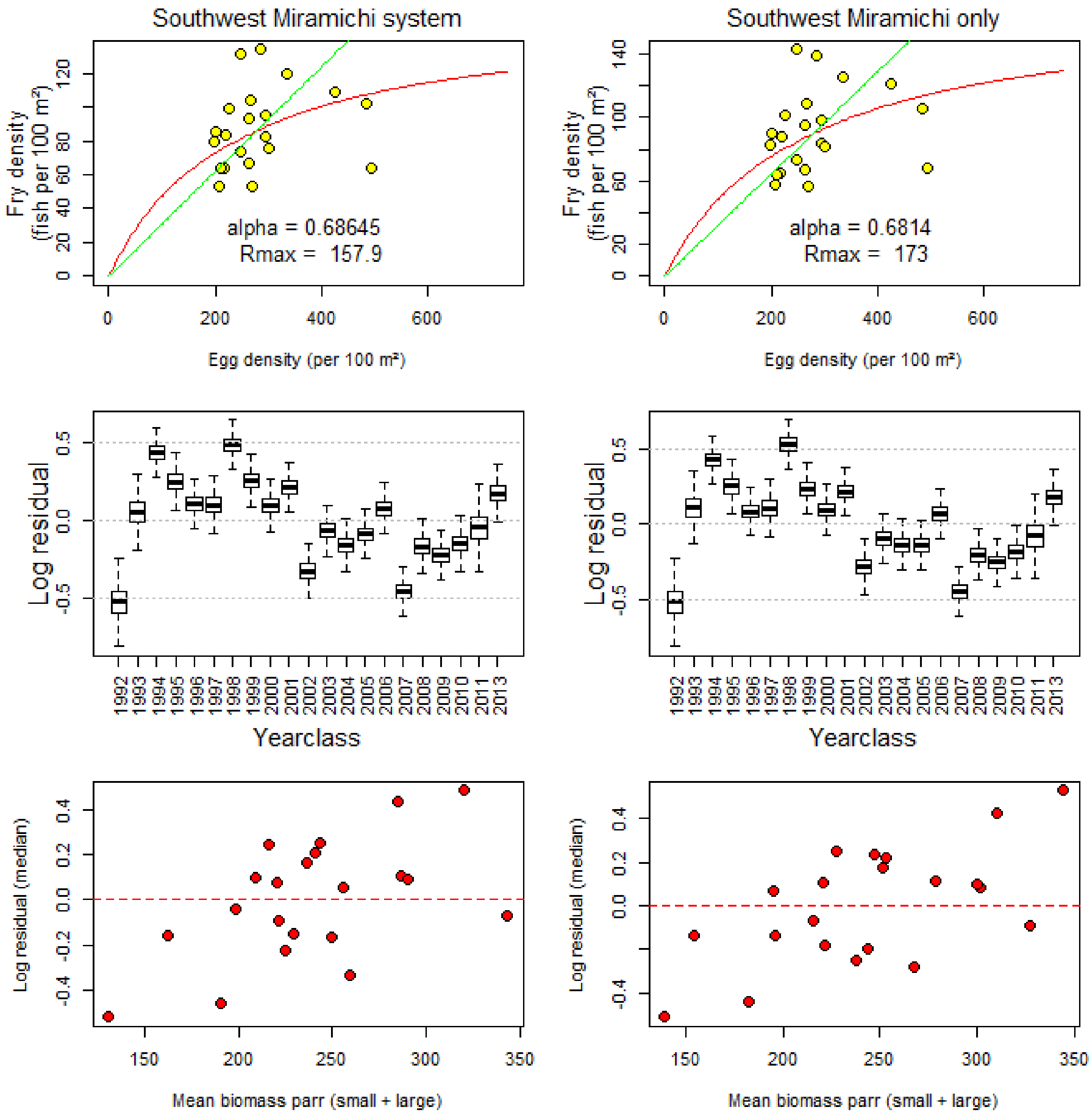

Figure 26. Stock and recruitment analysis (upper row) of egg deposition in year i to index of fry abundance in year i+1 for the Southwest Miramichi system (left column) and the Southwest Miramichi River only (right column). The red line in the upper row panels is the median Beverton-Holt stock and recruitment curve whereas the green line is the proportional linear fit (with multiplicative error) of the same data. The middle row is the residuals from the posterior distributions of the predicted abundance of fry by yearclass. The bottom row panels are the associations between the log residual (median) and the mean biomass of parr (small and large combined) as a potential explanatory variable of residual error in fry abundance after adjusting for egg deposition by yearclass. For the Southwest Miramichi system, the Beverton-Holt SR fit was statistically more likely than the linear proportion fit (DIC: 191.6, 202.4, respectively). For the Southwest Miramichi River, the Beverton-Holt SR fit was statistically more likely than the linear proportion fit (DIC: 193.9, 203.8, respectively).

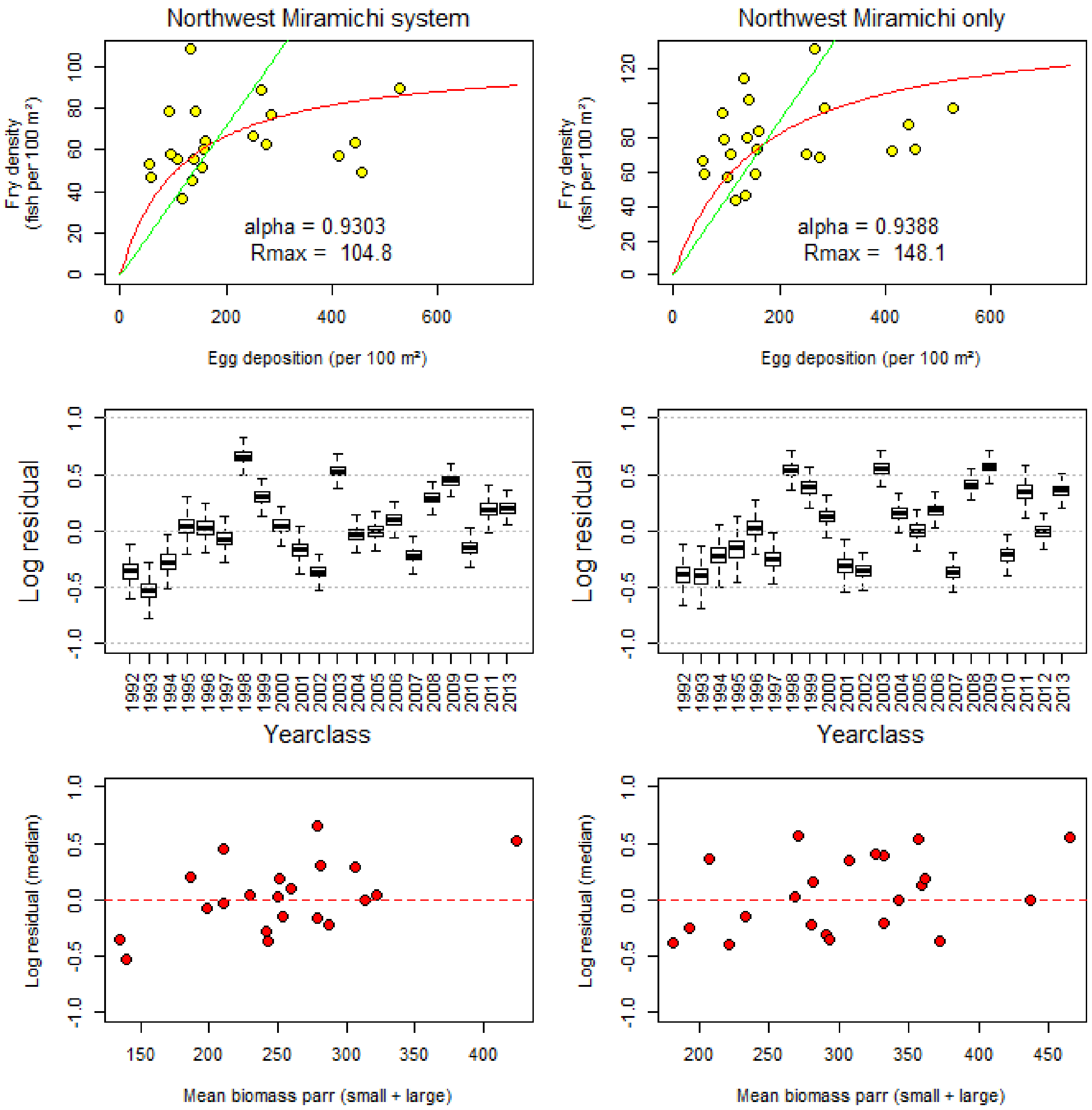

Figure 27. Stock and recruitment analysis (upper row) of egg deposition in year i to index of fry abundance in year i+1 for the Northwest Miramichi system (left column) and the Northwest Miramichi River only (right column). The red line in the upper row panels is the median Beverton-Holt stock and recruitment curve whereas the green line is the proportional linear fit (with multiplicative error) of the same data. The middle row is the residuals from the posterior distributions of the predicted abundance of fry by yearclass. The bottom row panels are the associations between the log residual (median) and the mean biomass of parr (small and large combined) as a potential explanatory variable of residual error in fry abundance after adjusting for egg deposition by yearclass. For the Northwest Miramichi system, the Beverton-Holt SR fit was statistically more likely than the linear proportion fit (DIC: 187.0, 216.5, respectively). For the Northwest Miramichi River, the Beverton-Holt SR fit was statistically more likely than the linear proportion fit (DIC: 208.7, 234.3, respectively).

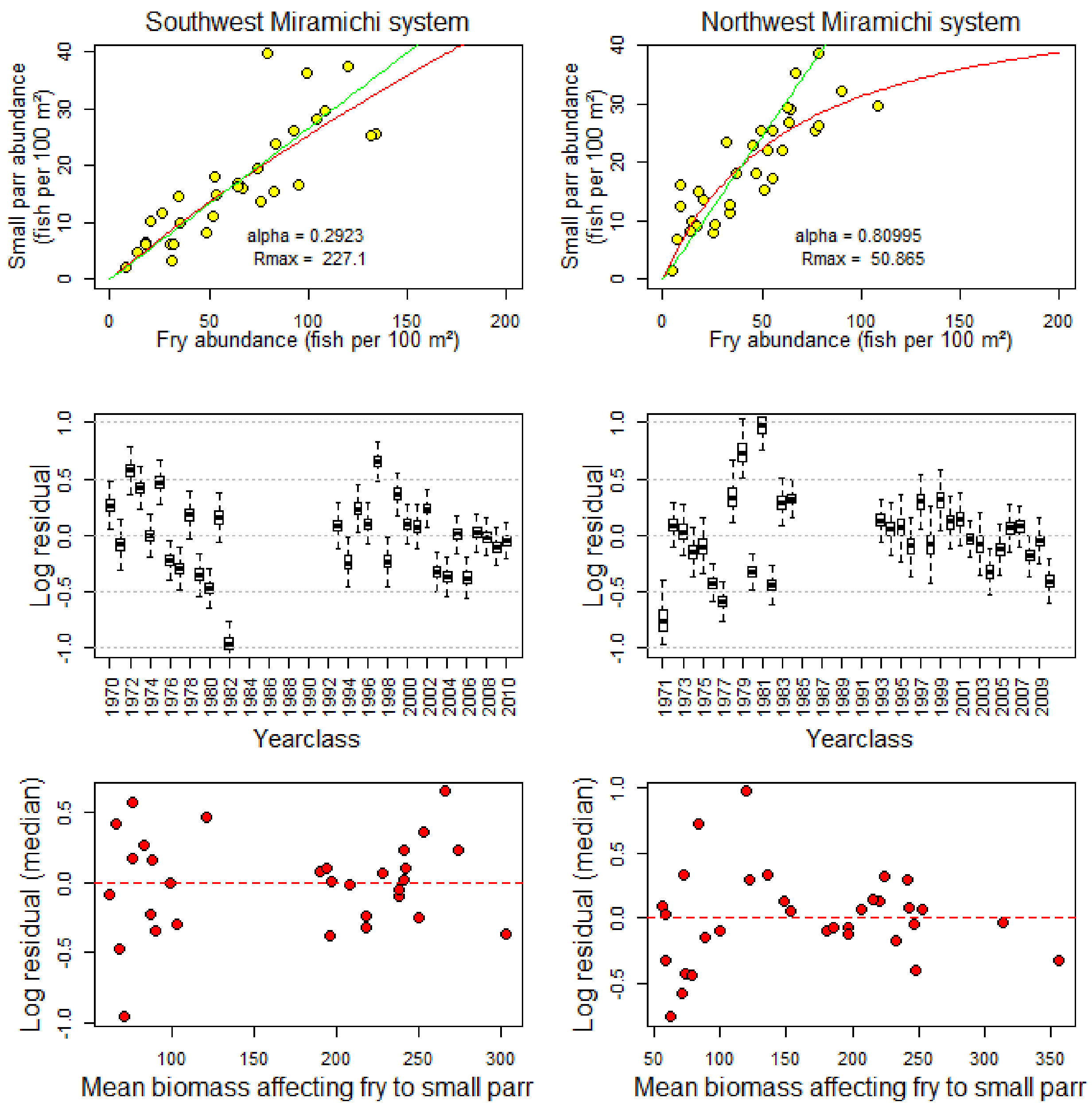

Figure 28. Stock and recruitment analysis (upper row) of fry abundance index in year i to index of small parr abundance in year i+1 for the Southwest Miramichi system (left column) and the Northwest Miramichi system (right column). The red line in the upper row panels is the median Beverton-Holt stock and recruitment curve whereas the green line is the proportional linear fit (with multiplicative error) of the same data. The middle row is the residuals from the posterior distributions of the predicted abundance of small parr by yearclass. The bottom row panels are the associations between the log residual (median) and the mean biomass of juveniles which would have interacted with the fry to small parr transition (average of the combined biomass of small parr and large parr in the fry year and the fry biomass and large parr biomass of the small parr sampling year) as a potential explanatory variable of residual error in small parr abundance after adjusting for fry abundance of the yearclass. For the Southwest Miramichi system, the linear proportional fit and the Beverton-Holt SR fits are indistinguishable (DIC: 187.2, 186, respectively). For the Northwest Miramichi system, the Beverton-Holt SR fit was statistically more likely than the linear proportion fit (DIC: 207.1, 222.2, respectively).

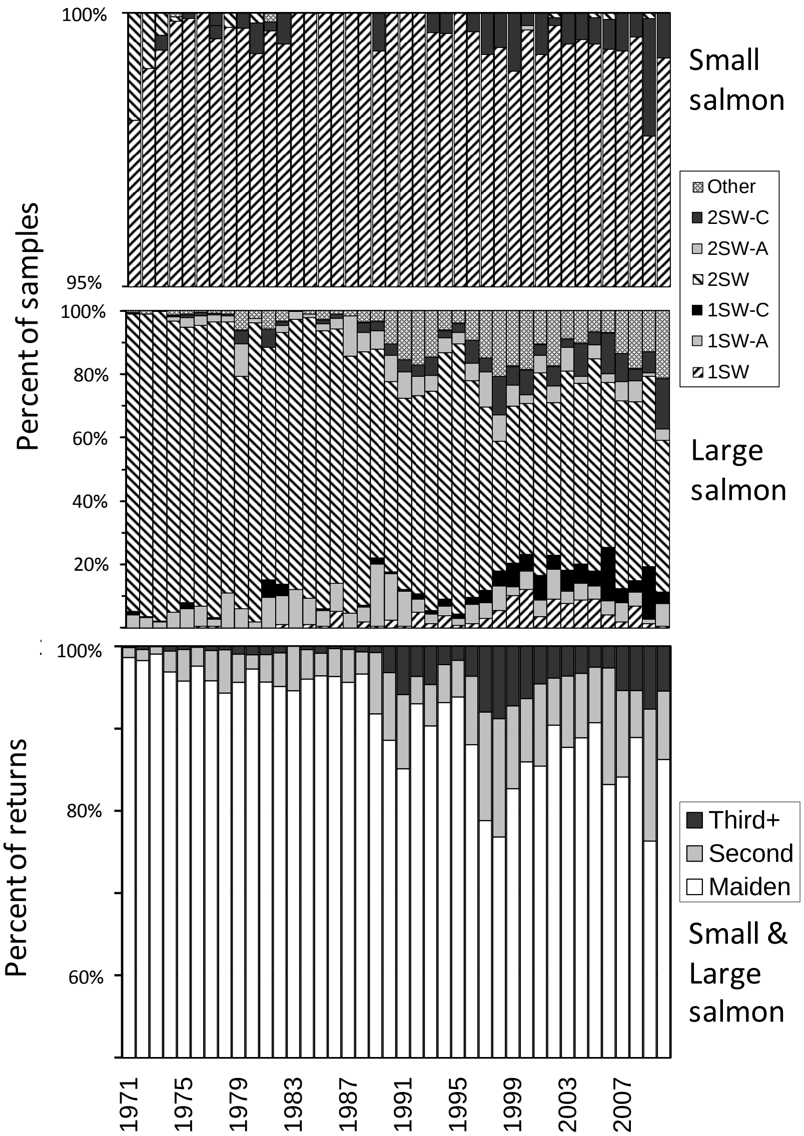

Figure 29. Percentage as maiden (1SW, 2SW), second time spawners (1SW-C, 1SW-A, 2SW-C, 2SW-A) and other spawning histories from small salmon (upper panel) and large salmon (middle panel) interpreted scale samples and percentage of estimated returns (size groups combined) which were maiden, second time spawners, and third plus time spawners (lower panel) in the Miramichi River, 1971 to 2010.

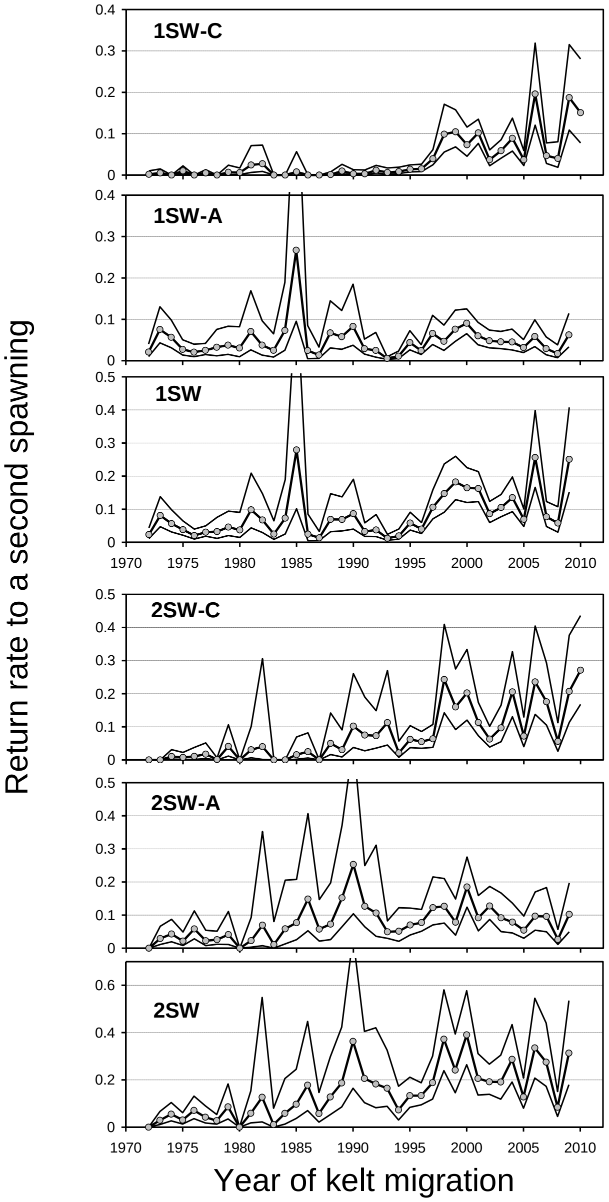

Figure 30. Return rates (median; 2.5 to 97.5 percentile range) to a second spawning as consecutives (1SW-C; 2SW-C), as alternates (1SW-A; 2SW-A) and combined (1SW; 2SW) by year of kelt migration (from Chaput and Benoit 2012). Kelt refers to the post-overwinter condition of salmon as they return to sea in the spring.

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et des océans

Canadian Science Advisory Secretariat (CSAS)

Research Document 2022/030

Gulf Region

Multi-species Considerations for Defining Fisheries Reference Points for Striped Bass (*Morone saxatilis*) from the Southern Gulf of St. Lawrence

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May 2022

Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

Published by:

Fisheries and Oceans Canada
Canadian Science Advisory Secretariat
200 Kent Street
Ottawa ON K1A 0E6

http://www.dfo-mpo.gc.ca/csas-sccs/
csas-sccs@dfo-mpo.gc.ca

ISSN 1919-5044
ISBN 978-0-660-43302-8 Cat. No. Fs70-5/2022-030E-PDF

Correct citation for this publication:

Chaput, G. 2022. Multi-species Considerations for Defining Fisheries Reference Points for Striped Bass (*Morone saxatilis*) from the Southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2022/030. vii + 50 p.

Aussi disponible en français :

*Chaput, G. 2022. Considérations plurispécifiques pour l’établissement de points de référence en matière de pêche du bar rayé (*Morone saxatilis*) dans le sud du golfe du Saint Laurent. Secr. can. des avis sci. du MPO. Doc. de rech. 2022/030. viii + 53 p.*

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ABSTRACT

The reference points and management strategies based on single species management approaches for the purpose of optimizing utility functions specific to Striped Bass do not take into account the interactions among multiple species of equally important conservation and fisheries values. Modifying the single species Striped Bass reference points to account for interactions with other species requires evidence of interactions between Striped Bass and the other species. The most direct interaction considered is the predation by Striped Bass on other species with a focus on Atlantic Salmon. Several data sources are presented and examined to inform on these associations, including recorded commercial landings of important diadromous species, indices based on catches at estuarine index trapnets operated by DFO Science, and studies directly related to predation and survival rates of Atlantic Salmon smolts during the seaward migration phase. A cohort model relating population specific indices of juvenile salmon abundance to adult returns is also presented. There is conflicting evidence of reductions in a few anadromous fish species abundance indicators associated with increased abundance of Striped Bass in the southern Gulf. There is direct evidence of predation by Striped Bass on gaspereau, Rainbow Smelt and Atlantic Salmon smolts. Several studies using acoustic tag technologies have inferred predation events and changes in estimated survival rates in the early phase of migration of Atlantic Salmon smolts through Miramichi Bay that point to Striped Bass predation as a likely driver of these variations. It is not clear from the available studies that reducing Striped Bass spawner abundances to a level of the mid 2000s, i.e., 100 thousand spawners or less, would improve the acoustic tagged smolt survival estimates or the population level relative survival rates derived from the cohort model, nor landings of gaspereau and Rainbow Smelt in the commercial fisheries.

INTRODUCTION

As the Striped Bass population of the southern Gulf increased in abundance, concerns have been expressed by Atlantic Salmon fishery advocates as well as some gaspereau (river herring, two species) and Rainbow Smelt commercial fishery interests that the rebuilding of Striped Bass stock in the southern Gulf has contributed to declines in abundances of Atlantic Salmon and other diadromous species. The impact of Striped Bass on these other species is indicated to be associated with high levels of predation by Striped Bass. One can readily find publications that consider correlations between abundance indices of Striped Bass and indices of species that are documented prey of bass to conclude that Striped Bass when abundant are impacting survival and productivity of these other species (see Grout 2006 for example). Similar concerns were expressed about the impact of the recovered Atlantic Coast Striped Bass on its prey-base and NEFSC (2019) summarize a number of analyses that examined the potential for Striped Bass to deplete prey populations along the Atlantic Coast. To date, no multi-species reference points or management plans have been proposed for the US situation.

DFO (2019) developed a policy to support rebuilding plans under the precautionary approach framework for stocks that are in the critical zone. DFO (2019) states that in cases where rebuilding of a stock has the potential to negatively impact the status of another, as in the case of rebuilding a predator species that could result in a decline of a prey species, rebuilding objectives need to be carefully developed through a balanced approach to ensure neither is depleted to a point of serious harm. Most importantly DFO (2019) acknowledge that it is not possible to simultaneously achieve yields corresponding to Maximum Sustainable Yield (MSY) predicted from single-species assessments for a system of multiple, interacting species and rebuilding efforts should be approached within an ecosystem context to the extent possible.

The reference points and management strategies developed for Striped Bass (DFO 2021; Chaput and Douglas 2022) are based on single species management approaches for the purpose of optimizing utility functions specific to Striped Bass. Modifying the single species Striped Bass reference points to account for interactions with other species requires evidence of interactions between Striped Bass and the other species. This working paper assembles the available information on status of diadromous species that are potential prey of Striped Bass to determine if there is any such evidence that would support modifying the single species reference values developed previously (DFO 2021).

BACKGROUND ON THE ECOSYSTEM OCCUPIED BY STRIPED BASS

The southern Gulf of St. Lawrence contains a diversity of diadromous fish species, many of which are at the northern limit of their species distribution. The Miramichi River, within the southern Gulf of St. Lawrence contains the largest abundance of these diadromous species including (Chaput 1995):

- The only confirmed spawning population of American Shad (*Alosa sapidissima*) in the southern Gulf (Chaput and Bradford 2003)
- Two species of river herring (Alewife, *Alosa pseudoharengus*; Blueback Herring, *A. aestivalis*) that spawn in the Miramichi and are fished commercially (Chaput and Atkinson 2001);
- Rainbow Smelt (*Osmerus mordax*) that are fished commercially in late fall and during the winter, under the ice (Chaput and LeBlanc 1996);

- Atlantic Tomcod (*Microgadus poulamon*) that spawn near the head of tide in winter under the ice (Bradford et al. 1997);
- Atlantic Salmon (*Salmo salar*) whose annual returns historically were the highest of individual rivers in eastern Canada (Moore et al. 1995);
- The only confirmed and annually predictable spawning area for the Striped Bass (*Morone saxatilis*) population of the southern Gulf of St. Lawrence (Douglas et al. 2011).
- American Eel (*Anguilla rostrata*) that rear in freshwater and estuarine areas of the Miramichi River and are fished commercially (Cairns et al. 2014).
- Sea Lamprey (*Petromyzon marinus*), an anadromous lamprey that spawns throughout the Miramichi River (Chaput 1995).

Assessments and stock status updates have been provided annually for Striped Bass (DFO 2020a) and Atlantic Salmon (DFO 2020b) but only infrequently for the other species. Life history and status information for a few species, American Shad and American Eel, have been reviewed in support of the development of COSEWIC status reports or for recovery potential assessments following a COSEWIC status assessment. Atlantic Tomcod has never been assessed even though it is an important commercial bycatch in the Rainbow Smelt fisheries of the southern Gulf (Chaput and LeBlanc 1991; Bradford et al. 1997). Sea Lamprey have never been assessed and are not commercially exploited (Chaput 1995).

Chaput (1995) indicated that the commercial landings of the diadromous species in the Miramichi River area represented approximately 50% of the total landings form the southern Gulf prior to 1940 but the contribution of the Miramichi to the total had declined to just over 30% by the early 1990s.

A summary of the life history characteristics of these species within the Miramichi River including general information on habitat, age at first maturity, fecundity, spawning season, proportion of time spent in freshwater, estuary and marine environments, size variations at maturity, and fisheries is provided in Chaput (1995). An attempt was also made to estimate the relative size, in number and biomass, of the annual returns to the Miramichi of the commercially exploited species based on fisheries landings, or index trapnet catches raised by assumed exploitation rates or efficiencies of trapnets. At the time of the publication, Chaput (1995) reported that the total adult biomass of diadromous fish migrating through the Miramichi estuary exceeded 16,000 t, with Rainbow Smelt accounting for 50% of the biomass and the gaspereau (two river herring species) representing just over 40% of the total biomass. At the time of the publication, Striped Bass were estimated to have been a very minor component of the total biomass, at < 0.1%. Based on the estimated mean weight at age and estimated spawner abundances at age of Striped Bass during 1996 to 2000, the estimated biomass of Striped Bass spawners is the range of 5 and 7 t for those years. Since 2015, the estimated biomass of Striped Bass spawners had increased to between 400 and 1,500 t. Based on the increased abundance of Striped Bass, and all else being equal for the other species, the proportion of the diadromous fish biomass comprised of Striped Bass would have increased from a negligible amount in the mid-1990s (< 0.1%) to between 3% and 10% since 2015.

The question regarding the impact of the increased abundance of Striped Bass on the other diadromous species is important in the context of establishing reference points to guide fisheries management of Striped Bass. The remainder of the document examines the evidence of potential impacts of an increasing Striped Bass population on other species and discusses options for reference points for Striped Bass that take account of these interactions.

STRIPED BASS PREDATOR/PREY INTERACTIONS WITH DIADROMOUS SPECIES OF FISHERIES INTEREST

Striped Bass is large bodied and piscivorous predator known to prey on valued anadromous fisheries species. The most important interaction is expected to be associated with predation by Striped Bass on these species. NEFSC (2019) summarized studies of Striped Bass along the eastern seaboard of the US and provide the same general descriptions of adult Striped Bass being generalist feeders on a variety of fish and invertebrates with the prey composition dependent upon the predator size (larger bass eat more fish), the time of year, and the foraging habitat. There is no indication from the literature of Striped Bass being a specialist feeder dependent on a particular prey; bass seemingly readily switch among prey based on availability. Andrews et al. (2017) provide information on diet of Striped Bass sampled in the Saint John River (NB) in an area downstream of a large hydro-electric dam. The size range of Striped Bass sampled by Andrews et al. (2017) is very large, essentially all Striped Bass exceeded 70 cm Total Length, and the most common prey item over all samples was gaspereau juveniles.

Vulnerability to predation by Striped Bass would depend on two main factors: body size of prey relative to gape size of the predator (Scharf et al. 2000), and overlapping spatial distribution of the prey and the predator. Based on these two considerations, the diadromous fish species and life stages, and their susceptibilities to predation by Striped Bass are summarized in Table 1.

DIET OF STRIPED BASS FROM THE SOUTHERN GULF OF ST. LAWRENCE

The best information available on Striped Bass diets in the southern Gulf is from the limited study conducted in 2013 to 2105 by DFO (2016) and summarized in Hanson (2020). Striped Bass were sacrifice sampled from angling and trapnet catches during the spawning period (May and June) in the Miramichi River and in the summer and early autumn from various shore locations in the southern Gulf of St. Lawrence (Table 2).

May and June in Miramichi

The diet of Striped Bass sampled in the Miramichi estuary during May and June was notably consistent between years with the majority (mean 68%, range 63%-77%) of Striped Bass stomachs empty (Figure 1). The highest proportion of empty stomachs occurred in late May and early June, when Striped Bass spawning is usually at its peak, and coincided with lower abundance of Rainbow Smelt and gaspereau at that time (Figure 2). This suggests that the majority of Striped Bass feeding in the Miramichi estuary during the spring occurs before and after the peak spawning time for Striped Bass.

Rainbow Smelt and gaspereau were the most frequently occurring prey in the samples. Despite their low occurrence in Striped Bass stomachs, gaspereau were the most important species in terms of prey weight (Figure 1). Gaspereau were absent from samples collected by angling which suggests that gaspereau were especially vulnerable to Striped Bass when both were captured in trapnets.

Striped Bass in the Miramichi River during the spring fed opportunistically and changed prey species as they became available (or unavailable) during different migration times (Figure 2). Rainbow Smelt were present when Striped Bass began feeding in the spring and were the first to be consumed, while gaspereau were the last to arrive in the estuary and also the last to be consumed. Small numbers of Atlantic Salmon smolts and in a low proportion of stomachs sampled were observed during the three years. Atlantic Salmon smolts were identified from samples collected during a relatively brief interval of time in late May to early June in the three years of sampling and the occurrence of smolts corresponded to the timing of the smolt migration in the Miramichi River. The highest occurrence of salmon smolts was approximately 30% of stomachs sampled in late May 2014; on all other sampling dates when salmon smolts were identified, they were identified from less than 10% of the stomachs sampled on that day (Figure 2).

Over all years and capture methods, the remaining prey species or prey categories were present in ≤ 2% of Striped Bass stomachs sampled and contributed on average ≤ 3% of the prey biomass during May and June in the Miramichi estuary (Figure 1).

Summer and Autumn Southern Gulf of St. Lawrence

Striped Bass were also sampled opportunistically during the months of June to October in 2013 to 2015 across the southern Gulf of St. Lawrence, from south of Chaleur Bay to Cape Breton (Table 2). Striped Bass were captured in a variety of habitats from a freshwater riverine location to saline coastal locations. The average fork length of Striped Bass sampled was 38.4 cm (range 21.3 – 73.1 cm). The diet of Striped Bass in the southern Gulf of St. Lawrence is diverse and consistent with the species that occupy estuarine and near shore coastal habitats (Hanson 2020). Eighteen species of fish, eight crustacean groups, three insect groups, marine worms, and a gastropod were identified in these stomach samples (Table 3).

EVIDENCE OF STRIPED BASS IMPACTS ON DIADROMOUS SPECIES OF FISHERIES INTEREST

As indicated previously, assessments and stock status have only been completed annually for Striped Bass (DFO 2020a) and Atlantic Salmon (DFO 2020b) however no assessment to date of Atlantic Salmon includes estimates of annual survival rates or considers the role that Striped Bass may play in the status of the resource. An updated assessment of the fisheries and status of the gaspereau stocks is anticipated in early 2021. Updates on status of the other species are not on the current schedule of assessments. For Atlantic Salmon, for which there have been studies on the survival rates of acoustically tagged smolts migrating through the Miramichi River (Chaput et al. 2018; Daniels et al. 2018, 2019).

The potential consequences of the increased abundance of Striped Bass on other diadromous species could potentially be indicated by correlations between Striped Bass abundance indicators and time series of recorded commercial harvests and to indicators of abundance based on catches at estuarine index trapnets, of other species.

INDICATORS OF CHANGES IN ABUNDANCE BASED ON LANDINGS

Annual recorded landings data of diadromous species by province within DFO Gulf Region were obtained from the DFO website for the years 1990 to 2018. For a number of years and species, the landings data are suppressed to meet confidentiality requirements. The recorded landings (t) by species / species group (gaspereau, Rainbow Smelt, American Eel) for the years 1990 to 2018 are summarized in Figure 3.

The gaspereau landings data from the DFO website for the 1990 to 1999 calendar years are essentially similar to the gaspereau landings data for Gulf NB reported in Chaput and Atkinson (2001), providing some level of confidence of the completeness of the aggregated data. Annual variations in the landings for the three species are likely in part due to differences in effort, changes in the number of active licences, and some differences in sales to buyers versus local sales (for bait) over time.

Gaspereau landings in Gulf NB have historically been dominated by the statistical districts that include the Miramichi River (Chaput 1995; Chaput and Atkinson 2001). There is a steep decline in recorded landings of gaspereau in the NB portion of the southern Gulf beginning in 2010 and continuing into 2017; landings data of 2015 were suppressed (Figure 3). In NS, gaspereau landings show a steep decline beginning in 1990 to 2000 with landings since 2000 remaining low although with a slight increase to 2018. Gaspereau landings in PEI are highly variable, but the lowest landings were consistently recorded during 2013 to 2018. The percentage decline in the mean landings for the recent period, 2011 to 2018, relative to the mean of an earlier period during 1995 to 2000 corresponding to low Striped Bass abundance was greatest for NB (74%) and least in PEI (125) (Figure 3).

For the NS area, Rainbow Smelt landings show a similar pattern of decline to gaspereau, with landings very low in the recent period 2011 to 2018 (Figure 3). Landings of smelt in NB show a steep decline beginning in 2008 to the lowest of the time series in 2015. In PEI, smelt landings have also declined since 2006 to reach the lowest levels of the times series in the recent five years. There was a large percentage decline in the mean landings during the recent period relative to the mean value of the earlier period in all three provinces, 70% in NB to 96% in NS (Figure 3).

American Eel show large annual variations, with only an observable decline in landings from NS (Figure 3). In PEI, recorded landings of American Eel currently surpass those of either Rainbow Smelt or gaspereau. In the other two provinces, the gaspereau harvests exceed those of Rainbow Smelt and American Eel. Changes in mean landings of the latter period relative to the earlier period ranged from a decrease of 58% in NS to an increase of 133% in PEI (Figure 3).

The only information on population size and fishing mortalities for these species is for the gaspereau stocks of the Miramichi River (NB) and the Margaree River (NS) up to the 1996 fishery year (Chaput and Atkinson 2001; Chaput et al. 2001). For the Miramichi, spawning escapements for both Alewife and Blueback Herring in the Miramichi River were decreasing and estimated fishing mortalities were high (above the reference F = 0.4 to 0.5) and increasing. Estimated fishing mortality rates in the Margaree River gaspereau fishery were also estimated to have been above the reference F = 0.5 (Chaput et al. 2001).

Associating the declines in gaspereau landings in Gulf NB to the increased abundance of Striped Bass is speculative. It is unclear what would be the cohort link between the species; if the predation effect is most important on the adult spawners or on the young of the year, then the lag between predation and recruitment would be 3 to 4 years, the dominant ages of maturity of gaspereau in this region. Hence, declines in landings that began in 2010 would have been associated with lower gaspereau spawner abundances of 2006 and 2007, corresponding to the years when Striped Bass spawner abundances were estimated to have been at low levels of 25 to 50 thousand spawners.

Rainbow Smelt are relatively short lived, the dominant ages in the fishery catches are 2 and 3 years old (Chaput and LeBlanc 1996). Using a similar premise to gaspereau of associating declines in Rainbow Smelt landings to increased predation by Striped Bass, then the link would have to be on young of the year recruitment rather than on post-spawned fish (Table 1). Therefore, the first year of low catches of smelt in NB recorded in 2011 would mechanistically have been the result of predation on young of the year smelt from the 2009 and 2010 year classes, when Striped Bass abundances were estimated at 50 to 60 thousand fish.

INDICATORS OF CHANGES IN ABUNDANCE BASED ON INDEX TRAPNET CATCHES

DFO has operated index estuary trapnets in the Northwest Miramichi (since 1998) and the Southwest Miramichi (since 1994) rivers for the purpose of monitoring the annual migrations and characteristics of diadromous and other fish species in the Miramichi River. The details of operation and the type of information obtained from the catches at the index trapnets are described by Hayward et al. (2014).

Briefly, live capture estuarine trapnets are installed in the tidal portion of the upper estuary of the river from mid to late May and are fished once a day, until mid to late October. All catches of fish are identified to species or species group (gaspereau), counted individually or batch estimated when there are large daily catches, and a subset or all are measured with additional data collected based on the species and the program objective. The majority of the fish are returned live to the water post-sampling. The important point is that these trapnets have been installed at the same location and monitored using similar procedures and protocols over the entire time series (Hayward et al. 2014). Annual start and end dates may vary somewhat and there are occasional, usually short term, periods that the traps are not operating due to high water discharges which can damage the gear or in recent years due to high water temperatures which would otherwise lead to stress and excess mortalities of the catches.

The data from these trapnets and supplementary catches from other fishing gears in the Miramichi are used in a mark and recapture experiment to estimate the annual returns of Atlantic Salmon by size group to each of the Northwest and Southwest Miramichi (DFO 2020b). The Atlantic Salmon assessment model also provides estimates of the annual catchabilities of each trapnet for Atlantic Salmon; the catchabilities vary annually but without a trend (DFO 2020c). The catches of Striped Bass in the spring and in the autumn months have also been presented as a supplementary index of trends in abundance of Striped Bass (DFO 2020a).

Figure 4 summarizes the total annual catches of gaspereau (Alewife and Blueback Herring combined), American Shad, Atlantic Salmon (anadromous adults size groups combined), and Striped Bass for each trapnet in the Northwest and Southwest Miramichi rivers. There are strong and consistent trends observed for these time series of data:

- Striped Bass catches were low, generally less than 500 fish annually, until 2010 when a rapid increase in annual catches were recorded at both facilities. The percent change in the mean abundance during 2015 to 2019 relative to the mean abundance during 1998 to 2002 was approximately 13,000% in the Northwest Miramichi and 7,300% in the Southwest Miramich (Figure 4). The annual catches are highest in June, and again in October, with particularly lower catches in August and September (Claytor 1996; Hayward 2001).
- Annual catches of American Shad also show an increase over the time period of monitoring, of 47% at the Northwest Miramichi facility and 259% at the Southwest Miramichi facility (Figure 4). The majority of these fish are spawners, catches are highest during the month of June and early July (Hayward 2001). Shad are a legally retained bycatch in the commercial gaspereau fishery and the fish are landed by the fishers.
- Catches of gaspereau are highly variable at the Northwest Miramichi trapnet, with an estimated decline of 27% in the mean catch of the 2015 to 2019 time period relative to the mean catch of the period 1998 to 2002 (Figure 4). The decline is more important in the Southwest Miramichi, at 86%, and the decline is generally continuous beginning in approximately 2005 (Figure 4). Whereas annual estimated catches of gaspereau at the Southwest Miramichi trapnet exceeded 150 thousand fish annually in the mid to late 1990s, estimated annual catches during 2011 to 2019 range from 11 to 35 thousand fish. Of note,

the commercial gaspereau fishery trapnets are located in the main stem of the Miramichi River and the Northwest Miramichi River; there were in some years one or two commercial gaspereau fishery trapnets in the Southwest Miramichi, of low catch intensity. Similarly, the Striped Bass spawner aggregations occur in the staging area below the confluence of the Northwest and Southwest Miramichi rivers and in the Northwest Miramichi River. Movements of Striped Bass do occur between the Southwest and Northwest Miramichi rivers in May and June but the bulk of the bass spawners are located in the Northwest Miramichi and downstream (Douglas et al. 2009).

- Atlantic Salmon (anadromous adults sizes combined) catches also show a decline at both facilities, but again with a more important decline at the Southwest Miramichi trapnet (43% decline) than in the Northwest Miramichi (27% decline) (Figure 4). The commercial gaspereau fishery is the only commercial fishery that can have an important bycatch of anadromous Atlantic Salmon adults returning to the Miramichi but in all fisheries in the DFO Gulf Region, Atlantic Salmon bycatch must be returned to the water as quickly as possible and in a manner that results in the least harm to the fish. In terms of Indigenous peoples fisheries, the most important fishery occurs in the Northwest Miramichi River, downstream of the Northwest Miramichi index trapnet whereas there are minimal Indigenous fisheries in the tidal waters of the Southwest Miramichi.

Collectively, Striped Bass predation and commercial fisheries removals of the diadromous fish would be expected to be most important in the Northwest Miramichi, however the declines in the indices of catches of gaspereau and salmon are more important in the Southwest Miramichi. The larger increased catches in the Northwest Miramichi of Striped Bass are expected given the estimated increase in the population size of the spawning stock located primarily in the Northwest Miramichi. The abundance indices of American Shad have increased at both facilities but with a larger increase in the Southwest Miramichi; there is a recognized shad spawning area in the Southwest Miramichi whereas the spawning area in the Northwest Miramichi is not known (Chaput and Bradford 2003).

SUMMARY OF INDICATORS OF CHANGES IN ABUNDANCE OF DIADROMOUS FISH

Gaspereau and Rainbow Smelt were important (occurrence) prey identified in Striped Bass stomachs sampled in May and June in the Miramichi River (Figures 1, 2). It has been argued that the declines in commercial landings of gaspereau and Rainbow Smelt in the Miramich area are directly associated with the increased predation pressure of a recovered Striped Bass stock and visually the declining trends are compelling. However, commercial fisheries harvests are generally not proportional to abundance, unless fishing effort and catchability are similar over time, which is almost never the case. In the Miramcihi River gaspereau fishery, there have been important changes in licence holders and fishing effort over the past decade, with effort (opening date of the fishery) and fishing practices modified to minimize the bycatch, handling, and discarding of Striped Bass. There have been equally important changes in licence holders in the commercial smelt fishery; fishers are getting older and there is less interest in continuing to fish using historical and labour intensive fishing methods.

Gaspereau abundance indices, either from reported commercial landings or from annual catches at the index estuarine trapnets of the Miramcihi show the largest declines occurring post-2010, approximating an inverse trend of increased abundance of Striped Bass. However, the decline in the trapnet indices for gaspereau is more important in the Southwest Miramichi which has less commercial fishery pressure than in the Northwest Miramichi. The gaspereau in the Southwest Miramichi would have less temporal overlap with Striped Bass aggregated on the spawning grounds of the Northwest Miramichi.

In a very simplistic way, landings in NB prior to the steep observed declines after 2005 were maintained at approximate levels of 3,000 t for gaspereau and 600 to 800 t for Rainbow Smelt. Prior to 2005, the corresponding Striped Bass spawner abundance was less than 25,000 fish and this level of Striped Bass abundance did not provide any fisheries opportunities for Striped Bass. As for Atlantic Salmon, the low indices of abundance recorded post 2010, particularly in the Southwest Miramichi correspond to the period when Striped Bass spawner abundances exceeded 100 thousand spawners. It was only after 2010 that re-opening of fisheries for Striped Bass were considered.

In the absence of assessments of population size, fishing mortality rates, and recruitment estimates for gaspereau and Rainbow Smelt, there is insufficient information to provide suitable guidance on reference points for Striped Bass that consider the potential predator effects on these species. There is substantially more information on the question of potential impacts of Striped Bass predation on Atlantic Salmon smolt survival and this is considered in greater detail in the next section.

ATLANTIC SALMON SPECIFIC INTERACTIONS

Adult anadromous Atlantic Salmon returning to DFO Gulf Region rivers, including the Miramichi River, range in size from approximately 50 cm to greater than 100 cm fork length. This fundamentally puts them outside the size range of potential prey of Striped Bass. The most likely interaction between Atlantic Salmon and Striped Bass is expected to occur at the juvenile smolt stage during the seaward outmigration phase. Atlantic Salmon smolts range in size from just over 10 cm fork length to generally less than 18 cm fork length (Chaput et al. 2016) which is suitable prey size range for most adult Striped Bass that exceed 40 cm fork length, based on observations that the diets of many marine fish are dominated by fish prey that are 10–20% of the predator’s length (Scharf et al. 2000).

The Atlantic Salmon smolt migration timing and corridor from freshwater to the sea has smolts from the Northwest Miramichi River in particular migrating through the spawning area and the staging areas of Striped B at approximately the same time as Striped Bass are aggregating and spawning in the Northwest Miramichi. As the spawning ground for Striped Bass is located in the Northwest Miramichi, the expectation is that smolts from the Northwest Miramichi would be most vulnerable to predation. Fortunately, there is an increasing body of research to draw on in understanding the potential interactions between Striped Bass and Atlantic Salmon smolts in the Miramichi River. In the following sections, we review several studies describing potential predator-prey interactions of Atlantic Salmon smolts and Striped Bass and examine evidence for population level effects of Striped Bass predation on abundance of anadromous adult Atlantic Salmon in the Miramichi River.

DIRECT EVIDENCE OF STRIPED BASS PREDATION ON ATLANTIC SALMON

Andrews at al. (2019) provide a literature review of available information regarding predation by Striped Bass on Atlantic Salmon, which is essentially very sparse.

DFO (2016) provides direct evidence of predation by Striped Bass on Atlantic Salmon smolts in the Miramichi River. A total of 48 Atlantic Salmon smolts were identified from 28 Striped Bass stomachs sampled during the 3-year diet study of Striped Bass in the Miramich River (Table 4). Many of these could only be confirmed based on otoliths. The majority of Striped Bass stomach samples that contained salmon smolts were collected by angling over a one (2015) or two week (2013, 2014) period in late May (Figure 2). The short duration of smolt predation by Striped Bass is consistent with the typical one to two week period corresponding to the peak smolt migration period (Chaput et al. 2002, 2016). The seaward migration of Atlantic Salmon smolts typically overlaps with the exit of post-spawned Rainbow Smelt through the estuary, returning to the ocean (Chaput et al. 2002). Atlantic Salmon smolts identified from sampled Striped Bass stomachs generally occurred as Rainbow Smelt presence declined and disappeared from the stomach samples (Figure 2). Additionally, one Atlantic Salmon, classified as a parr (non-migrating juvenile) was identified from the stomach of one Striped Bass captured in the Margaree River in the spring of 2014 (Table 3; DFO 2016).

INDIRECT EVIDENCE OF STRIPED BASS PREDATION BASED ON ACOUSTIC TAGGING AND TRACKING

Classification based on movement tracks

The use of acoustic tags and subsequent movement behaviours of tagged fish have been used to infer potential predation events of Striped Bass on acoustically tagged Atlantic Salmon smolts (Gibson et al. 2015; Daniels et al. 2018). Movement patterns of acoustically tagged Atlantic Salmon smolts migrating to sea and Striped Bass spawners in the Miramichi River were detected on common acoustic receiver deployments within the Miramichi River estuary in 2013 to 2016. Eight variables characterizing movement patterns were compiled including the average speed through system, the time between first and last detection within the study period, and the count of switches in upstream and downstream movement direction Movement patterns consistent with a smolt that had not been predated (training set) were described based on tracks of smolts subsequently detected at the Strait of Belle Isle array to the Labrador Sea. A Random Forest classification model was developed and the proportions of acoustically tagged Atlantic Salmon smolts that had movement patterns more similar to Striped Bass than to smolts that had been detected at the Strait of Belle Isle receiver array were estimated (Daniels et al. 2018). Atlantic Salmon smolts that had survived the exit to the Labrador Sea had movement patterns typically characterized by unidirectional downstream movements, in contrast to Striped Bass movement patterns which had more frequent up and downstream reversals (Daniels et al. 2018). Based on the classification model, the inferred percentages of smolts which displayed characteristic Striped Bass movement patterns, hence were concluded to have been eaten by Striped Bass with the tag present and transmitting in the body cavity of the Striped Bass, were highly variable ranging from 2.6% to 19.9% among years and between tag release locations within the Miramichi River. The inferred percentages of predation were higher for Northwest Miramichi tagged and released smolts (9.2 to 19.9%; versus 2.6 to 16.5% for the Southwest Miramichi) which is consistent with the higher estimated spatial and temporal overlap between the two species in the Northwest Miramichi (Table 5; Daniels et al. 2018).

Inference of predation based on predator tags

Daniels et al. (2019) tagged Atlantic Salmon smolts in the Northwest Miramichi River with novel acoustic tags, referred to as predator tags, that are used to directly detect the occurrence of a predation event. The predator tags will switch identification code triggered by a change in pH associated with the gastrointestinal tract. The tags (Amirix/Vemco V5 predator tags, 5.6 mm diameter by 12.7 mm length, 0.68 g in air) were small and programmed to transmit at random time intervals between 15 and 25 s at a frequency of 180 kHz. A total of 50 smolts were selected, ranging in fork length from 11.0 to 16.5 (0.5 cm bins), tagged and released between May 17 and May 29, 2017 (Daniels et al. 2019). Fish were released after a short recovery time, usually a minimum of one hour, with relatively small daily tag and release groups ranging from three to eight fish over a period of 13 days. Of the 50 smolts tagged and released, 41 were subsequently detected on downstream receivers in the Northwest Miramichi River and estuary. Of the 41 tags detected, 24 (59%) were detected with an identification code switch, indicating a predation event. The tags do not indicate what kind of predator would have consumed the acoustically tagged smolt. The high rate of predation inference was consistent with the high abundance of Striped Bass in the Miramichi River in 2017 although the authors caution against inferring mortality rates of wild and unmanipulated salmon smolts from acoustically tagged and tracked individuals (Daniels et al. 2019).

Time series estimates of relative survival from index rivers

A long term acoustic tagging and tracking study has been conducted by the Atlantic Salmon Federation (ASF) since 2003 in four rivers in the southern Gulf of St. Lawrence. A large part of the data from the ASF program were analysed by Chaput et al. (2018). The analyses included the annual tagging data from the Southwest Miramichi during 2003 to 2016, tagging data from the Northwest Miramichi during 2003 to 2008, 2013 to 2016, tagging data from the Restigouche River (Chaleur Bay) during 2004 to 2016 and tagging data from the Cascapedia River (Chaleur Bay) during 2006 to 2016 (Figures 5, 6). The value of this project in particular is the time series of relative survival rates which can derived from such data and the comparisons between two neighbouring bays (Miramichi, Chaleur) with different ecosystems, primarily Striped Bass abundant in Miramichi Bay and not so in Chaleur Bay.

During 2003 to 2016, a total of 2,862 Atlantic salmon smolts from four river populations and two neighbouring bays in eastern Canada were intercepted during their spring seaward migration and tagged with acoustic transmitters. The movements, detection rates and apparent survivals were estimated to the head of tide, at exit to the Gulf of St. Lawrence and at exit to the Labrador Sea, a migration covering a period of up to two months at sea and offshore marine distances of 800 km (Figure 5).

The study results can be summarized as follows (Chaput et al. 2018):

- Survival rates of “tagged smolts” through Chaleur Bay (Restigouche, Cascapedia) were relatively high (67% to 95%), with no change over time (Figure 7).
- Survival rates of “tagged smolts” through Miramichi Bay were lower (28% to 82%) and showed a decline in survival beginning in 2010 (Figure 7).
- The Northwest Miramichi River estimated survival rates through the bay actually show two clusters of survival rates corresponding to two experimental periods with different experimental conditions (Figure 7). During 2003 to 2008, the smolts were captured in the Little Southwest Miramichi, tagged, and released at that same location (Figure 5). During 2013 to 2016, the smolts were captured in the Northwest Miramichi, transported upstream a distance of less than 20 km before being tagged and released at the upstream site.
- The differences in apparent survival rates in two neighbouring coastal embayments have been hypothesized to be in part related to differences in predation pressure on migrating smolts from Striped Bass present in the Miramichi Bay during the smolt migration period but not in the Chaleur Bay (Chaput et al. 2018; Daniels et al. 2018).

The tagging and tracking experiments have continued into 2020 but the results have not been reported or peer reviewed. The multi-year and multi-river aspects of the ASF study provide particular advantages to describing and modelling smolt migrations and estimating survival rates that otherwise would not be possible from single year and single river experiments. Specifically, the time series estimates of survival rates can be used to examine possible associations with changes in abundance with other components of the ecosystem, such as predators of salmon smolts.

Figure 8 shows the scatter plot of Striped Bass spawner abundance estimates and the estimated survival rates of Northwest Miramichi and Southwest Miramichi acoustically tagged smolts. For both the Southwest Miramichi and Northwest Miramichi tagged smolts, the lowest survival rates from head of tide to bay exit were estimated in the recent period (2013 to 2016) when the estimated abundance of Striped Bass was greater than 100 thousand spawners. The linear association between bass abundance and survival is strong for the Northwest Miramichi data but weaker for the Southwest Miramichi smolts which show more variation in estimated survival over the time series (Figure 8). In showing these relationships, it is assumed that the time series of estimates of survival and Striped Bass abundance estimates are exchangeable. This is the case for the Striped Bass abundance estimates and the Southwest Miramichi smolt survival estimates, but it may not be for the Northwest Miramichi survival estimates. As indicated previously, during 2003 to 2008 the smolts were captured in the Little Southwest Miramichi, tagged and released at the point of capture whereas during 2013 to 2016, the smolts were captured in the Northwest Miramichi, transported upstream prior to being tagged and released.

An important concern regarding the use of marked animals to draw inferences on survival of unmarked/unhandled animals is the consequence of tagging and handling effects on the estimates of survival. It is extremely difficult to make the case that a tagged smolt would have the same mean probability of survival as an untagged smolt as the capture, handling, tagging procedures in addition to introducing stress and injury to individual animals (Amman et al., 2013) also interrupt the migration during a particularly sensitive period (Riley et al. 2007). The removal of individuals from schooling with conspecifics can result in increased vulnerability to predation (Furey et al. 2016).

If the experimental conditions of long term studies are standardized to ensure that the observations reflect to the extent possible the variations in the phenomenon of interest, rather than differences in experimental methodologies (design, technology), then the time series trends from acoustically tagged animals may well reflect the time series trends of un-manipulated animals. The case could be made that the data for the Southwest Miramichi experiment would meet the criteria of standardized experimental conditions, in which case, the estimated trends in survival rates of acoustically tagged smolts and the association with Striped Bass abundance may in fact correspond to trends in survival rates of un-manipulated Atlantic Salmon smolts from the Miramichi River.

The next section models the population dynamics of Atlantic Salmon in the Miramichi River using the annual indices of juvenile salmon abundance and the estimated returns of 1SW and 2SW salmon adults. The point of interest in this exercise is to estimate relative sea survival rates during the first year at sea and examine correlations of these to abundance estimates of Striped Bass, hence, we are looking for population level effects.

POPULATION LEVEL ESTIMATES OF RELATIVE SURVIVAL OF ATLANTIC SALMON

It has been shown that Striped Bass will consume Atlantic Salmon smolts in the Miramichi River (DFO 2016). Inferences of predation by Striped Bass on salmon smolts have also been reported based on acoustically tagged salmon smolt behaviours and using predator tag technologies (Gibson et al. 2015; Daniels et al. 2018, 2019), and from time series modelling of survival rates correlated to time series of Striped Bass spawner abundances (Chaput et al. 2018; this manuscript).

Based on estimates of survival of acoustically tagged salmon smolts in the Miramichi River, the pattern is of decreased survival during the early stage of migration in the Miramichi but no change for the rivers of Chaleur Bay. Whether these decreases or no change in survival rates of tagged smolts are reflected in the return rates as 1SW and 2SW fish over the entire period of marine life is not known. Atlantic Salmon marine return rates to adults in many monitored rivers of eastern Canada have declined over the past four decades (ICES 2020), however, since 1996 the return rates to 1SW salmon for four monitored rivers in the Maritime provinces and Quebec show variable but non-statistically significant linear trends (Figure 9).

There are very few reliable estimates of smolt production and marine return rates of Atlantic Salmon in the Miramichi River (Chaput et al. 2016), and too few over the time series of assessed Striped Bass spawner abundances with which to examine population level of predation by Striped Bass on Atlantic Salmon smolts.

The population dynamics of Atlantic Salmon in the Miramichi River were modelled using the annual indices of juvenile salmon abundance and the estimated assessed returns of 1SW and 2SW salmon adults. The point of interest in this exercise is to estimate relative sea survival rates during the first year at sea to see if there has been any trend in these rates and how these trends may correlate with the abundance estimates of Striped Bass. If predation by Striped Bass on Atlantic Salmon smolts is intense enough during the early post-smolt phase, we would expect to see a signal in the relative return rates of adult salmon.

Data

Annual indices of juvenile Atlantic Salmon, by size/age group (fry = young of the year, small parr = 1-year old juveniles; large parr = 2+ year old juveniles), were obtained by electrofishing at fixed sites throughout the Miramichi River. Sampling methods and models for estimating densities (number of fish per 100 m^2) by size / age group are described in Chaput et al. (2005) and Moore and Chaput (2007). The new Bayesian model developed by Dauphin et al. (2019) has not yet been applied to the time series of observations used in this analysis. The juvenile abundance time series for the four main branches (rivers that have a confluence in tidal waters) of the Miramichi River to 2019 are presented in DFO (2020b; Figure 10). Juvenile indices of abundance by size group for the Northwest Miramichi system (Little Southwest Miramichi habitat area = 807 ha; Northwest Miramichi habitat area = 823 ha) and for the Southwest Miramichi system (Renous habitat area = 582 ha; Southwest Miramichi habitat area = 2,953 ha) were obtained as a habitat weighted mean of the mean densities in the appropriate tributaries (Figure 11).

Adult Atlantic Salmon returns to each of the Northwest Miramichi and Southwest Miramichi rivers have been estimated since 1992 and the time series of estimates for small salmon (< 63 cm fork length, primarily 1SW salmon) and large salmon (>= 63 cm fork length, majority 2SW and repeat spawners) to 2019 are presented in DFO (2020b, 2020c). Sea age composition of adult salmon are determined from interpretation of scales which are sampled from returning adult salmon captured at the DFO index estuary trapnets of the Northwest and Southwest branches of the Miramichi. From these data and the estimated returns of large salmon, the returns of 2SW maiden salmon are estimated; 2SW salmon comprised annually varying proportions of the large salmon returns, between 0.41 and 0.85 during the period 1992 to 2019. The time series of estimated 1SW (small salmon) and 2SW salmon for the two main branches of the Miramichi River are shown in Figure 12.

The juvenile indices and the branch specific adult returns by sea age group time series use dn the population model begins in 1993 and extends to 2019.

Dynamic equations and cohort model

A cohort model was coded that tracks a cohort of salmon beginning at the juvenile fry stage (1993 to 2018), through small parr (1994 to 2018), large parr (1995 to 2018) and to returns of 1SW (1997 to 2019) and 2SW (1998 to 2019) adult salmon assessed in the river for each main branch of the Miramichi. Smolt output at age 2 in year y is an intermediate state derived from small parr abundance of year-1. Smolt output at age 3 in year y is an intermediate state derived from large parr abundance of year -1. Total smolts going to sea is the sum of smolts at age-2 and age-3 in year y and 1SW salmon returns in year y+1 are calculated from total smolts going to sea in year adjusted for marine survival in year and the proportion of the smolt cohort that matures at 1SW of age. The equations for the model are summarized in Table 6.

Three model variants were examined:

1. A river independent model in which the parameters were estimated independently for the Southwest and Northwest dynamics with no linkage between river dynamics (Appendix 1).
2. A model that assumed a common trend in smolt to 1SW survival for the two rivers but with a constant intercept shift for one of the rivers. All other parameters were set independent between the rivers.
3. A model that assumed a single common trend over time for the rivers in the proportion of small parr that become smolts age 2 and a single common trend for smolt to 1SW survival for the two rivers. All other parameters were set independent between the rivers.

Likelihoods and priors

The model was coded in OpenBugs (Appendix 1) and the posterior distributions of the parameters were obtained by MCMC sampling (Lunn et al. 2013).

Lognormal likelihoods for the abundance of fry, small parr, large parr, 1SW returns and 2SW returns were used.

Non-informative priors were set on the precisions (inverse variance) for each of the life stages, excluding fry. Fry were the initiating life stage; the likelihood of the mean fry index from sampling for river r and year t was assumed lognormal with $log(u.fry_{y,r})$ drawn from a prior distribution for year and river and a precision by river for all years set to the mean coefficient of variation of the fry abundance indices over years by river (cv = 0.14 for Southwest, 0.22 for Northwest).

Priors for the other parameters (Appendix 1) were variously set using a conjugate distributions or censored distributions (Table 7).

Results

All the models converged rapidly. A total of 100,000 iterations was used for a burn-in. A subsequent 25000 iterations with 2 chains and thinning by 10 were used to extract 5,000 MCMC samples to characterize the posterior distribution. All posterior parameter distributions were unimodal. Model diagnostics are summarized in Appendix 2 and the indicators of model adequacy to data are summarized in Table 8.

The model that fits the juvenile and adult return time series independently between rivers (model 1) was retained (Table 8; Appendix 2).

Conditional on model structure and assumptions, the fry and small parr survival rates are estimated to be higher in the Northwest Miramichi compared to the Southwest Miramichi (Figure 13). The proportion of smolts that mature as 1SW is also estimated to be higher for the Northwest Miramichi (Figure 13); this is consistent with other life history characteristics which indicate a higher proportion female in the 1SW maiden returns of the Northwest Miramichi compared to the Southwest Miramichi (Chaput et al. 2016).

The proportions of the small parr that were estimated to become smolt age 2 have declined in both the Northwest and Southwest Miramichi rivers (Figure 14). In terms of relative smolt production at age 2 and age 3, there has been a decline in the estimated relative number of smolts age 2 but no change in the estimated production of smolts age 3 (Figure 14). This is also consistent with data; generally small parr indices have been declining at a faster rate than the large parr indices (Figure 10).

The relative smolt to 1SW survivals show large variation in both the Southwest and Northwest Miramichi rivers, but without a statistically significant linear trend over the time series 1996 to 2018 (Figure 15). The term relative survival rate is used because the estimated smolt production is raised using the total habitat area of the rivers, which exaggerates the smolt production from juvenile indices; the juvenile indices are derived for specific components of the habitat, classic juvenile rearing habitat. The relative return rates are generally higher in the Southwest Miramichi relative to the Northwest Miramichi. Survival rates of the 2009 smolt migration year were high for both rivers.

When the relative survival rates are plotted against the corresponding Striped Bass spawner abundances for the year of smolt migration (and the year of potential predation by bass), there is an apparent decline in relative survival rates of smolts from the Southwest Miramichi, especially for the 2006 to 2018 migration years (the highest relative survival rates were estimated for the 2009 smolt migration year) associated with increasing Striped Bass abundance (Figure 15). However, low relative survival rates for the Southwest Miramichi were estimated in the late 1990s when Striped Bass abundances were low. The negative relationship between relative survival rates and Striped Bass spawner abundances is less clear to non-existent for the Northwest Miramichi smolts (Figure 15); in fact estimated relative survival rates have been variable over the entire time series with equally high and low relative survival rates in the late 1990s when Striped Bass abundances were low and again in the 2010s when Striped Bass abundances were high.

Relative survival rates from smolt to 1SW salmon are plotted against the survival rates through the bays of acoustically tagged smolts for the smolt migration years 2003 to 2016 of the Southwest Miramichi, and the smolt migration years 2003 to 2008 and 2013 to 2016 for the Northwest Miramichi (Figure 16). The survival rates of acoustically tagged smolts in the Southwest Miramichi are highly variable for corresponding low relative survival rates but generally, modelled relative survival rates from smolts to 1SW correspond to years when the survival rates of smolts through the bay were higher, with a statistically weak linear relationship ($p = 0.045$; Figure 16). For the Northwest Miramichi, there is no association between the survival rate estimates through the bay and the estimated relative survival rates from smolts to 1SW salmon (Figure 16).

Discussion

The cohort modelling of the juvenile to adult return indices for the Miramichi River provide indications at least for the Southwest Miramichi that relative survival rates of smolts to 1SW salmon may be negatively associated with Striped Bass abundances, at least for the period of 2006 to 2018. In part, this pattern matches the association between the acoustic tagged smolt derived survival rates through Miramichi Bay and Striped Bass abundances for the Southwest Miramichi, and suggests that early post-smolt survival perhaps driven by predation may define the total annual survival rate trends to 1SW adult returns. On the other hand, the relative survival rates of smolts to 1SW salmon for the Northwest Miramichi time series do not correspond to the acoustic tagged smolt derived survival rates nor are they associated with the variations in Striped Bass spawner abundances, thus providing no evidence of a survival variation driven by Striped Bass predation on smolts.

If we were to accept that there is a population level effect of predation on smolts by Striped Bass, it is not clear what would be an appropriate abundance of Striped Bass spawners that would not induce excessive early mortality on smolts. For the Southwest Miramichi, there have been equally low relative survival rates at very low and very high Striped Bass spawner abundances, with higher relative survivals of salmon at Striped Bass abundances of 20 to 100 thousand spawners. The relative survival rate estimated for the 2009 smolt cohort is anomalously high (Figure 15). However, similarly high survival rates of the 2009 smolt cohort relative to the river-specific time series were noted for the Nashwaak River (NB), LaHave River (NS) as well as in the two index rivers in Quebec (Figure 9) which indicates that in some years, broad ocean scale factors dominate the marine survival dynamics of Atlantic Salmon rather than nearshore or early post-smolt survival conditions.

CONCLUSIONS

The analyses of information provides conflicting evidence of reductions in a few anadromous fish species abundance indicators associated with increased abundance of Striped Bass in the southern Gulf. The recorded commercial landings of gaspereau and Rainbow Smelt have greatly declined in the Gulf NB portion of the southern Gulf of St. Lawrence, particularly since 2010. Higher landings of the period prior to 2010 were associated with Striped Bass spawner abundances of less than 5,000 to 50 thousand spawners.

The indices of abundance based on estuarine index trapnet catches in the two main branches of the Miramichi show a decline in total annual catches of gaspereau and adult Atlantic Salmon, with the more important declines in the Southwest Miramichi compared to the Northwest Miramichi. The declines in the Southwest Miramichi for those two species seemingly began in 2005, and has been less abrupt than indicated by the commercial landings of gaspereau. In contrast, American Shad catches have increased in both the Northwest and Southwest Miramichi, despite increased catches and spawner abundance estimates of Striped Bass.

There is direct evidence of predation by Striped Bass on Atlantic Salmon smolts and several studies using acoustic tag technologies have inferred predation events and changes in estimated survival rates in the early phase of migration through Miramichi Bay that point to Striped Bass predation as a likely driver of these variations in estimated survival rates. Based on acoustic tagging estimates of survivals through Miramichi Bay, the years when Striped Bass spawner abundances exceeded approximately 100 thousand spawners corresponded to year with visibly lower estimated survival rates.

Population level effects, described by estimates of relative survival rates of smolts in the first year at sea based on juveniles indices as proxies for smolt output, are contradictory between the two branches of the Miramichi. Whereas the relative survival rates of smolts from the Southwest Miramichi are associated with variations in Striped Bass abundance indices, it is not the case for the Northwest Miramichi which was expected to be more impacted by predation considering the spatial and temporal overlap of Striped Bass spawner aggregations and the smolt migration window. These analyses are not conclusive of effects or no effects of Striped Bass on smolt to adult returns and point to the need for conducting carefully designed ecological experiments to directly resolve the mechanisms and cause/effect of interactions between Striped Bass and other anadromous species in the Miramichi River.

Striped Bass abundances in the range of 100 thousand spawners in the past corresponded to high landings of gaspereau and smelt, and the highest survival rates of acoustically tagged smolts through Miramichi Bay. Setting a management objective for Striped Bass at approx. 100 thousand spawners, perhaps calling this a target reference point (rather than upper stock reference), will result in large reductions of the potential fisheries yield of Striped Bass. Based on the populations models and equilibrium abundance estimates presented in Chaput and Douglas (2022), maintaining spawner abundances at 100 thousand fish would be equivalent to a fishing rate that exceeds F = 0.50, yields that are substantially lower than what could be realized at MSY, and an abundance which is less than the single species Limit Reference Points regardless of the population model selected.

It is not clear from these time series of data, that reducing Striped Bass spawner abundances to the level of the early 2000s, i.e., less than 100 thousand spawners, would improve the acoustic tagged smolt survival estimates, the population level relative survival rates derived from the cohort model, or the landings trends of gaspereau and Rainbow Smelt in the commercial fisheries. It is not possible to suggest a reference level to address the multiple species concerns based on the information and analyses presented in this working document. Ultimately, the decision to use an alternate “target” Striped Bass reference point to account for the multi-species interactions will be made by fisheries managers. A carefully designed and monitored ecological experiment, could be envisioned to resolve the question of these species interactions. Such an experiment would require a long time series of monitoring and should include a control site, such as the Restigouche in which the Atlantic Salmon smolts and other species are not subject to Striped Bass predation in the spring, that is geographically proximate to the Miramichi River such that climate factors and other inriver dynamics closely resemble the conditions in the Miramichi River.

REFERENCES CITED

Ammann, A.J., Michel, C.J., and MacFarlane, R.B. 2013. The effects of surgically implanted acoustic transmitters on laboratory growth, survival and tag retention in hatchery yearling Chinook salmon. Environ. Biol. Fish 96: 135-143.

Andrews, S.N., Hirtle, S.V., Linnansaari, T., and Curry, R.A. 2019 Consumption of Atlantic Salmon Smolt by Striped Bass: A Review of the Predator–Prey Encounter and Implications for the Design of Effective Sampling Strategies. Fishes 2019, 4, 50.

Bradford, R.G., Chaput, G., Hurlbut, T., and Morin, R. 1997. Bycatch of striped bass, white hake, winter flounder, and Atlantic tomcod in the autumn “open water” smelt fishery of the Miramichi River estuary. Canadian Technical Report Fisheries and Aquatic Sciences No. 2195.

Cairns, D.K., Chaput, G., Poirier, L.A., Avery, T.S., Castonguay, M., Mathers, A., Casselman, J.M., Bradford, R.G., Pratt, T., Verreault, G., Clarke, K., Veinott, G., and Bernatchez, L. 2014. Recovery Potential Assessment for the American Eel (*Anguilla rostrata*) for eastern Canada: life history, distribution, reported landings, status indicators, and demographic parameters. DFO Can. Sci. Advis. Sec. Res. Doc. 2013/134. xiv + 157 p.

Chaput, G.J. 1995. Temporal Distribution, Spatial Distribution, and Abundance of Diadromous Fish in the Miramichi River Watershed, p. 121-139. *In* E.M.P. Chadwick [*editor*]. Water, science, and the public: the Miramichi ecosystem. Can. Spec. Publ. Fish. Aquat. Sci. 123.

Chaput, G., and Atkinson, G. 2001. The gaspereau fisheries (*Alosa pseudoharengus* and *A. aestivalis*) of the Miramichi River with updates on the fishery of Richibucto River of Gulf New Brunswick, 1997 to 2000. DFO Can. Sci. Advis. Sec. Res. Doc. 2001/047: 39 p.

Chaput, G., and Bradford, R.G. 2003. American shad (*Alosa sapidissima*) in Atlantic Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2003/009. iv + 71 p.t

Chaput, G., and Douglas, S. 2022. Fisheries Reference Points for Striped Bass (*Morone saxatilis*) from the Southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2022/029. xiv + 153 p.

Chaput, G., and LeBlanc, C. 1996. Anadromous rainbow smelt (*Osmerus mordax* Mitchill) from the Gulf of St. Lawrence – update of fishery and preliminary status of stocks. DFO Atl. Fish. Res. Doc. 96/109.

Chaput, G., LeBlanc, P., and Crawford, R. 2001. Assessment of the Margaree River gaspereau fishery, 1997 to 2000. DFO Can. Sci. Advis. Sec. Res. Doc. 2001/046: 35 p.

Chaput, G., Moore, D., and Peterson, D. 2005. Predicting Atlantic salmon (*Salmo salar*) juvenile densities using catch per unit effort open site electrofishing. Can. Tech. Rep. Fish. Aquat. Sci. No. 2600. v + 25 p.

Chaput, G., Hardie, P., Hayward, J., Moore, D., Sheasgreen, J., and NSPA. 2002. Migrations and biological characteristics of Atlantic salmon (*Salmo Salar*) smolts from the Northwest Miramichi River, 1998 to 2000. Can. Tech. Rep. Fish. Aquat. Sci. No. 2415. 70p.

Chaput, G., Douglas, S.G., and Hayward, J. 2016. Biological Characteristics and Population Dynamics of Atlantic Salmon (*Salmo salar*) from the Miramichi River, New Brunswick, Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2016/029. v + 53 p.

Chaput, G., Carr, J., Daniels, J., Tinker, S., Jonsen, I., and Whoriskey, F. 2018. Atlantic Salmon (*Salmo salar*) smolt and early post-smolt migration and survival inferred from multi-year and multi-stock acoustic telemetry studies in the Gulf of St. Lawrence, northwest Atlantic. ICES J. Mar. Sci. 76: 1107-1121.

Claytor, R.R. 1996. Weekly Fish Counts from In-River Traps, Counting Fences, Barrier Pools, and Fishways in Southern Gulf of St. Lawrence Rivers, From 1952 – 1993. Can. Data Rep. Fish. Aquat. Sci. No. 982.

Daniels, J., Chaput, G., and Carr, J. 2018. Estimating consumption rate of Atlantic salmon smolts (*Salmo salar*) by striped bass (*Morone saxatilis*) in the Miramichi River estuary using acoustic telemetry. Can. J. Fish. Aquat. Sci. 75: 1811–1822.

Daniels, J., Sutton, S., Webber, D., and Carr, J. 2019. Extent of predation bias present in migration survival and timing of Atlantic salmon smolt (*Salmo salar*) as suggested by a novel acoustic tag. Anim. Biotelemetry 7:16. https://doi.org/10.1186/s40317-019-0178-2.

Dauphin, G.J.R., Chaput, G., Breau, C., and Cunjak, R.A. 2019. Hierarchical model detects decadal changes in calibration relationships of single-pass electrofishing indices of abundance of Atlantic salmon in two large Canadian catchments. Can. J. Fish. Aquat. Sci. 76: 523–542.

DFO. 2016. Spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 2015. DFO Can. Sci. Advis. Sec. Sci. Resp. 2016/017.

DFO. 2019. Guidance for the development of rebuilding plans under the Precautionary Approach Framework: Growing stocks out of the critical zone.

DFO. 2020a. Update of spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence to 2019. DFO Can. Sci. Advis. Sec. Sci. Resp. 2020/009.

DFO. 2020b. Update of indicators of Atlantic Salmon (*Salmo salar*) in DFO Gulf Region Salmon Fishing Areas 15 - 18 for 2019. DFO Can. Sci. Advis. Sec. Sci. Resp. 2020/028.

DFO. 2020c. Update of indicators to 2019 of adult Atlantic Salmon for the Miramichi River (NB), Salmon Fishing Area 16, DFO Gulf Region. DFO Can. Sci. Advis. Sec. Sci. Resp. 2020/010.

DFO. 2021. Reference points for Striped Bass (*Morone saxatilis*) for the southern Gulf of St. Lawrence population. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2021/018.

Douglas, S.G., and G. Chaput. 2011. Information on the Striped Bass (*Morone saxatilis*) population of the southern Gulf of St. Lawrence relevant to the development of a 2nd COSEWIC status report for the species. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/098. iv + 16 p.

Douglas, S.G., Chaput, G., Hayward, J., and Sheasgreen, J. 2009. Prespawning, spawning, and postspawning behavior of striped bass in the Miramichi River. Trans. Am. Fish. Soc. 138: 121-134.

Furey, N.B., Hinch, S.G., Bass, A.L., Middleton, C.T., Minke-Martin, V., and Lotto, A.G. 2016. Predator swamping reduces predation risk during nocturnal migration of juvenile salmon in a high-mortality landscape. J. Anim. Ecol. 85: 948–959.

Gibson, A.J., Halfyard, E.A., Bradford, R.G., Stokesbury, M.J., and Redden, A.M. 2015. Effects of predation on telemetry-based survival estimates: insights from a study on endangered Atlantic salmon smolts. Can. J. Fish. Aquat. Sci. 72: 728–741.

Grout, D.E. 2006. Interactions between striped bass (*Morone saxatilis*) rebuilding programmes and the conservation of Atlantic salmon (*Salmo salar)* and other anadromous fish species in the USA. ICES J. Mar. Sci. 63: 1346-1352.

Hanson, J.M. 2020. Diet of Striped Bass in the Southern Gulf of St. Lawrence (Canada) with Emphasis on the Spring Spawning Aggregation. Northeastern Naturalist 27(3): 381-400.

Hayward, J. 2001. Weekly Fish Counts from In-River Traps and Barrier Pools in the Miramichi River, New Brunswick, 1994 to 1999. Can. Data Rep. Fish. Aquat. Sci. 1080: 104 p.

Hayward, J., Sheasgreen, J., Douglas, S., and Reid, J. 2014. Diadromous fish monitoring programs in the Miramichi River system in 2011. Can. Manuscr. Rep. Fish. Aquat. Sci. No. 3042: xiii + 102 p.

ICES. 2020. Working Group on North Atlantic Salmon (WGNAS). ICES Scientific Reports. 2:21. 357 pp.

Lunn, D., Jackson, C., Best, N., Thomas, A., and Spiegelhalter, D. 2013. The BUGS Book a Practical Introduction to Bayesian Analysis. CRC Press Taylor & Francis Group, LLC, Boca Raton, Florida. 381 pp.

Moore, D., and Chaput, G. 2007. Juvenile Atlantic salmon (*Salmo salar*) surveys in the Miramichi River Watershed from 1970-2004. Can. Data. Rep. Fish. Aquat. Sci. No. 1188. 117p.

Moore, D.S., G.J. Chaput, and P.R. Pickard. 1995. The effect of fisheries on the biological characteristics and survival of mature Atlantic salmon (*Salmo salar*) from the Miramichi River, p. 229-247. . In E.M.P. Chadwick [editor]. Water, science, and the public: the Miramichi ecosystem. Canadian Special Publication of Fisheries and Aquatic Sciences 123.

Northeast Fisheries Science Center (NEFSC). 2019. 66th Northeast Regional Stock Assessment Workshop (66th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 19-08; 1170 p.

Riley, W.D., Ibbotson, A.T., Gregory, S.D., Russell, I.C., Lauridsen, R.B., Beaumont, W.R.C., Cook, A.C., and Maxwell, D.L. 2018. Under what circumstances does the capture and tagging of wild Atlantic salmon *Salmo salar* smolts affect probability of return as adults?. J. Fish Biol. 93: 477-489.

Scharf, F. S., Juanes, F., and Rountree, R. A. 2000. Predator size–prey size relationships of marine fish predators: interspecific variation and effects of ontogeny and body size on trophic-niche breadth. Mar. Ecol. Prog. Ser. 208: 229–248.

TABLES

Table 1. Summary of fish species characteristics including life stage, body size, and location of most probable interaction with adult (age 3+) Striped Bass in the Miramichi River.

Species	Lifestage	Approx. body size (cm)	Location and time of overlap with Striped Bass
Rainbow Smelt	Spawners / postspawners	13 to 21	Estuary April to June
	Juveniles	< 10	Estuary and bay Autumn
Gaspereau	Spawners / postspawners	22 to 30	Estuary mid-May to early July
	Young of the year	< 8 cm	Lower portions of rivers, estuaries late summer onward
American Shad	Young of the year	< 8 cm	Lower portions of rivers, estuaries late summer onward
Atlantic Salmon	Smolt stage	10 to 18 cm	Estuary mid-May to early June; 3 to 4 week period
American Eel	Yellow phase	15 to 50	Estuaries April to October
Sea Lamprey	Ammocoetes	10 to 15	Estuaries May and June associated with metamorphosis to parasitic stage and migration to the sea
Atlantic Tomcod	Adult	14 to 25	No overlap with Striped Bass; migrate to estuaries in late autumn, spawn in the winter, return to Miramichi Bay and southern Gulf by April

Table 2. Summary of sampling effort for Striped Bass diets during 2013 to 2015. Samples are summarized by season (spring = May and June in Miramichi only; other = all samples other than those collected in the Miramichi during spring), capture location, capture date range, and capture method (angling or trapnet). GNS = Gulf Nova Scotia, NNB = northern New Brunswick, and SENB = southeastern New Brunswick. Summary table is from DFO (2016).

				Capture date range		Capture method	
Year	Season	Region	Location	Min	Max	Angling	Trapnet
2013	spring	Miramichi	Northwest	5-May	26-Jun	153	320
			Southwest	25-Jun	26-Jun	0	30
			Main Miramichi	1-May	17-May	77	0
	other	Miramichi	Northwest	24-Sep	30-Sep	0	9
			Southwest	23-Sep	10-Oct	0	76
Subtotal 2013						230	435
2014	spring	Miramichi	Northwest	9-May	25-Jun	178	295
			Southwest	15-May	20-Jun	34	30
			Main Miramichi	22-May	29-May	78	0
	other	GNS	Margaree	6-Jun	6-Jun	0	23
		Miramichi	Main Miramichi	6-Oct	22-Oct	18	0
		NNB	Shippagan	6-Oct	6-Oct	1	0
			Burnt Church	7-Oct	16-Oct	3	0
			Inkerman	16-Oct	16-Oct	3	0
		SENB	Cocagne	10-Jul	11-Jul	11	0
			Cote-Ste-Anne	28-Sep	28-Sep	13	0
Subtotal 2014						339	348
2015	spring	Miramichi	Northwest	15-May	29-Jun	162	299
			Southwest	12-May	2-Jun	8	12
			Main Miramichi	20-May	22-May	143	0
			Striper cup	30-May	31-May	25	0
	other	GNS	Margaree	6-Jun	20-Oct	81	9
			Antigonish	22-Jun	22-Jun	4	0
			Pictou	23-Jun	30-Jul	34	0
			Grand Etang	24-Aug	24-Aug	8	0
		Miramichi	Main Miramichi	5-Aug	14-Sep	10	0
		NNB	Burnt Church	16-Jun	7-Oct	26	0
			Tracadie	14-Aug	14-Aug	1	0
			Inkerman	4-Oct	14-Oct	8	0
		SENB	Little Bouctouche	25-Jul	27-Sep	73	0
			Bouctouche	27-Sep	11-Oct	23	0
			Cocagne	28-Sep	13-Oct	26	0
			St. Edouard	15-Oct	20-Oct	7	0
Subtotal 2015						639	320
Grand total 2013-2015						1,208	1,103

Table 3. Diet, as % occurrence of prey species or prey categories, of Striped Bass (21 to 73 cm fork length) collected from various locations throughout the southern Gulf of St. Lawrence in 2013-2015, other than in the Miramichi estuary during May and June.The % occurrence has been rounded to the nearest percentage and the total may be greater than 100% due to individual stomachs containing more than one prey type.

	% Occurrence			
Prey	2013	2014	2015	Combined
Decapoda (*Crangon septemspinosa, Neomysis* spp., *Palaemoneter vulgaris*)	1	38	39	32
Crustacean (Green crab, Rock crab)	1	0	1	1
Insect (Anisoptera, Ephemeroptera, Plecoptera)	1	1	2	1
Gastropod (Common periwinkle)	0	0	1	0
Other invertebrates (Gammarid amphipod, isopod, mysid, polychaete)	0	3	6	4
Atlantic Silverside	0	1	12	8
Stickleback (3-spine, 4-spine, *Gasterosteus* spp.)	0	13	9	8
Mummichog	0	1	5	3
Speckled Trout	0	0	5	3
Flatfish (Smooth Flounder, Winter Flounder)	2	0	4	3
Eels (American Eel, sea lamprey ammocoete)	1	0	4	3
Rainbow Smelt	0	0	3	2
Atlantic Tomcod	1	6	0	1
American Sand Lance	0	4	1	1
White Hake	1	6	0	1
Atlantic Salmon parr	0	1	0	0
Other fish (Atlantic Herring, Cunner, Greenland Cod, Pipefish)	0	3	1	1
Unidentified fish remains	5	3	7	6
Number of stomachs processed	85	72	310	467
% Empty	86	53	35	47

Table 4. The number of Atlantic Salmon smolts identified in Striped Bass stomach samples collected from the Miramichi River in May and June 2013 to 2015. The collection date, location, and method of capture for the individual Striped Bass containing Atlantic Salmon smolts are identified. The location refers to where the stomach samples containing salmon smolts were collected. "NW comm gasp" refers to samples from a commercial gaspereau trapnet in the Northwest Miramichi River. Striper cup refers to stomach samples extracted from incidental mortalities during the live release only Striped Bass fishing derby in the Miramichi on 30 and 31 May 2015. All other dates and locations are detailed in Table 1. Data are from DFO (2016).

Year	Date	Location	Stomach samples	Stomach samples with smolts	Total number of smolts identified	
					Angling samples	Trapnet samples
2013	10-May	Cassilis	15	1	0	1
	14-May	Beaubear's Island	25	1	1	0
	16-May	Beaubear's Island	20	1	1	0
	27-May	NW comm gasp	32	1	0	3
	28-May	Millstream	30	4	13	0
	29-May	NW comm gasp	30	1	0	1
	all other dates	all other locations	428	0	0	0
Sub total			580	9	15	5
2014	23-May	Beaubear's Island	21	2	2	0
	26-May	Hackett's Beach	13	4	4	0
	28-May	Strawberry Marsh	30	3	3	0
	2-Jun	Cassilis	64	2	3	0
	4-Jun	Cassilis	14	1	0	1
	5-Jun	Cassilis	31	1	1	0
	all other dates	all other locations	442	0	0	0
Sub total			615	13	13	1
2015	26-May	NW comm gasp	31	2	0	3
	28-May	Millstream	32	1	1	0
	28-May	NW comm gasp	30	2	0	8
	30-May	Striper cup	25	1	2	0
	all other dates	all other locations	531	0	0	0
Sub total			649	6	3	11
Grand total			1,844	28	31	17

Table 5. Summary of number of Atlantic Salmon smolts tagged with acoustic tags and released in the Northwest Miramichi and Southwest Miramichi rivers during 2013 to 2016 and the inferred percentages of the tagged smolts by river and year which had characteristic Striped Bass movements through the estuary areas of the Miramichi River. Data are from Daniels et al. (2018).

Year	River	Number of smolts tagged and released	Inferred % of tagged smolts showing Striped Bass movement patterns	Striped Bass spawner abundance estimates (median, 5th to 95th percentile; thousand)
2013	Northwest	40	19.4%	255 (67 to 864)
	Southwest	65	16.5%	
2014	Northwest	50	19.9%	138 (79 to 250)
	Southwest	80	9.0%	
2015	Northwest	80	9.2%	299 (146 to 675)
	Southwest	80	2.6%	
2016	Northwest	60	17.6%	318 (160 to 633)
	Southwest	59	8.2%	

Table 6. Cohort dynamics model equations of Atlantic Salmon juveniles to adult returns. The notations in bold identify the data.

$\boldsymbol{Psm_{r,t+1}} \sim \boldsymbol{Fry_{r,t}} * S_r^{fry}$

With $Fry_{r,t}$ the end of summer / fall index of fry (fish per 100 m²) for river r in year t,

S_r^{fry} the survival rate [0,1] of fry to small parr for river r (assumed similar over years but differing by river),

$Psm_{r,t+1}$ the end of summer / fall index of small parr (fish per 100 m²) for river r in year t+1,

$Sm_{r,t+1}^{a.2} \sim \boldsymbol{Psm_{r,t}} * S'^{psm}_{r,t} * pSm_r^{a.2} * \boldsymbol{Hab_r}$

With $Sm_{r,t}^{a.2}$ the total number of smolts at age 2 leaving river r in year t+1,

$Psm_{r,t}$ the small parr index (fish per 100 m²) for river r in year t,

S'^{psm}_{r} the survival rate of small parr from summer/fall to smolt in the spring, expressed as $e^{-Z/2}$ with $e^{-Z} = S_r^{psm}$ the annual survival from small parr to large parr,

$pSm_{r,t}^{a.2}$ the relative proportion of small parr that survived to the spring that become smolts age 2 in river r in year t,

Hab_r is the habitat area of river r (in 100 m²).

$\boldsymbol{Plg_{r,t+1}} \sim \boldsymbol{Psm_{r,t}} * S_r^{psm} * \left(1 - pSm_{r,t}^{a.2}\right)$

$Plg_{r,t+1}$ the large parr index in the summer / fall in river r in year t+1 from survivors of small parr in river r and year t that did not become smolts age 2, and other parameters as above.

$Sm_{r,t+1}^{a.3} \sim \boldsymbol{Plg_{r,t}} * S'^{psm}_{r,t} * \boldsymbol{Hab_r}$

With $Sm_{r,t+1}^{a.3}$ the total number of smolts at age 3 leaving the river r in year t+1, and other parameters as above.

$Sm_{r,t} = Sm_{r,t}^{a.2} + Sm_{r,t}^{a.3}$

With $Sm_{r,t}$ the total number of smolts (ages 2, 3) leaving freshwater in river r in year t

$\boldsymbol{A.1sw_{r,t+1}} \sim Sm_{r,t} * S1sw_{r,t} * p1sw_r$

With $A.1sw_{r,t+1}$ the abundance of 1SW returning to river r in year t+1;

$Sm_{r,t}$ the total number of smolts (ages 2, 3) leaving freshwater in river r in year t;

$S1sw_{r,t}$ the marine survival rate from smolt to 1SW from river r and migration year t (model dependent)

$p1sw_r$ the probability of maturing to 1SW for smolts from river r.

$\mathbf{A.}\boldsymbol{2sw_{r,t+2}} \sim Sm_{r,t} * S1sw_{r,t} * \left(1 - p1sw_r\right) * S2sw_r$

With $A.2sw_{r,t+2}$ the abundance of 2SW salmon returning to river r in year t+2;

$S2sw_r$ the survival rate of a 1SW non-maturing in the second year at sea for river r, and other parameters as above.

Simplifying assumptions

- S_r^{psm} differs by river r but is similar over years;
- $S'^{psm}_{r,t}$ Large parr to smolt age 3 survival is similar to small parr survival, but for half the year ();
- $pSm_{r,t}^{a.2}$, the proportion of small parr that become smolt age 2 is either similar for the rivers but annually variable or annually variable and different between rivers (dependent upon model);
- $S2sw$ the sea survival rate in the second year at sea is assumed known and similar for rivers and years. Because there are three marine dynamics parameters ($S1sw, p1sw, S2sw$) but only two observations (returns of 1SW and 2SW), one of the parameters must be fixed. Following the modelling approach of ICES (2020), the sea survival in the second year is fixed at 0.70 ($e^{-0.03*12}$), assumed an instantaneous monthly mortality rate of 0.03 per month for 12 months of life at sea.

Indices:

- r in 1:2 (Southwest Miramichi, Northwest Miramichi)
- y in 1993 to 2019

Table 7. Parameters and their prior distributions for the Atlantic Salmon cohort model.

Parameter	Prior
S_r^{fry}	Beta(2,3)
S_r^{psm}	Beta(3,2)
$pSm_{r,t}^{a.2}$	Beta(5,5)
$Log(u.fry_{y,r})$	N(2,10) C(1,)
$S1sw_{y,r} = e^{-Z_{r,t}}$	Z ~ N(1,10) C(0.1,)
$p1sw_r$	Beta(5,5)
$S2sw$	Beta(72,28)
Precision ($1/\sigma^2$)	σ ~ U(0,5)

Table 8. Summary of the Atlantic Salmon cohort model structures and diagnostics.

Model	Effective number of parameters	DIC value	Comment
1	67	2719	DIC values of this model cannot be compared to model 2 and model 3 as the estimates between rivers are fully independent. The fits to juvenile indices and to adult returns are better than models 2 and 3. Temporal trend in residuals of juveniles and adult returns
2	80	2700	Some misfitting of juvenile indices, temporal trend in residuals of juveniles and 1SW returns. Some systematic misfitting of 1SW returns to both rivers but not as severe as in model 3.
3	73	2677	Reasonable fit to juvenile indices. Some temporal trends in residuals of juveniles and 1SW returns but fewer than in model 2. Systematic misfitting of 1SW returns for the Southwest Miramichi.

FIGURES

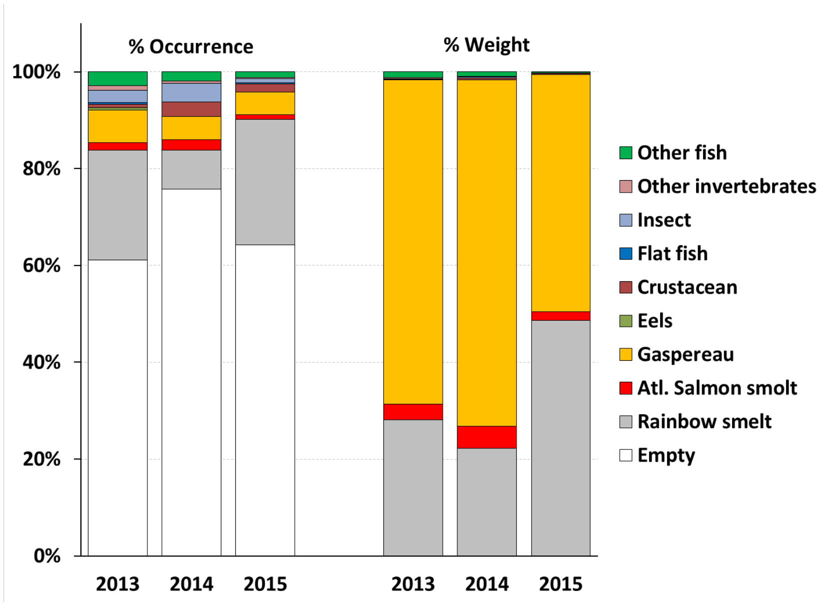

Figure 1. Summary of the percent occurrence of prey species or prey categories (left) and their corresponding percent weight (right) in the stomachs of Striped Bass collected from the Miramichi River in May and June, 2013 to 2015. The figure is copied directly from DFO (2016).

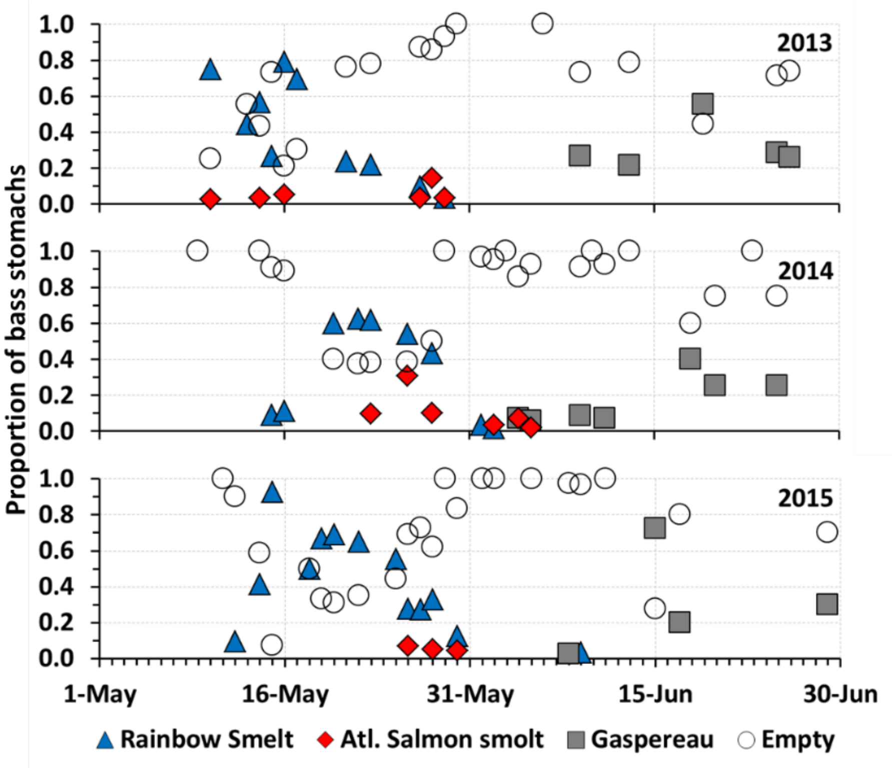

Figure 2. The proportion of Striped Bass stomachs by date that were empty, contained Rainbow Smelt, contained Atlantic Salmon smolt, and/or gaspereau in the May and June period from the Miramichi River in 2013 to 2015. Only dates for which ≥3 stomach samples were collected are shown. Proportions may not add to one because not all prey groups are shown or multiple prey groups may occur in one sample. The figure is copied directly from DFO (2016).

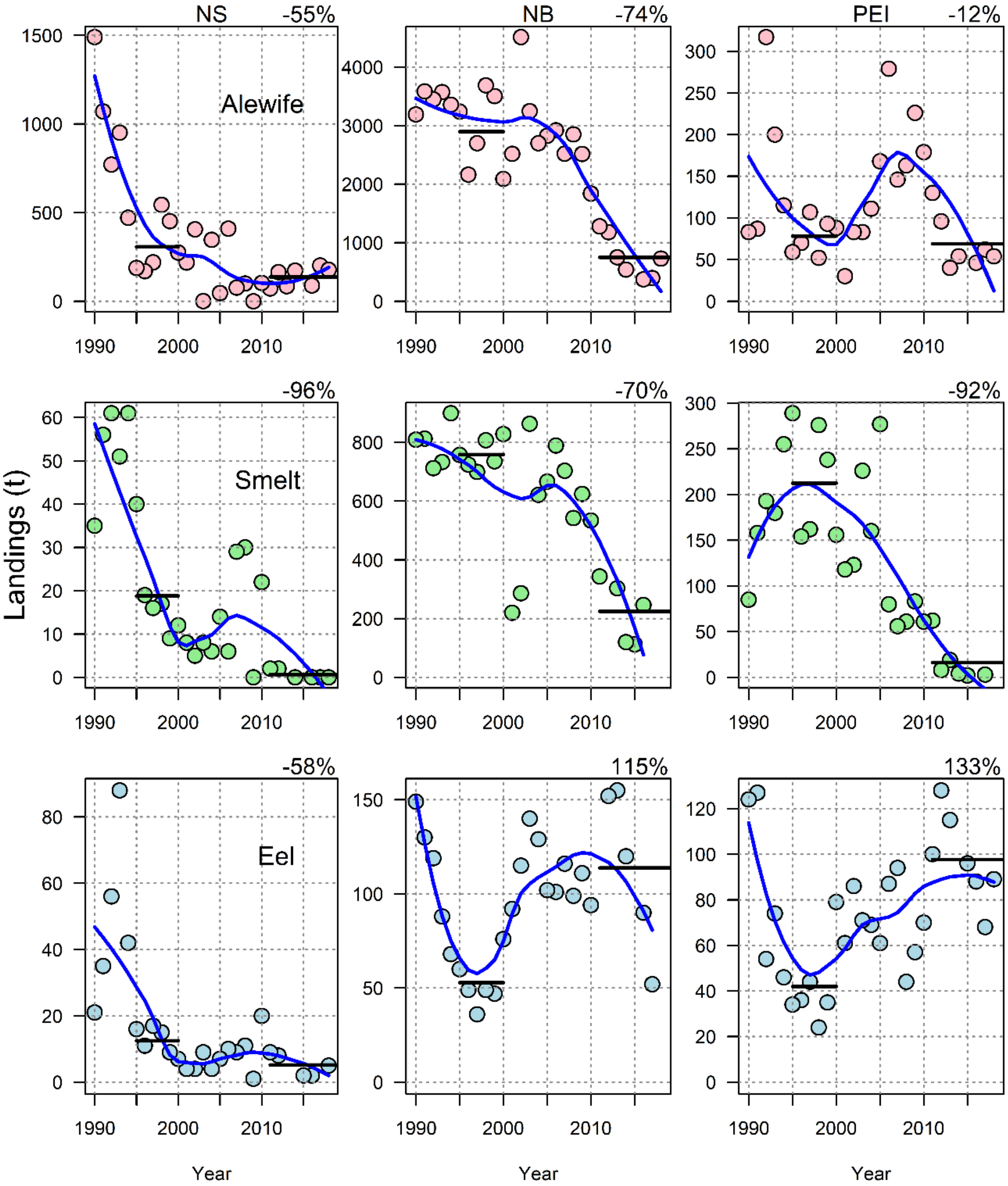

Figure 3. Summary of recorded landings (t) of three diadromous fish species (gaspereau = Alewife and Blueback Herring, smelt = Rainbow Smelt; Eel = American Eel) by province within DFO Gulf Region, 1990 to 2018. Some data are missing due to confidentiality restrictions. A value of 0 represents a landings record < 0.5 t. The blue line in each plot is a loess smoother using a span value of 0.8. The mean landings for the periods 1995 to 2000 and 2011 to 2018 are shown as black horizontal lines and the percent change of the 2011 to 2018 period relative to the 1995 to 2000 period is shown in the top right above each panel.

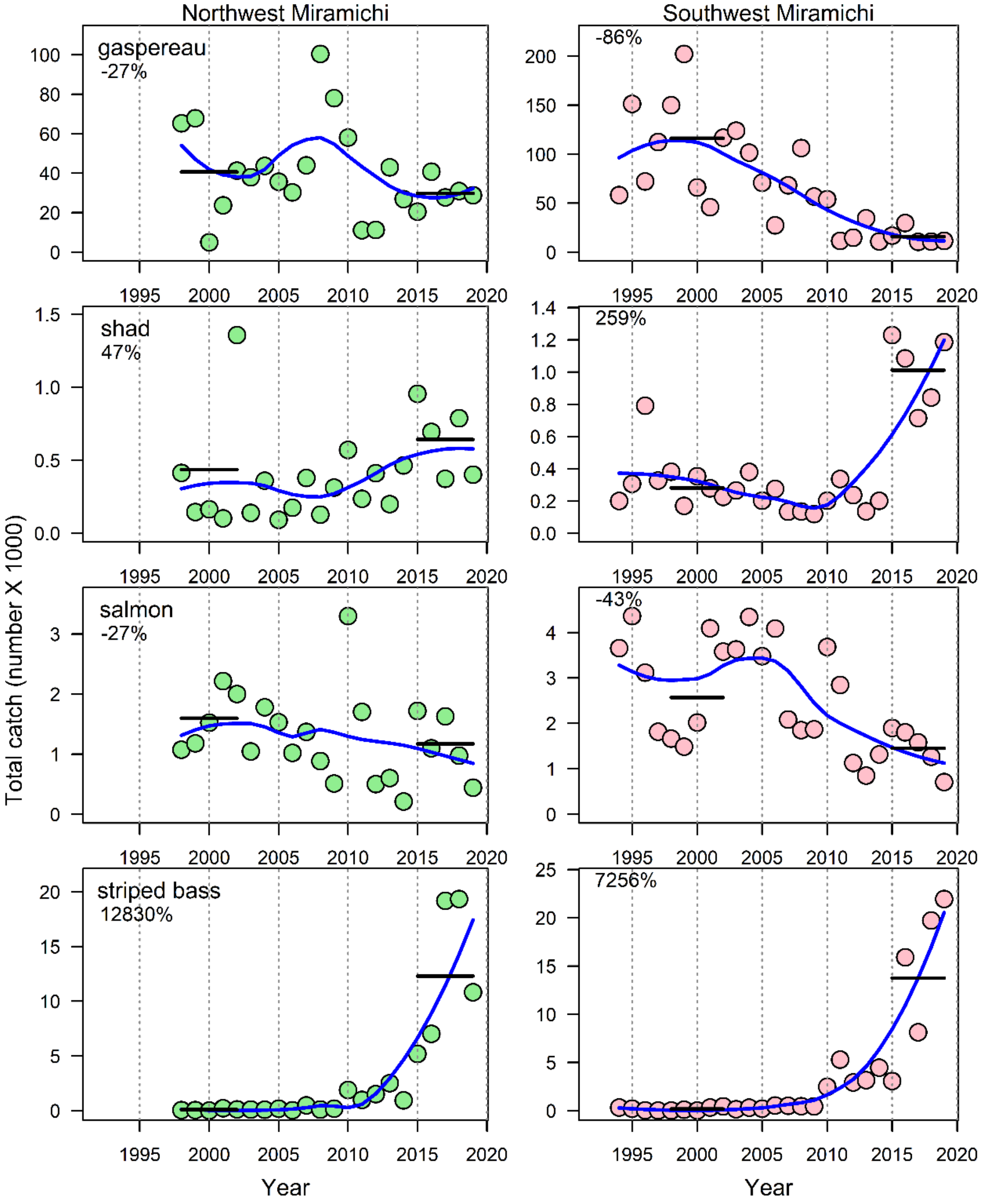

Figure 4. Summary of total catches (number, thousands) of diadromous fish species (gaspereau = Alosa pseudoharengus and A. aestiavalis, top row; shad = A. sapidissima, second row; salmon = Salmo salar (adults), third row; striped bass = Morone saxatilis, fourth row) at the DFO index estuary trapnets in the Northwest Miramichi (left column) and the Southwest Miramichi (right column), 1994 (1998 for Northwest Miramichi) to 2019. Total catches are not corrected for dates of operation which can vary between years and between trapnets. The blue line in each plot is a loess smoother using a span value of 0.8. The mean catches for the periods 1998 to 2012 and 2015 to 2019 are shown as black horizontal lines and the percent change of the 2015 to 2019 period relative to the 1998 to 2002 period is shown in the top left corner of each panel.

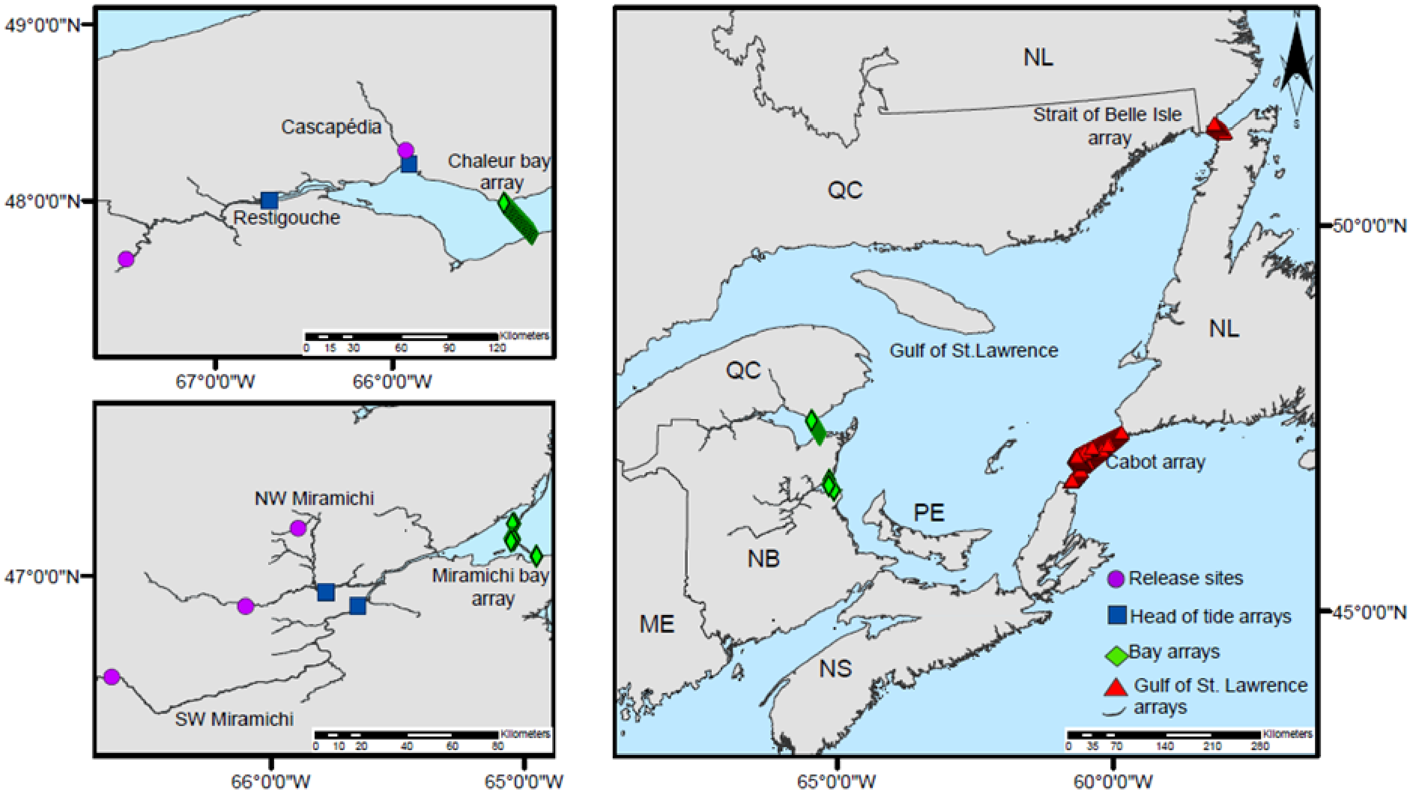

Figure 5. Geographic references of the Atlantic Salmon smolt tagging and tracking experiment conducted by the Atlantic Salmon Federation in four rivers of the southern Gulf of St. Lawrence. The release locations by study river, the head of tide receiver locations, and the respective bay receiver lines are shown. The right panel shows the bay exits and the Strait of Belle Isle receiver array. The figure is from Chaput et al. (2018).

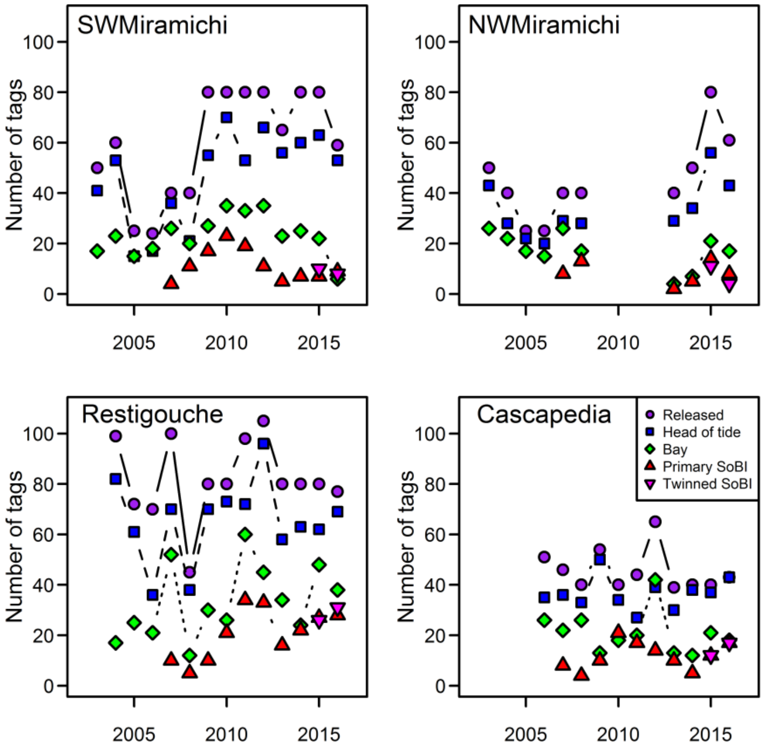

Figure 6. Summary of the number of tagged fish released and number of tags detected at the respective receiver lines for four rivers during 2003 to 2016. The figure is from Chaput et al. (2018).

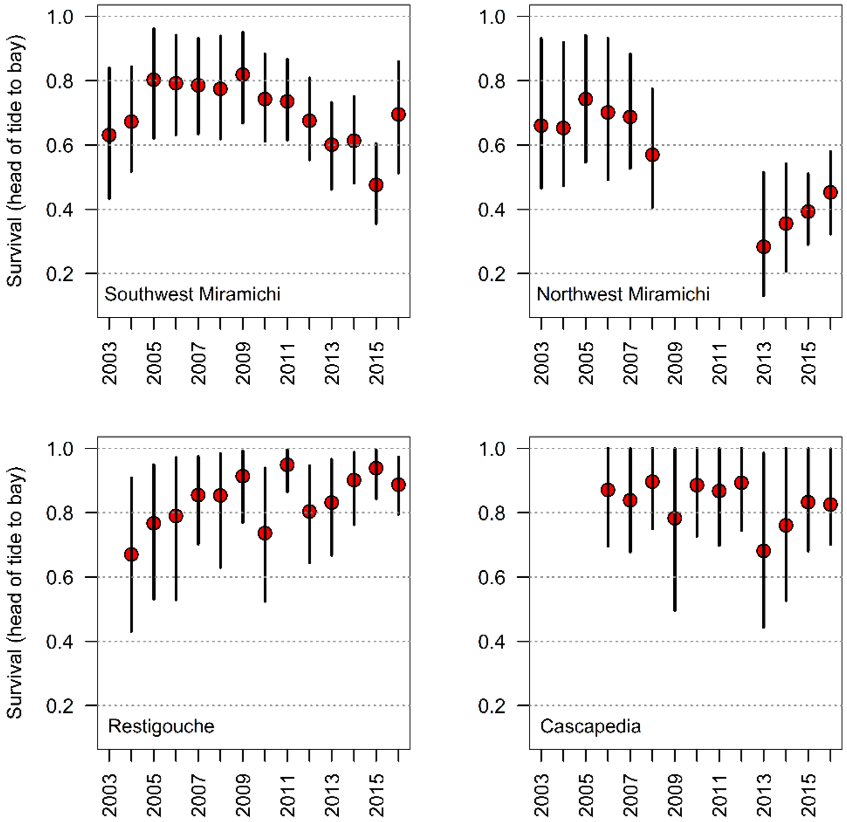

Figure 7. Posterior distributions (median, 5th to 95th percentile range) of estimated survival rates through the bay (head of tide to bay exit, Miramichi in upper row, Chaleur in lower row) of acoustically tagged smolts from four rivers of the southern Gulf of St. Lawrence. The survival rates are for a smolt of centered length 14.6 cm fork length. Data are summarized from the study reported by Chaput et al. (2018).

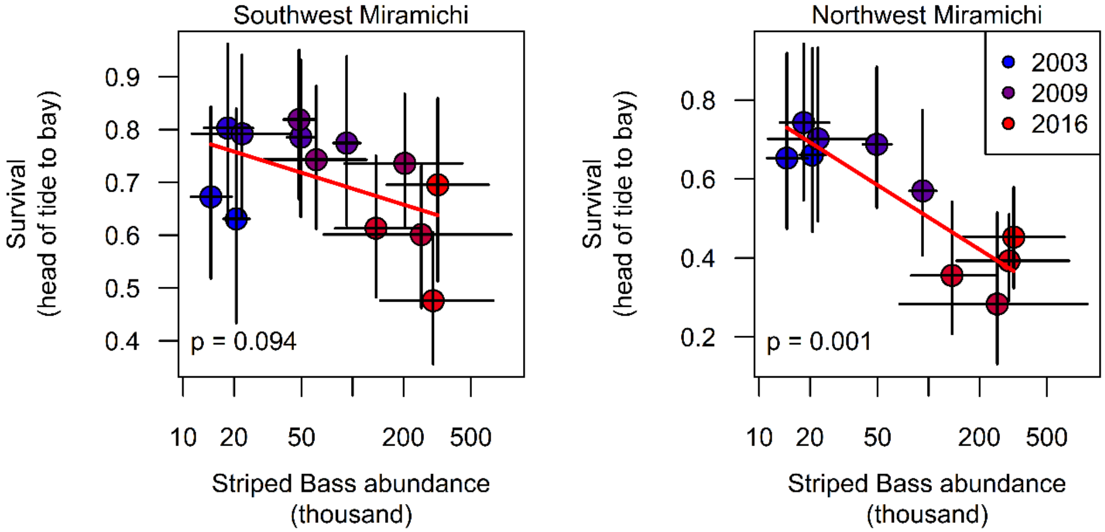

Figure 8. Correlations between the estimated survival rates from head of tide to bay exit of acoustically tagged smolts (Southwest Miramichi left panel, Northwest Miramichi right panel) and the estimated spawner abundance of Striped Bass (log scale) in the Miramichi River, 2003 to 2016. For both the survival rates and spawner abundance values, the symbol is the median and the black lines are the respective 5th to 95th percentile range of the estimates. The linear relationship (red line) and the corresponding p-value of the slope of the regression = 0 is shown in the lower left corner of each panel. Survival rate data are from the study results of Chaput et al. (2018)mfork length. The survival rates are for a smolt of centered length 14.6 cm fork length. Data are summarized from the study reported by Chaput et al. (2018).

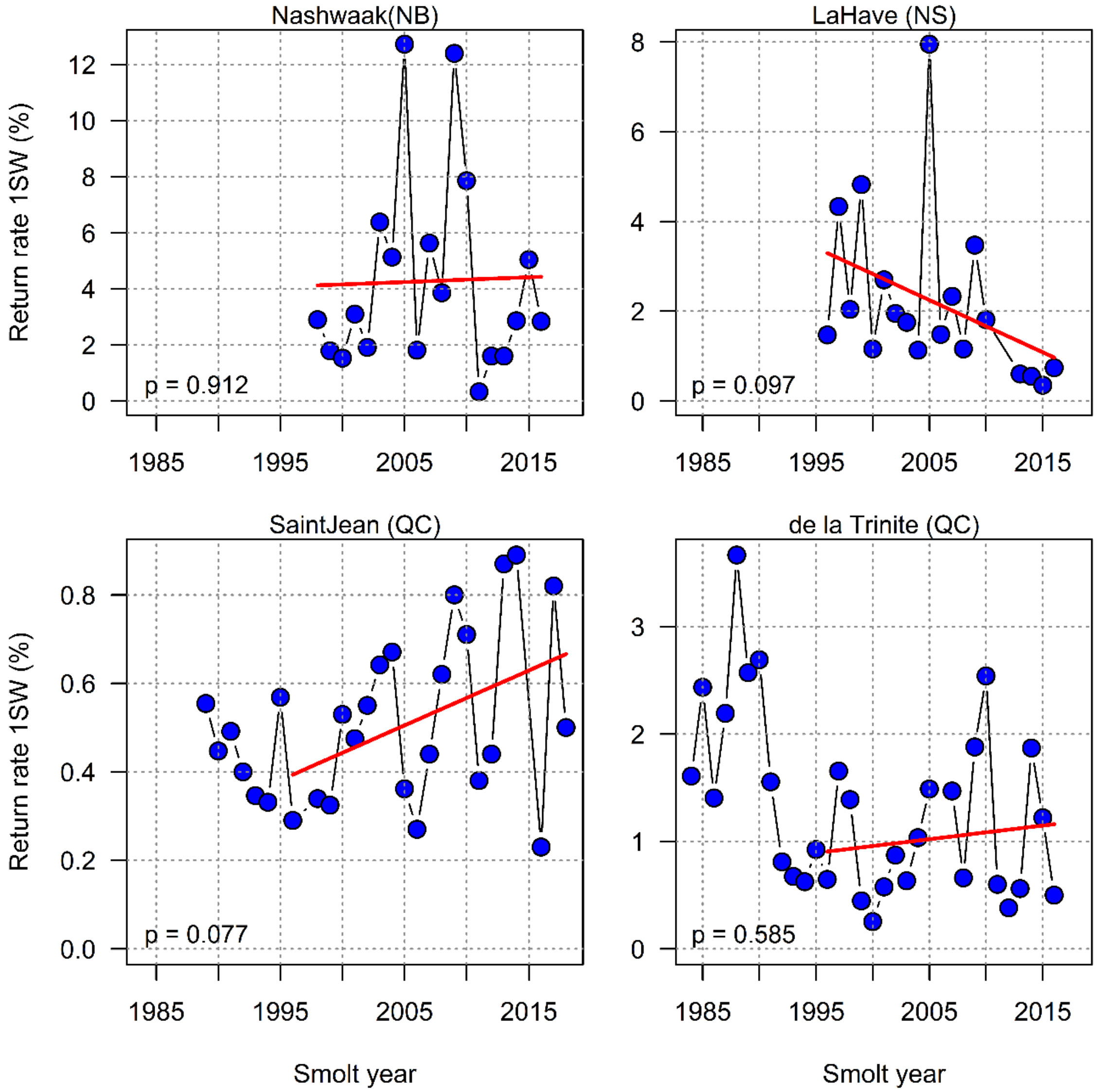

Figure 9. Point estimates of the return rate (%) by year of smolt migration of 1SW Atlantic Salmon to four index rivers of eastern Canada. The red line shows the linear fit to the time series (1996 to present) and the p-value for the null hypothesis of the slope = 0 is shown in the lower left corner of each panel.

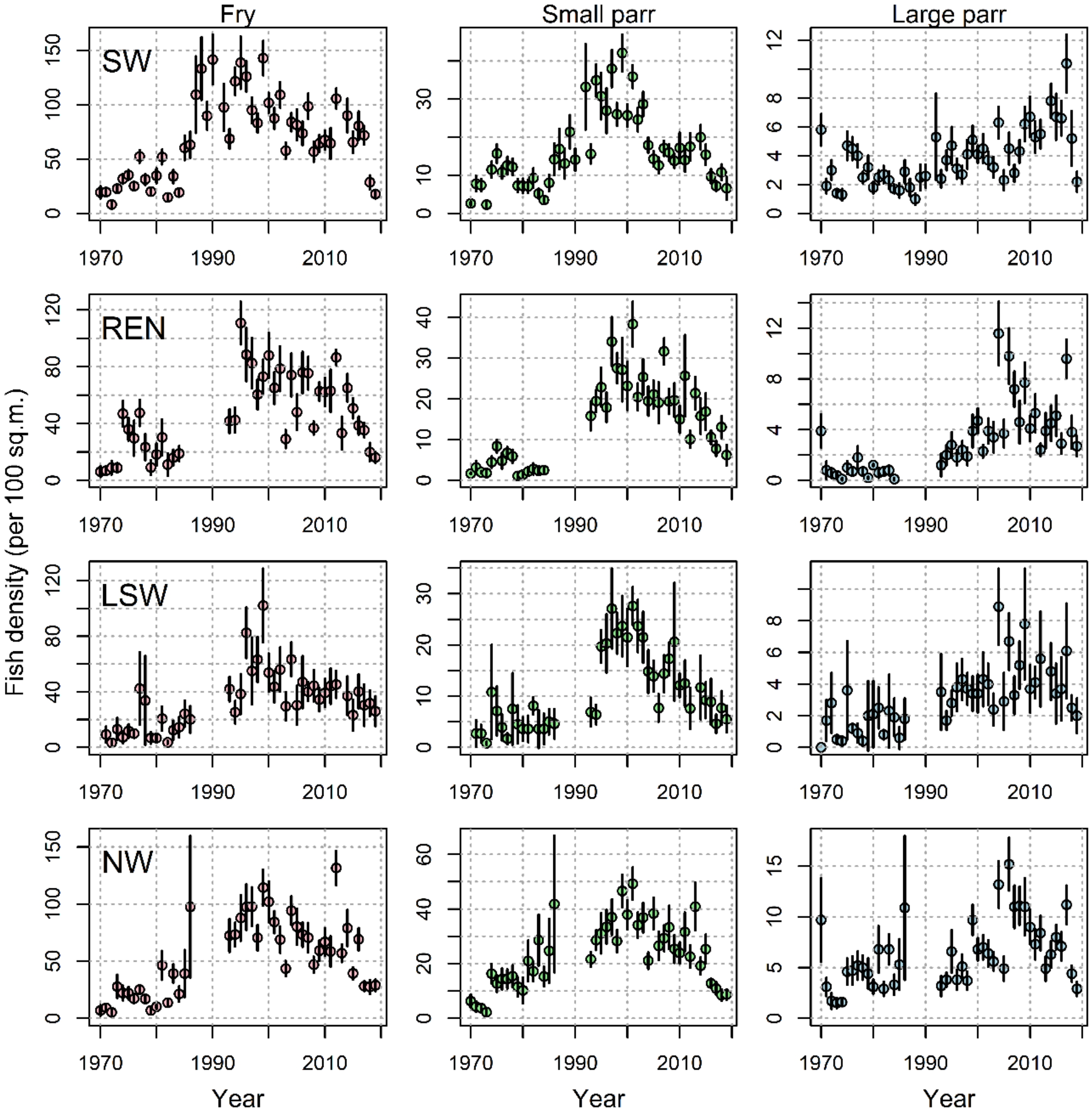

Figure 10. Time series of juvenile Atlantic Salmon indices (fish per 100 m²; mean and one standard deviation error bar) by life stage (columns) in the four main tributaries (rows; SW = Southwest Miramichi; REN = Renous; LSW = Little Southwest Miramichi; NW = Northwest Miramichi) for the years 1970 to 2019. Only the years in which at least four sites were sampled within each of the tributaries are shown. Figure is available in DFO (2020b).

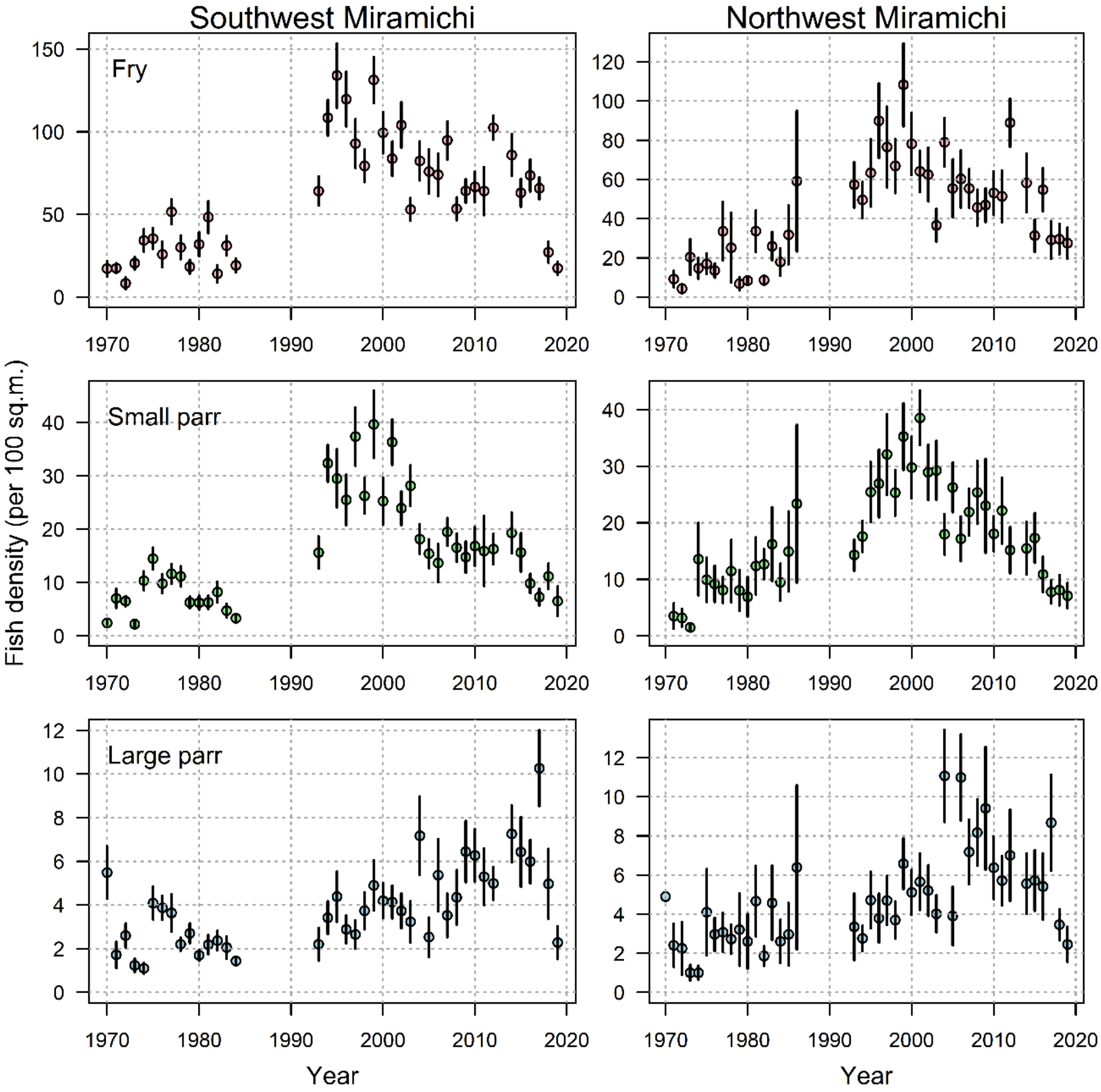

Figure 11. Habitat weighted juvenile Atlantic Salmon abundance indices (fish per 100 m²; mean and one standard deviation error bar) by life stage (rows) in the two main branches of the Miramichi River (columns), 1970 to 2019. The standard deviations for the main branches are calculated as the mean of the standard deviations from the tributary estimates shown in Figure 10.

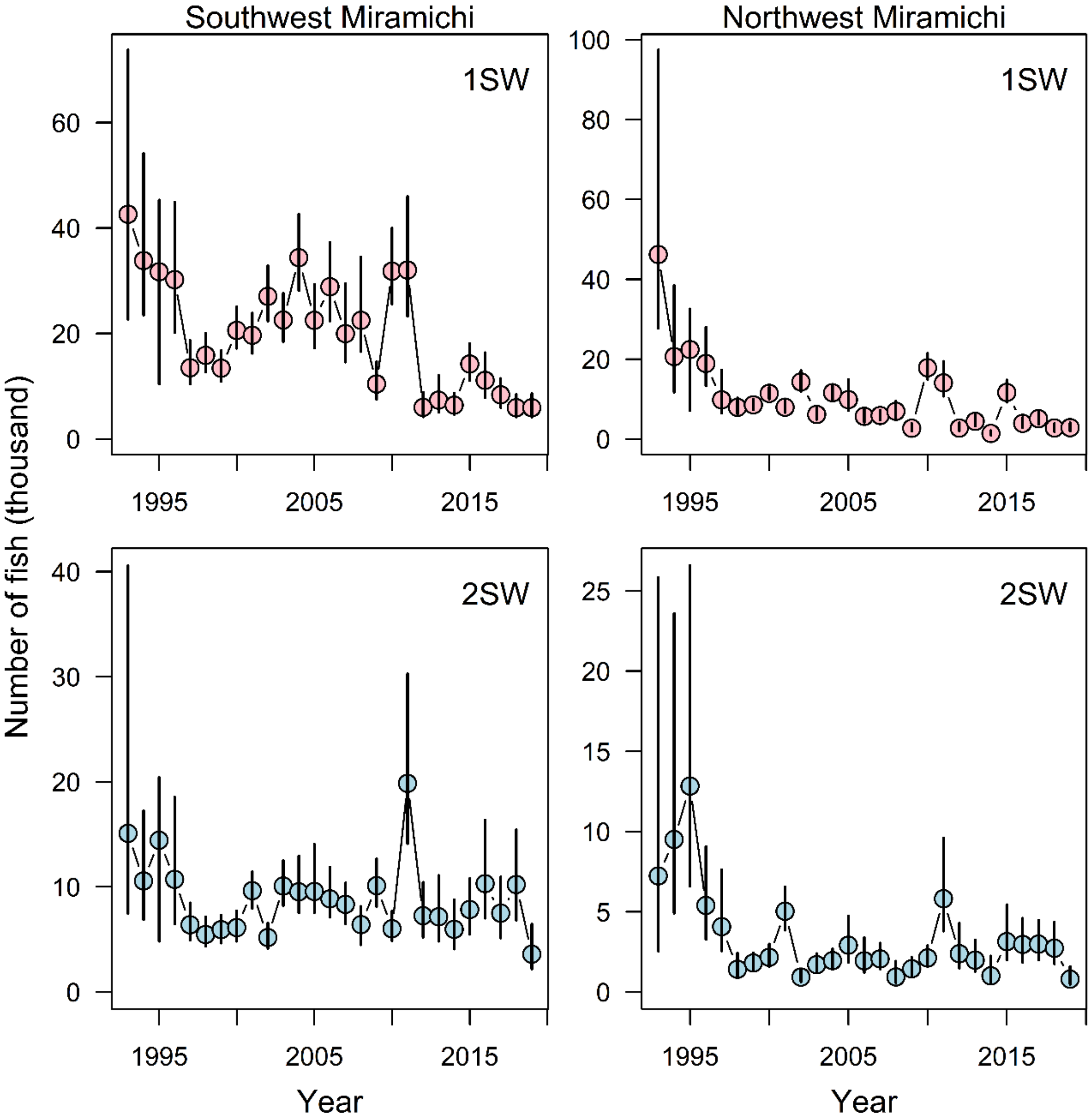

Figure 12. Estimated returns (before inriver fisheries; median and the 5th to 95th percentile range) of Atlantic Salmon by sea age group (1SW upper row, 2SW bottom row) to the Southwest Miramichi (left column) and the Northwest Miramichi (right column) for the assessment period 1993 to 2019.

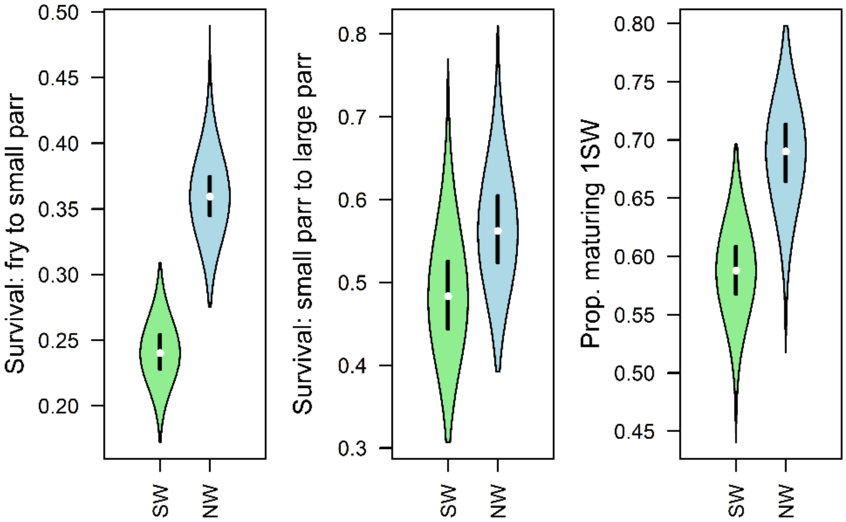

Figure 13. River specific posterior distributions of parameters that were common across years but differed between rivers.

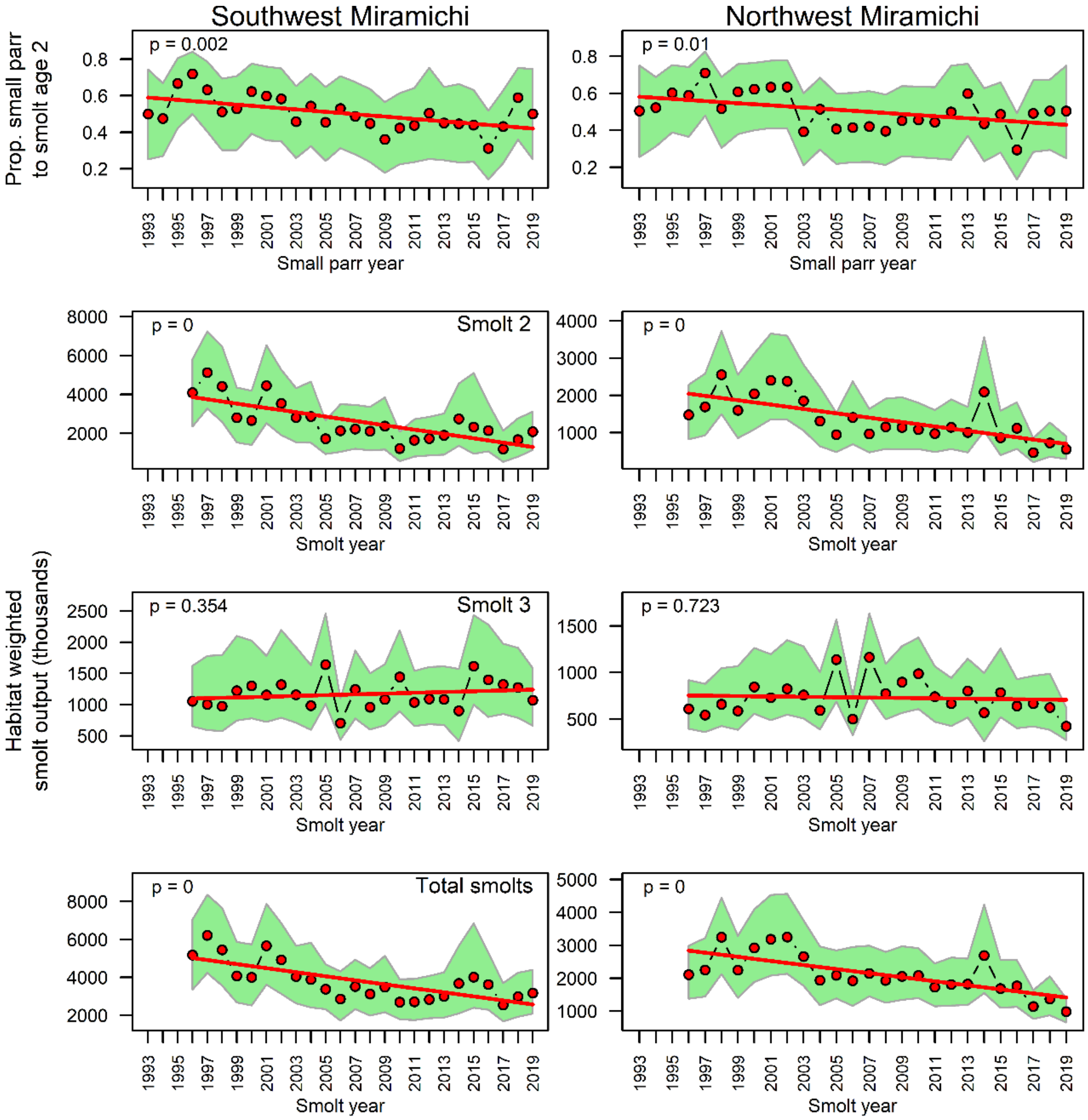

Figure 14. Posterior distributions of the annual proportions of small parr that smolt at age 2, by river (upper row) and estimated relative smolt production at age 2, age 3 and total smolts by year and river (second to fourth rows) for the Southwest and Northwest Miramichi. The red symbols and shaded polygons are the median and the 5th to 95th percentile range of the posterior distribution. The solid horizontal line is the linear regression of the medians and the p-value for the null hypothesis of the slope = 0 is shown in the lower left corner of each panel.

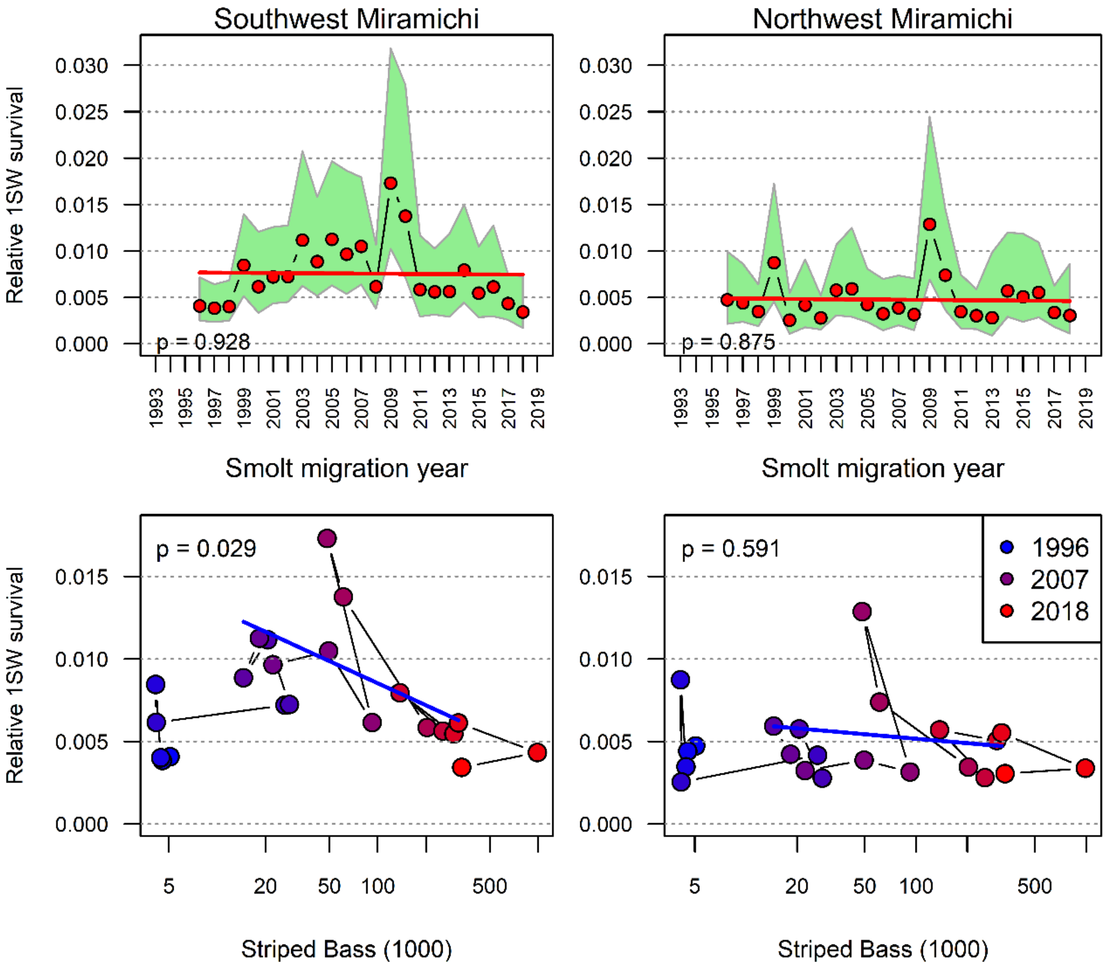

Figure 15. Posterior distributions of the relative survival rates from smolt to 1SW maiden returns to the Southwest (left column) and Northwest (right column) Miramichi rivers for the smolt migration years 1996 to 2018 (top row). The bottom row shows the relative smolt to 1SW survival rates by river plotted against the estimated (log scale) Striped Bass spawner abundances in the Miramichi River for the smolt and Striped Bass spawning years 1996 to 2018. The solid blue line is the linear regression of relative survival rates to log of Striped Bass abundances for the 2003 to 2016 years corresponding to the acoustic tagged smolt survival time series of the Miramichi River (see Figures 7 and 8).

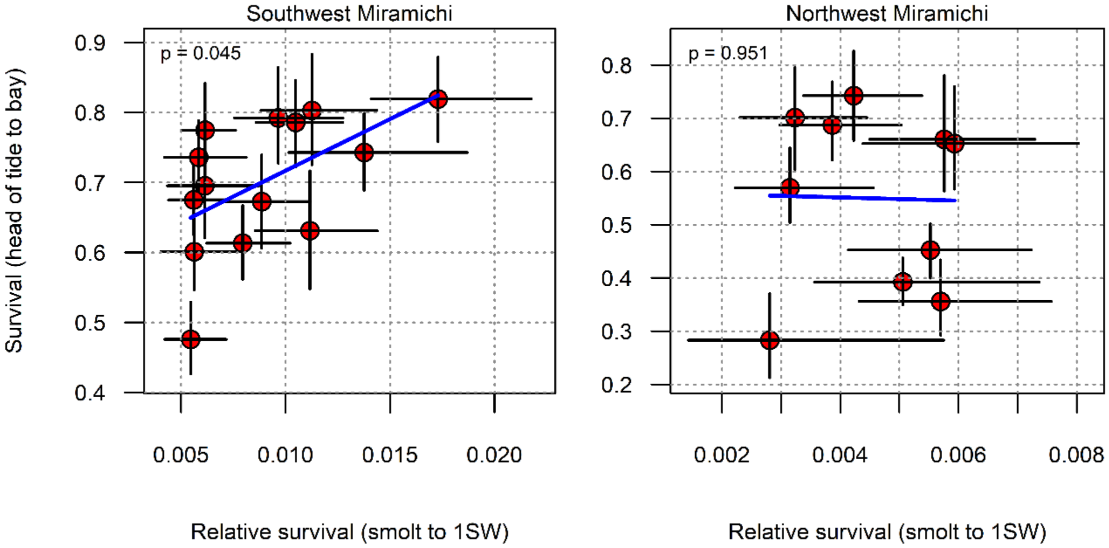

Figure 16. Scatter plot of the posterior distribution (medians) of the relative survival rates from smolt to 1SW maiden returns and the estimated survival rates through the bays of acoustically tagged smolts for the Southwest (left panel; 2003 to 2016 smolt years) and Northwest (right panel; 2003 to 2008, 2013 to 2016 smolt year). The error bars are the 25th to 75th percentile ranges of the posterior distributions. The blue line is the linear relationship and the p-value of the null hypothesis of slope = 0 is shown in the upper left corner of each panel.

APPENDICES

APPENDIX 1. OPENBUGS MODEL CODE OF THE COHORT MODEL RUNNING INDEPENDENT ESTIMATES BY RIVER

```
# habitat area in 100's of square meters to translate parr to similar scale as maiden adult returns
# data variable names
# Parr.sm = small parr index (fish per 100 sq. m.)
# Parr.lg = large parr index (fish per 100 sq. m.)
# N.1sw = number of maiden 1SW adult returns
# N.2sw = number of maiden 2SW adult returns
# cv.fry is the mean coefficient of variation of the annual fry index from sampling by river
# S.fry is the fry to small parr survival rate
# S.psm is the small parr to large parr survival rate
# p.Sm2 is probability small parr becomes smolt age 2
# p.1sw probability smolt matures at 1sw
# S.1sw  survival rate for 1sw year
# S.2sw survival rate in the second year at sea = exp(-0.03*12)
model {
# priors for life history parameters
for (r in 1:2){
  tau.fry[r] <- pow(cv.fry[r],-2)
  S.fry[r] ~ dbeta(2,3)
  Z.fry[r] <- -log(S.fry[r])
  S.psm[r] ~ dbeta(3,2)
  Z.psm[r] <- -log(S.psm[r])
  p.1sw[r] ~ dbeta(5,5)  # prob maturing at 1SW
  S.2sw[r] ~ dbeta(72,28) # single essentially fixed parameter
  tau.parr.sm[r] <- pow(sig.psm[r],-2)
  sig.psm[r] ~ dunif(0,5)
  tau.parr.lg[r] <- pow(sig.plg[r],-2)
  sig.plg[r] ~ dunif(0,5)
  tau.1sw[r] <- pow(sig.1sw[r],-2)
  sig.1sw[r] ~ dunif(0,5)
  tau.2sw[r] <- pow(sig.2sw[r],-2)
  sig.2sw[r] ~ dunif(0,5)
   }  # end river loop
for (y in 1:Y){
  for (r in 1:2){
     log.u.fry[y,r] ~ dnorm(2,0.01) C(1, )
     p.Sm2[y,r] ~ dbeta(5,5)   # prop small parr becomes smolt age 2, by river
     Z.1sw[y,r] ~ dnorm(0.1,0.01) C(0.1, )
     S.1sw[y,r] <- exp(-Z.1sw[y,r])
     }  # end river loop
  } # end year loop
# life history dynamic equations for fry and parr
for (r in 1:2){  #begin river loop
# year loop for fry
 for (y in 1:Y){
    fry[y,r] ~ dlnorm(log.u.fry[y,r], tau.fry[r])
    log(u.fry[y,r]) <- log.u.fry[y,r]
    res.fry[y,r] <- log(fry[y,r]/u.fry[y,r])
    }  # end year loop for fry
# year loop for small parr
   for (y in 2:Y){
     parr.sm[y,r] ~ dlnorm(log.u.psm[y,r], tau.parr.sm[r])
     log(u.parr.sm[y,r]) <- log.u.psm[y,r]
     log.u.psm[y,r] <- log.u.fry[y-1,r] + log(S.fry[r])
     res.parr.sm[y,r] <- log(parr.sm[y,r]/u.parr.sm[y,r])
     tot.parr.sm[y,r] <- u.parr.sm[y,r] * hab[r]
    } # end year loop for small parr
```

```
# year loop for large parr
  for (y in 3:Y){
    parr.lg[y,r] ~ dlnorm(log.u.plg[y,r], tau.parr.lg[r])
    log(u.parr.lg[y,r]) <- log.u.plg[y,r]
    log.u.plg[y,r] <- log.u.psm[y-1,r] + log(S.psm[r] * (1-p.Sm2[y-1,r]))
    res.parr.lg[y,r] <- log(parr.lg[y,r]/u.parr.lg[y,r])
    tot.parr.lg[y,r] <- u.parr.lg[y,r] * hab[r]
    } # end year loop
# smolts
  for (y in 4:Y){
     Sm.2[y,r] <- tot.parr.sm[y-1,r] * exp(-Z.psm[r]/2) * p.Sm2[y-1,r]
     Sm.3[y,r] <- tot.parr.lg[y-1,r] * exp(-Z.psm[r]/2)
     Sm[y,r] <- Sm.2[y,r] + Sm.3[y,r]
   }  # end year loop
# 1sw loop
  for (y in 5:Y){
    N.1sw[y,r] ~ dlnorm(log.u.1sw[y,r], tau.1sw[r])
    log(u.1sw[y,r]) <- log.u.1sw[y,r]
    log.u.1sw[y,r] <- log(Sm[y-1,r]) + log(S.1sw[y-1,r]) + log(p.1sw[r])
    res.1sw[y,r] <- log(N.1sw[y,r]/u.1sw[y,r])
   }  # end year loop
# 2sw loop
  for (y in 6:Y){
    N.2sw[y,r] ~ dlnorm(log.u.2sw[y,r], tau.2sw[r])
    log(u.2sw[y,r]) <- log.u.2sw[y,r]
    log.u.2sw[y,r] <- log(Sm[y-2,r]) + log(S.1sw[y-2,r]) + log(1-p.1sw[r]) + log(S.2sw[r])
    res.2sw[y,r] <- log(N.2sw[y,r]/u.2sw[y,r])
   }  # end year loop
} # end river loop
}  # end model
```

APPENDIX 2. DIAGNOSTICS OF COHORT MODEL 1

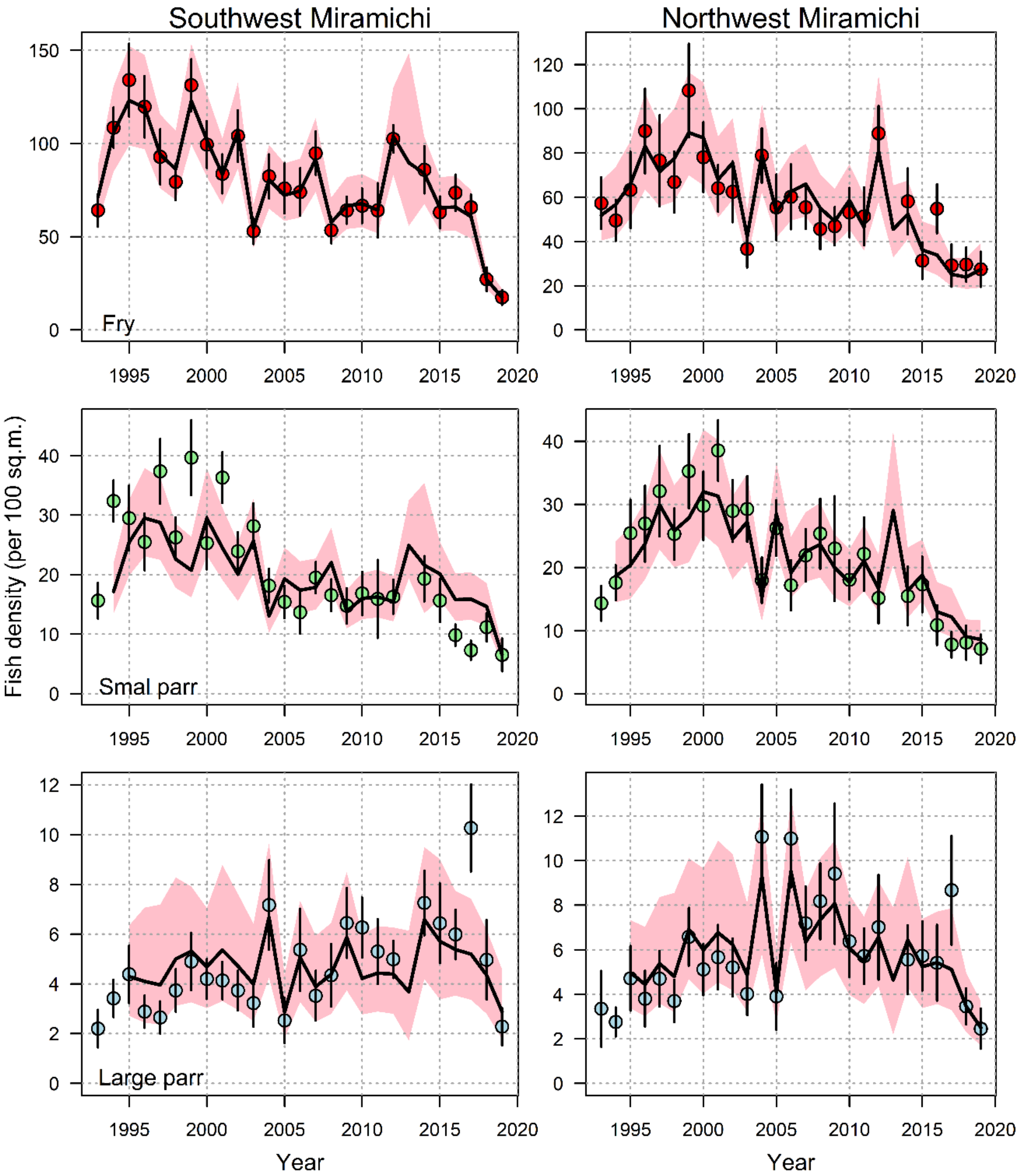

Figure A2.1. Fits to juvenile indices. The symbols and vertical bars are the mean and one standard deviation distributions from sampling. The black line and the shaded polygon are the median and 5th to 95th percentile range from the posterior distributions.

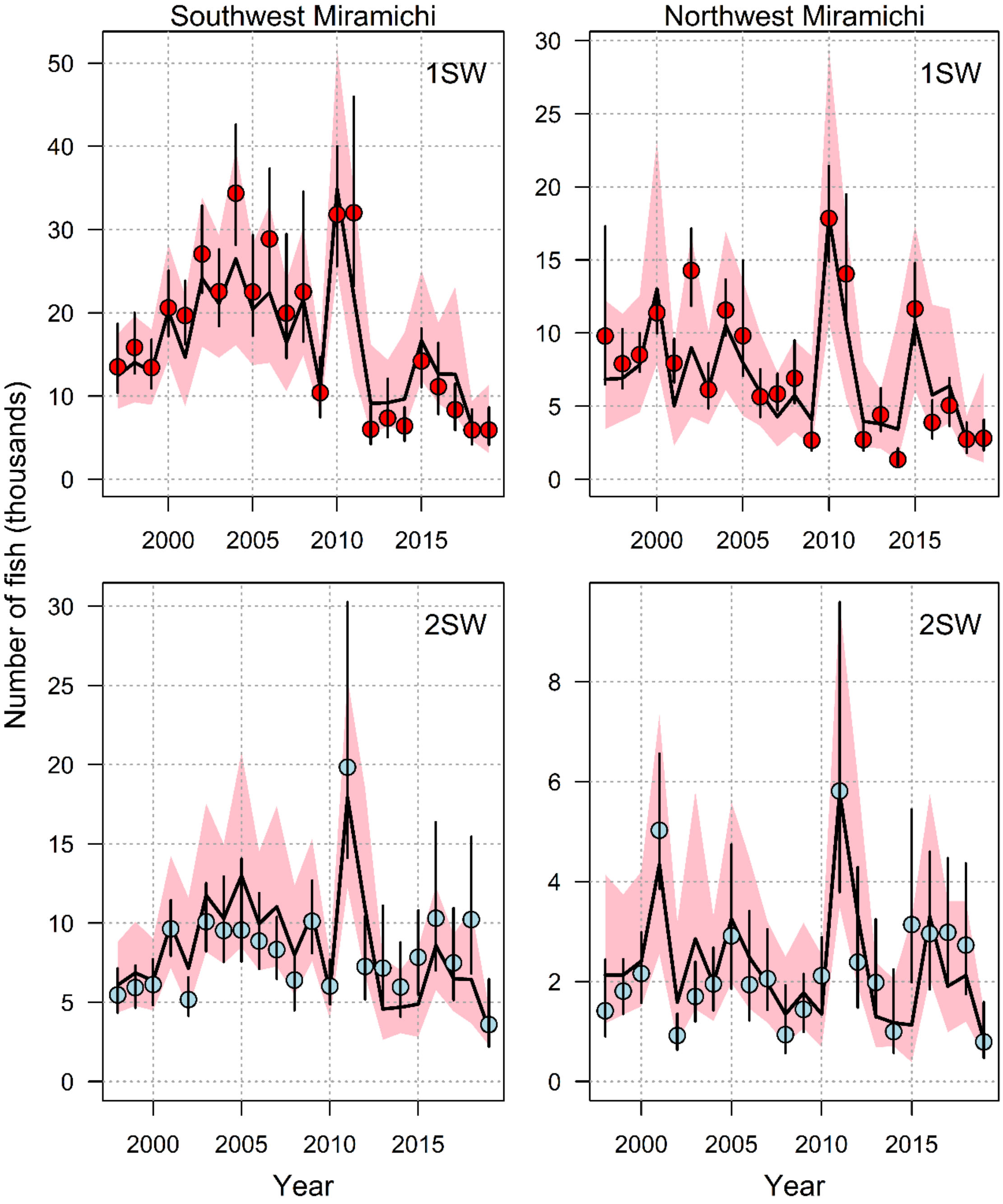

Figure A2.2. Fits to 1SW and 2SW return estimates in the Southwest Miramichi (left column) and Northwest Miramichi (right column). The symbols and vertical bars are the mean and one standard deviation distributions from sampling. The black line and the shaded polygon are the median and 5th to 95th percentile range from the posterior distributions.

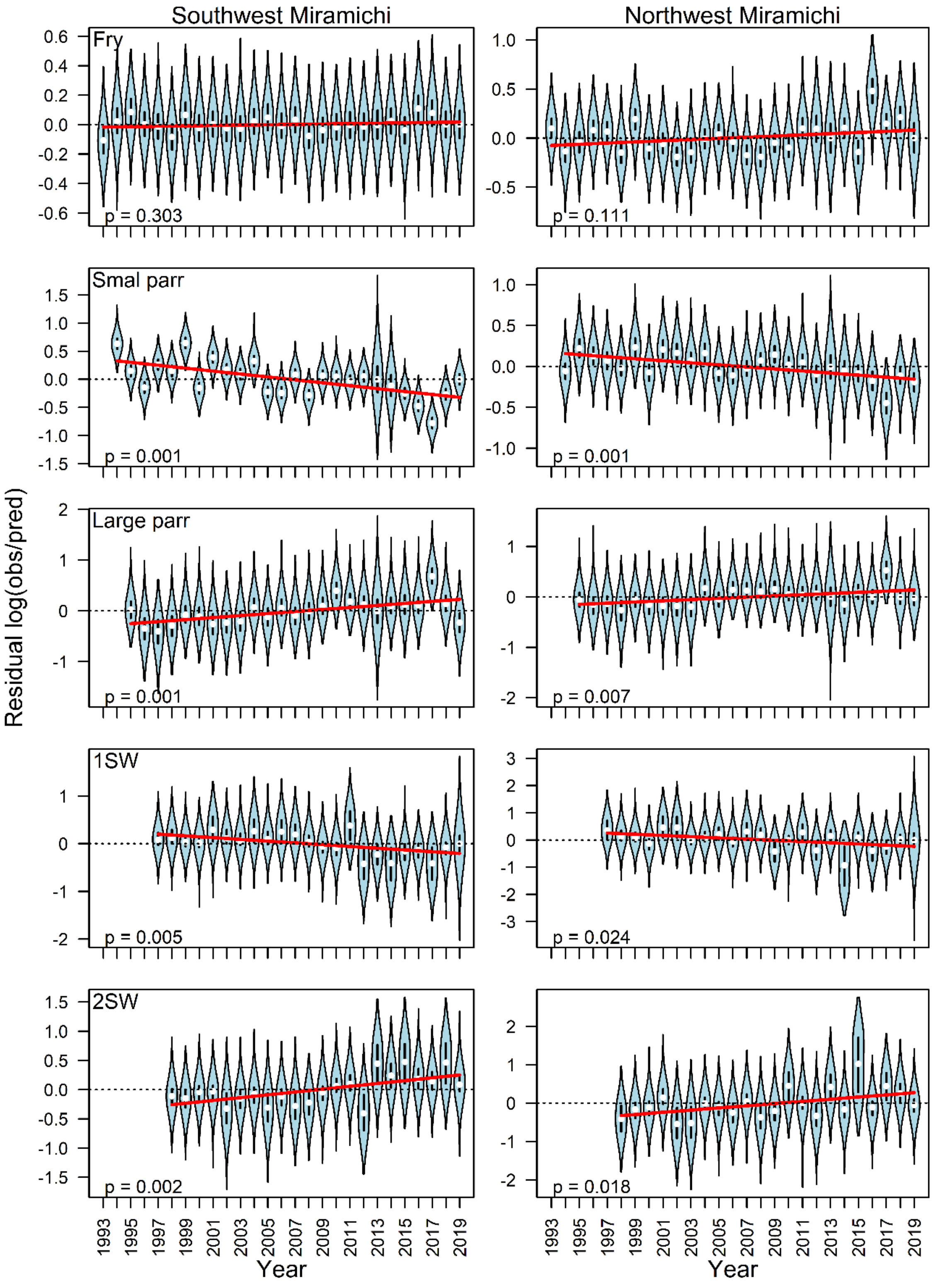

Figure A2.3. Residuals from the fits of model 1. The solid red line is the linear trend of the median of the posterior distribution of the annual residual and the p-value for the null hypothesis of the slope = 0 is shown in the lower left corner of each panel.

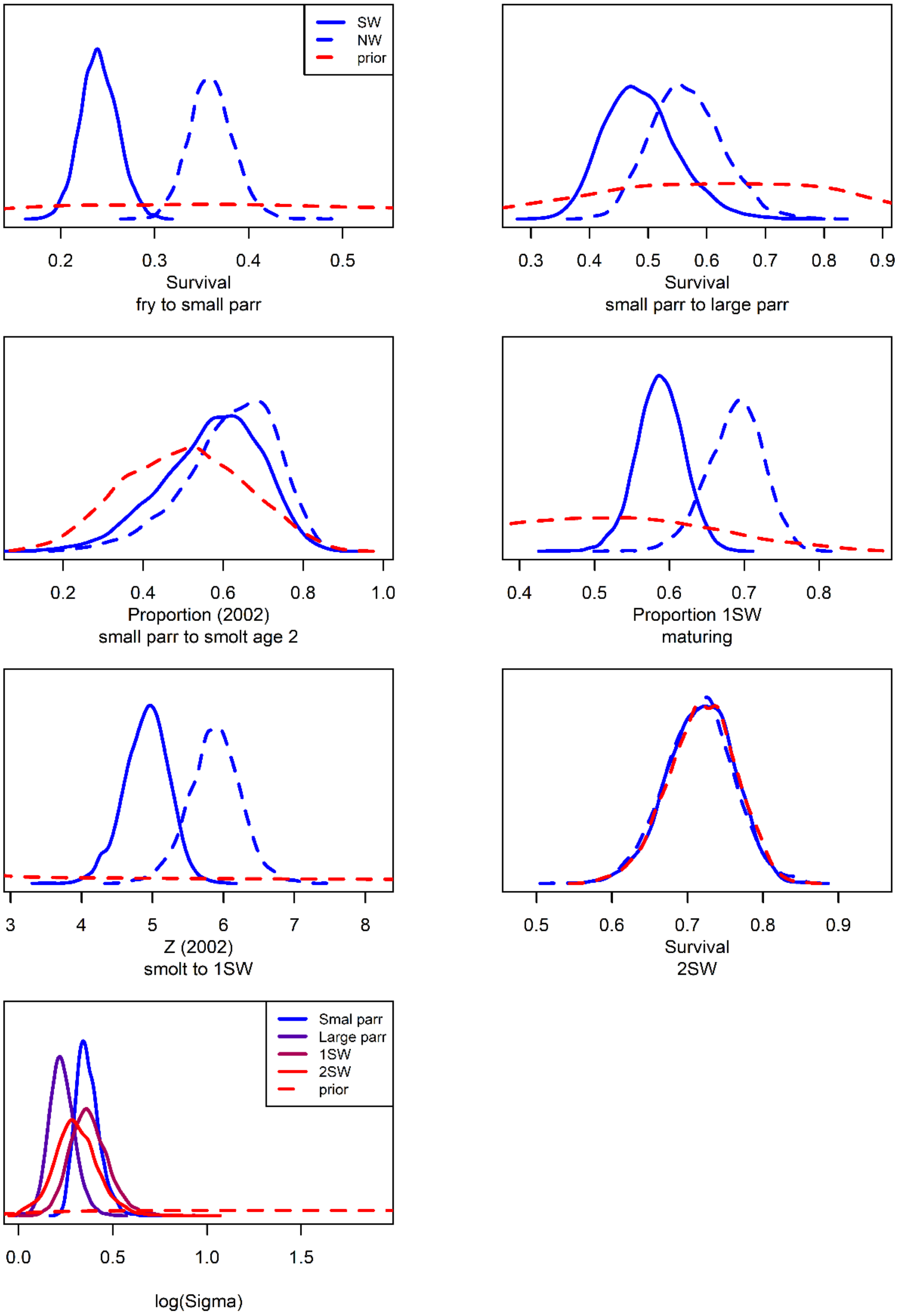

Figure A2.4. Prior versus posterior distributions of the model parameters.

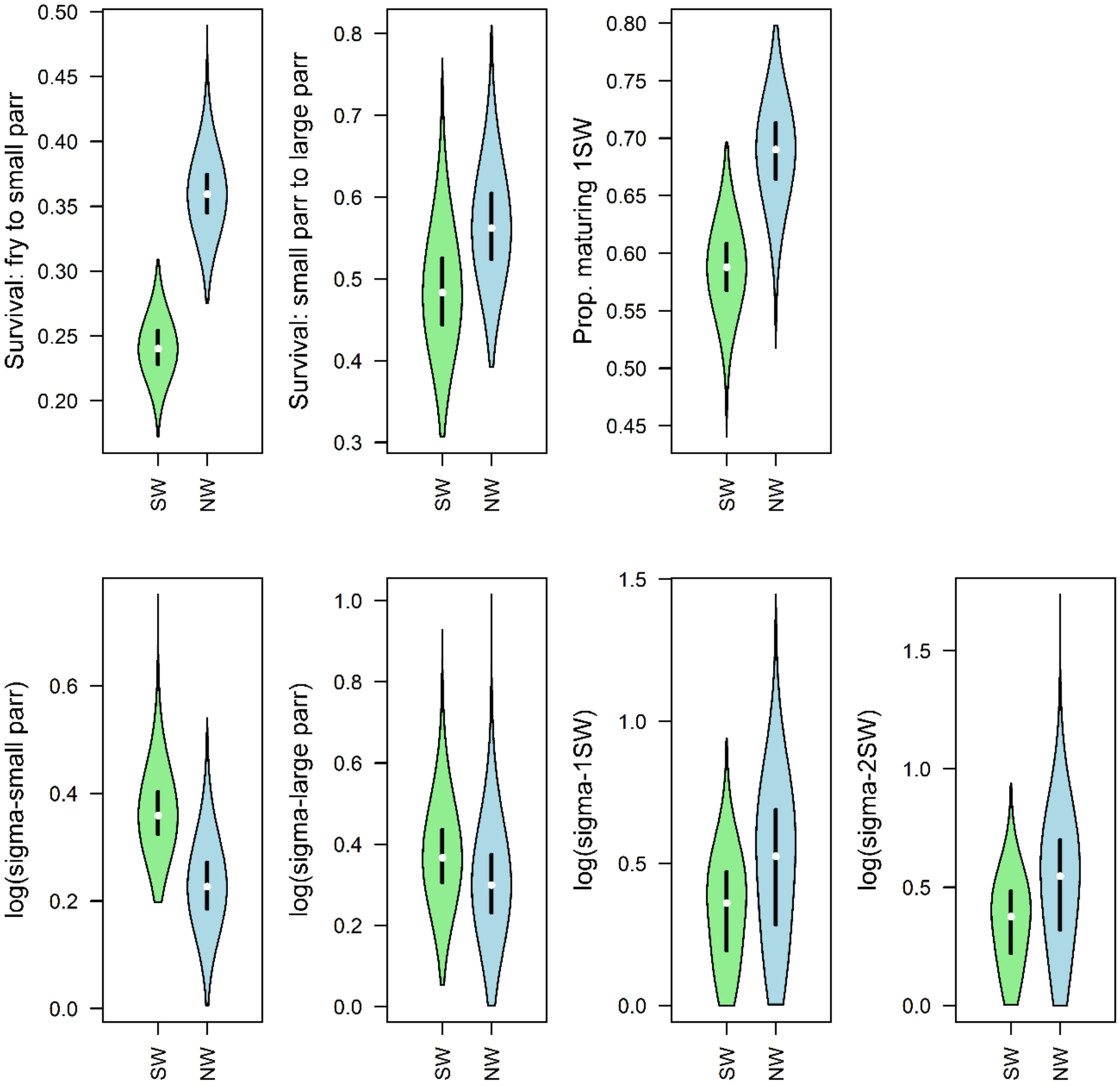

Figure A2.5. Posterior distributions of common over years but river specific parameters.

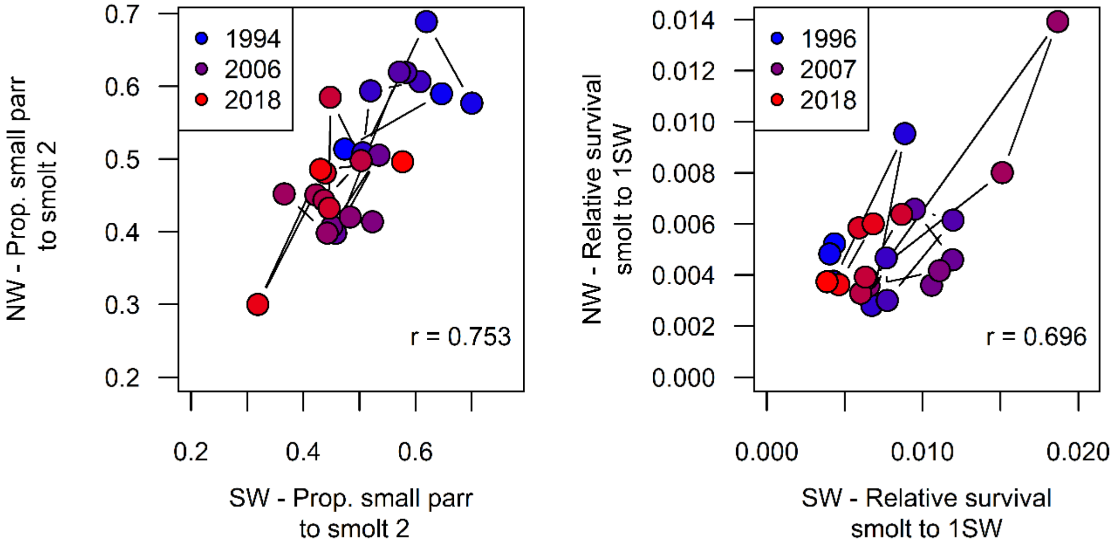

Figure A2.6. Correlations of parameter estimates between rivers.

IS STRIPED BASS (*MORONE SAXATILIS*) PREDATION THE CAUSE OF THE DECLINE IN AND LOW ATLANTIC SALMON (*SALMO SALAR)* RETURNS TO THE MIRAMICHI RIVER?

Introduction

Significant concern exists within Conservation organizations and Indigenous Peoples in New Brunswick about the severe decline in and low returns of Atlantic salmon to the Miramichi River. While the decline in salmon returns is common to all parts of the Miramichi, the lowest returns are to the Northwest Miramichi and its various tributaries. Conservation organizations, Indigenous Interests and the University of New Brunswick (UNB) have investigated the decline in and low salmon returns to the Miramichi and have arrived at the same conclusion that Striped Bass predation of migrating salmon smolts is a principal contributor to the current low salmon returns to the Miramichi River (Daniels et al. 2018; Wilbur and Collins 2024; and K. Phillips, pers. comm.[1]).

The Northwest Miramichi is the main spawning ground for Striped Bass in the Gulf of St. Lawerence, and its spawning occurs during the spring period when salmon smolts make their migration to sea (DFO 2023a). Despite the apparent evidence from acoustic tagging studies that Striped Bass predation is a main contributor to the current downturn in Small and Large salmon returning to the Miramichi system (Daniels et al. 2018; Wilbur and Collins 2024; and K. Phillip, pers. comm.[1]), the Department of Fisheries and Oceans (DFO) refuses to accept this explanation and instead claims Atlantic salmon are declining everywhere in Eastern Canada, including rivers where Striped Bass are absent (Minister of DFO, pers. comm.[2]). In an assessment of available data Chaput (2022) concluded that it's not clear that reducing Striped Bass abundance to the levels of the early 2000s (i.e., less than 100,000 spawners) would improve smolt survival and salmon returns to the Miramichi River. Advocates for stopping the decline and restoring the salmon stocks of the Mirmaichi River system have requested of DFO that measures be

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[1] Results of acoustic tagging studies on the Miramichi River, carried out 2021-2023 by the Canadian Rivers Institute, Atlantic Salmon Federation (ASF), Miramichi Salmon Association (MSA) and Anqotum Resource Management, provided by K. Phillips, Canadian Rivers Institute, UNB (*karl.phillips@unb.ca*).

[2] Stated in a letter (date Feb. 13, 2024) from The Honourable Diane Lebouthillier, P.C., M.P. to representatives of a number of New Brunswick organizations concerned about the decline in and low Atlantic salmon returns to the Miramichi River system, and the lack of action by DFO to address the apparent main cause of the decline (i.e., the massive Striped Bass spawning population in the Northwest Miramichi).

undertaken to reduce the Bass population on the Northwest Miramichi to 100,000 spawners (J. Bagnall, pers. comm.[3]). DFO refuses to implement any measures that would significantly reduce the Bass spawning population.

Striped Bass Abundance

Below is a graph of DFO's estimates of the numbers of Striped Bass spawning in the Northwest Miramichi River. The data for the plot were provided by DFO Science and as shown in DFO (2023a).

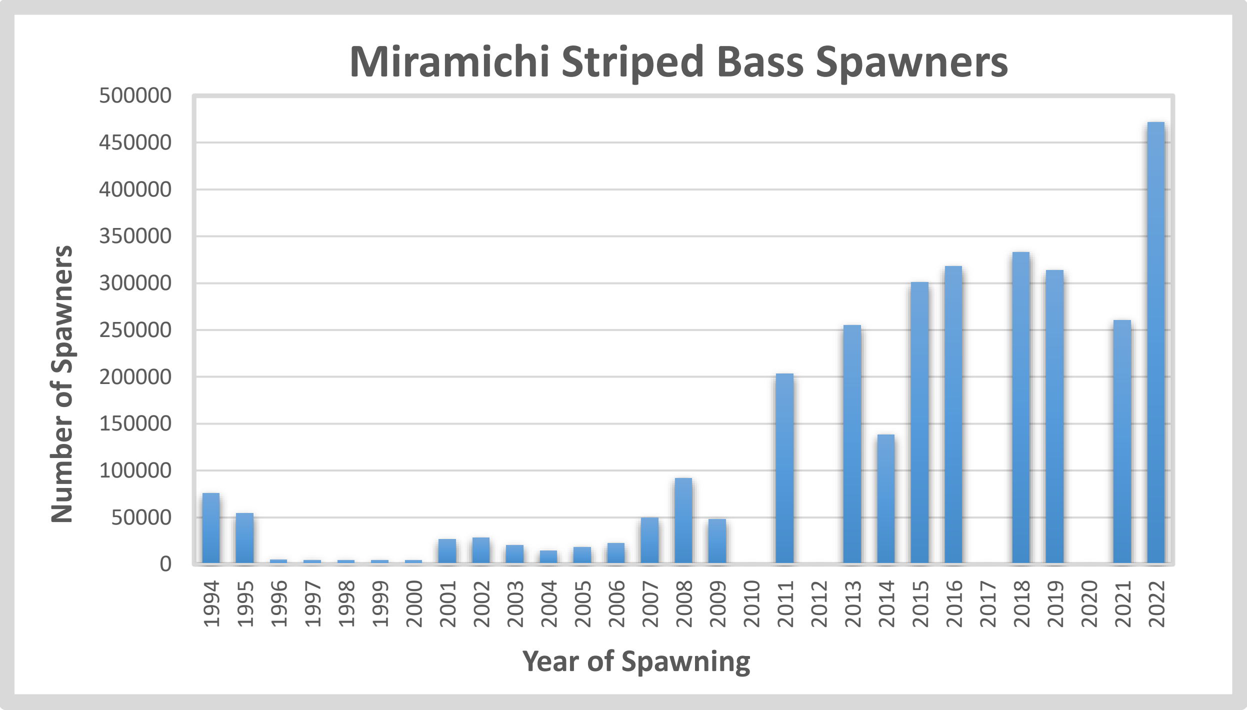

Figure 1. Estimates of the Number of Striped Bass spawning in the Northwest Miramichi River system for years 1994 through 2022. Estimates of the Numbers of Spawners in 2010, 2012, 2017 and 2020 are not available/included. The estimate for 2010 was incomplete, no estimates were made of the 2012 and 2020 populations, and the estimate for 2017 (994,000 spawners) was excluded because it was excessively greater than can be accounted for in population estimates in prior years (i.e, 2015 and 2016) and also, in later years (i.e., 2018 and 2019).

DFO reported the first major increase in the Striped Bass spawner population to have occurred in 2011 (DFO 2013). The spawner population in 2022 is estimated

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[3] J. Bagnall (Fisheries Committee Chair and Past President, New Brunswick Salmon Council), acting on behalf of multiple Conservation and Indigenous interest groups, made oral presentations to representatives of DFO.

to be close to 500,000 (DFO 2023a). While no official report has be made as to the abundance of spawners in 2023, DFO has stated that the number is around 500,000 spawners.

Atlantic Salmon Abundance

Atlantic salmon abundance in the Miramichi River system is assessed against measures of adult salmon returns or smolt survival for salmon populations of other rivers emptying into the Gulf of St. Lawrence. The data sets for the graphs below include counts and population estimates of the numbers of Small (< 63 cm fork length) and Large (= or > 63 cm fork length) salmon returning, recreational catches of Small plus Large salmon, and smolt-to-adult survival rates for one river population. The plots vary in length as a result of data availability.

Figures 2 and 3 below are Counts of Small and Large Salmon at Salmon Protection Barriers operated by the MSA for the New Brunswick Department of Natural Resources and Energy Development (NB DNRE). Data on the counts were provided by NB DNRE.

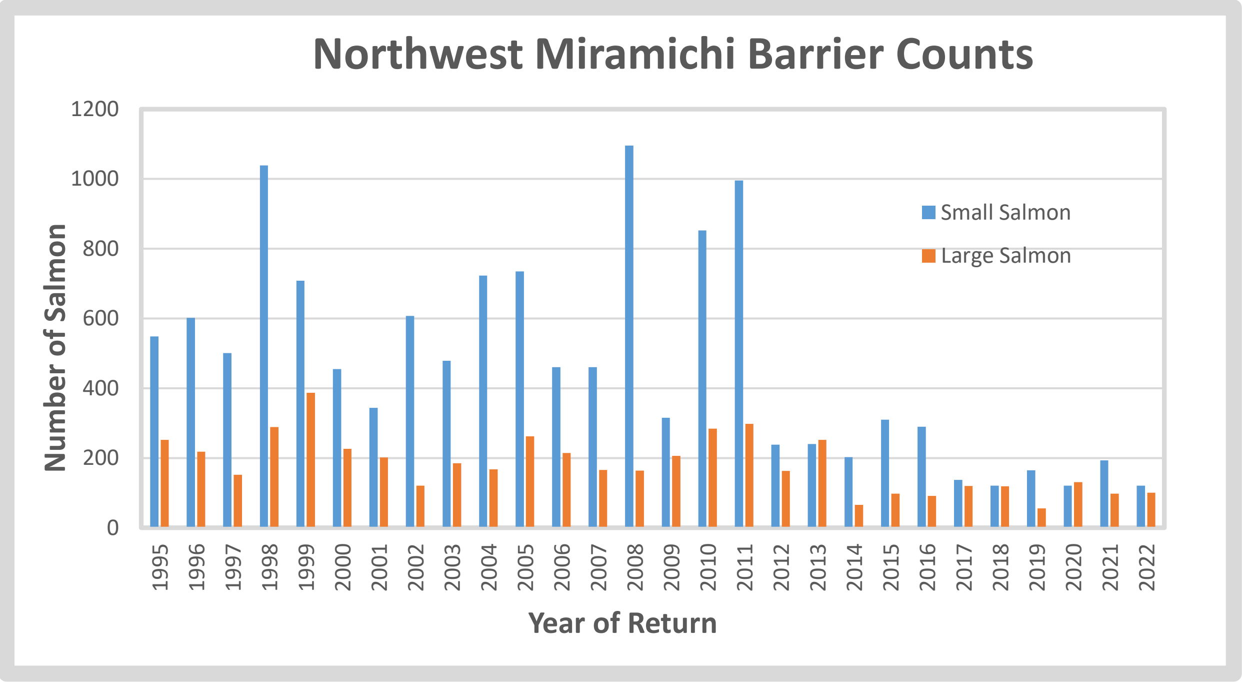

Figure 2. Annual Counts of Small and Large Salmon at the Salmon Protection Barrier on the Northwest Miramichi River for years 1995 through 2022.

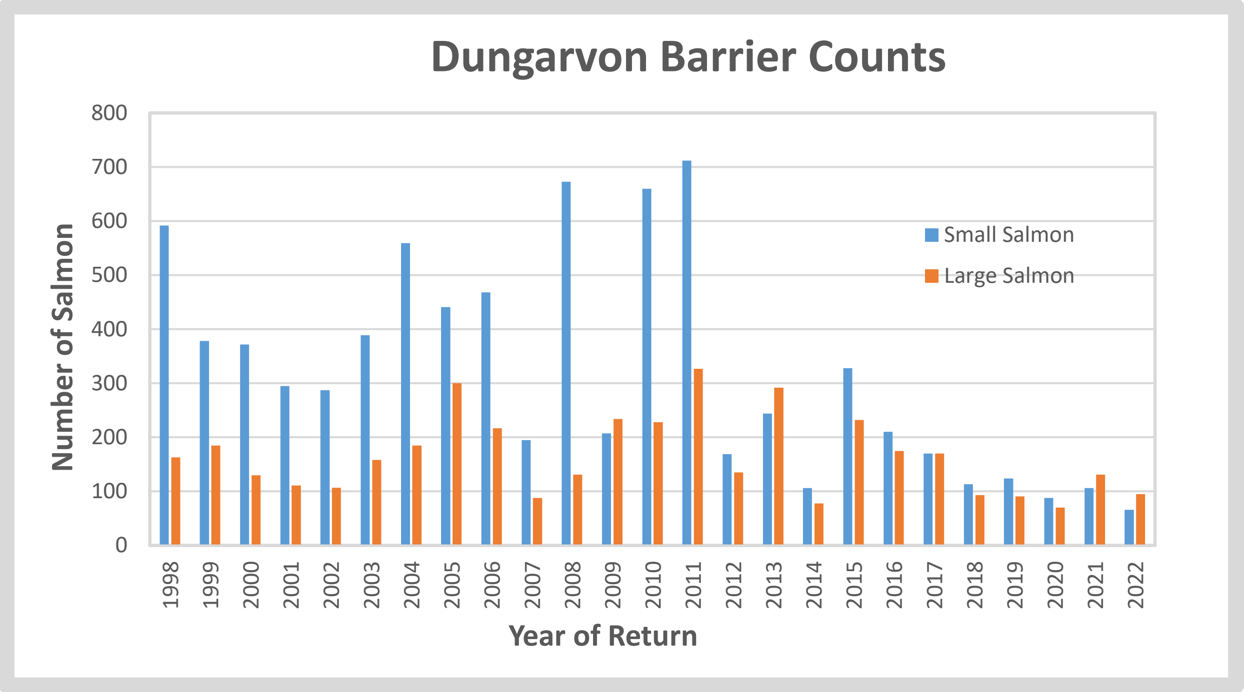

Figure 3. Annual Counts of Small and Large Salmon at the Salmon Protection Barrier on the Dungarvon River for years 1998 through 2022.

Counts at both Barriers show the same pattern for Small Salmon for which Counts drop off to lower levels beginning in 2012, a pattern that appears to persist and even worsen in the later years (i.e., in years 2018 through 2022). Numbers of Large Salmon are considerably fewer than the Numbers of Small Salmon throughout the time series, and the drop-offs in their Counts are later in time for the Barrier on the Northwest (2014) (Fig. 2), and not as apparent for the Dungarvon Barrier (Fig. 3).

The next two graphs are DFO Science's estimates of the Numbers of Small and Large Salmon returning to the Northwest Miramichi and the Southwest Miramichi, respectively (Fig. 4 and 5). The data for both plots were from various DFO Science documents (DFO 2015, 2020, 2022 and 2023b) and provided by DFO Science. No estimates were made of the Numbers of Small and Large Salmon returning in 2020.

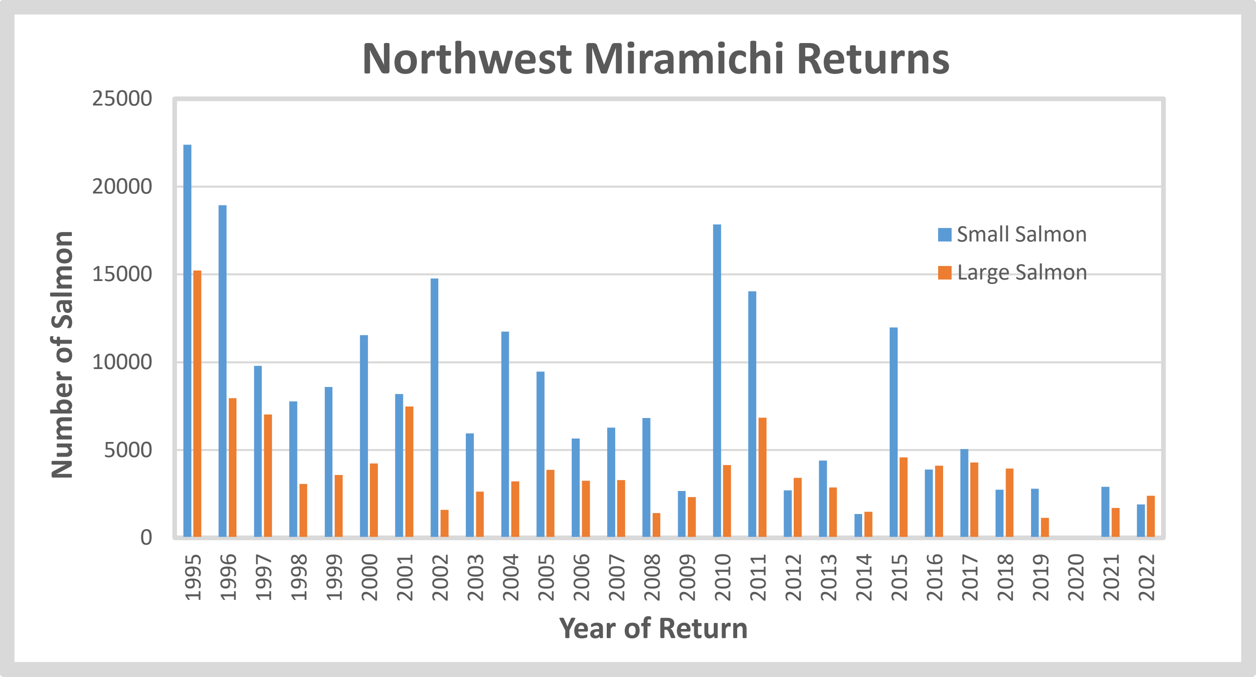

Figure 4. Annual estimates of the Number of Small and Large Salmon Returns to the Northwest Miramichi River for years 1995 through 2022.

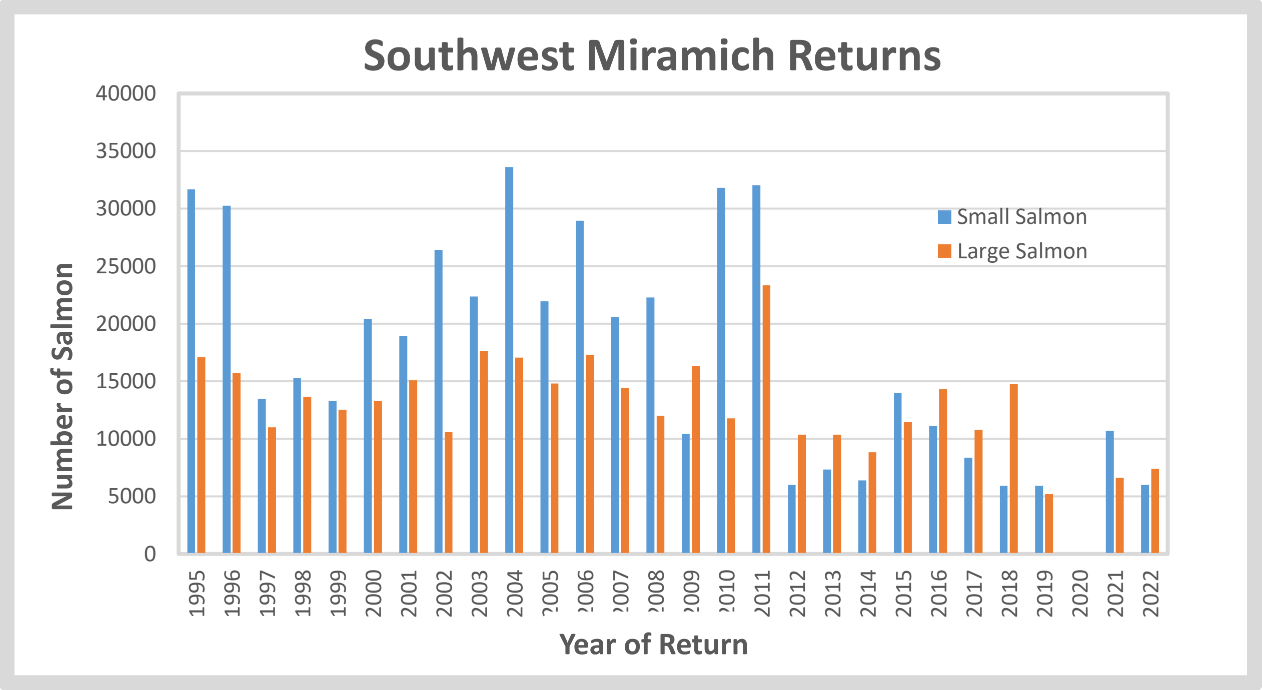

Figure 5. Annual estimates of Small and Large Salmon Returns to the Southwest Miramichi River for years 1995 through 2022.

Figures 2, 3, 4 and 5 confirm that both the Northwest and Southwest Miramichi have been experiencing lower Returns of Small and Large Salmon since 2011.

To assess the claim of DFO that the decline in and low salmon returns to Miramichi since 2011 are consistent with declines in Atlantic salmon everywhere in Eastern Canada (Minister of DFO, pers. comm.[4]), measures of adult salmon abundance or smolt survival for salmon populations of river systems within the Gulf of St. Lawrence are presented in graph-form below. The assessment is limited to salmon populations within the Gulf because they likely experienced the same at-sea perils as salmon from the Miramichi. The first of the plots is for the Margaree River (Figure 6). The Margaree population is presented as being representative of Southern Gulf salmon populations outside the Miramichi River system. The graph presents DFO Science's estimates of Small and Large Salmon Returns to the Margaree River. The data were taken from DFO assessment documents (DFO 2015, 2020, 2022 and 2023b) and provided by DFO Science.

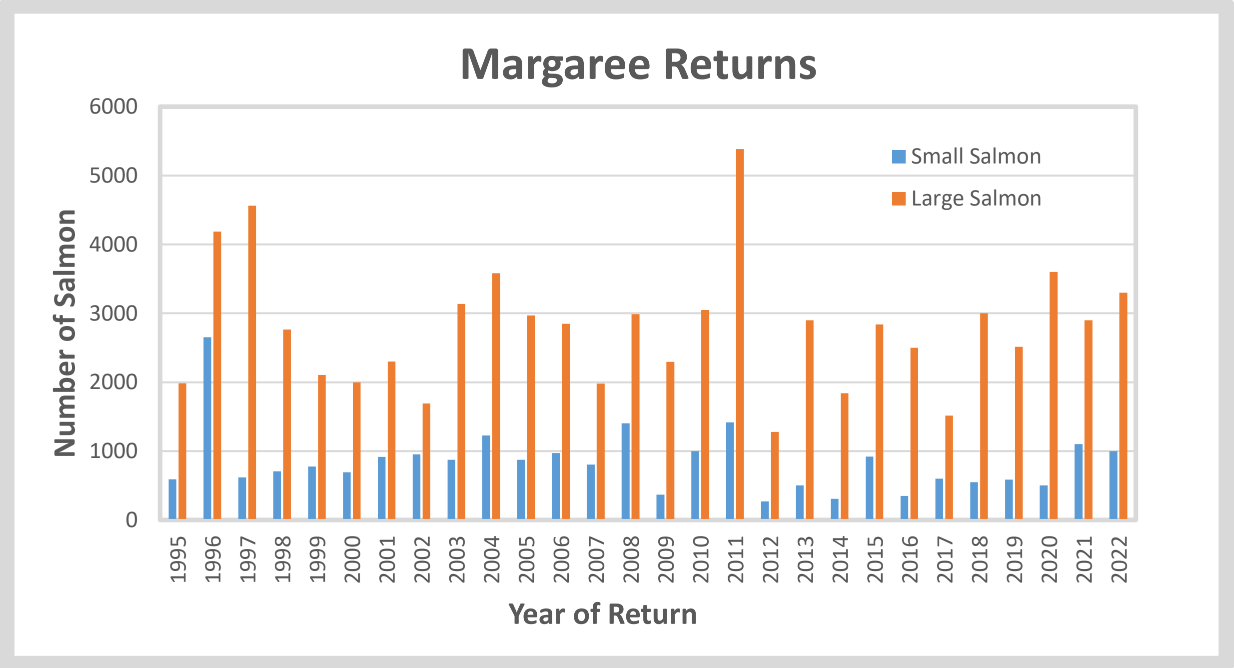

Figure 6. Annual estimates of the Numbers of Small and Large Salmon Returns to the Margaree River for years 1995 through 2022.

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[4] Stated in a letter (date Feb. 13, 2024) from The Honourable Diane Lebouthillier, P.C., M.P. to representatives of a number of New Brunswick organizations concerned about the decline in and low Atlantic salmon returns to the Miramichi River system, and the lack of action by DFO to address the apparent main cause of the decline (i.e., the massive Striped Bass spawning population in the Northwest Miramichi).

Both Small and Large Salmon Returns to the Margaree show no decline over the period 1995 through 2022. This is confirmed by DFO's statement '*The changes in estimated returns (to the Margaree) over the recent 12-year period do not result in a statistically significant trend for either Small or Large salmon.'* (DFO 2023b).

Figure 7 below is for the West Coast of Newfoundland's Western Arm Brook. The graph details annual Smolt-to-Adult Survival rates for smolts emigrating from Western Arm Brook for years 1994 through 2021. No survival rate was determined for smolts emigrating in 2020. Data for the graph was extracted from a plot of the same in the document detailing DFO's most recent stock assessment of Newfoundland and Labrador salmon stocks (DFO 2024).

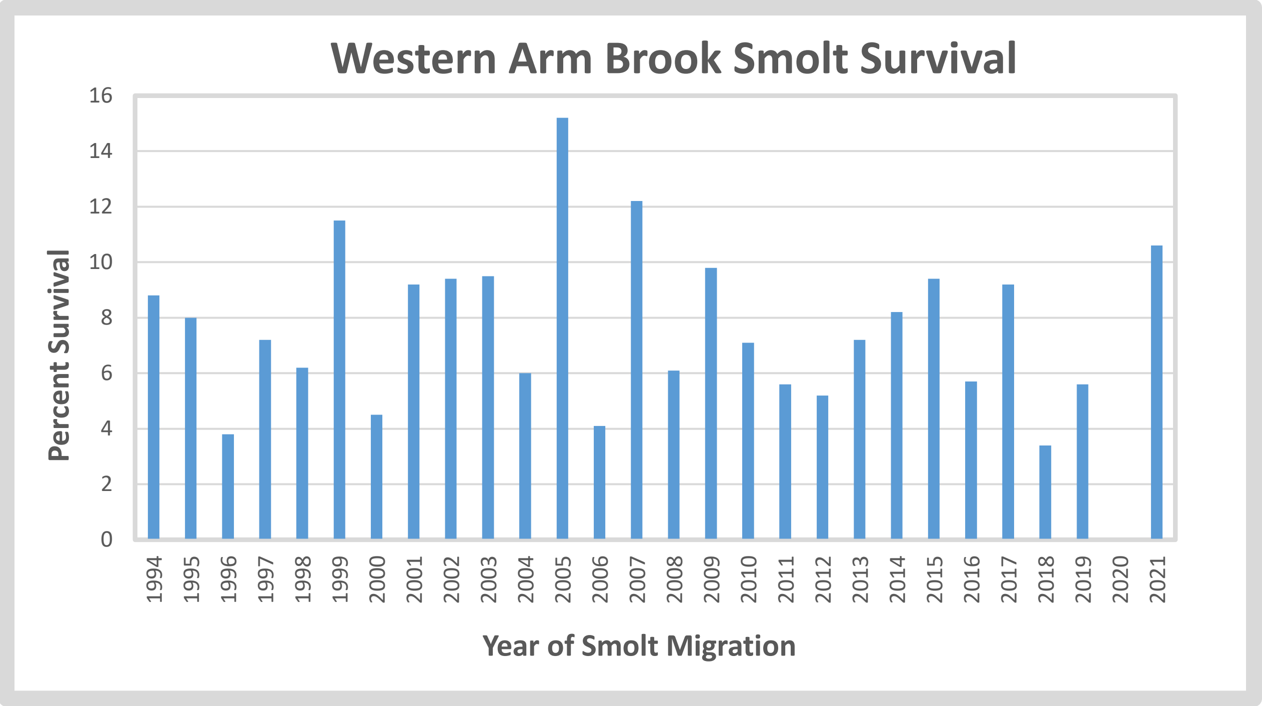

Figure 7. Percent survival rates for Atlantic salmon smolts emigrating from Western Arm Brook, Newfoundland in years 1994 through 2021.

While Smolt-to-Adult Survival varied from less than 4% to over 15% over the period 1994 through 2021, no trend in smolt survival is apparent over both the complete duration of monitoring or over the most recent 15 years.

Figures 8 and 9 below show the Numbers of Small and Large Salmon Returns to the Cascapedia and Saint-Jean rivers on the Gaspe, Quebec (MFFP 2024).

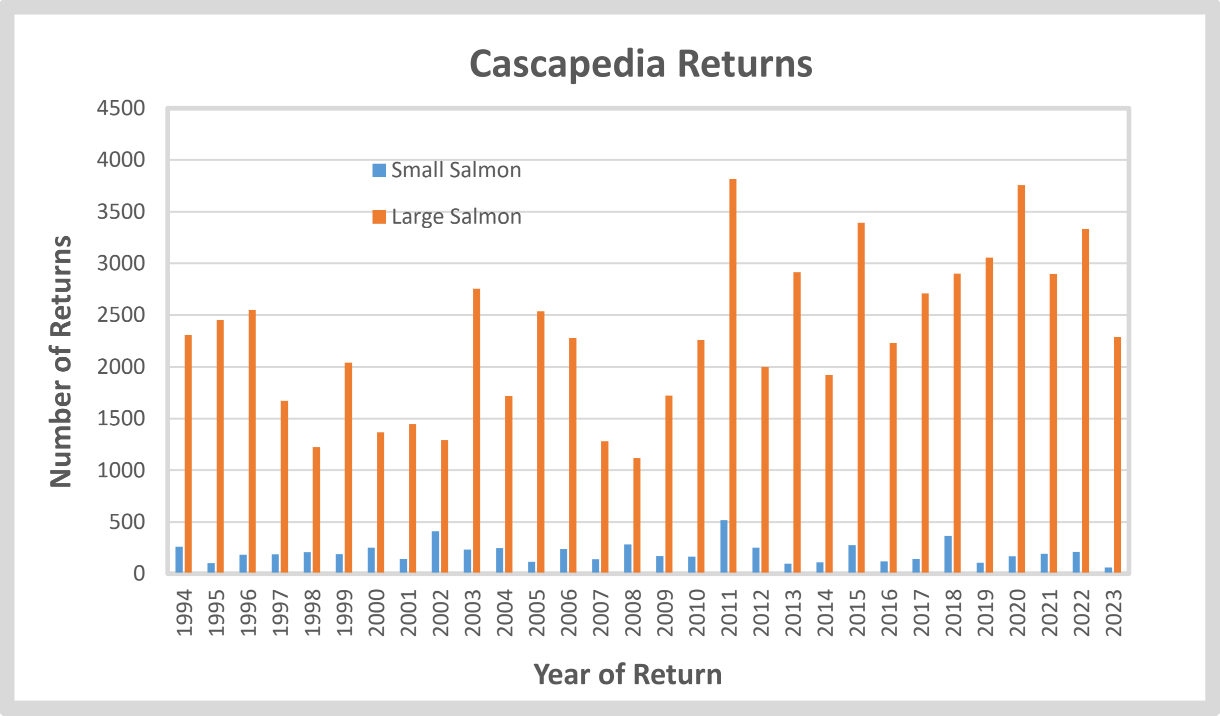

Figure 8. Numbers of Small and Large Atlantic Salmon Returns to the Cascapedia River, Gaspe, Quebec for years 1994 through 2023.

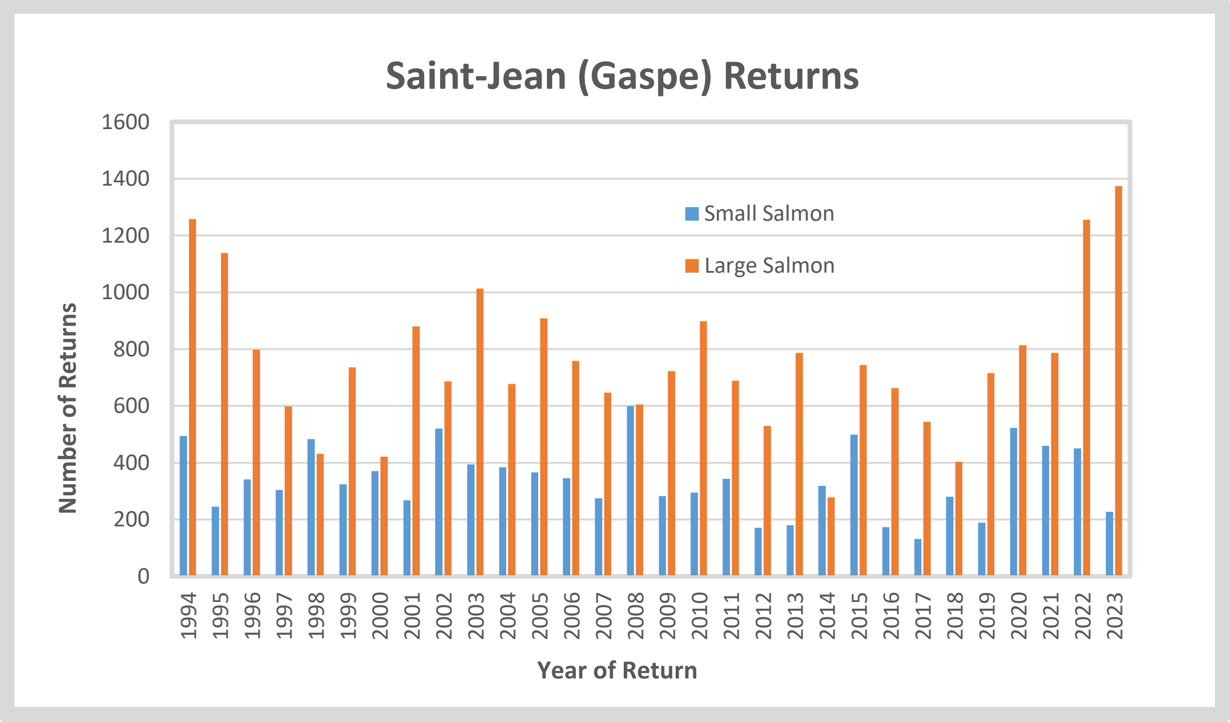

Figure 9. Numbers of Small and Large Atlantic Salmon Returns to the Saint-Jean River, Gaspe, Quebec for years 1994 through 2023.

Figures 8 and 9 show the Numbers of Salmon returning to the Cascapedia and Saint-Jean rivers to be variable. They show no decline in Salmon Returns over the time frame of the series (i.e., 1994 through 2023) and in particular, no downturn since 2011. To the contrary, Salmon Returns to the Cascapedia are generally greater since 2010.

Figures 9, 10 and 11 below depict Recreational Catches of Atlantic salmon for rivers in the Province of Quebec's three composite areas bordering the Gulf of St. Lawrence and where salmon angling takes place (i.e., the Gaspe, Anticosti Island and the North Shore) (MFFP 2024). The catches are the sum of the three size class designations for Adult Salmon for Quebec's recreational fishery for salmon (i.e., Grilse, Small salmon and Large salmon), and include both Retained and Released fish. Released salmon entered the Catch figures in the mid-to-late 1990s with the imposition of Provincial Regulations Limiting the Retention of Large salmon.

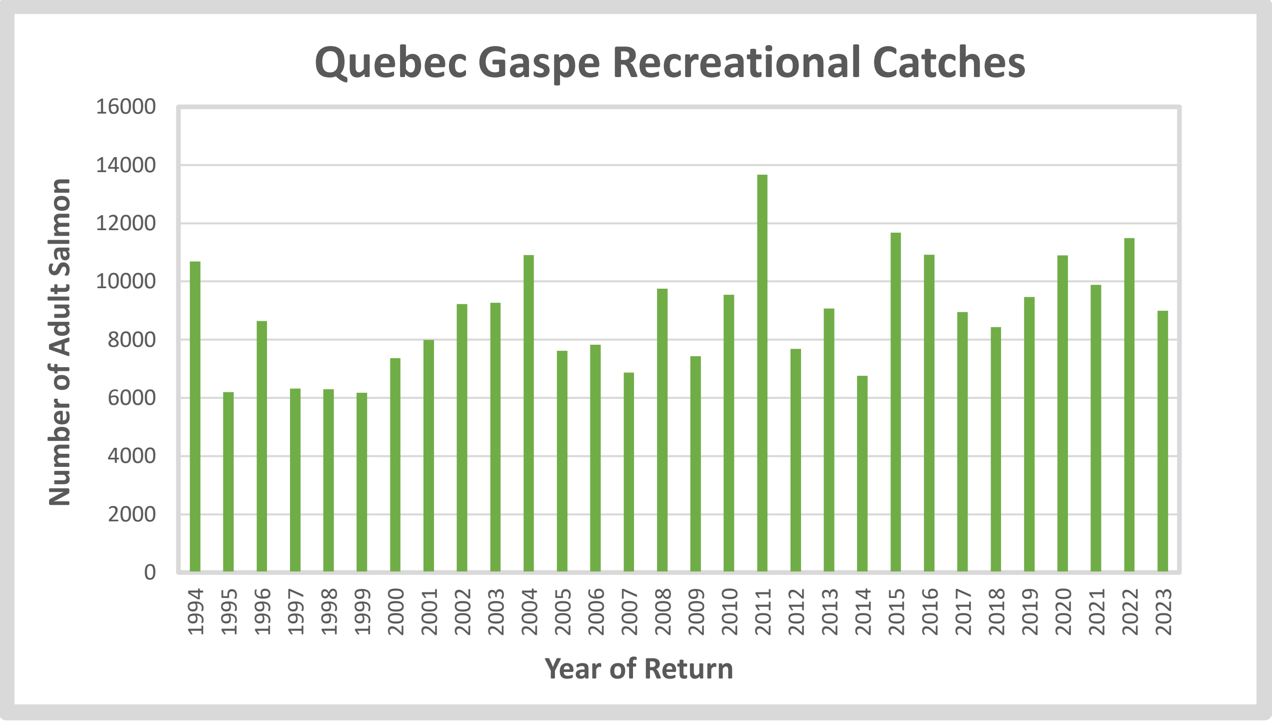

Figure 9. Catch Numbers (retained and released) for Adult Salmon in the Recreational Fishery on the Gaspe, Quebec for year 1994 through 2023.

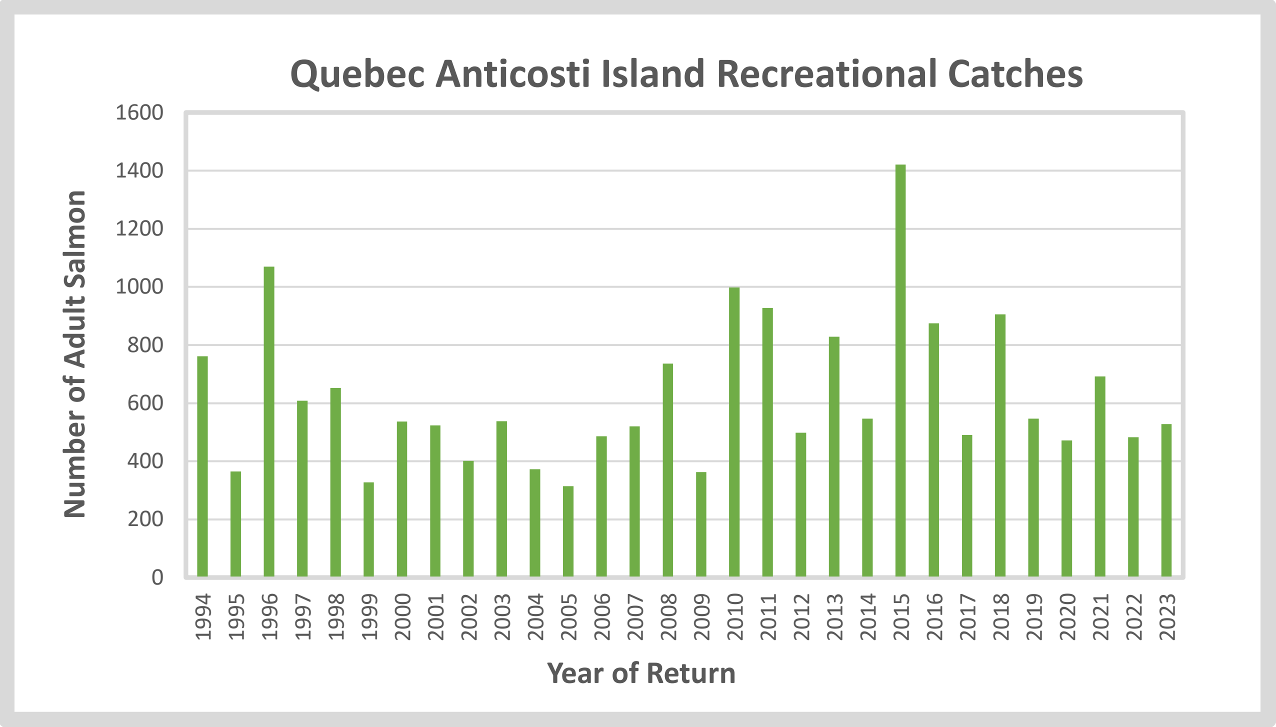

Figure 10. Catch Numbers (retained and released) for Adult Salmon in the Recreational Fishery on Anticosti Island, Quebec for years 1994 through 2023.

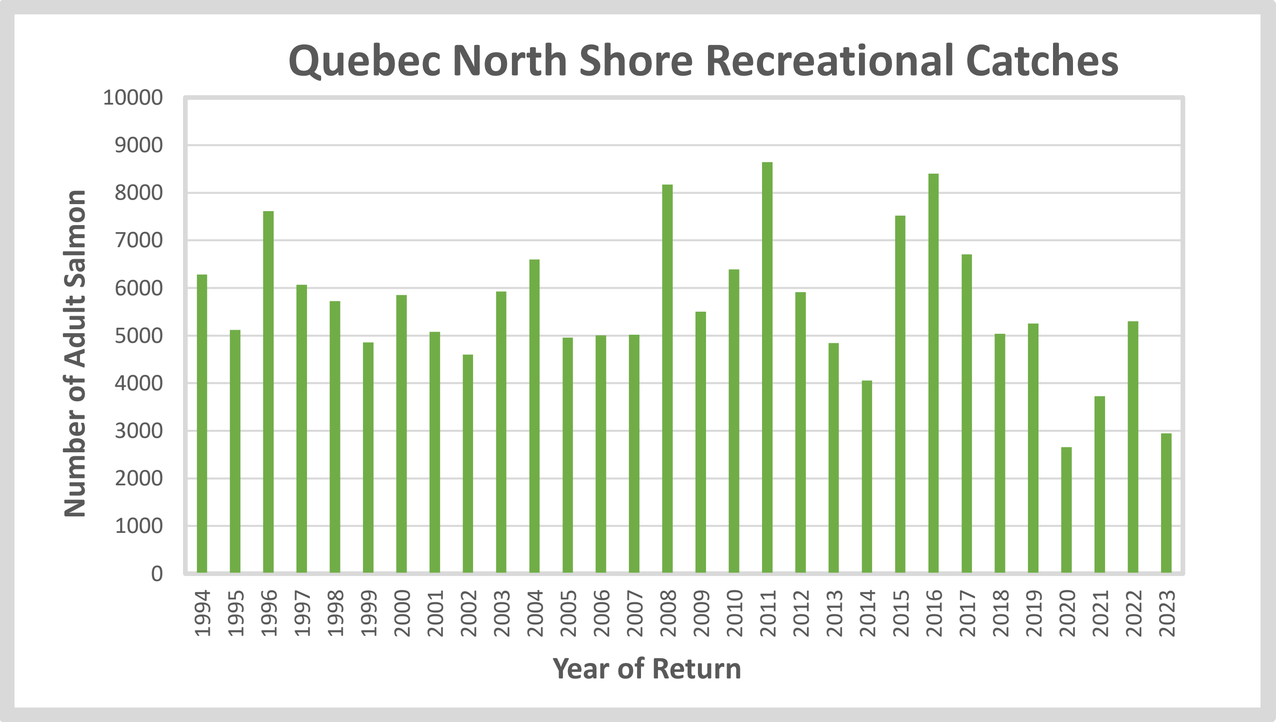

Figure 11. Catch Numbers (retained and released for Adult Salmon in the Recreational Fishery on the North Shore, Quebec for years 1994 through 2023.

Both the Gaspe and Anticosti Island Fishing Areas show relatively uniform annual catches of salmon over the time span shown (1994 through 2023). The catches on the North Shore are also relatively uniform over time except for the most recent four years (2020 through 2023) when catches for three of the years appear lower. Catches in all three Areas appear uniform through 2019.

Analyses and Discussion

The timing and declines in salmon returns to the Miramichi River system shown in Figures 2, 3, 4, and 5 are supported by telemetry studies reported by Chaput et al. (2018) which show lower survival rates for smolts of the Northwest Miramichi passing through the Miramichi estuary and Bay for the years 2013 through 2016 than for smolts migrating in the years 2003 through 2008. The studies demonstrated a similar pattern for smolts emigrating from the Southwest Miramichi although the survival rates for the later period were less depressed than for smolts of the Northwest Miramichi. The same study showed no difference in survival rates between the two periods for smolts migrating through the tidal waters of the Restigouche and Cascapedia river systems.

Excluding the salmon populations of the Miramichi River System (Figures 2, 3, 4 and 5), none of the graphs for salmon populations in the Gulf of St. Lawrence show a major decline over the past decade in adult salmon abundance (Figures 5, 7, 8, 9, 10, 11 and 12) or survival at sea (Figure 6). This discrepancy between the decline in salmon returns to the Miramichi system and the absence of decline in the salmon populations of other Gulf rivers strongly suggests that the cause of the decline in salmon returns to the Miramichi River is local in origin.

Claims made that salmon populations of some rivers emptying into the Gulf of St. Lawrence and outside the Miramichi system are experiencing low returns of Atlantic salmon were investigated by searching data for Quebec rivers (MFFP 2024). While the claims are correct, the rivers with low salmon returns are relatively few and the patterns of their downturn do not align with the significant decline in salmon returns to the Miramichi River system beginning in 2012 (Figures 2, 3, 4 and 5). Given that the Quebec rivers experiencing low returns of salmon are in the midst of rivers for which good and stable abundances of salmon are reported (MFFP 2024), and shown here (Figures 7, 8, 9, 10, 11 and 12), it seem logical that the causes of their low returns of salmon are of local origins.

Adult salmon population numbers of rivers of the Outer Bay of Fundy (e.g., Saint John River) and along the Atlantic coast of Nova Scotia (e.g., LaHave and St. Mary's rivers) continue to remain low after the severe declines they experienced in marine survival beginning in the early-1990s (DFO 2023c). Their declines were more severe than that of salmon populations of rivers emptying into the Gulf of St. Lawrence (Fig. 2, 3, 4, 5, 6, 7, 8, 9, 10 and 11). The degree of their declines, and general lack of recovery experienced by these southern populations, do not coincide with those of Gulf salmon populations. Further, the general trends in adult salmon returns to both Newfoundland and Labrador rivers, and smolt-to-adult survival rates for populations of rivers on Newfoundland, do not show the sharp decline and continuing low levels since 2011 like that evident for populations of the Miramichi River system (DFO 2024).

The pattern in the decline in and low returns of salmon to both the Northwest and Southwest systems (Fig. 2, 3, 4, and 5) is consistent with the build-up in the Northwest Miramichi's Striped Bass spawning population (Fig. 1). Results from the continuing acoustic tagged smolt study on the Miramichi are showing strong evidence of significant predation of smolts by Striped Bass in the lower Miramichi River (Daniels et al. 2018; Wilbur and Collins 2024; and K. Phillip, pers. comm.[5]). A very low survival rate of 6.4% was recorded for 249 acoustic tagged smolts, migrating from the Northwest and Southwest Miramichi systems, to the Gulf of St. Lawrence in 2023 (K. Phillips, pers. comm.[5]). While the losses of 93.6% of the tagged smolts included all types of losses, the digestion sensory telemetry system employed enabled the determination that 65.1% of the tagged smolts were lost to cold-blooded predators (K. Phillips, pers. comm.[5]). As there is no obvious evidence of any abundant cold-blooded predator in the tidal waters of the Miramichi in May other than Striped Bass, it is likely that the majority of the losses to cold-blooded predators are due to Striped Bass predation.

Further supporting the conclusion that Striped Bass predation is the main cause of the recent decline in salmon returns to the Miramichi systems is the timing of the major decline being 2012 (Fig. 2, 3, 4 and 5), the year following 2011 when

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[5] Results of acoustic tagging studies on the Miramichi River, carried out 2021-2023 by the Canadian Rivers Institute, ASF, MSA and Anqotum Resource Management, provided by K. Phillips, Canadian Rivers Institute, UNB (*karl.phillips@unb.ca*).

the Striped Bass population dramatically increased to more than 200,000 spawners (Fig. 1).

While returns of salmon to the Miramichi River system have continued to remain low, and even decline further, the Striped Bass population has continued to expand to close to 500,000 spawners in 2022 (DFO 2023b), and seemly to the same level or greater in 2023 according to DFO. Of immediate concern are the extremely low returns of salmon to the Northwest Miramichi. Consistent with these low returns are salmon egg deposition levels below 100 eggs per 100 m^2 of rearing habitat in recent years (DFO 2023b), and as low as 51 eggs per 100 m^2 in 2019 (DFO 2020). Such low egg deposition levels put at risk the recovery of the Northwest salmon population, a risk that will heighten if this pattern of low salmon returns persists or worsens. This critical state in the Northwest Miramichi salmon population depicts the urgency for action be taken to reduce the Striped Bass spawning population size in the Miramichi River system. A significant reduction in the Bass population is also required to prevent harm to the Southwest salmon population and to enable its recovery.

The recent massive expansion in the Northwest Miramichi Striped Bass spawning population coincides with the rising annual sea temperatures (May-November) in the Gulf of St. Lawrence (Galbraith et. al. 2022). Given this improvement in marine conditions for Striped Bass recruitment, the numbers of Striped Bass are likely to continue to increase further in the Miramichi, and to become more abundant in the Southern Gulf through the establishment of new populations in other rivers. Reports of Striped Bass spawning in the Southwest Miramichi and Tabusintac River (McGee 2020) are evidence of spawning outside the Northwest Miramichi. As well, anecdotal reports of the presence of Striped Bass juveniles in the Hillsborough River annually over the past eight years further support the likelihood that the numbers of Striped Bass will continue to increase in the Southern Gulf. The extension of the spawning of Striped Bass to these rivers, and possibly to others yet to be discovered, along with the massive increase in the Northwest spawning population (Fig. 1), reflect a large and growing abundance of Striped Bass in the Southern Gulf of St. Lawrence and their improved harmonization with the warming environmental conditions in the Gulf (Galbraith et al. 2022). Given the resilience of the Northwest population and the magnitude in abundance and expanse of Striped Bass in the Southern Gulf, it is likely that a major reduction in the Northwest spawner population could be made without harm to its sustainability at a designated lower level, or to its regrowth if such were desired in the future.

Conclusions

1. Atlantic salmon returns to the Miramichi River are low and the salmon population of the Northwest Miramichi is at/or approaching risk of population-level damage that endangers its recovery.
2. DFO's claim that the decline in and low returns of salmon to Miramichi River system are like that common to salmon populations throughout Eastern Canada, is not valid.
3. The cause(s) of the decline in and low salmon returns to the Miramichi system is/are local in origin and the massive Striped Bass spawning population in the Northwest Miramichi is a main contributing factor.
4. Further increases in the numbers of Striped Bass may occur in the Miramichi and are likely in the Southern Gulf as a result of changes in environmental conditions in the Gulf being more favorable to Striped Bass recruitment.
5. Without action being taken to reduce the numbers of Northwest Miramichi Striped Bass, an annual population of 500,000 spawners, or more, is likely.
6. A significant reduction in the Miramichi Striped Bass spawning population is urgently required to curtail the threat that its large numbers pose to the recovery of the salmon populations of the Miramichi River system.
7. It is likely that the Northwest Striped Bass population could be reduced to 100,000 spawners with no harm to the population given its resilience and the growing expansion of Bass spawning to other rivers in the Southern Gulf.

Sources of Information

Chaput, G. 2018; Carr, J., Daniels, J., Tinker, S., Jansen, I., and Whoriskey, F. 2018. Atlantic salmon (*Salmo salar*) salt and early post-smolt migration and survival inferred from multi-year and multi-stack acoustic telemetry studies in the Gulf of St. Lawrence, Northwest Atlantic. ICES J. Marine Science. Vol. 76 (4), 1107-1121.

Chaput, G. 2022. Multispecies considerations for defining Fisheries Reference Points for Striped Bass (*Morone saxatilis*) from the Southern Gulf of St. Lawrence. DFO Sci. Advis. Sec. Res. Doc. 2022/030. 50p.

Daniels, J., Chaput, G., and Carr, J. 2018. Estimating consumption rate of Atlantic salmon (*Salmo salar*) smolts by Striped Bass (*Morone saxatilis*) in the Miramichi River estuary using acoustic telemetry. Can. J. Fish. Aquat. Sci. 75(11), 1811-1822.

DFO. 2013. Update to 2012 on spawner abundance and biological characteristics for Striped Bass (*Morone saxatilis*) in the Southern Gulf of St. Lawrence. DFO Sci. Advis. Sec. Sci. Resp. 2013/010.

DFO. 2015. Status of Atlantic salmon (*Salmo salar* L) stocks in rivers of Nova Scotia flowing into the Gulf of St. Lawrence (SFA 18), 2012-2013. DFO Sci. Advis. Sec. Sci. Res. Doc. 2015/043. 40p.

DFO. 2020. Update of indicators of Atlantic salmon (*Salmo salar*) in DFO Gulf Region Fishing Areas 15-18. DFO Sci. Advis. Sec. Sci. Resp. 2020/028.

DFO. 2022. Update of indicators of Atlantic salmon (*Salmo salar*) in DFO Gulf Region Fishing Areas 15-18. DFO Sci. Advis. Sec. Sci. Resp. 2022/021.

DFO. 2023a. Update of spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the Southern Gulf of St. Lawrence to 2022. DFO Sci. Advis. Sec. Sci. Resp. 2023/004.

DFO. 2023b. Update of stock status of Atlantic salmon (*Salmo salar*) in DFO Gulf Region Fishing Areas 15-18. DFO Sci. Advis. Sec. Sci. Resp. 2023/035.

DFO. 2023c. Stock status update of Atlantic salmon to 2021 in Salmon Fishing Areas 19-21 and 23. DFO Sci. Advis. Sec. Sci. Resp. 2023/019.

DFO. 2024. Stock assessment of Newfoundland and Labrador of Atlantic salmon (*Salmo salar*) in 2022 (SFA 1-14B). DFO Sci. Advis. Sec. Sci. Resp. 2024/015.

Galbraith, P.S., Chassé, J., Dumas, J., Shaw, J.-L., Caverhill, C., Lefaivre, D., and Lafleur, C. 2022. Physical oceanographic conditions in the Gulf of St. Lawrence during 2021. DFO Sci. Advis. Sec. Res. Doc. 2022/034. 83p.

McGee, K. 2020. Striped Bass spawning survey 2020. Study carried out by MSA in cooperation with Miramichi Watershed Management Committee, Northumberland Salmon Protection Association and DFO Science Branch. Report available from MSA.

MFFP. 2024. Bilan de l’exploitation du saumon au Québec en 2023. Governement du Québec Ministère des Forêts, de la Faune et des Parcs, Secteur de la faune et parcs. 303p.

Wilbur, N., and Collins, S. 2024. *Written Presentation to* Eastern New Brunswick Coastal Inland Recreational Fisheries Advisory Committee, and submitted to Fisheries and Oceans Canada (Gulf Region). Submission made by Nathan Wilbur and Serge Collins, ASF, Jan. 17, 2024.

Prepared by: John Ritter (May 28, 2024)

Fisheries and Oceans Canada Pêches et Océans Canada
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Research Document 2006/041 **Document de recherche 2006/041**

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Assessment of Status and Recovery Potential for Striped bass (*Morone saxatilis*) in the Southern Gulf of St. Lawrence

Évaluation de la situation et du potentiel de rétablissement du bar rayé (*Morone saxatilis*) dans le sud du golfe du Saint-Laurent

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ISSN 1499-3848 (Printed / Imprimé)

Canada

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ABSTRACT

The COSEWIC's recommended "Threatened" listing for striped bass in the southern Gulf required that DFO's Section 73 permitting framework be applied to determine if incidental harm would jeopardize survival or recovery of the species. Three primary indicators were explored to determine the status of striped bass in the southern Gulf. First, mark-recapture experiments conducted on striped bass returning to the Northwest Miramichi River to spawn indicated an average of 22,000 mature individuals each year since 2001. This represented a modest increase from a low of approximately 4,000 annually in the 1998-2000 period but not as high as the peak level of 50,000 spawners observed in the mid 90's. Secondly, analyses of the striped bass bycatch in the gaspereau fishery of the Northwest Miramichi River indicated an average of 84 bass per net per day over the last 5 years and closely correlated with estimates of population size derived from mark-recapture experiments. Lastly, spawning success measured from catches of young-of-the-year in the fall open-water smelt fishery (1991-1998) and beach seine surveys (2001-2005) were weakly correlated with spawner abundance estimates and suggests that year-class success can be determined by environmental factors. Secondary indicators of status such as the truncated and unchanged age and size distributions for spawning striped bass since the early 90's supported the high natural mortality estimate of 0.54-0.59 derived for this population after the commercial fishery closure in 1996. Tagging studies continue to define the whole of the southern Gulf as the area of occupancy for this population of striped bass.

A discrete life history model was used to propose reference levels for recovery of the southern Gulf population. The recovery objectives parallel the precautionary approach benchmarks of critical, cautious, and healthy zones. We propose an S_{opt} value of 21,600 spawners as the recovery limit for southern Gulf striped bass and the 50%SPR value of 31,200 spawners as the recovery target, the latter being the value for managing any directed fisheries. The S_{eq} value (spawners at replacement in the absence of fisheries) was estimated at 63,000 fish. We discuss the need to implement compliance rules and suggest that a 6 year sliding window may be appropriate with the objective of exceeding the recovery limit in 5 of 6 years. Under present conditions, including bycatch of YOY and continued illegal removals of adult bass, there is a low probability (18%) that the southern Gulf striped bass will be above the recovery limit by 2015. If the total mortality on adults is reduced to $Z = 0.6$ from the current condition of $Z = 0.8$ and YOY bycatch is eliminated, there is a greater than 95% chance that the population will be above the recovery limit by 2015.

The Northwest Miramichi River remains the only confirmed spawning location for striped bass in the southern Gulf. Because striped bass occupy all of southern Gulf but yet continue to show high fidelity to the Northwest Miramichi, the colonization or establishment of new spawning locations may not be a realistic recovery objective.

Quantitative estimates of mortality were not possible for each of the major threats believed to be limiting the rebuilding of this population. Illegal harvests are believed to be the single greatest cause of mortality for the population. Total accumulated mortality does not seem to jeopardize the survival, but under present conditions, recovery above the proposed limit is unlikely.

Mitigation measures are discussed. Recovery efforts for southern Gulf striped bass should focus on reducing adult mortality and YOY bycatch and protecting the striped bass habitat and spawning grounds of the Miramichi system.

RÉSUMÉ

La recommandation du COSEPAC d’inscrire le bar rayé du sud du Golfe sur la liste des espèces « menacées » exigeait l’application du cadre d’autorisation du MPO en vertu de l’article 73 afin de déterminer si la survie ou le rétablissement de l’espèce risque d’être compromis par une activité qui la toucherait de manière incidente. Trois indicateurs principaux ont été examinés afin d’établir la situation du bar rayé dans le sud du Golfe. Premièrement, des expériences de marquage et de recapture menées sur les bars rayés remontant la rivière Miramichi Nord-Ouest pour frayer ont donné une moyenne de 22 000 poissons matures, chaque année, depuis 2001. Ce total représente une modeste hausse par rapport au creux d’environ 4 000 par année pendant la période de 1998 à 2000, mais il n’est pas aussi élevé que le sommet de 50 000 géniteurs observé au milieu des années 1990. Deuxièmement, des analyses des prises accidentelles de bars rayés au cours de la pêche du gaspareau dans la rivière Miramichi Nord-Ouest ont indiqué une moyenne de 84 bars par filet par jour au cours des cinq dernières années; ces résultats ont été reliés à l’estimation de l’effectif de la population à partir des expériences de marquage et de recapture. Enfin, on a établi une faible corrélation entre le succès de la ponte, mesuré à l’aide des prises des jeunes de l’année au cours de la pêche de l’éperlan d’automne en eau libre (1991-1998) et des relevés à la senne de rivage (2001-2005), et l’estimation de l’abondance des géniteurs; ces résultats semblent montrer que le succès des classes d’âge peut être déterminé par des facteurs environnementaux. Les indicateurs secondaires de la situation, tels que la répartition tronquée et inchangée selon l’âge et la taille des bars en frai depuis le début des années 1990, appuient l’estimation élevée du taux de mortalité naturelle de 0,54 à 0,59 établie pour cette population après la fermeture de la pêche en 1996. Les études de marquage continuent de définir l’ensemble du sud du Golfe comme la zone occupée par cette population de bar rayé.

Un modèle discret du cycle biologique a été utilisé afin de proposer des niveaux de référence pour le rétablissement de la population du sud du Golfe. Les objectifs de rétablissement correspondent aux points de référence de l’approche de précaution pour les zones essentielles, prudentes et saines. Nous proposons une valeur S_{opt} de 21 600 géniteurs comme limite de rétablissement pour le bar rayé du sud du Golfe et une valeur de 31 200 géniteurs représentant 50 % des géniteurs par recrue comme cible de rétablissement, celle-ci étant la valeur utilisée pour la gestion de toute pêche dirigée. La valeur de S_{eq} (géniteurs de remplacement en l’absence d’exploitation) a été estimée à 63 000 poissons. Nous examinons la nécessité d’appliquer des règles de conformité et proposons une fenêtre mobile de 6 ans comme appropriée, l’objectif étant de dépasser la limite de rétablissement en 5 ou 6 ans. Dans les conditions actuelles, y compris les prises accidentelles de jeunes de l’année et le prélèvement illégal continu de bars adultes, il subsiste une faible probabilité (18 %) que le bar rayé du sud du Golfe dépasse la limite de rétablissement d’ici 2015. Si le taux de mortalité totale des adultes est réduit à $Z = 0,6$ par rapport à la situation actuelle de $Z = 0,8$ et que les prises accidentelles de jeunes de l’année sont éliminées, la probabilité que la population dépasse la limite de rétablissement d’ici 2015 serait de plus de 95 %.

La rivière Miramichi Nord-Ouest demeure la seule frayère confirmée du bar rayé dans le sud du Golfe. Puisque le bar rayé occupe tout le sud du Golfe, mais continue néanmoins d’afficher une grande fidélité à la Miramichi Nord-Ouest, la colonisation ou l’établissement de nouvelles frayères n’est peut-être pas un objectif de rétablissement réaliste.

Il n’a pas été possible de faire une estimation quantitative de la mortalité pour chacune des principales menaces au rétablissement de cette population. Les prises illégales constitueraient la plus grande limitation au rétablissement de cette population. La mortalité totale cumulative ne

semble pas menacer la survie de la population, mais, dans les conditions actuelles, le rétablissement au-delà de la limite proposée est peu probable.

Les mesures d’atténuation sont décrites. Les efforts de rétablissement du bar rayé du sud du Golfe devraient porter principalement sur la réduction de la mortalité des adultes et des prises accidentelles de jeunes de l’année et sur la protection de l’habitat du bar rayé et des frayères dans le réseau de la Miramichi.

RATIONALE

The 2002 review of striped bass (*Morone saxatilis*) by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) resulted in the division of 3 “designatable units” (DU) with listings of extirpated for Quebec’s St. Lawrence population and threatened for those of the southern Gulf of St. Lawrence (southern Gulf) and the Bay of Fundy (COSEWIC 2004). If the Governor in Council accepts COSEWIC’s recommendations, striped bass will be listed on, and afforded protection under, Canada’s *Species at Risk Act* (SARA) by May/June 2007.

Section 73 of the SARA authorizes competent Ministers to permit otherwise prohibited activities affecting a listed wildlife species, any part of its critical habitat, or the residence of its individuals but only after certain preconditions have been met. DFO Science has developed the framework to evaluate these preconditions and determine whether or not incidental harm permits should be issued (DFO, 2004a). The results from the application of this framework to striped bass in the southern Gulf is presented in this document and forms part of the “Recovery Potential Assessment” (RPA) for the population.

The organization of this document reflects a series of questions that were posed in the remit for the striped bass RPA held in Moncton, between November 30 and December 2, 2005 (Appendix A). The series of questions were developed to address the requirements of the Section 73 permitting framework and were divided into 3 phases. Phase I covered an update of the species status and trends, an evaluation of recovery for the population and a general time frame to reach it. Major potential threats and possible mitigation measures were discussed in Phase II and III, respectively.

LIFE HISTORY

Mitochondrial and nuclear DNA analyses indicated that striped bass in the southern Gulf are distinct and isolated from neighbouring striped bass populations in the Bay of Fundy and the USA (Wirgin et al. 1993, Wirgin et al. 1995; Diaz et al. 1997; Robinson et al. 2004). Because the genetic make-up of Quebec’s St. Lawrence population was not determined prior to their extirpation (Beaulieu 1985), their relatedness to striped bass in the southern Gulf remains unknown.

Further evidence indicating the isolated nature of the southern Gulf population stems from conventional tagging studies initiated in the early 1980s. With the exception of one reported recapture from the Chesapeake Bay area (Hogans and Melvin 1984), no southern Gulf tagging study has produced a recaptured striped bass from outside of the region. Tags have been returned from as far north as Percé, Quebec (Bradford and Chaput 1996), and tags placed on striped bass in the Margaree River Cape Breton, have been recaptured on the spawning grounds of the Northwest (NW) Miramichi. None of the nearly 10,000 tags applied to southern Gulf bass have been returned from waters of the Bay of Fundy. Recent recaptures from locations in Gulf Nova Scotia are updated in the “Secondary Indicators of Status” section under “Area of Occupancy”.

Striped bass in the southern Gulf are anadromous and return annually to the NW Miramichi River to spawn. Despite recent attempts to locate additional spawning grounds (Robinson et al. 2001; AVC Inc. 2003), the NW Miramichi River remains the only location in the southern Gulf where striped bass eggs and larvae have been collected (Robichaud-LeBlanc et al. 1996). During an early study of striped bass in the southern Gulf, Hogans and Melvin (1984) report on a spawning population in the Kouchibouguac River situated in southeastern New Brunswick. Their conclusion of a spawning population was based on the observation of presumed spawning activity for 3 days during the second week in May but was not confirmed with the collection of eggs or larvae. Furthermore, 8 striped bass tagged during early May of the same study were recaptured only weeks later during the spawning run to the Miramichi system (Hogans and Melvin 1984). Finally, without evidence of eggs or larvae, Rulifson and Dadswell (1995) speculated on 4 southern Gulf rivers (besides the Miramichi) believed to sustain a spawning population of striped bass: Nepisiguit, Tabusintac, Kouchibouguac, and Richibucto.

Age to sexual maturation for southern Gulf bass is believed to occur at ages 4 to 5 for females and earlier for males at ages 3 to 4. Fecundity analyses of southern Gulf striped bass have been limited to a single study involving 8 females captured during the early spring of 1983 in the Kouchibouguac River (Hogans and Melvin 1984). From these data, a female striped bass with a fork length of 50 cm would produce about 96,000 eggs, similar to about 89,000 eggs that a Shubenacadie-Stewiacke bass of the same length would produce (Paramore 1998). The high fecundity of striped bass is critical to compensate for the high mortality incurred by the eggs after release. Spawning occurs in early June only weeks after ice out when river temperatures are nearing 15^{o}C (Scott and Scott 1988). A gradual increase in water temperature is believed to promote spawning and sudden drops in water temperature can have devastating effects on egg and larval survival (Rutherford and Houde 1995). Many researchers believe that striped bass recruitment is largely dependant on the conditions experienced by eggs and larvae during the first weeks after spawning (Rutherford and Houde 1995; McGovern and Olney 1996; Ulanowicz and Polgar 1980).

Spawning occurs near the head of tide and at the surface of the water. The spawning act is obvious and can vary from a gentle swirling motion of several fish, to an aggressive behaviour that splashes water high into the air. The eggs and milt are broadcast simultaneously by the females and males respectively, and fertilization occurs in the water column. The eggs are semibuoyant and need to remain in suspension until hatching is complete. Depending on water temperatures and conditions, eggs require 48 to 72 hours to hatch (Peterson et al. 1996; Scott and Scott 1988). Robichaud-LeBlanc et al. (1996) found the highest concentration of striped bass eggs and larvae directly upriver of the salt wedge in the NW Miramichi.

Young-of-the-year (YOY) striped bass remain in the mid-channel portion of the NW Miramichi until the end of June (Robichaud-LeBlanc et al. 1998). Larval yolk reserves get exhausted within 14 days post-hatch at which time exogenous feeding begins. By early July, underyearling bass have moved to nearshore habitats of the estuary where they will remain and grow rapidly during the summer, attaining lengths between 8 and 20 cm by October (this document). Young-of-the-year exhibit a downstream range extension throughout the summer, and by late July, can be captured along coasts both north and south of the Miramichi system (Robichaud-Leblanc et al. 1998; Douglas et al. 2003; Robinson et al. 2004). Underyearling bass of Miramichi origin are abundant in many estuaries of the southern Gulf by the end of their first growing season.

After summer feeding migrations, both YOY and adult striped bass either remain or return to estuaries in the fall to spend the winter. Striped bass of every age and size are known to overwinter in many southern Gulf estuaries. The winter season is considered stressful as these fish remain under the cover of ice and fast during that time. It appears that YOY striped bass in the southern Gulf which do not attain a fork length of 11cm during their first growing season have poor overwinter survival (Bernier 1996).

FISHERIES MANAGEMENT

Striped bass in the southern Gulf are managed as a single unit. The sale of striped bass bycatch from commercial fisheries was prohibited after 1996 and when the interim conservation requirement of 5,000 female striped bass was not achieved between 1997 and 2000, recreational and First Nation fisheries were suspended. Those regulations remain in effect.

1 – INDICATORS OF STATUS

Three primary and five secondary indicators of status were explored for southern Gulf striped bass. Deviations and updates from the last assessment (Douglas et al. 2003) are discussed.

PRIMARY INDICATORS OF STATUS

1. Spawner abundance – Mark and recapture estimates

The gaspereau (*Alosa aestivalis* and *Alosa pseudoharengus*) fishery of the Northwest Miramichi has been used to assess the spawning run of striped bass to that river since 1993. Detailed sampling protocols are available from previous assessments (Bradford et al. 1995; Bradford and Chaput 1998; Douglas et al. 2001) but can be briefly summarized by efforts to mark adult striped bass early in the year prior to spawning (mid May) and the subsequent monitoring of their bycatch throughout the remainder of the gaspereau fishery. Biological characteristics of the population including fork length (nearest mm), age, and sex were also collected annually during gaspereau trapnet monitoring.

Two indicators of stock status from this monitoring program include a spawner abundance estimate based on mark-recapture and another on catch per unit effort (CPUE). Start dates for mark-recapture experiments have been similar throughout the time series while end dates were chosen on the basis of a decline in overall striped bass catches, a decline in the number of recaptured striped bass, and an increase in the number of spent fish (Table 1.1).

Peak spawner abundance was estimated at 50,000 fish in 1995 but fishery harvests during that same year accounted for most of the decrease into 1996 (Bradford and Chaput 1997) (Fig. 1.1). The collapse between 1995 and 1996 and the low spawner abundance estimates during the 1998-2000 period prompted closures of the commercial, recreational, and Aboriginal fisheries. The complete closures of all fisheries have likely helped the population stabilize to an average of 22,000 fish in each of the last five years (2001-2005) (Table 1.2). Strongest contributions to the spawning run of striped bass in recent years have been from the 2001 year-class (Fig. 1.2). The contribution of the 2002 year-class in 2005, primarily as males, was weak and provides little hope for a strong showing of the female component of this year-class in 2006 and beyond (Fig. 1.2).

2. Spawner abundance – CPUE-commercial gaspereau fishery

An annual index of abundance was derived using a general linear model treating striped bass catch per 24 hours per trapnet as the response variable and year as the explanatory variable. The log link was used because a Poisson distribution was assumed for the response variable (Venables and Ditchmount 2004). The analysis was performed using PROC GENMOD in SAS and the variance of the estimates was generated using a Pearson chi-square correction for overdispersion (SAS 2005). This is an alternate treatment from that presented in Douglas et al. (2003) and is considered more appropriate for CPUE data (Maunder and Punt 2004).

Striped bass catches are highest in late May and early June and decline rapidly by mid June after spawning is complete (Fig. 1.3). There has been limited control on the timing of the collection of the fishery data relative to the striped bass spawning period in the NW Miramichi but it is assumed that the peak spawning period was encompassed in all years. We explored three groupings of the data: 1) data from the entire sampling period, 2) data constrained to end on the date used for the mark-recapture experiment, and 3) data constrained to the period encompassing the upper quartile of the catch rates (Table 1.1). The third option was considered because of the interest in estimating the maximum abundance of spawners and the difficulty in collecting abundance data for that time period only.

The abundance of bass in the gaspereau fishery of the NW Miramichi has important within season and annual variability (Table 1.3; Fig. 1.3). Regardless of the method used to group the data, peak abundance was estimated for the years 1994, 1995 and 2001 to 2003 and the lowest abundance in 1993 and the 1996 to 2000 time period (Fig. 1.4). The median annual catch rate (CPUE) was positively correlated with the spawner estimates derived from mark-recapture experiments (Fig. 1.5) with 1995 being an outlier year. When the 1995 data point is excluded, the correlation is greater than 0.8.

Sources of Uncertainty

Mark recapture estimates of spawner abundance for the period 2004-2005 are considered to be conservative. It is believed that many striped bass had moved through the fishing area and were on the spawning grounds before a large number of tags could be applied. Fewer striped bass were recaptured in 2004 and 2005 than in previous years which resulted in wider confidence intervals than the previous 3 years (Table 1.2).

Striped bass of all sizes were harvested in the gaspereau fisheries of the Miramichi system between 1993 and 1995 (Table 3 in Bradford and Chaput 1998). Neither the mark and recapture estimate nor the abundance index estimate has been corrected for removals from the fishing area in those years.

Estimates of spawner abundance only reflect the number of adult striped bass returning to the NW Miramichi to spawn and do not represent the entire biomass of the population. Not all adult striped bass return to the NW Miramichi to spawn on a yearly basis. Adult striped bass have been sampled in the Hillsborough River PEI (AVC Inc. 2003) and the Kouchibouguac River (Hogans and Melvin 1984) during May and June and fishery officers report striped bass in spring gaspereau fisheries of Kent Co., the Tracadie, Pokemouche, and Margaree rivers, Pictou harbour, and Wallace Bay, NS (Chiasson et al. 2002). Results from an acoustic tracking survey in 2003-2005 indicated that 6 of 26 striped bass that overwintered in the Miramichi system left soon after ice out and did not return to spawn later in the season (this document) (Table 1.4).

3. Young-of-the-year abundance as index of spawning success

Indices of YOY in the fall and in the summer have been obtained over different time periods in the Miramichi. An abundance index of YOY in the fall was obtained from sampling the bycatch of the commercial open-water rainbow smelt (*Osmerus mordax*) fishery of the Miramichi estuary during 1991 to 1998. A summer beach seine index has been adopted since 2001.

Fall YOY Index

Details of the sampling procedure for the fall smelt fishery were described in Bradford et al. (1997b). Sampling in 1991 commenced later and was less intense than subsequent years. Sampling ceased after 1998 following a change in management which delayed the opening of the smelt fishery in the Miramichi to November 1, a measure which was considered by the fishermen to have been a direct consequence of the science related activities to monitor bycatch. An annual index of abundance was derived using a general linear model treating the catch per 24 hour per net as the response variable and year as the explanatory variable. The log link was used because a Poisson distribution was assumed for the response variable (Venables and Ditchmount 2004). The analysis was performed using PROC GENMOD in SAS and the variance of the estimates was generated using a Pearson chi-square correction for overdispersion (SAS 2005).

Catches of YOY in the smelt fishery declined through the season in some years but showed important annual variability (Fig. 1.6). Peak abundance was estimated for the years 1995 and 1996 with the lowest abundances in 1993, 1997 and 1998 (Fig. 1.6). The mean annual catch rate (CPUE) is positively correlated (R = 0.66) to the female spawner estimates derived from mark and recapture and less so for the total spawner abundance (Fig. 1.7). When female spawner abundance was at or above 5,000 fish, there was a high YOY index in the fall smelt fishery, as previously indicated by Bradford and Chaput (1997). This observation supports the premise that spawner abundance is an important component of recruitment to the fall YOY stage of striped bass.

Summer Beach Seining Index

Beach seining at index sites of the Miramichi began late in the 2000 season and only complete surveys between 2001 and 2005 were used in the analysis. The same five or six beaches were seined on a weekly basis during the months of July and August in all 5 years. Sites covered the Miramichi estuary from nearly complete freshwater of the upper estuary to sites with salinities around 20 ppt of the lower estuary at Loggieville where the river widens into Miramichi Bay. The seine was fabricated of 6.4mm mesh, measured 30m long x 1.8m deep, and was equipped with a cone shaped bag in the centre that measured 1.8x1.8x1.8m. Single sweeps were made at each site during day light hours and high tide was often targeted. When conditions permitted, half of the seine was pulled into the water perpendicular to shore then swept in an arc formation back to shore.

Catch per unit effort analyses were restricted to the July surveys only because 1) YOY are readily captured in nearshore habitats of the Miramichi by this time, 2) most YOY have not yet extended their distribution outside of the Miramichi system, and 3) catches of YOY by beach seine in the Miramichi substantially decrease by August.

Mean CPUE estimates were highly variable between years ranging from a high of 139 YOY per sweep to a low of 4 YOY per sweep in 2003 and 2004, respectively. Captures of YOY striped bass in 2004 were so few that plans to collect 2,000 individuals for the St. Lawrence estuary (Quebec) restocking initiative were cancelled, while collections were completed in 2002, 2003 and 2005. Further evidence of a very low abundance of YOY (year-class failure) in 2004 were provided from the DFO sponsored Community Aquatic Monitoring Program (CAMP) which reported the capture of only one YOY outside of the Miramichi system during region wide beach seine surveys (J. Weldon DFO pers. comm.).

Several more years of beach seine data will be required to determine the correlation between YOY and spawners but the limited data set indicates that environmental factors may play an important role in year-class success. Good numbers of female spawners in 2002 and 2004 produced the lowest YOY CPUE estimates of the time series (Fig. 1.8). Furthermore, the poor YOY index in 2002 appears to have manifested itself with the poor recruitment of 3 year old fish in 2005 (Fig. 1.2). These data would agree with several US studies that have demonstrated that recruitment is largely determined in the first few days after spawning as a result of variable environmental conditions affecting survival (Richards and Rago 1999).

SECONDARY INDICATORS OF STATUS

1. Size structure

Fork lengths of adult striped bass sampled on the spawning grounds of the NW Miramichi have not changed during the 1993-2005 monitoring period (Table 1.5). Striped bass with fork lengths of 40-50 cm are most abundant in Miramichi samples and similar to fork lengths reported by Chaput and Robichaud (1995) from samples collected at Millbank (Miramichi) between 1975-1982, and from samples collected in the Kouchibouguac River during the early 80's (Hogans and Melvin 1984). Occasionally striped bass with fork lengths between 65 and 75 cm are sampled and only rarely from fish > 75 cm. Complete fishery closures in the 1996-2000 period have not produced an increased length or age distribution.

2. Sex ratio

Male striped bass have nearly always outnumbered female striped bass on the spawning grounds of the NW Miramichi (Table 1.2). This phenomenon can be partly explained by the earlier maturation schedule of males at age 3 versus females at age 4. Furthermore, striped bass that are not ripe enough to expel either milt or eggs at the time of sampling are considered to be female. The proportion of males declines with age and males are rare beyond age 6 years (Table 1.6).

3. Age structure

The age structure of striped bass sampled during their spawning migration to the NW Miramichi between 1994 and 2005 is predominantly comprised of 3 to 5 year old bass (Table 1.7). The closure of fisheries between 1996-2000 has had little effect on the age distribution of this population of striped bass. Striped bass greater than age 6 are rare and those over age 10 are negligible.

4. Mortality estimates

Adult (age 3 and older) mortality

Scales collected from adult striped bass during their spawning migration between 1994 and 2005 were interpreted for ages. Field sampling during the week prior to the main run of gaspereau was considered to be the most representative of the age and length composition because fishermen were often contracted to provide access to striped bass with the agreement that complete catches could be sampled. For years 1998-2005, only ages interpreted from scales taken during the initial sampling events were used to determine proportion at age. For 1994 to 1997, proportion at age was determined using age-length keys from data taken throughout the sampling season applied to the lengths taken during the initial sampling events.

The numbers of spawners at age were calculated using the proportion at age and the spawner abundance as estimated either by mark-recapture or CPUE analysis (see Primary Indicators of Status above). The instantaneous mortality rate (Z) was calculated from the standard equation described by Ricker (1975):

$$Z = -Ln\left(\frac{N_{t,i}}{N_{t-1,i-1}}\right)$$

where $N_{i,t}$ = abundance of spawners at age i in year t

Annual survival (range 0 to 1) is calculated as e^{-Z} and annual mortality is 1 – S.

Although the commercial fishery has been closed since 1996, mortality estimates after that date contain an important but undetermined amount of fishing mortality (F). Illegal harvests of striped bass are believed to be substantial throughout the southern Gulf (see Phase II below). Because there are no legal directed fisheries on striped bass adults, we consider the mortality estimates presented here as equivalent to natural mortality and indicative of the underlying conditions of recent years.

Numbers at age derived from spawner abundance estimated either by mark-recapture or CPUE produced similar mortality rate estimates (Table 1.8). Positive values of Z ranged from a low of 0.07 to a high of 3.41, corresponding to annual rates of 7% to 97% (Table 1.8). Negative estimates of Z were frequent at age 3 and were not unexpected given the presumed maturity schedules for male and female bass at ages 3 to 5. There was a consistent bias in the spawner abundance estimates of 2000-2001 and 2004-2005 as evidenced by the negative Z values along the diagonals of the age by year-class matrix (Table 1.8). This bias is the result of either an underestimate of spawners for 2000 and 2004, an overestimate in 2001 and 2005, or both. Alternatively, proportionally more fish at age may have recruited to the spawning grounds in 2001 and 2005 relative to the previous years. Similar bias was noted for the CPUE series (Table 1.8).

Based on the average abundance at ages 3 to 7 years over the period 1997 to 2005, the mortality of adult striped bass is in the order of 0.5 to 0.6 (Z = 0.8 to 0.9; Fig. 1.9). Mortality estimates of recent years were as high as those during the period 1994-1996 when striped bass were commercially exploited (Fig. 1.10). Recruitment of strong year-classes is obvious at age 3 and often increases the following year when the age 4 female component returns to spawn for the first time. By age 5, year-class abundance is diminishing rapidly and nearly nonexistent after age 7. The high mortality estimates are consistent with the observed truncated age and length distributions of spawners in the NW Miramichi.

Natural mortality factors

The southern Gulf of St. Lawrence striped bass is the most northern spawning population in North America (Douglas et al. 2003). Environmental conditions of the southern Gulf are characterized as relatively warm in the summer with cold winters defined by extensive ice cover inshore and offshore and complete ice cover in rivers and estuaries for upwards of four months (December to March). These conditions pose particular challenges to striped bass populations in Canada that are not experienced by populations along the eastern seaboard of the United States. Adult and juvenile bass overwinter in the upper portions of estuaries where feeding is believed to cease when temperatures fall below 10°C in October-November (Robichaud-LeBlanc et al. 1997). The overwinter survival depends upon obtaining sufficient energy reserves before the period of fasting and upon suitable temperature/salinity conditions for osmoregulation (Hurst and Conover 1998). Both juvenile and adult bass are subjected to overwinter mortality. YOY striped bass (13.2 - 15.1cm) were retrieved from frozen-over holes drilled in the ice during a winter tracking study in late winter 2004; the cause of their death or an explanation of the circumstances that lead to their entrapment in an 8 inch auger hole is unknown. Kills of striped bass associated with the loss of a thermal refuge in the cooling tail race of a generating station in Trenton (Nova Scotia) during the winter of 2004 provide evidence of the susceptibility of all size groups of bass to sudden changes in conditions and exposure to lethal temperatures in the winter. Adult bass mortalities have been reported from some southern Gulf estuaries shortly after ice-out in the spring.

The overwintering period may represent a specific constraint on survival of striped bass. The causes of overwinter mortality could include starvation, size-dependent predation, or physiological intolerance to reduced temperatures (Sogard 1997). It has been suggested that the survival of young-of-the-year striped bass in the first winter may be conditioned by the size attained at the end of the first growing season (Chaput and Robichaud 1995), as was observed in white and yellow perch (Johnson and Evans 1991). Preliminary work provided evidence in support of size-based overwinter survival in striped bass of the Miramichi River (Bernier 1996), with few bass less than 10 cm fork length at the end of the first year of growth (quantified by back-calculation from scales) estimated to have been present in the survivors sampled at two years of age. Chaput and Robichaud (1995) presented back-calculated size at age for male and female striped bass in which few fish were estimated to have been less than 10 cm fork length at the end of their first year. Environmental conditions which would therefore affect size of YOY into the first winter are of interest.

Winter conditions

High discharge events during the winter period (December to March) have presumably displaced juvenile bass downstream to locations in Miramichi Bay which they would normally not inhabit during those months (Hanson and Courtenay 1995). This displacement may expose bass to sub-zero temperatures and osmoregulatory stressful conditions. Mean winter air temperatures (Nov. to March) in the Miramichi area ranged between -8 to -4ºC during 1961 to 2003 (Fig. 1.11; Appendix B). The warmest period of the time series was observed in 2001/02 resulting from a succession of four winters of warmer conditions commencing in 1997/98 (Fig. 1.11). Mid-winter (Jan. to March) daily peak flows in the Miramichi are generally less than 100 $m^3 s^{-1}$ with very high flow events (>= 400 $m^3 s^{-1}$) recorded in 7 of 42 winters between 1962 and 2003, four of these were recorded between 1996 and 2000, corresponding to the warmer recent period (Fig. 1.11). The potential to displace bass into cold and stressful osmoregulatory conditions was particularly acute in the recent decade.

There is a significant negative correlation between mean winter air temperature and the number of days of ice cover in the Miramichi River; an increase in mean air temperature of 1ºC reduces the duration of ice cover by about one week (Fig. 1.12; Appendix B). There is large annual variability in the duration of ice cover, ranging from a low of 102 days in 1999/2000 to a high of over 170 days during 1972/1973 (Fig. 1.12). Duration of ice cover was somewhat higher in the 1960s and 1970s than in the recent decade. Years of shorter ice duration may not necessarily be positive for striped bass survival as increases in water temperatures may result in a more rapid depletion of energy reserves. A number of studies have shown that size-dependent survival may be expressed during intermediate winter conditions rather than during mild or extremely severe winters (Sogard 1997 and references within). The association between overwintering conditions and survival of southern Gulf striped bass is unknown.

Preliminary studies on size dependent overwinter survival of striped bass from the southern Gulf suggest that few YOY striped bass less than 10 cm fork length survive their first winter (Bernier 1996). An analysis of the length distributions of YOY bass in the fall compared with back-calculated lengths of the survivors at age 2 years of the same cohort show a shift in size distribution pre and post winter in two of the three years with no observed shift in the size distributions of the 1991 cohort which was large bodied (Fig. 1.13). Assuming that only size-dependent mortality was occurring in the first winter, the relative losses of the cohorts ranged from 0% for 1991, to over 80% in 1992 and 1993. Based on the observed length frequency distributions of YOY bass between 1991 and 1998, the additional size-dependent mortalities over the first winter would have been most important for the 1992, 1993, and 1996 cohorts and least important for the large-bodied 1991, 1995, 1998, and 1999 cohorts (Fig. 1.14).

Summer conditions

The size of YOY striped bass at the end of the growing season is largely determined by the conditions favourable to growth. Growth is strongly correlated to water temperature (Dey 1981; Secor et al. 2000). An index of growth potential for striped bass YOY was developed using the air temperature time series from Chatham (Miramichi). Indices based on mean summer temperature and degree days (June to September) are strongly correlated (Fig. 1.15). The mean summer temperature for the period 1960 to 2003 was 17.0ºC, within a range of 14.9ºC in 1986 to a high of 18.7ºC in 1999. The modal length of juvenile bass in the fall of 1991 to 1999 is positively correlated to the mean air temperature and degree day indices with the large bodied 1999 cohort associated with the maximum mean temperature and degree days indices of the time series (Fig. 1.15). Faster growth and larger body size are expected to be positive for survival of bass during their first year.

Early life stage survival

The high fecundity, early age at maturity and iteroparity features of striped bass are adaptive traits indicative of high early life stage mortality. Year-class variability in striped bass has been observed to be high and largely determined during the egg and larval stages and influenced by environmental factors (see references within Richards and Rago 1999). Instantaneous daily rates of mortality ($M\ d^{-1}$) between the egg and the 8mm larval stage have been estimated to vary between 0.11 and 0.34 with survival after 20 days varying between 0.03% and 11% (Rutherford et al. 1997). Rutherford and Houde (1995) indicated that spawning success of striped bass in eastern U.S. was largely dependent on water temperatures. They reported that a storm which lowered river temperatures below 12ºC caused complete mortality of eggs and larvae and eliminated more than 50% of the season’s production. Larval growth was positively correlated to water temperatures. There is minimal information with which to examine early life stage survival variability in the Miramichi River. A cooling event in late May 2004 on the spawning grounds is suspected of having contributed to high mortality of eggs and early larvae and subsequently a low index value of YOY abundance in the summer beach seine survey (Fig. 1.8).

Parasites disease

Philometra rubra is a common nematode occurring in striped bass in the southern Gulf (Hogans 1984). It is possible that striped bass weakened or compromised by infection of *P. rubra* can succumb when subjected to other stresses such as secondary bacterial or viral infection, unusually cold water or pollution (J. Melendy DFO parasitologist pers. comm.).

Three dead YOY and two adult striped bass were collected before or just after ice out from the Miramichi estuary. DFO’s fish health unit positively identified the North American strain of the viral hemorrhagic septicaemia virus (VHSV) and nodavirus in all five fish. VHSV could not be confirmed as the cause of death for these fish.

Lymphocystis is a common chronic and usually non-fatal infection caused by an iridovirus that results in uniquely hypertrophied cells of the skin and fins. The condition is much like that of warts in that the lesions are macroscopic, occur mostly in the periphery of the vascular system and have a cauliflower appearance. Transmission of the disease is facilitated by increased fish density, trauma during spawning, netting or tagging practices, pollution, or disruption of the protective mucous layer by external parasites. Striped bass infected with lymphocystis are common in the southern Gulf and infection rates of 1% to 8% have been noted in May on the spawning grounds of the NW Miramichi between 2001 and 2005.

5. Area of occupancy (Fig. 1.16)

Douglas et al. (2003) presented tagging information that indicated striped bass utilized the entire southern Gulf. Tagging studies of southern Gulf striped bass have shown no mixing with the Bay of Fundy populations and a distribution of recaptured fish between Percé, Quebec and the Margaree River in Cape Breton (Fig. 1.16). More recently, 3 of 57 striped bass tagged in Wallace Bay, NS in the fall 2001 were recaptured on the spawning grounds of the NW Miramichi in June 2003 (n=2) and June 2004 (n=1). Additionally, 1 of 29 bass tagged in East River, Pictou Co. NS during the autumn of 2002 and 2003 was recaptured in each of the 2003 and 2004 spawning runs to the NW Miramichi.

Recent surveys in the Kouchibouguac and Richibucto rivers (Robinson et al. 2001), the Tabusintac River (DFO unpublished) and the Hillsborough River in PEI (AVC Inc. 2003) have failed to find evidence of striped bass eggs and larvae. The only confirmed spawning location for striped bass in the southern Gulf remains the NW Miramichi.

Recent studies tracking the movements of spawners in the Miramichi provide further evidence of the discrete spawning site in the southern Gulf. Eight striped bass implanted with acoustic pingers monitored on the spawning grounds of the NW Miramichi between May 29 and June 21 were subsequently detected off the coast of Val Comeau (north of the Miramichi) between June 18 and July 3, 2004. The northward post-spawning migration from the Miramichi corroborates all of the information (published and anecdotal) on the presence of striped bass in northern areas in early summer but discounts the notion of spawning at any of those locations (Rulifson and Dadswell, 1995). One of the eight striped bass detected at Val Comeau was detected approximately one month later in Bay of Chaleur on July 15 during a simultaneous Atlantic salmon (*Salmo salar*) smolt tracking study (P. Brooking ASF pers. comm.).

YOY continue to move out of the Miramichi system only weeks after spawning and occupy much of the southern Gulf by the end of their first growing season. In 2003, YOY striped bass were captured by beach seine in the surf off Miscou Island and at the mouth of the Little Buctouche River by mid August in northern and southern New Brunswick, respectively (Fig. 1.16).

RECOVERY

ISSUE

The striped bass population in the southern Gulf met COSEWIC's criteria (Appendix C) for Endangered B2ac(iv), but was designated as threatened, B2ac(iv); D2, and "because of the high degree of resilience evident in recent spawner abundance estimates" (COSEWIC 2004). COSEWIC's threatened designation was largely attributed to the single spawning location for striped bass in the southern Gulf which is well below the "Threatened" criteria of <10 and the "Endangered" criteria of < 5 (Appendix C). Additionally, the large number of striped bass confined to the Miramichi system each spring, increases their susceptibility to "the effects of human activities or stochastic events and becoming highly endangered in a very short period of time" (COSEWIC 2004). Fluctuations in the number of mature individuals (Criterion B2c(iv)) also factored into their assessment.

SMALL DISTRIBUTION

There is presently no information that would indicate that there is, or was, more than one spawning location for striped bass in the southern Gulf. Hogans and Melvin (1984) and Rulifson and Dadswell (1995) speculate on other spawning populations in the southern Gulf based on the presence of adults and juveniles in various estuaries other than the Miramichi. The presence of adult or YOY striped bass alone in southern Gulf estuaries is insufficient evidence for multiple spawning areas. Recent studies have shown that underyearling bass extend their distribution out of the Miramichi River and into neighbouring estuaries soon after hatching (Robinson et al. 2004; Douglas et al. 2003) and adult striped bass tagged throughout the southern Gulf have been recaptured on the spawning grounds of the NW Miramichi (this document). Adult striped bass are found throughout the year in many southern Gulf estuaries but no spawning has been documented outside of the NW Miramichi. Directed monitoring for striped bass eggs and larvae has been limited (Douglas et al. 2003).

Striped bass have colonized new areas, either naturally (since the last ice age) or through human intervention (Scofield 1931). Striped bass continue to demonstrate high fidelity to the NW Miramichi despite their presence in, and potential colonization of many estuaries of the southern Gulf. The establishment of several new spawning locations in the southern Gulf is not likely a realistic recovery objective for this population of striped bass. An experiment is presently taking place to re-introduce striped bass into the St. Lawrence River. Should the experiment be successful, it will provide evidence that where habitat is appropriate, striped bass from the southern Gulf have retained the capacity to establish self-sustaining populations. Since the stock for the re-introduction program is from the Miramichi, this would provide a second spawning location for the southern Gulf striped bass unit.

The single spawning location for southern Gulf bass will always meet COSEWIC's "Endangered" criteria for Small Distribution (Appendix C). COSEWIC assessed the shortnose sturgeon (*Acipenser brevirostrum*) with its similar limited distribution and single spawning location in the Saint John River, New Brunswick (COSEWIC 2005). The COSEWIC criteria for small distribution were not strictly adhered to for the shortnose sturgeon and the species' was designated as one of "Special concern" (COSEWIC 2005).

FLUCTUATION IN MATURE INDIVIDUALS

Large fluctuations in abundance of striped bass on the spawning grounds in the southern Gulf during 1993 to 2005 are, in part, associated with the removal of over 30,000 animals during the 1995 and 1996 fishing seasons. Year-class variability has been observed to be high in this species, determined largely within the first few weeks during the egg and larval stages and influenced by environmental factors (see references within Richards and Rago 1999). It has been acknowledged that recovery of juvenile production is not guaranteed by increased spawning stock, but in the Chesapeake Bay experiment, increases in spawning biomass resulted in improvements in recruitment. The high levels of spawners were considered to have been a major factor in the establishment of above average year classes in two of the eight years (Richards and Rago 1999).

RECOVERY DEFINITION

A national workshop was recently held to consider the issue of what comprises "recovery" for aquatic species (DFO 2005). The intent of the workshop was to develop guidelines to help science advisors in the provision of consistent interpretations of recovery in the development of recovery plans. The SARA does not define recovery but expert groups must reach a consensus on the biological characteristics which would constitute recovery of the species or populations. The workshop participants concluded that the "precautionary framework" consisting of three zones (healthy, cautious, critical) appears suitable as a starting framework for incorporating recovery definitions. The discussions of the workshop centered around where the recovery definition would be placed relative to the precautionary framework zones as well as the biological attributes which might be used to characterize recovery. There was a strong consensus from the workshop that recovery would be well above the level which would ensure that COSEWIC considers the population neither Threatened nor Endangered. The qualitative conclusion from the workshop appears to have been that recovery plans which aim to increase abundance, for example, to the cautious-healthy boundary in the precautionary framework would most likely be acceptable as a definition of recovery by species assessment committees such as that of the COSEWIC (DFO 2005).

The attributes used to define recovery and assess status relative to the recovery objective should be defined in terms of quantities which can be monitored. The workshop considered that direct measures of abundance and total range occupied would be the preferred currencies for specifying recovery objectives because these correspond to COSEWIC assessment criteria and to reference objectives used in fisheries management. Supplementary attributes could include fragmentation or recovery of habitat, age and size composition, and genetic diversity.

Defining when a species or population is recovered also requires a compliance rule, i.e. how consistently the attribute(s) remains in the recovered state. Requiring that the attribute(s) always be above the recovery level before the species or population can be considered "recovered" is synonymous with treating the reference level as a limit. The compliance rate in these situations has to be very high, i.e. a very low probability (<5%) of the attribute falling below the limit. On the other hand, if it is acceptable that the attribute be sometimes above or sometimes below the reference level without a trend, then that is a candidate for a target, i.e. an objective to aim for with a probability of attainment of about 50%. It could be argued that recovery as a limit may be the point where a population or species is assessed as being above threatened or endangered whereas recovery as a target would be when the status is assessed to be above "special concern". A desirable feature of the recovery objective and its compliance rule is that the species status assessment be robust to uncertainty in assessments and dynamics and have sufficient inertia to preclude rapid and frequent changes in status.

In the eastern United States, recruitment overfishing was implicated as a factor in the decline of striped bass and an intensive fisheries management plan was put in place to restore the populations (Richards and Rago 1999). Projection models indicated that a fishing mortality (F) target of F= 0.25 was required for stock rebuilding and measures were introduced to reduce the fishing rates (Richards and Rago 1999). The plan had the defined objective of protecting 95% of the females of the 1982 and subsequent year classes until 95% had an opportunity to spawn at least once. A recovery attribute was defined based on an index of juvenile recruitment. The populations of striped bass were to be considered recovered (end point decision rule) when the juvenile index, calculated as the three-year running average, exceeded the approximate long-term average (1954 to 1984) of 8.0 fish per unit of effort (Richards and Rago 1999). The definition of a clear decision rule was critical in the implementation of the management plan but the decision rule had some shortcomings. It was realized later that the populations could be considered recovered after a single annual index of 24 was measured even if the two preceding indices were zero (Richards and Rago 1999). As well, there was no accounting of the precision of the index itself. An example of an alternative decision rule could have been a recovery value of the annual index other than the mean (for example, the median or a lower percentile) and a decision rule, for example, that would have required that the annual index be above a value for three consecutive years (Fig. 2.1). This would have been a more cautious rule to ensure that recovery status was not based on the presence of a single strong year class.

RECOVERY OBJECTIVES FOR SOUTHERN GULF STRIPED BASS

The potential and time frame for recovery were examined using a discrete life history model (Appendix D). Mortality, fecundity, and stock and recruitment dynamics were modeled using general life history information of the species and observed or assumed values specific to the southern Gulf striped bass. The choice of parameter values in the model were governed by observations on characteristics of the population and balancing of life stage abundances. The characteristics of the southern Gulf population of particular interest included:

- prior information on abundance of adult bass and spawners,
- relative age structure of the spawners, and
- sex ratio of spawners.

Specific assumptions and functional relationships are described in Appendix D.

Perceptions of historical abundance

The maximum recorded annual landing of southern Gulf striped bass was 61.4 metric tons in 1917 with the maximum in the most recent 30 years (1968 to 1996) of 47.1 metric tons (LeBlanc and Chaput 1991; Douglas et al. 2003). Landings were recorded from a large number of statistical districts and seasons. The annual mean weight of adult striped bass on the spawning grounds between 1994 and 2005 has varied from a low of 1 kg in 1994 to a high of 1.9 kg in 1996. This range of average weights was combined with a range of exploitation rates between 0.3 to 0.5 to estimate historical abundances of adult bass in the southern Gulf. High exploitation rates have been documented previously for southern Gulf bass when removals between May 1995 and May 1996 were in excess of half of the 1995 spawning stock (Bradford and Chaput 1997).

Using the historical maximum landing of 61.4 t, the abundance of adult-sized (3 year and older) striped bass in the southern Gulf was between 65,000 and 200,000 fish. The maximum recorded harvest of the last three decades would suggest a range of 50,000 to 160,000 fish. Estimates of the total abundance of bass age 3 years and older are not available for the southern Gulf because only the abundance on the spawning grounds in the NW Miramichi is estimated and only a portion of the striped bass aged 3 years and older are considered to be on the spawning grounds.

Estimates of the spawning stock

Estimates of the abundance of spawners in the Northwest Miramichi are available for the period 1993 to 2005. A peak abundance of over 50,000 bass was estimated in 1995 with a low of 3,400 fish in 1998 (Table 1.2).

Relative age distribution of spawners

During 1994 to 2005, the most abundant age groups on the spawning grounds have been age 3 and 4 year olds with 99% of the fish aged 3 to 7 years (Table 1.7). The oldest fish sampled was interpreted at 13 years old. This constricted age distribution for spawners is indicative of a high mortality rate. For this stock, the average instantaneous mortality of bass age 3 and older over all the years sampled is estimated at about 0.8 annually (Fig. 1.9). A rate of $M = 0.6$ applied to a stable age distribution starting at age 3 results in less than 2% of the population alive at age 11.

Sex ratios on the spawning ground

Sampling on the spawning grounds has consistently shown a disproportionate number of males relative to females (Table 1.6). Confirmed male bass (by extrusion of milt) have represented between 37% and 94 % of sampled fish. Confirmed female bass (as determined by extrusion of eggs or residual fluids) were generally low with the highest proportion (8 to 11%) of all samples in 2004 and 2005. The bias toward males on the spawning grounds is consistent with earlier maturation schedule for males and partial recruitment of females at all ages. Differential mortality among males and females may also produce this biased sex ratio but this dynamic was not explored in the discrete model.

Recovery objectives

We ran the life history model without stochasticity (Fig. 2.2) over a range of egg depositions to derive four spawning stock reference levels: spawners at equilibrium in the absence of fisheries (Seq), the spawning stock which produced the maximum gain (Sopt), spawning stock at a fishing rate which resulted in 50% and 30% spawning per recruit (50%SPR, 30%SPR). The mortality rate and life history parameters were assumed as:

$S_0 = 0.001$ and $YOY_{cap} = 1.5$ (million)
M = 1.5 for the six months of overwintering for YOY
M = 1 for age 1 bass
M = 0.6 for age 2 and older bass
Maturation schedule as in Appendix D
Fecundity as in Appendix D based on mean length, mean weight at age

The 30%SPR and Sopt have been defined as limit reference points (Mace 1994; Potter 2001). The 50%SPR level has been proposed as a precautionary reference point (Mace 1994). We propose Sopt as the recovery limit for the southern Gulf striped bass and spawners for 50%SPR as the recovery objective for directed fisheries. Under the equilibrium conditions, the loss in lifetime eggs due to fishing YOY in the fall occurs at an F value five times greater than the F value for fishing bass age one year and older (Figure 2.3).

Since the parameters for the Beverton-Holt compensatory function are not known, we ran the simulations under lower and higher average YOY production (1, 1.5, 2 million YOY capacity) and for lower and higher density independent survival (0.0005, 0.001, 0.002). Based on the prior expectation of adult abundance being in the range of 100,000 bass, the YOY production capacity of 1.5 million and the density independent survival rate of 0.1% were retained as suitable values for deriving the reference levels (Table 2.1). The S_{eq} value (spawners at replacement in terms of lifetime egg production) was estimated at 63,000 fish. The proportion female is 0.34.

We propose the S_{opt} value of 21,600 spawners as the recovery limit for the southern Gulf striped bass and the 50%SPR value of 31,200 spawners as the recovery target, the latter being the value for managing any directed fisheries (Table 2.2). We see no reason for using 30%SPR as the recovery limit because it involves higher fishing rates, lower abundance of both spawners and adult bass, and lower yield than S_{opt}.

COMPLIANCE RULES

The definition of a compliance rule was based on observed age structure on the spawning grounds and responsiveness to changes in status. In the context of a limit reference level, the compliance rule should respond rapidly but not necessarily to abundance falling below the limit. The rule would allow for rapid identification when the status falls below the limit but a slower response concluding that the resource is above the limit (see example in Fig. 2.1). A target reference level can be more responsive since it is an objective to aim for rather than a point to avoid (limit).

Based on the observed age structure on the spawning grounds (99% of spawners between 3 and 7 years), we suggest a six year sliding window to assess status. For the southern Gulf striped bass, we suggest that the reasonable compliance rule for the limit reference point be that the stock attribute is above the level in at least 5 of 6 consecutive years. The compliance rule for the recovery target is when the stock is above the target objective in at least 3 of 6 consecutive years. The status can be summarized using traffic lights (Caddy 2002) with the overall status of the resource determined first by its status relative to the limit reference point and secondly by the status relative to the target reference point conditional on the limit status.

Traffic light	Status	Compliance rule
Red	Below the recovery limit	Attribute < limit reference in > 1 of 6 years
Yellow	Above limit, below target	Attribute > limit reference in >= 5 of 6 years AND Attribute < target reference in > 3 of 6 years
Green	Above target	Attribute > limit reference in >= 5 of 6 years AND Attribute > target reference in > 2 of 6 years

The limit reference defines the border between the red and yellow traffic light zones. Red is assigned a value of 0, yellow a value of 1. The target reference level defines the border between the yellow and green zones with yellow a value of 1 and green a value of 2. The overall status is the product of the limit and target colour values with the red zone a product of 0, the yellow zone a product of 1, and the green zone a product of 2. An example of the application of this traffic light summary calculation is shown in Fig. 2.4.

Recovery attributes of abundance

Possible recovery attributes of abundance of striped bass in the southern Gulf include the number of spawners (see above), an index of the spawner abundance (CPUE index) and relative spawning success (juvenile indices). Additional attributes could include the age structure of the spawners (range of ages, relative contributions of year-classes).

Spawner abundance

Estimates of the spawning stock in the NW Miramichi are available for the period 1993 to 2005. A peak abundance of over 50,000 bass was estimated in 1995 with a low of 3,400 fish in 1998. Between 1994 and 2005, the spawner abundance has been consistently below the recovery limit. Based on the previous 5 years and the compliance rule, the stock will be below the recovery limit until at least 2009 (Fig. 2.4). Considering the uncertainty in the estimates of spawner abundance (Table 1.2), the lower confidence interval of the estimates (i.e. 97.5% chance that the abundance was greater than the lower interval) should be used to evaluate compliance relative to the recovery limit.

Sampling on the spawning grounds has consistently shown a disproportionate number of males relative to females. Bradford and Chaput (1998) indicated that the abundance of young-of-the-year striped bass as measured in the open-water fall smelt fishery increased dramatically when female spawner abundance was estimated to have been above 5,000 fish and this was subsequently suggested as an interim conservation level (Douglas et al. 2001). The interim conservation threshold of 5,000 females which has been used to close all directed fisheries on striped bass equates on average to about 15,000 spawners. As the estimate of female abundance is even more uncertain than the estimate for total spawners, we recommend using the latter as the spawner attribute.

Index of spawner abundance

An index of spawner abundance was obtained from the commercial gaspereau fishery in the Northwest Miramichi. Details of the sampling procedure and analyses are described in the section on Primary Indicators of Status above.

The mean annual catch rate (Ln(CPUE)) is positively correlated to the spawner estimates derived from mark and recapture with 1995 being an outlier year (Fig. 2.5). When the observation for 1995 is excluded, the correlation is greater than 0.9. Using the reference levels for spawners derived previously, the equivalent index for S_{opt} would correspond to a catch rate value of 4.02 (LnCPUE) or 56 bass per trapnet per 24 hour period whereas 50%SPR spawner reference level would be demarcated by a catch rate value of 5.15 (LnCPUE) or 173 bass per trapnet per 24 hour period (linear regression of LnCPUE on spawner abundance excluding the 1995 data point).

Using the traffic light boundaries established for the spawner estimates, the boundary for the red/yellow zone could correspond to a catch rate value of 3.8 (LnCPUE) or 46 bass per trapnet per 24 hour period whereas the yellow/green zone would be demarcated by a catch rate value of 6.2 (LnCPUE) or 486 bass per trapnet per 24 hour period (linear regression of LnCPUE on spawner abundance excluding the 1995 data point) (Fig. 2.5).

Index of spawning success

Indices of young-of-the-year in the fall and in the summer have been obtained over different time periods in the Miramichi (see section on Primary Indicators of Status above). The fall index was obtained from bycatch sampling of the commercial open water rainbow smelt fishery for the period 1991 to 1998. The mean annual catch rate (CPUE) is positively correlated (R = 0.66) to the female spawner estimates derived from mark and recapture but less so for the total spawner abundance. When female spawner abundance was at or above 5,000 fish, there was a high young-of-the-year index in the fall smelt fishery (Bradford and Chaput 1997) which supports the premise that spawner abundance is an important component of recovery and maintenance of striped bass.

A summer beach seine index has been developed covering a shorter but more recent time period (see section on Primary Indicators of Status above). Several more years of the summer index and spawner abundance estimates will be required before the functional relationship can be described. However, the data thus far illustrates that survival in the early egg and larval stages can dramatically influence recruitment. This was evidenced in 2004 when the summer YOY index was very low relative to the estimated abundance of spawners that year (see section on Primary Indicators of Status above).

TIME FRAME FOR RECOVERY

Recruitment for striped bass is considered to be strongly influenced by environmental factors (Richards and Rago 1999). In the Chesapeake Bay stock, high recruitment indices were noted in only four of the nine years when spawning stock biomass was high (1989 to 1997; Fig. 7 in Richards and Rago 1999). Year class failures are also possible. Despite an above average abundance of spawners in 2004, the resulting year class is expected to be weak as a result of poor survival in the first few weeks after spawning when water temperatures cooled. We used the discrete life history model with stochasticity to describe the potential and the general time frame for recovery (Appendix D). The results of the simulations are specific to the assumptions regarding the stock and recruitment dynamic, survival, and maturation schedule.

With stochastic variation in survival and in the absence of directed fisheries on any life stage, the mean abundance of spawners reaches a ceiling of 60,000 fish with the median abundance of 55,000 fish (Fig. 2.6). The 2.5th percentile of the spawner abundance levels at about 25,000 spawners although the minimum value in any simulation occasionally falls below 10,000 fish. Individual simulation trajectories illustrate the annual variability in YOY abundance, spawner abundance, and 3+ adult bass abundance, expected from the life history model conditional upon the assumed parameters of the model (Fig. 2.6; Appendix D). The variability in abundance over time in any single simulation is quite broad, determined largely by the assumed variation in egg to YOY recruitment (Fig. 2.7).

In the absence of fisheries removals and assuming M at age as above, there is a greater than 90% chance that by 2011, the abundance of spawners will be greater than S_{opt} (21,600) in six consecutive years (Table 2.3). There is a 90% chance that the abundance will be greater than S_{opt} in 5 of 6 consecutive years by 2010. There is a 92% chance that the abundance of spawners will be above the limit and above the target (in GREEN) during the next ten years (Table 2.4).

There continues to be losses of striped bass as legal bycatch of young-of-the-year striped bass and in illegal fisheries on adult bass (see Phase II below). We consider these current conditions to be the starting point for evaluating persistence and potential to recover. Average Z for ages 3 to 7 is 0.8 between 1997 and 2005 (Fig. 1.9) and consequently we ran the life history model with M = 0.6 and F = 0.2 for striped bass aged 2 years and older. We assumed full recruitment to the illegal fisheries for those ages. Young of the year bycatch was modeled at F = 0.1 on all bycatch fisheries in the summer and fall. Under these conditions, the population trajectory for spawners leveled out at a median value of just under 13,000 adults within a 2.5 to 97.5 percentile range of 4,000 to 38,000 fish (Fig. 2.8). The minimum value in any simulation is about 1,000 fish. Removals of adult bass are in the order of 7,300 fish (2.5 to 97.5 interval range of 2,200 to 24,000 fish) whereas losses in the YOY bycatch fisheries are in the order of 33,000 fish (2.5 to 97.5 interval range of 6,000 to 157,000 fish). The probability of the stock recovering above the limit of 21,600 spawners in at least 5 years out of 6 (YELLOW) between 2006 and 2015 is only 18% (Table 2.5).

If exploitation on adult bass was reduced to zero (with M = 0.6) and exploitation in the bycatch fisheries set at the 50%SPR reference level (F=0.444), there is a 44% chance that the spawner abundance will remain below the recovery limit (RED) if the 5 of 6 years compliance rule is used (Table 2.4, 2.6).

Directed fishing on adults at the 50%SPR rate (F = 0.089) will result in a 31% chance that the spawner abundance will remain below the recovery limit (RED) (Table 2.4, 2.7). Finally, fishing at the combined 50%SPR rates for young-of-the-year bycatch (F=0.220) and adults (F=0.044) is expected to result in a 37% chance that the spawner abundance will remain below the recovery limit between 2006 and 2015 (Table 2.4, 2.8).

Using the 5 of 6 years compliance rule, a very low bycatch mortality of juveniles (F = 0.1) and no increased mortality on adults may provide a 95% chance of the spawning stock recovering out of the RED zone between 2006 and 2015.

RESIDENCE AND CRITICAL HABITAT

Landings data, tagging data (conventional and acoustic), beach seine surveys, fishery officer reports, field observations, traditional ecological knowledge, and anecdotal information indicate that striped bass are widespread as adults, juveniles, and YOY throughout the southern Gulf.

With the exception of the freshwater habitat, striped bass are present at some time of the year in every estuary, lagoon, inlet, and coast of the southern Gulf.

Although the whole of the southern Gulf is crucial for striped bass life history events associated with feeding, rearing, and overwintering, it does not fall within the definition of "residence" described by the "den" and "nest" examples in the SARA s.2(1). The transient nature of striped bass during all life stages and the passive drift of eggs and larvae out of their area of production do not meet the criteria of containment. Although the southern Gulf as a whole must be technically disqualified as striped bass "residence", the Miramichi estuary to the head of the tide encompasses the only known spawning location for striped bass in the southern Gulf and as such is critical to the persistence of the population.

The Northwest Miramichi is presently the only known spawning location for striped bass in the southern Gulf (see Life History section). This phenomenon has been and continues to be demonstrated annually with conventional tagging studies (Douglas et al. 2003; this document) and unsuccessful attempts at finding spawned eggs and larvae in other southern Gulf estuaries during spring (Robinson et al. 2001; AVC Inc. 2003, DFO unpublished). The Miramichi estuary is habitually occupied each spring when the spawning migration of striped bass arrives.

The extent of the spawning grounds of the Northwest Miramichi was further refined in time and space through acoustic tracking studies in 2003 and 2004. Striped bass captured on the spawning grounds of the NW Miramichi were implanted with acoustic transmitters (N = 19 in 2003, N = 21 in 2004) and their movements monitored with stationary receivers placed throughout the Miramichi estuary and inner bay in 2003 and additionally in coastal waters off northern NB in 2004. Implanted striped bass spent an extended period staging at Strawberry Marsh, adjacent to the confluence of the Northwest and Southwest Miramichi rivers prior to spawning in early May. Striped bass activity was next highest during early June, between the areas of the Northwest Millstream and Cassilis on the NW Miramichi, an area traditionally known to encompass the spawning grounds. The time that implanted striped bass spent in this section of the river was assumed to represent spawning and was consistent with egg and larval density distributions during the 1992 spawning run (Robichaud-LeBlanc et al. 1996). Implanted striped bass made excursions up the Southwest Miramichi to the head of tide in both years but the amount of time spent in that branch was substantially less than that of the NW Miramichi. There was relatively little activity recorded on receiver arrays in the middle and lower sections of the estuary, and no detections were recorded at arrays anchored in the main shipping channels between the barrier islands (Fig. 2.9). In 2004, 8 implanted striped bass that exhibited spawning behaviour in the Northwest Miramichi were detected in coastal waters at Val Comeau and one was later detected in Chaleur Bay (Fig. 1.16). These data may provide managers with protection options that could include area and time closures targeted at staging and spawning bass.

The Miramichi estuary and specifically the Northwest Miramichi also proved to be an important overwintering area for implanted striped bass. Twenty-six of the implanted striped bass returned to the Miramichi estuary during late autumn in 2003 and 2004 (Table 1.4). Overwintering striped bass were monitored under the ice by stationary receivers anchored to the streambed. Again striped bass remained in the area of Strawberry Marsh until after ice cover during late December. In both years, there was a slow progressive movement up the Northwest Miramichi in tidal waters until they reached the same area where they had spawned only months previously.

Striped bass are known to overwinter in many estuaries of the southern Gulf (Rulifson and Dadswell 1995) but assessment of overwintering habitat has been limited to the Kouchibouguac

River (Bradford et al. 1997a). The relative importance of estuaries in terms of overwintering locations for striped bass is unknown. It has always been perceived that the choice of an overwintering location has been opportunistic and in response to cooling ocean temperatures encountered by migrating adult bass (Bradford and Chaput 1996). In years of high YOY abundance coupled with their annual widespread range extension out of the Miramichi, it is extremely likely that every estuary in the southern Gulf harbours striped bass through the winter months. It is believed that striped bass would chose alternate overwintering sites if conditions deteriorated but the sudden loss of many overwintering sites could be disastrous for southern Gulf striped bass.

2 - HUMAN – INDUCED MORTALITY

MAJOR POTENTIAL SOURCES OF MORTALITY / HARM

The list of threats put forward by COSEWIC (2004) for striped bass in the southern Gulf included "bycatch in various fisheries such as gaspereau, and rainbow smelt and illegal takes particularly during ice fishing". To the extent possible, we explore this list of limiting factors, as well as others in the southern Gulf. Little or no quantitative information exists to derive mortality estimates for each of the potential sources of harm and therefore we assigned qualitative ranks based on field observations by DFO Science and Conservation and Protection (C&P) staff, and discussions with commercial fishermen, First Nations, non-government organizations (NGO), and the public. Ranks used to characterize the potential sources of mortality were low, moderate, high, or uncertain. The rank of "no indication" (NI) was used when there was no evidence in support of striped bass mortality associated with the human induced factors explored as a requirement of the RPA exercise (DFO 2004b). Evidence in support of many of the assigned ranks were discussed below while all mortality factors and associated ranks were summarized in Table 2.9.

Directed fishing – high

Striped bass of the southern Gulf are managed under the *Canada Fisheries Act* and the *Maritime Provinces Fishery Regulations*. Current protection for the species prohibits the retention, possession, or sale of any wild striped bass in the region. Although legal fisheries for striped bass are currently closed, illegal fisheries and black markets for their sale are extensive. Striped bass angling, frequently under the guise of targeting other legal species in tidal waters, is widespread throughout the southern Gulf.

The worst accounts of illegal gillnet fisheries for striped bass originate from Kent County, NB. Credible reports of gillnetting striped bass by the thousands are common, especially from the mouths of the Kouchibouguac and Black rivers and at Rivière au Portage, during early May, when striped bass are leaving their winter refuges to begin their spawning migration to the Miramichi. The fish's nearshore schooling behaviour is easily exploited by individuals setting gillnets in water no deeper than that required to tend them with chest waders. Hogans and Melvin (1984) cautioned that if poaching continued at the high levels observed in the Kouchibouguac National Park in 1983-84, the population would decline.

Striped bass are gillnetted intensively again in the fall and through the winter under the ice, as they move into estuaries for the winter months. Much of the autumn gillnetting occurs in the Richibucto River system. In recent years, we have received reports of striped bass being offered for sale for $5.00 each and available throughout Kent county in any quantity desired (Kouchibouguac National Park warden pers. comm.). Furthermore, poaching with gillnets is extensive in the areas of Tabusintac, Tracadie and Pokemouche (e.g. one net was seized recently with 19 adult striped bass (DFO fishery officer, pers. comm.)). Credible accounts of severe poaching exist for the upper reaches of the Pugwash River, NS just before freeze-up. Given the widespread distribution of striped bass in the southern Gulf, it may be reasonable to assume that gillnetting occurs or has occurred in other locations of the southern Gulf.

Anglers either kill or cause harm to striped bass when they are hooked and released. US studies on hook and release of striped bass indicate that mortality generally increases with water temperature and type of bait (Wilde et al. 2000). The Atlantic States Marine Fisheries Commission currently assumes a hooking mortality rate of 8% (Diodati and Richards 1996) which infers that well over 1 million striped bass died along the US Atlantic coast between 1996 and 2000. Millard et al. (2005) suggest that a hook and release mortality rate closer to 16% in freshwater river systems may be more appropriate. There is no doubt that mortality associated with the hooked and released component of any striped bass recreational fishery is substantial and needs to be considered when managing the resource.

Angler accounts of hooking and releasing 75 bass per day are common during May at the confluence of the SW and NW Miramichi rivers, a location known as Strawberry Marsh, as well as 15km upstream, directly on the spawning grounds of the NW Miramichi. After a single patrol in May 2005, fishery officers reported 34 anglers in the Strawberry Marsh area hooking and releasing striped bass, and one individual was charged with the illegal retention of two bass (F. Butler, DFO C&P, pers. comm.). Although angling for striped bass is widespread throughout the southern Gulf, striped bass are particularly targeted in the staging areas only days before spawning and only meters away from the spawning grounds. Anecdotal accounts are made of increased catches along the coasts of New Brunswick as the bass leave the Miramichi after spawning and begin their summer coastal migrations.

Miscou Island is a well known striped bass angling area. DFO fishery officers estimate that a minimum of 1,000 striped bass are angled from Miscou beaches annually (Miscou C&P, pers. comm.). Evidence to corroborate these claims surfaced in 2003 when an angler from Miscou returned an acoustic pinger that had been implanted in the body cavity of a striped bass two months earlier. Reports from a reliable source indicate that one Miscou angler completed the 2005 season with a total take of 125 bass.

Angling for striped bass continues to occur regularly in many locations of the southern Gulf including: Bathurst harbour, the Tabusintac and Tracadie rivers, all of the tributaries and rivers of Kent Co., NB, Mabou harbour in Cape Breton, and the Hillsborough river in PEI. Offenders with striped bass angled from Pictou Harbour were charged in 2004.

Bycatch in gaspereau fisheries – moderate

Landings data prior to the commercial fishery closure in 1996, reveal that striped bass bycatch in the gaspereau fisheries of the southern Gulf can be substantial (LeBlanc and Chaput 1991). The highest level of striped bass bycatch is believed to exist in the gaspereau fishery of the Miramichi system and can be attributed to their annual spawning migration to the NW Miramichi. With the exception of the Miramichi, fishery officers reported that striped bass bycatch is minimal in the larger gaspereau fisheries of the Margaree, Buctouche, Richibucto, Kouchibouguac, Pokemouche, and Big Tracadie rivers, the Pictou and Pugwash harbours, and Wallace Bay (Table 7 in Chiasson et al. 2002). This list does not exhaust the locations of all potential effort and gear for gaspereau in the southern Gulf (Appendix E). For example, striped bass are intercepted annually in the Napan, Black (n=2), St. Louis, and Tabusintac rivers of NB, as well as, the Pisquid and Hillsborough rivers of PEI (AVC Inc. 2003). Given the widespread and roaming behaviour of striped bass in the southern Gulf, any gear set for gaspereau at any location has the potential of harming the species (Appendix E) (DFO 2000a; DFO 2001a; DFO 2002a).

The commercial fishery closure in 1996 eliminated the direct mortality of striped bass bycatch, but any interference with natural behaviour or migrations will cause harm to the species. Survival is believed to be high when striped bass are culled quickly and efficiently from catches of gaspereau. DFO Science has used the gaspereau fishery of the NW Miramichi extensively for striped bass assessment purposes and has yet to witness or report on mortality from improperly handled fish (see previous assessments). Whether the striped bass bycatch is treated similarly throughout the other southern Gulf gaspereau fisheries is unknown. One commercial fisherman from the Richibucto River was charged in 2003 with a poorly culled catch where approximately 40 juvenile bass were mixed in with the gaspereau.

Bycatch in rainbow smelt fisheries – high

The only quantitative bycatch study completed for the Rainbow smelt fishery was limited to the Miramichi estuary during the open-water season between October 15 and the end of November in 1994 and 1995. Bradford et al. (1997b) estimated that between 100-500 thousand YOY striped bass were captured as bycatch in the open water fall smelt fisheries of 1994 and 1995. The high bycatch levels of striped bass in the Miramichi were consistent with levels ranked by fishery officers for that same smelt fishery (Chiasson et al. 2002). Fishery officers also ranked striped bass bycatch as highest in the Tabusintac, and Wallace areas, and as moderate throughout Kent Co. NB, and in the East and West rivers of PEI (Chiasson et al. 2002). Fishery officers from the Neguac NB detachment commented that “after a rain in the fall there is nothing to see thousands of small striped bass in a single smelt net. Fishers try their best not to harm the small bass but when they lift the box or bag net, 50% of the bass are already dead due to the current pushing the small juvenile fish against the back of the net” (Chiasson et al. 2002).

There is a large amount of gear in the rainbow smelt fishery in the both the fall and winter seasons throughout the southern Gulf (Appendix F).

Culling efforts by fishermen using box and bag nets is generally considered to be poor and mortality is essentially 100% in gillnets. Fishers are often constrained by the timing of the tides and entire catches are loaded before culling begins. Frigid air temperatures that instantaneously freeze striped bass out of the water or gulls feasting on striped bass trying to escape provides little chance of survival.

Based on the estimates in Bradford et al. (1997b) for a portion of the smelt fishing activity, bycatch in the order of 100s of thousands, particularly when juvenile abundance is high, is entirely realistic. The proportion of the total juvenile stock intercepted in these fisheries remains unknown. Bycatch concerns in smelt fisheries date to the 1930s when Huntsman (1945) identified this fishery as having a particular negative effect on striped bass abundance.

Bycatch in American eel fisheries – moderate

Little is known about the level of bycatch in the American eel fishery of the southern Gulf (Bradford et al. 1995). Fishery officers ranked bass bycatch as minimal in eel fisheries of Cascumpec PEI, the Richibucto River system in NB, and Antigonish and Pomquet harbour in Gulf NS (Chiasson et al. 2002). Areas where striped bass bycatch was considered to be high and moderate were the Tabusintac River and Merigomish Harbour, respectively. Fishery officers report that striped bass bycatch didn’t occur in the eel fisheries of the Miramichi, Tracadie, or Pokemouche systems even though it is common knowledge that it does (Chiasson et al. 2002). With the exception of years with low spawner success, YOY striped bass are captured by beach seine along most coasts of the southern Gulf. It is reasonable to assume that the large amount of gear targeting eels in the southern Gulf captures several thousand striped bass annually (Appendix G) (DFO 2000c; DFO 2001c; DFO 2002c).

Bycatch in coastal fisheries – uncertain
Bradford and Chaput (1998) reported two tag returns from striped bass captured in the herring fishery off Escuminac, NB, in spring 1997. Herring gillnets in this fishery are often set in a few meters of water next to shore, so it is likely that striped bass undergoing their spring spawning migration to the Miramichi are intercepted more frequently than previously documented. Herring gear have small mesh and would be expected to have a larger impact on the younger age groups.

Bycatch in Aboriginal fisheries – high
Prior to 2000, First Nations retained striped bass for food, social, and ceremonial purposes. Allocations were suspended when the conservation requirement of 5,000 female spawners was not achieved for several consecutive years in the late 90's. Although the striped bass allocation was suspended, seasons and locations that coincided with traditional Atlantic salmon fisheries were not altered. Since gillnets do not select by species, striped bass continue to be killed and discarded during those fisheries that target Atlantic salmon. Aboriginal gillnet fisheries believed to have the greatest impact on striped bass survival include those near the spawning grounds on the NW Miramichi at Eel Ground and Red Bank. Striped bass returning to estuaries in the fall to overwinter are intercepted by Elsipogtog (formerly Big Cove) and Esgenoôpetij (Burnt Church) First Nation targeting late run Atlantic salmon in the Richibucto and Tabusintac rivers, respectively. Pictou Landing First Nation intercept striped bass in the East River of Gulf NS.

Reliable striped bass harvest statistics from any First Nation fishery have never been provided. One 24 hr gillnet catch sampled in June 2003 at Eel Ground yielded 80 adult striped bass and no other species of fish. Gillnets are generally tended once a day and it is unlikely that more diligent monitoring of the gear would improve the poor survival probability of striped bass enmeshed in gillnets.

Striped bass are intercepted by First Nation trapnets set for food, science, or commercial purposes (Table 2.10). When striped bass allocations were in place, First Nations often harvested striped bass captured in trapnets. Efficient culling practices in trapnet fisheries are believed to cause little mortality.

Habitat alterations under permit - low
There are no major habitat alteration issues believed to threaten bass in the southern Gulf. DFO Habitat is cognisant of the specialized spawning grounds for striped bass in the NW Miramichi and has recently refused permission for the construction of a wharf and marina in that area. Applications for large floating docks near the spawning grounds of the NW Miramichi have also been denied. Dredging activities are localized and generally of a low scale, directed at opening channels near community wharves for access by relatively small inshore fleets. Plans for any habitat alterations in the southern Gulf need to consider the widespread use of all estuaries by striped bass.

Power generating stations - uncertain
The effect of power generating stations (PGS) on striped bass in the southern Gulf is unknown. It is known, however, that large numbers of striped bass are drawn to the thermal effluent of the PGS at Trenton NS, Dalhousie and Belledune NB, during late fall and winter. Anglers target these warm water effluents because of the large concentrations of striped bass which continue

to feed at that time of year. Well over 1,000 striped bass were estimated to have died at the outflow of the Trenton PGS in February 2004. The cause of the fish kill was believed to be the result of an acute reduction in water temperature when the PGS went off line and the thermal discharge was turned off (C. MacInnis DFO, pers. comm.).

Industrial and municipal waste water - uncertain

Waste water effluents from industrial and municipal facilities are widespread throughout the southern Gulf, but their effect on striped bass or striped bass habitat is unknown. Sites of particular interest are those along the Miramichi River including UPM-Kymenne's pulp paper mills (both bleached kraft and ground wood) at Newcastle and Nelson (closed in 2004) respectively, and the 7 sewage treatment facilities between the head of tide at Red Bank and the inner Miramichi Bay (Robichaud-LeBlanc et al. 2000). The discharge from the UPM-Kymenne operation is in the Northwest Miramichi, in the Strawberry Marsh area, the location believed to function as the primary pre-spawning staging area for striped bass. This warm effluent produces a localized open water area even in the coldest months of the year. It is expected that millions of eggs, larvae, and free swimming YOY are exposed to a wide array of chemicals in the Miramichi environment. Many of the chemicals used in present day operations are known to contain endocrine disrupting compounds such as estrogens and testosterones that eventually make their way into rivers via effluents (Wayne Fairchild, DFO, pers. comm.). Burton et al. (1983) demonstrated significant mortality of striped bass larvae after a 72-h exposure to bleached kraft mill effluent. Environmental impact assessments for new industries in any estuary of the southern Gulf need to consider potential effects on striped bass and their habitat.

Other pulp and paper operations in the southern Gulf include Atholville (Restigouche River), Dalhousie and Bathurst (Chaleur Bay), and Pictou Harbour (Nova Scotia). The Pictou Harbour area is a well known overwintering site for striped bass and is located in the same embayment as the thermal power generating station at Trenton, NS.

Scientific research - low

Any scientific research program that requires fixed or mobile gear anywhere in the southern Gulf will harm striped bass. However, gear that traps fish instead of entangling fish has been shown to cause less mortality. Trapnets similar to those used in the gaspereau fisheries of the southern Gulf have been used by DFO, First Nations, and NGO's to assess different anadromous stocks throughout the region and all have recorded striped bass catches (Hayward 2001). Trapnets used for science based activities on a relatively consistent basis are listed in Table 2.10.

Extracting scales for ageing purposes and applying tags to striped bass as part of the annual DFO stock assessment activity disrupts the mucous membrane covering the fish and increases the likelihood of infections. There has been no evaluation of mortality from tagging procedures used on southern Gulf striped bass. Special striped bass research requests are often granted that have recently included a tracking study that required incisions into the body cavity of striped bass and fin clipping for genetic analysis.

Striped bass YOY are susceptible to injury or death from beach seining early in the growing season when fish are fragile, or if catches are large, or if the catch includes a lot of debris. DFO Science has developed a beach seine index of YOY striped bass in the southern Gulf that has entailed over 100 sweeps annually between 2000 and 2005. DFO Stewardship has recently started the Community Aquatic Monitoring Program (CAMP) where community groups throughout the southern Gulf beach seine approximately 5 sites per watershed each month between May and October. Effort in this program is not consistent but 100 beach seine sweeps annually throughout the region would be a reasonable estimate of effort with a maximum combined catch in the range of a few thousand individuals.

An action plan developed between DFO Gulf Region and the Province of Quebec to re-introduce striped bass to the St. Lawrence Estuary required the use of Miramichi progeny as seed stock. The approximately 2,000 YOY striped bass that have been removed from the Miramichi system in each of 1999, 2002, 2003, and 2005, have been either stocked directly into the St. Lawrence estuary or are being raised for broodstock.

CONCLUSION

Individual mortality estimates of the potential threats listed above are not possible at this time, but the cumulative mortality has the potential of being important. There is little doubt that poaching alone removes several thousand adult striped bass annually, conceivably half of the adult population each year. The extent of striped bass mortality associated with the rainbow smelt and American eel fisheries of the southern Gulf is poorly understood but it is reasonable to assume that hundreds of thousands, and in exceptional cases, a million YOY striped bass could be captured in these fisheries annually.

Although the aggregate mortality of all of these factors is high, striped bass in the southern Gulf have persisted, although at much lower levels of abundance than might be expected if anthropogenic related mortalities were reduced. Our analysis and perceptions indicate that the aggregate sum of the current mortality rates imposed on this population is well above the value which would allow abundance to increase and be maintained above the recovery objective. In the absence of any reductions in mortality, the abundance of spawners is expected to remain low and highly variable. Any additional mortality will preclude the recovery potential for southern Gulf striped bass.

3 - MITIGATION AND ALTERNATIVES

The persistent reports of thousands of bass being gillnetted and angled year-round in numerous estuaries and inshore coastal areas of the southern Gulf indicate to us that these illegal fisheries are the largest threat to recovery of the population. We suggest that efforts be focused, first, on substantially reducing the illegal and bycatch mortality on adult striped bass, followed by measures to reduce the bycatch of YOY bass in estuarine fisheries. Because of the large amounts of YOY available to capture, the bycatch which numbers in the hundreds of thousands appears to be impressive. The Spawner per Recruit (SPR) analysis presented in this document puts the relative losses in perspective; a loss of 300 thousand YOY bass in the bycatch fisheries has the same impact on lifetime production of eggs as the removal of 18,500 age one and older bass. These values are less than the realized removals of bass in these fisheries during the mid 1990s.

Although angling for striped bass occurs throughout the region, the elimination of angling on the staging and spawning grounds of the NW Miramichi in May and June should be operationally feasible as it entails the closure of a relatively small section of the Miramichi River. The acoustic tracking study demonstrated that striped bass staged in the area of Strawberry Marsh between the middle and end of May. Spawning for this population is generally during the first 10 days of June in the vicinity of the Northwest Millstream, upstream to Cassilis. The closure of these two areas in May and June each spring would substantially reduce mortality on the striped bass spawning component. “Hotspot” angling areas could also be targeted for special monitoring.

Coincident with the prohibition of angling in tidal waters in the time and area stated above, a delay in the opening of the Eel Ground First Nation gillnet fishery targeting Atlantic salmon in the Northwest Miramichi to possibly June 15 would substantially reduce the mortality on striped bass spawners during the peak spawning period of late May to early June. Delaying the season would reduce bycatch and eliminate handling of undesired fish in the gillnet fishery and possibly have minimal impact on overall catches of Atlantic salmon. Data from trapnets fished by the Eel Ground First Nation in the same area as the gillnets indicated that few Atlantic salmon were captured prior to June 15 and the peak of the early run never occurred before the first week in July (Hayward 2001).

Gaspereau fisheries in the Miramichi River, particularly those in the Northwest Miramichi, have the potential to intercept large numbers of bass spawners. In recent years, with the prohibition on retention and sale of all striped bass in these fisheries, the industry has delayed the opening of the gaspereau fisheries into late May to maximize their fishing opportunities for the target species, gaspereau. Handling practices have improved in these fisheries as science staff collaborate regularly with the fishermen for assessment and research purposes. Furthermore, there has been a concerted effort on the part of the fishermen to cull rapidly to reduce harm. Other than an outright closure on this fishery, we cannot think of any additional measures which would further reduce the low impact this fishery is currently having on striped bass spawners.

As a result of studies conducted on the spatial and temporal characteristics of the bycatch of YOY bass in the open water smelt fishery of Miramichi Bay, the season opening was delayed by two weeks, from Oct. 15 to Nov. 1, in 1999. This measure was estimated to potentially reduce the YOY striped bass bycatch in this fishery by as much as 50% (Bradford et al. 1997b). However, in the absence of similar detailed studies in other areas of the southern Gulf, the delayed opening of the season was restricted to the small study area in Miramichi Bay. Important bycatch issues remain in many other locations (Chiasson et al. 2002). At a minimum, precautionary approach principles would favour a delay in the opening of the season in all of these areas. This would be supported by targeted research to assess the spatial and temporal distributions of YOY bass to ascertain whether alternative or additional interventions would be required.

We suggest that mortality of striped bass at all life stages would be substantially reduced if fishers used gear that allowed for the live release of non-targeted species rather than entanglement gears (i.e. gillnets). In estuaries that are used extensively by striped bass in the fall and winter, several First Nation gillnet fisheries operate regularly. These fisheries occur in the estuaries of the Tabusintac, Kouchibouguac, Richibucto, Buctouche, Pictou Harbour, and Miramichi.

CONCLUSION

Because there are no baseline mortality estimates for striped bass bycatch in any fishery, the effect of implementing the mitigation measures described above will be difficult to gauge. We argue, however, that implementation of any mitigation measure can only improve this population's potential for recovery.

ACKNOWLEDGEMENTS

We thank Noella McDonald (DFO Moncton) for age interpretation of striped bass scales. We thank Harry Collins and Daryl Sullivan of the Miramichi River Environmental Assessment Committee for completing the beach seine survey in 2005.

LITERATURE CITED

AVC Inc. 2003. Do striped bass spawn in the Hillsborough River? Phase II of a feasibility study for a bass enhancement project. Report prepared for Hillsborough River and Area Development Corporation 42 p.

Beaulieu, H. 1985. Rapport sur la situation du bar rayé (*Morone saxatilis*). Association des Biologistes du Quebec et Ministère du Loisir, de la Chasse et de la Pêche. 53 p.

Bernier, R. 1996. Relation entre la taille automnale et la survie hivernale de bar rayé (*Morone saxatilis*) de la rivière Miramichi. Thèse d'Initiation à la Recherche. Université de Moncton, Moncton, N.B. 24 p.

Bradford, R.G. and Chaput, G. 1996. Status of striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 1995. DFO Atlantic Fisheries Res. Doc. 96/62: 36 p.

Bradford, R.G. and Chaput, G. 1997. Status of striped bass (*Morone saxatilis*) in the Gulf of St. Lawrence in 1996 and revised estimates of spawner abundance for 1994 and 1995. DFO Atlantic Fisheries Res. Doc. 97/16: 31 p.

Bradford, R.G. and Chaput, G. 1998. Status of striped bass (*Morone saxatilis*) in the Gulf of St. Lawrence in 1997. DFO Can. Stock Assess. Res. Doc. 98/35: 25 p.

Bradford, R.G., Chaput, G., and Tremblay, E. 1995. Status of striped bass (*Morone saxatilis*) in the Gulf of St. Lawrence. DFO Atlantic Fisheries Res. Doc. 95/119: 43 p.

Bradford, R.G., Tremblay, E., and Chaput, G. 1997a. Winter distribution of striped bass (*Morone saxatilis*) and associated environmental conditions in Kouchibouguac National Park during 1996-1997. Parks Canada – Eco. Monit. Data Rep. 003: iv + 59 p.

Bradford, R.G., Chaput, G., Hurlbut, T., and Morin, R. 1997b. Bycatch of striped bass, white hake, winter flounder, and Atlantic tomcod in the autumn "open water" smelt fishery of the Miramichi River estuary. Can. Tech. Rep. Fish. Aquat. Sci. 2195: vi + 37 p.

Burton, D.T., Hall, L.W.Jr., Klauda, R.J., and Margrey, S.L. 1983. Effects of treated leached kraft mill effluent on eggs and prolarvae of striped bass *Morone saxatilis*. Water Resour. Bull. 19: 869-879.

Caddy, J.F. 2002. Viewpoint: limit reference points, traffic lights, and holistic approaches to fisheries management with minimal stock assessment input – a personal viewpoint. J. Fish. Res. 56: 133-137.

Chaput, G.J. and Robichaud, K.A. 1995. Size and growth of striped bass, (*Morone saxatilis*), from the Miramichi River, Gulf of St. Lawrence, Canada. *In* Water, science, and the public: the Miramichi ecosystem. *Edited by* E.M.P. Chadwick. Can. Spec. Publ. Fish. Aquat. Sci. 123: 161-176.

Chiasson, G., Gallant, P.A., and Mallet, P. 2002. Traditional and local knowledge: estuarine fisheries by-catch in the southern Gulf of St. Lawrence; ecosystem based fisheries management considerations. Can. Man. Rep. Fish. Aqua. Sci. No. 2613: vi + 45 p.

Chow, V.T., Maidment, D.R., and Mays, L.W. 1988. Applied Hydrology. McGraw-Hill Book Company, New York. 572 p.

COSEWIC. 2004. COSEWIC assessment and status report on the striped bass *Morone saxatilis* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vii + 43 p.

COSEWIC. 2005. COSEWIC assessment and update status report on the shortnose sturgeon *Acipenser brevirostrum* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vi + 27 p.

Dey, W.P. 1981. Mortality and growth of young-of-the-year striped bass in the Hudson river estuary. Trans. Amer. Fish. Soc. 110: 151-157.

DFO, 2000a. Integrated fisheries management plan gaspereau (alewife) Prince Edward Island 2000-2004 (inclusive) Gulf fisheries. 15 p.

DFO, 2000b. Integrated fisheries management plan rainbow smelt Prince Edward Island 2000 to 2004 (inclusive) Gulf fisheries. 15 p.

DFO, 2000c. Integrated fisheries management plan eels Prince Edward Island 2000-2004 (inclusive) Gulf fisheries. 15 p.

DFO, 2001a. Integrated fisheries management plan gaspereau eastern New Brunswick area Gulf region 2001-2006. Fisheries and Oceans Canada Gulf Region Eastern NB Area: vi + 42 p.

DFO, 2001b. Integrated fisheries management plan smelt eastern New Brunswick area Gulf region 2001-2006. Fisheries and Oceans Canada Gulf Region Eastern NB Area: iv + 63 p.

DFO, 2001c. Integrated eel fishery management plan eastern New Brunswick area Gulf region 2001-2006. Fisheries and Oceans Canada Gulf Region Eastern NB Area: vi + 48 p.

DFO, 2002a. Integrated fisheries management plan alewife (gaspereau) Gulf area of Nova Scotia Inland and Coastal Fisheries 2002-2005. Fisheries and Oceans Canada Gulf Region Gulf NS Area: i + 18 p.

DFO, 2002b. Integrated fisheries management plan rainbow smelt Gulf area of Nova Scotia Inland and Coastal Fisheries 2002-2005. Fisheries and Oceans Canada Gulf Region Gulf NS Area: i + 21 p.

DFO, 2002c. Integrated fisheries management plan American eels Gulf area of Nova Scotia Inland and Coastal Fisheries 2002-2005. Fisheries and Oceans Canada Gulf Region Gulf NS Area: i + 21 p.

DFO, 2004a. Revised framework for evaluation of scope for harm under section 73 of the species at risk act. DFO Can. Sci. Advis. Sec. Stock Status Report 2004/048: 7 p.

DFO, 2004b *In Prep*. Proceedings of the National Science Advisory Meeting on Section 73 Permits under the Species At Risk Act March 8-10, 2004. DFO Can. Advis. Sec. Proceed. Ser. 2004/005.

DFO, 2005. A framework for developing science advise on recovery targets for aquatic species in the context of the *Species at Risk Act*. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2005/054: 16 p.

Diaz, M., Leclerc, G.M., and Ely, B. 1997. Nuclear DNA markers reveal low levels of genetic divergence among Atlantic and Gulf of Mexico populations of striped bass. Trans. Am. Fish. Soc. 126: 163-165.

Diodati, P.J. and Richards, R.A. 1996. Mortality of striped bass hooked and released in salt water. Trans. Amer. Fish. Soc. 125: 300-307.

Douglas, S.G., Chaput, G.C., and Bradford, R.G. 2001. Status of striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 1999 and 2000. DFO Can. Sci. Adv. Sec. Res. Doc. 2001/058: 34 p.

Douglas, S.G., Bradford, R.G., and Chaput, G. 2003. Assessment of striped bass (*Morone saxatilis*) in the Maritime Provinces in the context of species at risk. DFO Can. Sci. Adv. Sec. Res. Doc. 2003/008: iii + 49 p.

Goodyear, C.P. 1985. Toxic materials, fishing, and environmental variation: simulated effects on striped bass population trends. Trans. Am. Fish. Soc. 114: 107-113.

Hanson, M.J. and Courtenay, S.C. 1995. Seasonal abundance and distribution of fishes in the Miramichi Estuary. *In* Water, science, and the public: the Miramichi ecosystem. *Edited by* E.M.P. Chadwick. Can. Spec. Publ. Fish. Aquat. Sci. No. 123: 141-160.

Hayward, J. 2001. Weekly fish counts from in-river traps and barrier pools in the Miramichi River, New Brunswick, 1994 to 1999. Can. Dat. Rep. Fish. Aquat. Sci. 1080: x + 94 p.

Hilborn, R. and Walters, C.J. 1992. Quantitative fisheries stock assessment: choice, dynamics, and uncertainty. Chapman and Hall, London. 570 p.

Hogans, W.E. 1984. Helminths of striped bass (*Morone saxatilis*) from the Kouchibouguac River, New Brunswick. J. Wildl. Dis. 68: 61-63.

Hogans, W. and Melvin, G. 1984. Kouchibouguac National Park striped bass (*Morone saxatilis* Walbaum) fishery survey. Aquatic Industries Limited. St. Andrews, N.B. 91 p.

Huntsman, A.G. 1945. Miramichi fisheries investigations twenty-five years ago. In: The Commercial and the World, February 8, 1945, page 6, Chatham, N.B.

Hurst, T.P. and Conover, D.O. 1998. Winter mortality of young-of-the-year Hudson River striped bass (*Morone saxatilis*): size-dependent patterns and effects on recruitment. Can. J. Fish. Aquat. Sci. 55: 1122-1130.

Johnson, T.B. and Evans, D.O. 1991. Behaviour, energetics, and associated mortality of young-of-the-year white perch: climate warming and invasion of the Laurentian Great Lakes. Trans. Amer. Fish. Soc. 119: 301-313.

LeBlanc, C.H. and Chaput, G.J. 1991. Landings of estuarine fishes in the Gulf of St. Lawrence 1917 – 1988. Can. Data Rep. Fish. Aquat. Sci. 842: viii + 101 p.

Mace, P.M. 1994. Relationships between common biological reference points used as threshold and targets of fisheries management strategies. Can. J. Fish. Aquat. Sci. 118: 179-187.

Maunder, M.N. and Punt, A.E. 2004. Standardizing catch and effort data: a review of recent approaches. Fish. Res. 70: 141-159.

McGovern, J.C. and Olney, J.E. 1996. Factors affecting survival of early life stages and subsequent recruitment of striped bass on the Pamunkey River, Virginia. Can. J. Fish. Aquat. Sci. 53: 1713-1726.

Millard, M.J., Mohler, J.W., Kahnle, A., and Cosman, A. 2005. Mortality associated with catch and release angling of striped bass in the Hudson River. North Amer. J. Fish. Manag. 25: 1533-1541.

Myers, R.A., Bridson, J., and Barrowman, N.J. 1995. Summary of worldwide spawner and recruitment data. Can. Tech. Rep. Fish. Aquat. Sci. 2020: iv + 327 p.

Paramore, L.M. 1998. Age, growth, and life history characteristics of striped bass, *Morone saxatilis*, from the Shubenacadie-Stewiacke River, Nova Scotia. M.Sc. thesis, East Carolina University, Greeneville, NC. 91 p.

Peterson, R.H., Martin-Robichaud, D.J., Harmon, P., and Berge, Å. 1996. Notes on striped bass culture, with reference to the Maritime Provinces. Department of Fisheries and Oceans, Halifax, Nova Scotia. p. 35.

Potter, T. 2001. Past and present use of reference points for Atlantic salmon. *In* Stock, recruitment, and reference points: assessment and management of Atlantic salmon. *Edited by* E. Prévost and G. Chaput. INRA Editions, Paris. 195-223 p.

Richards, R.A. and Rago, P.J. 1999. A case history of effective fishery management: Chesapeake bay striped bass. North Amer. J. Fish. Manag. 19: 356-375.

Ricker, W. 1975. Computation and interpretation of biological statistics of fish populations. Bull. Fish. Res. Board Can. 191: 382 p.

Robichaud-LeBlanc, K.A., Courtenay, S.C., and Locke, A. 1996. Spawning and early life history of a northern population of striped bass (*Morone saxatilis*) in the Miramichi River estuary, Gulf of St. Lawrence. Can. J. Zool. 74: 1645-1655.

Robichaud-LeBlanc, K.A., Courtenay, S.C., and Hanson, J.M. 1997. Ontogenetic diet shifts in age-0 striped bass, Morone saxatilis, from the Miramichi River estuary, Gulf of St. Lawrence. Can. J. Zool. 75: 1300-1309.

Robichaud-LeBlanc, K.A., Courtenay, S.C., and Benfey, T.J. 1998. Distribution and growth of young-of-the-year striped bass in the Miramichi River estuary, Gulf of St. Lawrence. Trans. Am. Fish. Soc. 127: 56-69.

Robichaud-LeBlanc, K.A., Bradford, R.G., Flecknow, J., and Collins, H. 2000. Bibliography of Miramichi, southern Gulf of St. Lawrence striped bass and North American studies on the effects of domestic and industrial effluent on striped bass. Can. Manuscr. Rep. Fish. Aquat. Sci. 2547: iii + 31 p.

Robinson, M., Courtenay, S., Benfey, T., Maceda, L., and Wirgin, I. 2004. Origin and movements of young-of-the-year striped bass in the southern Gulf of St. Lawrence, New Brunswick. Trans. Amer. Fish. Soc. 133: 412-426.

Robinson, M., Courtenay, S., Benfey, T., and Tremblay, E. 2001. The fish community and use of the Kouchibouguac and Richibucto estuaries by striped bass during the spring and summer of 1997 and 1998. Parks Canada – Tech. Rep. Eco. Sci. 032: iv + 83 p.

Rulifson, R.A. and Dadswell, M.J. 1995. Life history and population characteristics of striped bass in Atlantic Canada. Trans. Am. Fish. Soc. 124: 477-507.

Rutherford, E.S. and Houde, E.D. 1995. The influence of temperature on cohort-specific growth, survival, and recruitment of striped bass, *Morone saxatilis*, larvae in Chesapeake Bay. Fish. Bull. U.S. 93: 315-332.

Rutherford, E.S., Houde, E.D., and Nyman, R.M. 1997. Relationship of larval-stage growth and mortality to recruitment of striped bass, Morone saxatilis, in Chesapeake Bay. Estuaries 20: 174-198.

SAS 2005. Statistical Analysis Software. Cary, NC, USA.

Scofield, E.C. 1931. The striped bass of California (*Roccus lineatus*). Division of fish and game of California. Fish Bull. 29. 83 p.

Scott, W.B. and Scott, M.G. 1988. Atlantic fishes of Canada. Can. Bull. Fish. Aquat. Sci. 219: 731 p.

Secor, D.H., Gunderson, T.E., and Karlsson, K. 2000. Effect of temperature and salinity on growth performance in anadromous (Chesapeake Bay) and nonanadromous (Santee-Cooper) strains of striped bass (*Morone saxatilis*). Copeia 2000(1): 291-296.

Sogard, S.M. 1997. Size-selective mortality in the juvenile stage of teleost fishes: a review. Bull. Mar. Sci. 60(3): 1129-1157.

Ulanowicz, R.E. and Polgar, T.T. 1980. Influences of anadromous spawning behaviour and optimal environmental conditions upon striped bass (*Morone saxatilis*) year-class success. Can. J. Fish. Aquat. Sci. 37: 143-154.

Venables, W.N. and Ditchmount, C.M. 2004. GLMs, GAMx and GLMMs: an overview of theory for applications in fisheries research. Fisheries Research 70: 319-337.

Wilde, G.R., Muoneke, M., Bettoli, P., Nelson, K., and Hysmith, B. 2000. Bait and temperature effects on striped bass hooking mortality in freshwater. North Amer. J. of Fish. Manag. 20: 810-815.

Wirgin, I.I., Ong, T., Maceda, L., Waldman, J.R., Moore, D., and Courtenay, S. 1993. Mitochondrial DNA variation in striped bass (*Morone saxatilis*) from Canadian rivers. Can. J. Fish. Aquat. Sci. 50: 80-87.

Wirgin, I.I., Jessop, B., Courtenay, S., Pedersen, M., Maceda, S., and Waldman, J.R. 1995. Mixed-stock analysis of striped bass in two rivers of the Bay of Fundy as revealed by mitochondrial DNA. Can. J. Fish. Aquat. Sci. 52: 961-970.

Table 1.1. Sampling details and start and end dates for the three groupings of the CPUE data from the striped bass bycatch of the NW Miramichi gaspereau fishery. Sampling intensity refers to the proportion of catches sampled relative to the total number of catches landed during a sampling event.

	Sampling period					Upper quartile dates	
Year	First	Last	Catches sampled	Sampling intensity	Mark and recapture end date	First	Last
1993	28-May	18-Jun	46	7.7%		28-May	16-Jun
1994	24-May	18-Jun	50	7.7%	12-Jun	24-May	15-Jun
1995	24-May	21-Jun	60	7.7%	9-Jun	24-May	11-Jun
1996	24-May	19-Jun	72	8.3%	10-Jun	31-May	13-Jun
1997	4-Jun	20-Jun	60	8.2%	20-Jun	4-Jun	14-Jun
1998	21-May	17-Jun	83	8.4%	4-Jun	21-May	11-Jun
1999	21-May	18-Jun	139	8.0%	8-Jun	21-May	15-Jun
2000	25-May	21-Jun	102	9.0%	21-Jun	25-May	16-Jun
2001	25-May	20-Jun	32	11.0%	11-Jun	25-May	1-Jun
2002	23-May	19-Jun	56	9.4%	4-Jun	23-May	3-Jun
2003	24-May	23-Jun	75	9.3%	11-Jun	24-May	11-Jun
2004	24-May	25-Jun	86	9.0%	11-Jun	24-May	17-Jun
2005	24-May	24-Jun	85	9.6%	8-Jun	1-Jun	13-Jun

Table 1.2. Spawner abundance estimates for striped bass on the spawning grounds of the NW Miramichi. Estimates based on mark recapture experiments between 1993 and 2005.

	Spawner abundance estimates for year												
Estimate	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**
Total spawners (mode)	5,500	29,000	50,000	8,090	8,000	3,400	3,940	3,900	24,000	29,000	21,000	15,000	20,000
95% confidence limit (lower)	4,550	23,000	35,000	6,275	5,800	2,900	3,450	2,850	18,000	25,500	17,000	10,000	11,500
95% confidence limit (upper)	7,300	47,000	175,000	13,370	17,500	4,800	4,430	5,250	33,000	32,500	27,000	24,500	45,500
Proportion mature males	0.94	0.92	0.63	0.37	0.69	0.83	0.69	0.64	0.77	0.58	0.51	0.69	0.40
Proportion mature females (minimum)	na	na	na	na	na	na	0.03	0.04	0.02	0.01	0.00	0.08	0.11
Proportion mature females (maximum)	0.06	0.08	0.37	0.63	0.31	0.17	0.31	0.36	0.23	0.42	0.49	0.31	0.60
Mature males (minimum)	5,170	26,680	31,500	2,993	5,520	2,822	2,719	2,496	18,480	16,820	10,710	10,350	8,000
Mature females (minimum)	na	na	na	na	na	na	118	156	480	290	0	1,200	2,200
Mature females (maximum)	330	2,320	18,500	5,097	2,480	578	1,221	1,404	5,520	12,180	10,290	4,650	12,000

Table 1.3. Mean annual catch per unit of effort (Ln(CPUE), standard error of Ln(CPUE) and back transformed median CPUE of striped bass in the NW Miramichi, derived from the general linear model, in three sampling periods. All season refers to the CPUE derived using the observations over the entire sampling period. Mark and recapture refers to the mean CPUE based on the sampling interval from the start of sampling to the end date of the mark and recapture experiment. Upper quartile is the mean CPUE derived using the period which encompassed the upper quartile of the CPUE observations annually. Samples where catch = 0 is the percent of all the samples for the entire sampling period when no bass were observed in the catches at the trapnets.

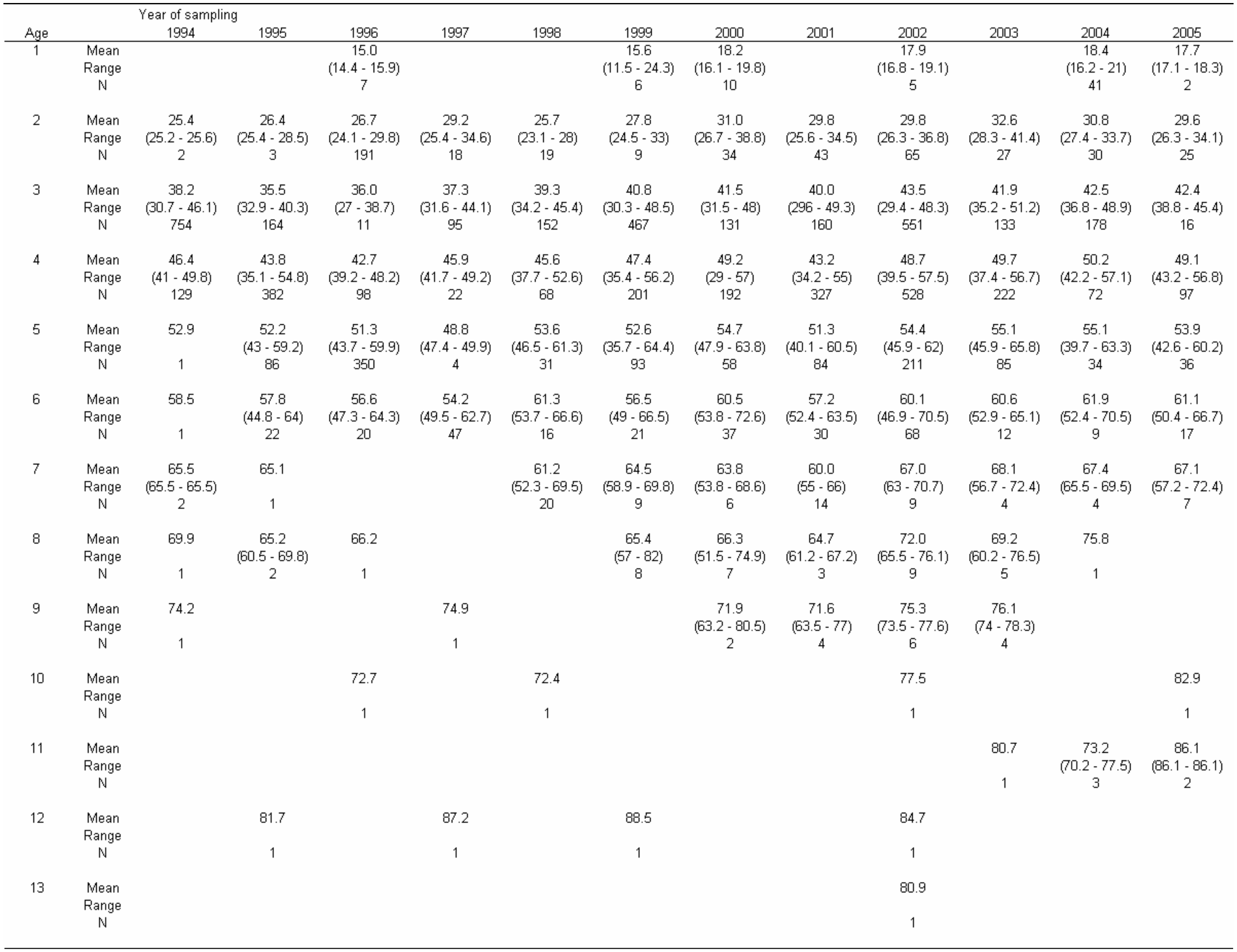
	Ln (CPUE)			CPUE			Samples where catch=0
Year	All season	Mark and recapture	Upper quartile	All season	Mark and recapture	Upper quartile	
1993	1.39	1.39	1.58	4	4	5	28.3%
1994	4.19	4.42	4.32	66	83	75	0.0%
1995	3.70	4.28	4.04	41	72	57	1.7%
1996	2.19	2.49	2.53	9	12	13	4.2%
1997	1.60	1.60	1.77	5	5	6	11.7%
1998	2.34	2.68	2.48	10	15	12	4.8%
1999	1.99	2.38	2.07	7	11	8	17.3%
2000	1.89	1.89	1.99	7	7	7	3.9%
2001	4.36	4.56	5.23	78	95	186	0.0%
2002	4.52	5.28	5.51	91	196	247	0.0%
2003	3.66	4.21	4.21	39	67	67	1.3%
2004	2.53	3.34	2.92	13	28	19	5.8%
2005	2.98	3.59	3.42	20	36	30	5.9%

	Standard error		
Year	All season	Mark and recapture	Upper quartile
1993	0.55	0.51	0.48
1994	0.13	0.13	0.12
1995	0.15	0.15	0.14
1996	0.29	0.32	0.28
1997	0.43	0.40	0.39
1998	0.25	0.26	0.23
1999	0.23	0.23	0.20
2000	0.29	0.27	0.26
2001	0.15	0.14	0.14
2002	0.10	0.11	0.10
2003	0.14	0.14	0.13
2004	0.23	0.25	0.20
2005	0.18	0.19	0.17

Table 1.4. Spawning and overwintering history for striped bass implanted with acoustic pingers in 2003 and 2004.

	2003			2004		
	Males	**Females**	**Total**	**Males**	**Females**	**Total**
Tagged in spring	7	12	19	9	12	21
Known removals	0	1	1	0	0	0
Tags available for detection	7	11	18	9	12	21
Overwintered in Miramichi (tagging year)	7	7	14	5	7	12
Overwintered in the Miramichi (tagging year) and spawned after ice out	4	6	10	4	6	10
Overwintered elsewhere and spawned after ice out	0	2	2	0	3	3
Consecutive spawns	4	8	12	4	9	13
Overwintered in the Miramichi and left after ice out	3	1	4	1	1	2
Missing	0	2	2	2	4	6

Table 1.5. Fork lengths at age for striped bass sampled during their spawning run to the NW Miramichi between 1994 and 2005.



		Year of sampling											
Age		1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
1	Mean			15.0			15.6	18.2		17.9		18.4	17.7
	Range			(14.4 - 15.9)			(11.5 - 24.3)	(16.1 - 19.8)		(16.8 - 19.1)		(16.2 - 21)	(17.1 - 18.3)
	N			7			6	10		5		41	2
2	Mean	25.4	26.4	26.7	29.2	25.7	27.8	31.0	29.8	29.8	32.6	30.8	29.6
	Range	(25.2 - 25.6)	(25.4 - 28.5)	(24.1 - 29.8)	(25.4 - 34.6)	(23.1 - 28)	(24.5 - 33)	(26.7 - 38.8)	(25.6 - 34.5)	(26.3 - 36.8)	(28.3 - 41.4)	(27.4 - 33.7)	(26.3 - 34.1)
	N	2	3	191	18	19	9	34	43	65	27	30	25
3	Mean	38.2	35.5	36.0	37.3	39.3	40.8	41.5	40.0	43.5	41.9	42.5	42.4
	Range	(30.7 - 46.1)	(32.9 - 40.3)	(27 - 38.7)	(31.6 - 44.1)	(34.2 - 45.4)	(30.3 - 48.5)	(31.5 - 48)	(296 - 49.3)	(29.4 - 48.3)	(35.2 - 51.2)	(36.8 - 48.9)	(38.8 - 45.4)
	N	754	164	11	95	152	467	131	160	551	133	178	16
4	Mean	46.4	43.8	42.7	45.9	45.6	47.4	49.2	43.2	48.7	49.7	50.2	49.1
	Range	(41 - 49.8)	(35.1 - 54.8)	(39.2 - 48.2)	(41.7 - 49.2)	(37.7 - 52.6)	(35.4 - 56.2)	(29 - 57)	(34.2 - 55)	(39.5 - 57.5)	(37.4 - 56.7)	(42.2 - 57.1)	(43.2 - 56.8)
	N	129	382	98	22	68	201	192	327	528	222	72	97
5	Mean	52.9	52.2	51.3	48.8	53.6	52.6	54.7	51.3	54.4	55.1	55.1	53.9
	Range		(43 - 59.2)	(43.7 - 59.9)	(47.4 - 49.9)	(46.5 - 61.3)	(35.7 - 64.4)	(47.9 - 63.8)	(40.1 - 60.5)	(45.9 - 62)	(45.9 - 65.8)	(39.7 - 63.3)	(42.6 - 60.2)
	N	1	86	350	4	31	93	58	84	211	85	34	36
6	Mean	58.5	57.8	56.6	54.2	61.3	56.5	60.5	57.2	60.1	60.6	61.9	61.1
	Range		(44.8 - 64)	(47.3 - 64.3)	(49.5 - 62.7)	(53.7 - 66.6)	(49 - 66.5)	(53.8 - 72.6)	(52.4 - 63.5)	(46.9 - 70.5)	(52.9 - 65.1)	(52.4 - 70.5)	(50.4 - 66.7)
	N	1	22	20	47	16	21	37	30	68	12	9	17
7	Mean	65.5	65.1			61.2	64.5	63.8	60.0	67.0	68.1	67.4	67.1
	Range	(65.5 - 65.5)				(52.3 - 69.5)	(58.9 - 69.8)	(53.8 - 68.6)	(55 - 66)	(63 - 70.7)	(56.7 - 72.4)	(65.5 - 69.5)	(57.2 - 72.4)
	N	2	1			20	9	6	14	9	4	4	7
8	Mean	69.9	65.2	66.2			65.4	66.3	64.7	72.0	69.2	75.8	
	Range		(60.5 - 69.8)				(57 - 82)	(51.5 - 74.9)	(61.2 - 67.2)	(65.5 - 76.1)	(60.2 - 76.5)		
	N	1	2	1			8	7	3	9	5	1	
9	Mean	74.2			74.9			71.9	71.6	75.3	76.1		
	Range							(63.2 - 80.5)	(63.5 - 77)	(73.5 - 77.6)	(74 - 78.3)		
	N	1			1			2	4	6	4		
10	Mean			72.7		72.4				77.5			82.9
	Range												
	N			1		1				1			1
11	Mean										80.7	73.2	86.1
	Range											(70.2 - 77.5)	(86.1 - 86.1)
	N										1	3	2
12	Mean		81.7		87.2		88.5			84.7			
	Range												
	N		1		1		1			1			
13	Mean									80.9			
	Range												
	N									1			

Table 1.6. Sex ratios at age for striped bass sampled during the spawning run to the NW Miramichi between 1995-2005.

		Proportion at age		
Year	**Age**	**Female**	**Male**	**Juvenile**
2005	3	0.00	1.00	0.00
	4	0.55	0.45	0.00
	5	0.73	0.27	0.00
	6	0.88	0.13	0.00
	7	1.00	0.00	0.00
2004	2	0.00	0.11	0.89
	3	0.12	0.88	0.00
	4	0.61	0.39	0.00
	5	0.81	0.19	0.00
	6	1.00	0.00	0.00
	7	1.00	0.00	0.00
	11	0.00	1.00	0.00
2003	2	0.21	0.21	0.57
	3	0.24	0.76	0.00
	4	0.54	0.46	0.00
	5	0.78	0.22	0.00
	6	1.00	0.00	0.00
	7	1.00	0.00	0.00
	8	0.50	0.50	0.00
	9	1.00	0.00	0.00
2002	1	0.00	0.00	1.00
	2	0.00	0.08	0.92
	3	0.21	0.79	0.00
	4	0.54	0.46	0.00
	5	0.82	0.18	0.00
	6	0.92	0.08	0.00
	7	1.00	0.00	0.00
	8	1.00	0.00	0.00
	9	1.00	0.00	0.00
	10	1.00	0.00	0.00
	12	1.00	0.00	0.00
	13	1.00	0.00	0.00
2001	2	0.09	0.07	0.84
	3	0.08	0.90	0.03
	4	0.16	0.84	0.00
	5	0.42	0.58	0.00
	6	0.76	0.24	0.00
	7	0.67	0.33	0.00
	8	1.00	0.00	0.00
	9	0.75	0.25	0.00
2000	1	0.00	0.00	1.00
	2	0.00	0.06	0.94
	3	0.26	0.74	0.00
	4	0.37	0.63	0.00
	5	0.50	0.50	0.00
	6	0.72	0.28	0.00
	7	0.50	0.50	0.00
	8	1.00	0.00	0.00
1999	1	0.00	0.00	1.00
	2	0.00	0.00	1.00
	3	0.20	0.80	0.00
	4	0.49	0.51	0.00
	5	0.70	0.30	0.00
	6	0.86	0.14	0.00
	7	0.89	0.11	0.00
	8	0.88	0.13	0.00
1998	2	0.00	0.00	1.00
	3	0.25	0.75	0.00
	4	0.59	0.41	0.00
	5	0.63	0.38	0.00
	6	1.00	0.00	0.00
	7	1.00	0.00	0.00
1997	2	0.67	0.00	0.33
	3	0.92	0.08	0.00
	4	0.23	0.77	0.00
	5	0.75	0.25	0.00
	6	0.87	0.13	0.00
	9	1.00	0.00	0.00
	12	1.00	0.00	0.00
1996	1	0.00	0.00	1.00
	2	0.00	0.00	1.00
	3	0.82	0.09	0.09
	4	0.65	0.35	0.00
	5	0.82	0.18	0.00
	6	0.80	0.20	0.00
	8	1.00	0.00	0.00
	10	1.00	0.00	0.00
1995	2	0.00	0.00	1.00
	3	0.44	0.56	0.00
	4	0.23	0.77	0.00
	5	0.83	0.17	0.00
	6	1.00	0.00	0.00
	7	1.00	0.00	0.00
	8	1.00	0.00	0.00
	12	1.00	0.00	0.00

Table 1.7. Abundance estimates of striped bass at age in the spawning population of the Northwest Miramichi, 1994 to 2005. Estimates of abundance were based on mark-recapture experiments.

	Year of spawning											
Age	**2005**	**2004**	**2003**	**2002**	**2001**	**2000**	**1999**	**1998**	**1997**	**1996**	**1995**	**1994**
3	1127	9651	6826	11490	5872	1240	2300	1977	4471	185	12462	24596
4	11268	3314	10318	11131	12272	1844	996	726	1035	1648	29027	4208
5	4225	1570	3283	4457	3049	484	456	341	188	5887	6535	33
6	2254	291	313	1373	1092	272	104	199	2212	336	1672	33
7	1127	116	104	190	452	30	45	156	0	0	76	65
8	0	0	104	190	113	30	40	0	0	17	152	33
9	0	0	52	106	151	0	0	0	47	0	0	33
10	0	0	0	21	0	0	0	0	0	17	0	0
11	0	58	0	0	0	0	0	0	0	0	0	0
12	0	0	0	21	0	0	0	0	47	0	76	0
13	0	0	0	21	0	0	0	0	0	0	0	0
Total	20000	15000	21000	29000	23000	3900	3940	3400	8000	8090	50000	29000

Table 1.8. Estimates of Z and corresponding M values for striped bass year-classes 1991-2001 based on two indicators of stock status.

			Estimates of Z and M for year-class										
Indicator		**Age**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**
Mark-recapture													
	Z	3	-0.17	2.02	-1.72	1.82	0.69	0.22	-2.29	-0.64	0.11	0.72	-0.15
		4	1.60	2.17	1.11	0.46	0.72	-0.50	1.01	1.22	1.88	-0.24	
		5	0.98	-0.06	1.19	0.52	-0.81	0.80	2.66	2.42	-0.36		
		6	2.65	1.50	1.24	-0.51	1.75	2.58	0.99	-1.35			
		7	1.37	0.39	-1.32	0.87	0.60						
		8		-1.61	0.07	1.29							
		9		1.96									
	M	3	-0.14	0.86	-4.60	0.84	0.50	0.20	-8.90	-0.90	0.10	0.51	-0.17
		4	0.79	0.89	0.67	0.37	0.51	-0.65	0.64	0.71	0.85	-0.28	
		5	0.62	-0.06	0.70	0.40	-1.26	0.55	0.93	0.91	-0.44		
		6	0.93	0.78	0.71	-0.66	0.83	0.92	0.63	-2.88			
		7	0.75	0.32	-2.74	0.58	0.45						
		8		-3.98	0.06	0.73							
		9		0.86									
CPUE													
	Z	3	0.52	1.99	-0.86	-0.14	1.14	0.66	-3.13	-1.13	0.86	1.26	-0.12
		4	1.57	3.03	-0.84	0.92	1.16	-1.34	0.52	1.97	2.42	-0.21	
		5	1.84	-2.01	1.65	0.96	-1.65	0.31	3.41	2.96	-0.33		
		6	0.69	1.95	1.68	-1.34	1.26	3.33	1.53	-1.32			
		7	1.83	0.83	-2.15	0.37	1.35						
		8		-2.44	-0.43	2.04							
		9		1.47									
	M	3	0.41	0.86	-1.36	-0.15	0.68	0.48	-21.78	-2.10	0.58	0.72	-0.13
		4	0.79	0.95	-1.33	0.60	0.69	-2.80	0.41	0.86	0.91	-0.23	
		5	0.84	-6.47	0.81	0.62	-4.19	0.26	0.97	0.95	-0.38		
		6	0.50	0.86	0.81	-2.82	0.72	0.96	0.78	-2.74			
		7	0.84	0.56	-7.60	0.31	0.74						
		8		-10.46	-0.53	0.87							
		9		0.77									

Table 2.1. Recovery objectives for southern Gulf striped bass based on the discrete life history model.

	Density independent survival (S_o)	YOY_{cap} (millions)		
		1.0	**1.5**	2.0
Spawners at equilibrium eggs (S_{eq})	0.0005	26,400	39,300	52,100
	0.001	42,500	**63,300**	84,100
	0.002	50,500	74,500	100,100
Spawners for maximum gain of eggs (S_{opt})	0.0005	11,200	16,000	22,400
	0.001	14,400	**21,600**	28,800
	0.002	12,800	21,000	27,200
3+ abundance at S_{eq}	0.0005	45,100	67,300	89,500
	0.001	72,500	**108,500**	144,600
	0.002	86,200	129,000	172,100
3+ abundance at S_{opt}	0.0005	25,900	37,400	51,800
	0.001	47,300	**70,900**	94,600
	0.002	61,400	95,000	125,700

Table 2.2. Possible reference points for striped bass from the southern Gulf based on 50%SPR, S_{opt}, and 30%SPR. Reference levels are for values of S_0 = 0.001 and YOY_{cap} = 1.5 (Table 2.1).

Reference level	Spawners	F		Yield			Mid-year 3+ abundance
		YOY	Age 1+	YOY (millions)	Age 1+ (number)	Age 1+ (weight, t)	
50% SPR	31,200	0.444	0.000	0.303	0	0.0	54,000
		0.000	0.089	0.000	18,500	15.7	64,200
		0.220	0.044	0.167	7,800	7.1	59,100
S_{opt}	21,600	0.640	0.000	0.336	0	0.0	37,400
		0.000	0.131	0.000	21,600	17.5	47,500
		0.320	0.064	0.195	8,400	7.4	42,000
30% SPR	20,000	0.695	0.000	0.341	0	0.0	34,000
		0.000	0.143	0.000	22,300	17.7	44,000
		0.345	0.069	0.199	8,400	7.4	39,000

Table 2.3. Status of the resource in 2005 and the probability of a change of status and the year when status changed for the period 2006 to 2015 under the conditions of no mortalities in any fisheries.

Compliance rule At least 6 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0.000
Frequency of status change	Probability of change over next ten years	
0	0.049	
1	0.923	
2	0.029	
Year first changed occurred		
2011	0.928	
2012	0.023	
2013	0.020	
2014	0.015	
2015	0.015	

Compliance rule At least 5 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0.000
Frequency of status change	Probability of change over next ten years	
0	0.017	
1	0.966	
2	0.016	
3	0.001	
Year first changed occurred		
2010	0.907	
2011	0.054	
2012	0.012	
2013	0.009	
2014	0.009	
2015	0.009	

Table 2.4. Traffic light summary of potential for recovery of striped bass from the southern Gulf for the period 2006 to 2015.

	Probability of being in zone during 2006 to 2015		
	RED Spawners <= 21,600 in > 1 of 6 years	**YELLOW** Spawners > 21,600 in >= 5 of 6 years AND Spawners <= 31,200 in > 3 of 6 years	**GREEN** Spawners > 21,600 in >= 5 of 6 years AND Spawners > 31,200 in >= 3 of 6 years
No directed fisheries	**0.017**	0.058	0.925
Assumed current conditions YOY (F=0.1) and adults (F = 0.2)	**0.820**	0.116	0.065
Fishing on YOY only (50%SPR F = 0.444)	**0.437**	0.252	0.312
Fishing on adults only (50%SPR F = 0.089)	**0.306**	0.238	0.456
Fishing at 50%SPR on YOY (F=0.22) and adults (F = 0.044)	**0.369**	0.248	0.383

Table 2.5. Status of the resource in 2005 and the probability of a change of status and the year when status changed for the period 2006 to 2015 under the assumed current fishing conditions on YOY in bycatch fisheries (F = 0.1) and on adult bass (F = 0.2).

At least 6 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0
Frequency of change of status	Probability of change over next ten years	
0	0.899	
1	0.053	
2	0.049	
Year first change occurred		
2011	0.571	
2012	0.101	
2013	0.118	
2014	0.106	
2015	0.105	

At least 5 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0
Frequency of change of status	Probability of change over next ten years	
0	0.820	
1	0.102	
2	0.077	
3 to 4	0.001	
Year first change occurred		
2010	0.418	
2011	0.219	
2012	0.104	
2013	0.096	
2014	0.084	
2015	0.079	

Table 2.6. Status of the resource in 2005 and the probability of a change of status and the year when status changed for the period 2006 to 2015 under the conditions of mortalities on young-of-the-year in bycatch fisheries at the 50%SPR rate (F = 0.444).

At least 6 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0
Frequency of change of status	Probability of change over next ten years	
0	0.572	
1	0.321	
2	0.107	
Year first change occurred		
2011	0.661	
2012	0.083	
2013	0.084	
2014	0.090	
2015	0.081	

At least 5 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0
Frequency of change of status	Probability of change over next ten years	
0	0.436	
1	0.455	
2	0.106	
3 to 4	0.003	
Year first change occurred		
2010	0.555	
2011	0.176	
2012	0.067	
2013	0.071	
2014	0.064	
2015	0.067	

Table 2.7. Status of the resource in 2005 and the probability of a change of status and the year when status changed for the period 2006 to 2015 under the conditions of directed mortalities on adult bass (age 1 and older) at the 50%SPR rate (F = 0.089).

At least 6 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0
Frequency of change of status	Probability of change over next ten years	
0	0.446	
1	0.437	
2	0.118	
Year first change occurred		
2011	0.718	
2012	0.070	
2013	0.073	
2014	0.072	
2015	0.068	

At least 5 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0
Frequency of change of status	Probability of change over next ten years	
0	0.306	
1	0.583	
2	0.108	
3 to 4	0.004	
Year first change occurred		
2010	0.621	
2011	0.165	
2012	0.057	
2013	0.056	
2014	0.051	
2015	0.051	

Table 2.8. Status of the resource in 2005 and the probability of a change of status and the year when status changed for the period 2006 to 2015 under the conditions of mortalities on young-of-the-year in bycatch fisheries and adult bass at the 50%SPR rate (F = 0.22, F = 0.044).

At least 6 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0
Frequency of change of status	Probability of change over next ten years	
0	0.506	
1	0.379	
2	0.116	
Year first change occurred		
2011	0.690	
2012	0.074	
2013	0.080	
2014	0.082	
2015	0.075	

At least 5 of 6 years > 21,600	Status in 2005 relative to compliance rule for the recovery limit	
	Below	Above
	1.000	0
Frequency of change of status	Probability of change over next ten years	
0	0.369	
1	0.521	
2	0.107	
3 to 4	0.004	
Year first change occurred		
2010	0.588	
2011	0.171	
2012	0.064	
2013	0.060	
2014	0.058	
2015	0.059	

Table 2.9. Summary of human induced mortality factors for southern Gulf striped bass, their associated relative rank, cause, effect and potential alternatives to the activity. The rank "NI" signifies no indication.

Potential sources of mortality/harm	**Source**	**Relative rank**	**Cause**	**Effect**	**Alternatives or mitigation**
Domestic					
Directed Fishing	Illegal (poaching)	High	Targeted captures	Direct mortality	Education, Increased enforcement
Bycatch in Fisheries	Commercial Gaspereau	Moderate	Incidental captures	Direct mortality, Handling related mortality	Season / area closures, Gear modifications, Best management practice in effect
	Commercial Rainbow smelt	High	Incidental captures, Inefficient culling	Direct mortality, Handling related mortality	Season / area closures, Gear modifications
	Commercial American Eel	Moderate	Incidental captures, Inefficient culling	Direct mortality, Handling related mortality	Season / area closures, Gear modifications
	Commercial Herring	Uncertain	Incidental captures in gillnets, Inefficient culling	Direct mortality, Handling related mortality	
	Recreational	Uncertain	Incidental captures	Direct mortality, Hook and release mortality	Season / area closures, Gear modifications

Table 2.9. (continued).

Potential sources of mortality/harm	Source	Relative rank	Cause	Effect	Alternatives or mitigation
Bycatch in Fisheries	Aboriginal (food, social, ceremonial)	High	Incidental captures, Inefficient culling	Direct mortality, Handling related mortality	Season / area closures, Gear modifications, Best management practices
Fisheries Impacts on Habitat	Illegal (poaching)	Low	Installation of fixed gear, Boat use	Obstruction of natural migrations and behaviour, Introduction of petroleum products and bi-products	Education, Enforcement
	Commercial	Low	Installation of fixed gear, Boat use	Obstruction of natural migrations and behaviour, Introduction of petroleum products and bi-products	Season / area closures, Gear modifications
	Recreational	NI	Boat use	Introduction of petroleum products and bi-products	Season / area closures, Gear modifications
	Aboriginal (food, social, ceremonial)	Low	Installation of fixed gear, Boat use	Obstruction of natural migrations and behaviour, Introduction of petroleum products and bi-products	Season / area closures, Gear modifications

Table 2.9. (continued).

Potential sources of mortality/harm	Source	Relative rank	Cause	Effect	Alternatives or mitigation
Direct Mortality Under Permit	NI				
Habitat Alterations Under Permit	Municipal, provincial, and federal dredging activities	Low	Dredging of navigation channels, wharf construction, and deposition of sediments	Mortality of benthos and habitat destruction	Select deposition sites which do not impact striped bass habitat, Forbid activities near essential habitats
	Municipal waste water treatment facilities, Pulp & paper mills, Power generating facilities	Uncertain	Discharge of effluents, (waste water, heat)	Mortality of all life stages	
Ecotourism and Recreation	Private companies and public at large	Uncertain	Boat and recreational vessel use	Mortality of eggs and larvae on spawning grounds, Disturbance of fish aggregations, Introduction of petroleum products and bi-products	
Shipping, Transport and Noise	Municipal, provincial, federal, and private transport activities (land and water based)	Low	Transport of hazardous materials, toxic substance spills Boat use	Mortality of all life stages, Introduction of petroleum products and bi-products	Best management practices

Table 2.9. (continued).

Potential sources of mortality/harm	Source	Relative rank	Cause	Effect	Alternatives or mitigation
Fisheries on Food Supplies	Commercial, recreational, Aboriginal fisheries for: Gaspereau, American shad, American eel, Rainbow smelt, Atlantic tomcod, Atlantic silverside, Mackerel, Herring, Crab sp. etc.	Low	Mortality of striped bass prey species	Mortality associated with starvation, reduced growth and/or reproductive effort	Management plans in place for directed fisheries
Aquaculture	Private shellfish culture companies	Uncertain	Introduction of excessive fecal matter Boat use	Mortality of benthos and habitat destruction, Introduction of petroleum products and bi-products	
Scientific Research	Government, university, community groups, First Nations	Low	Installation of fixed gear, use of mobile gear, Manipulation and collection of striped bass, Boat use	Handling related mortality, increased stress, and disease transfer, Obstruction of natural migrations and behaviour, Introduction of petroleum products and bi-products	Permitted activities under section 52 of *Fisheries Act*, Proper handling techniques, Education
Military Activities		NI			
Non-domestic		NI			

Table 2.10. Non-commercial fishing gear used on a regular basis throughout the southern Gulf known to intercept striped bass. The exception is Eel Ground FN who fish trapnets commercially for gaspereau early in the season but switch to food and science based activities by the end of June.

Organization	Watershed	Gear (target species)	Quantity	Activity
Dept. Fisheries & Oceans	Miramichi R.	trapnets (all species)	2-3	science
Dept. Fisheries & Oceans	Southern Gulf	beach seine (all species)	>100	science
Miramichi Salmon Assoc.	Miramichi R.	trapnets (salmon smolts)	1	science
Various NGOs	Southern Gulf	beach seine (all species)	>100	science
Eel Ground First Nation	Miramichi R.	trapnets (gaspereau, salmon)	3-5*	food, science
Eel Ground First Nation	Miramichi R.	gillnets (salmon)	11	food
Eel Ground First Nation	Miramichi R.	fyke nets (American eel)	1	food
Red Bank First Nation	Miramichi R.	fyke nets (American eel)	2	food
Red Bank First Nation	Miramichi R.	trapnets (salmon)	2	food, science
Red Bank First Nation	Miramichi R.	gillnets (salmon)	4	food
Burnt Church First Nation	Miramichi Bay / Tabusintac R.	trapnets (American eel)	10	food
Burnt Church First Nation	Miramichi Bay / Tabusintac R.	trapnets (salmon)	2	food, science
Burnt Church First Nation	Miramichi Bay / Tabusintac R.	gillnets (salmon)	41**	food
Burnt Church First Nation	Miramichi Bay / Tabusintac R.	box nets (rainbow smelt)	unlimited	food
Eel River Bar First Nation	Eel R.	trapnets (salmon)	3	food
Eel River Bar First Nation	Eel R.	gillnets (salmon)	30	food
Pabineau First Nation	Nepisiguit R	trapnets (salmon)	1	food
Big Cove First Nation	Richibucto R.	trapnets (gaspereau)	6	food, science
Big Cove First Nation	Richibucto R.	box nets (rainbow smelt)	6	food
Indian Island First Nation	Richibucto R.	fyke nets (American eel)	2	food
Indian Island First Nation	Richibucto R.	gillnets (salmon)	15	food
Indian Island First Nation	Richibucto R.	box nets (rainbow smelt)	6	food
Buctouche First Nation	Buctouche R.	fyke nets (American eel)	1	food
Buctouche First Nation	Buctouche R.	trapnets (trout)	1	food

* includes 1 partial counting fence at Big Hole Tract

** includes 3 gillnets for kelts

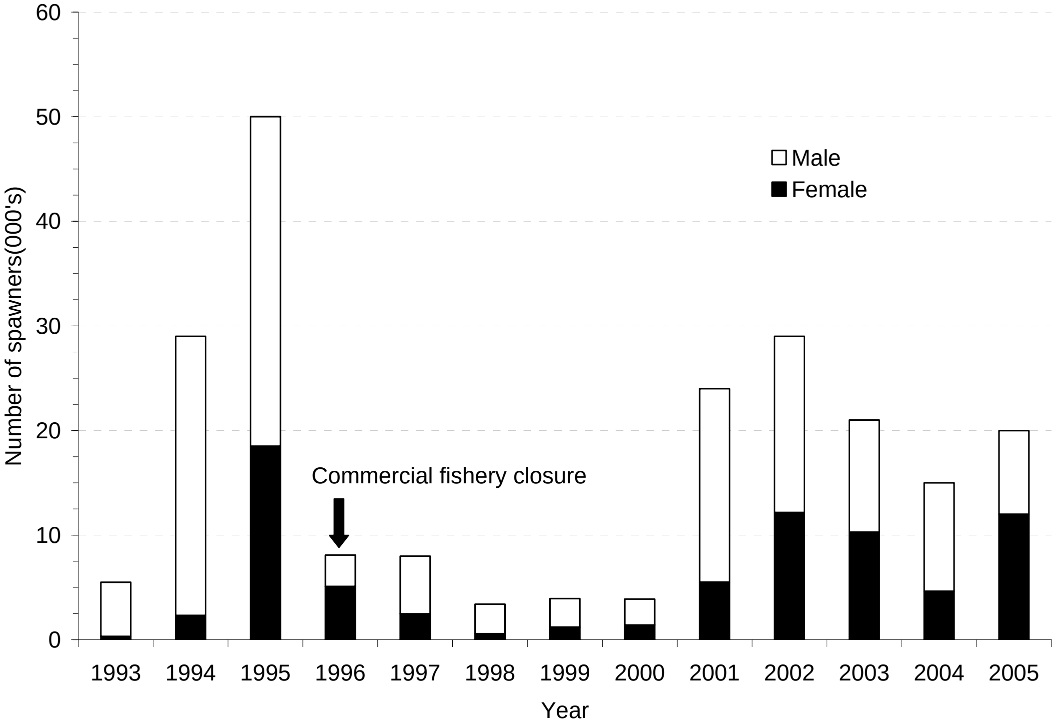


Figure 1.1. Spawner abundance estimates from mark-recapture experiments between 1993 and 2005 on the Northwest Miramichi River.

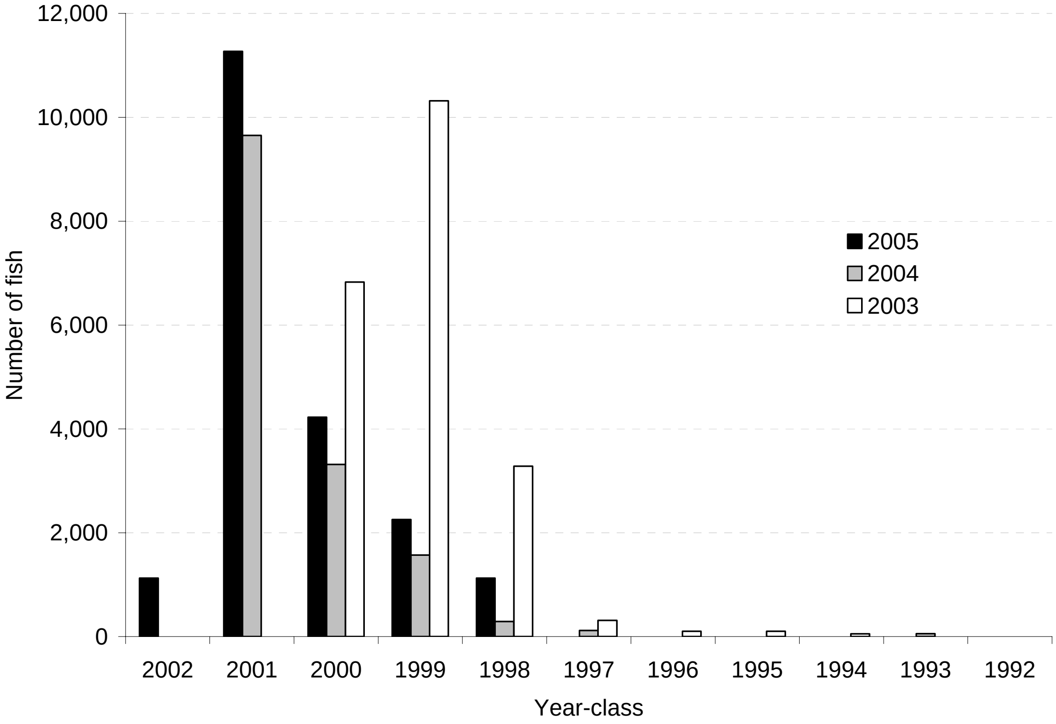


Figure 1.2. Year-class contributions to the spawning run of striped bass to the Northwest Miramichi between 2003 and 2005.

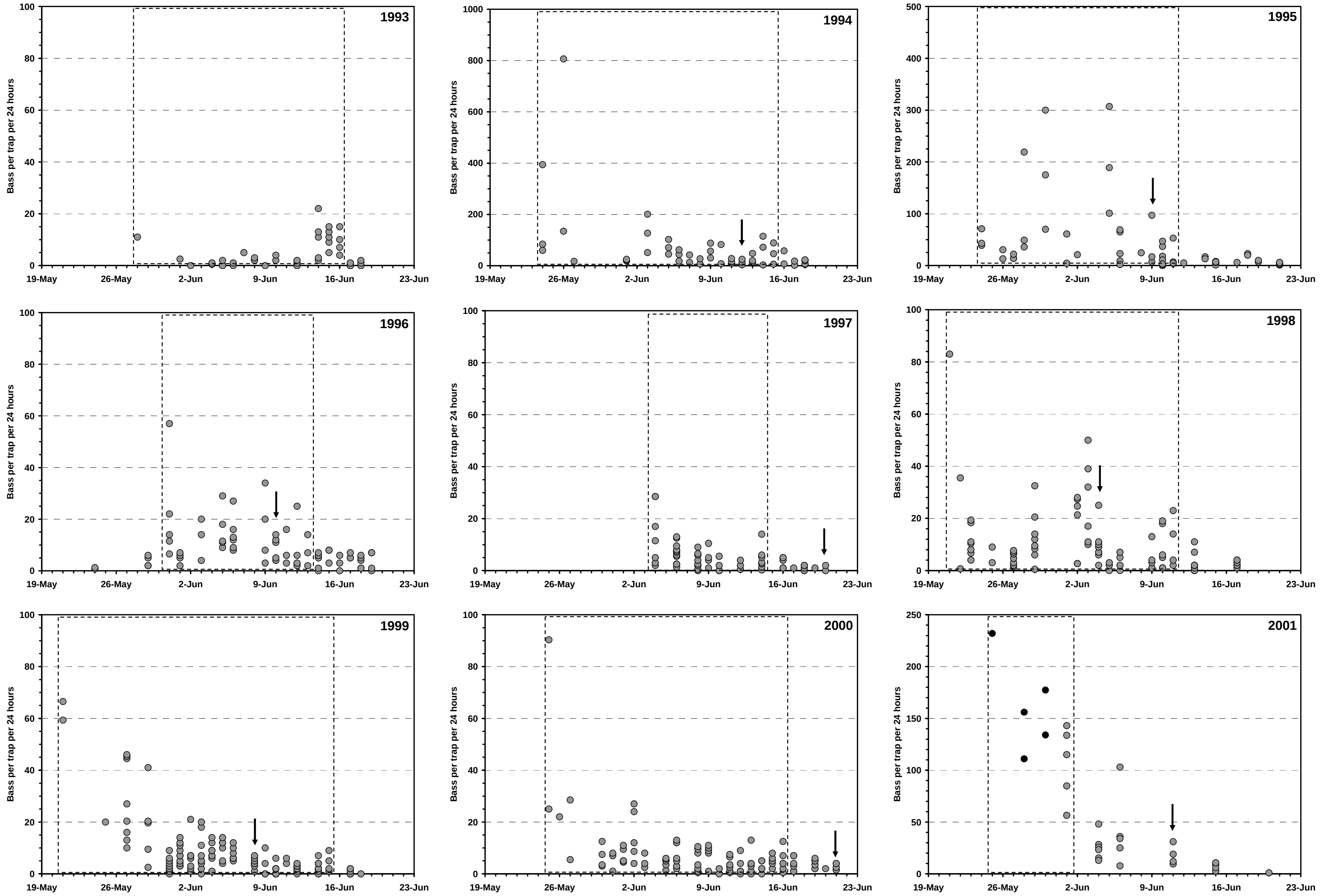


Figure 1.3. Catch of striped bass per net per 24 hour period in the gaspereau fishery of the Northwest Miramichi, 1993 to 2005. The arrow defines the date at which the mark and recapture experiment was considered complete. The dashed rectangle represents the period encompassing the upper quartile of the annual CPUE data. Darkened points represent catches on dates when an individual fisher was contracted to fish his gear specifically for marking purposes early in the season.

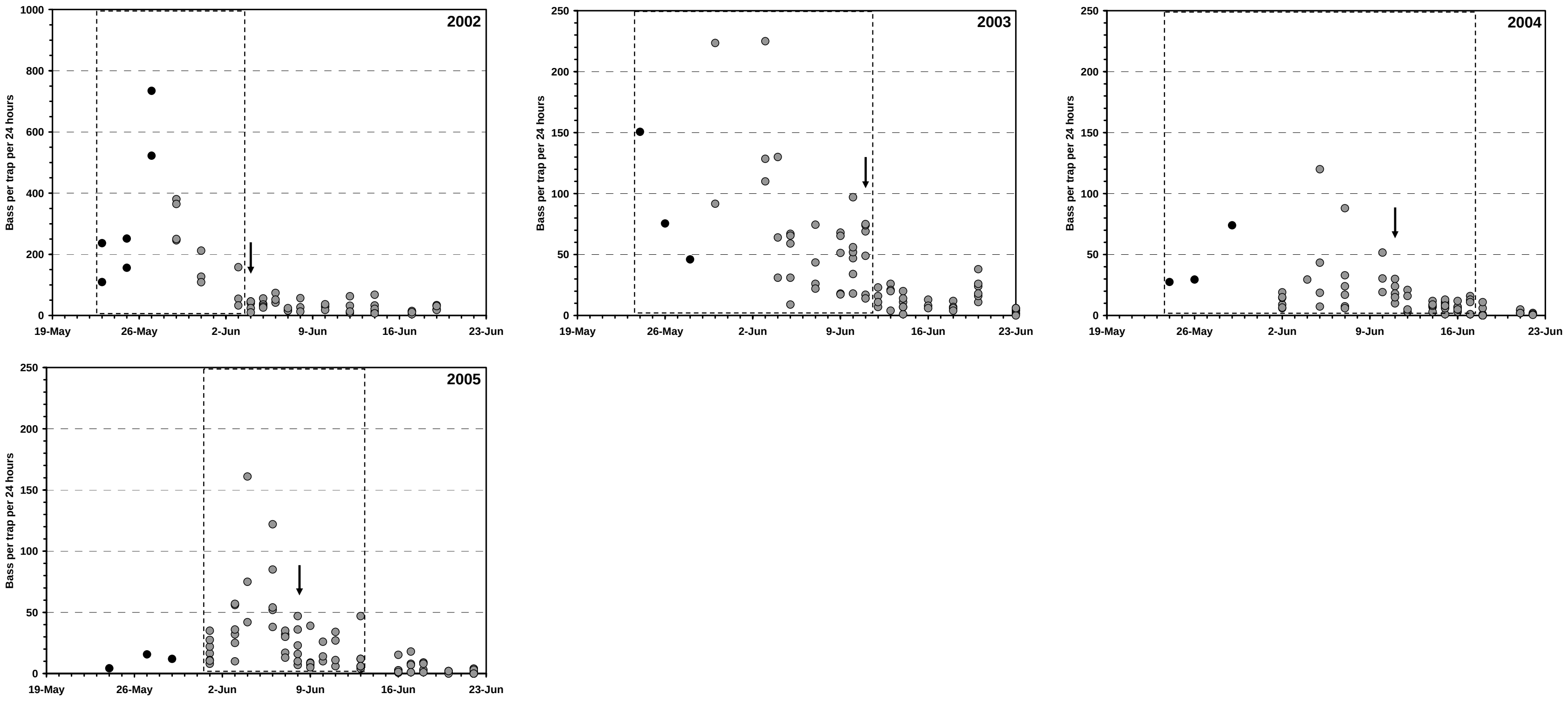


Figure 1.3 (continued). Catch of striped per net per 24 hour period in the gaspereau fishery of the Northwest Miramichi, 1993 to 2005. The arrow defines the date at which the mark and recapture experiment was considered complete. The dashed rectangle represents the period encompassing the upper quartile of the annual CPUE data. Darkened points represent catches on dates when an individual fisher was contracted to fish his gear specifically for marking purposes early in the season.

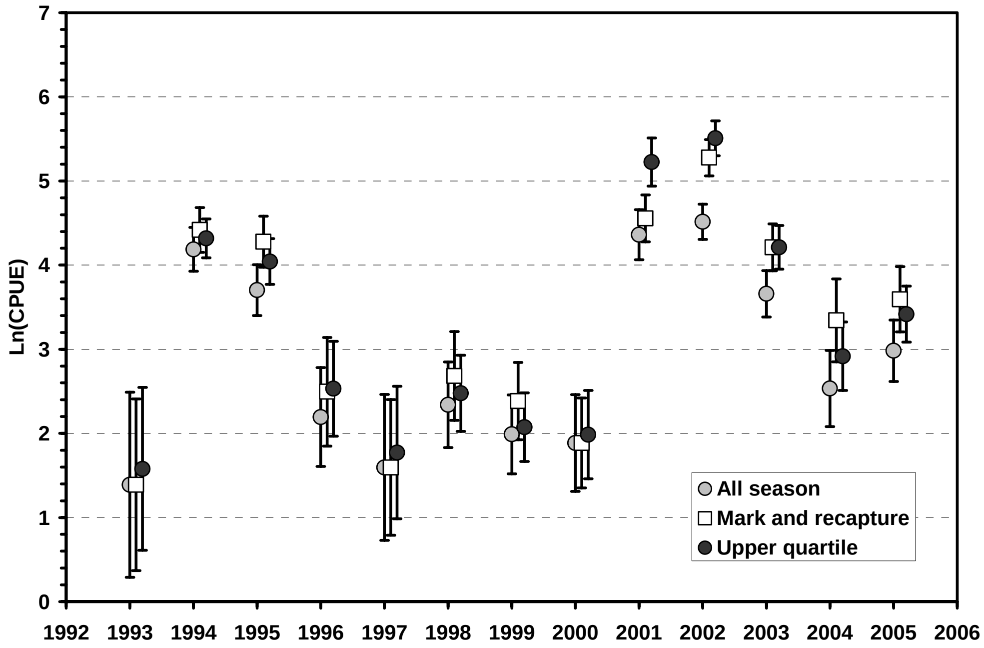


Figure 1.4. Abundance index (Ln(CPUE) mean +/- 2 standard errors) of striped bass spawners in the gaspereau trapnets of the Northwest Miramichi, 1993 to 2005. All season refers to the CPUE derived using the observations over the entire sampling period. Mark and recapture refers to the CPUE based on the sampling interval from the start of sampling to the end date of the mark and recapture experiment. Upper quartile is the CPUE derived using the period which encompassed the upper quartile of the CPUE observations annually.

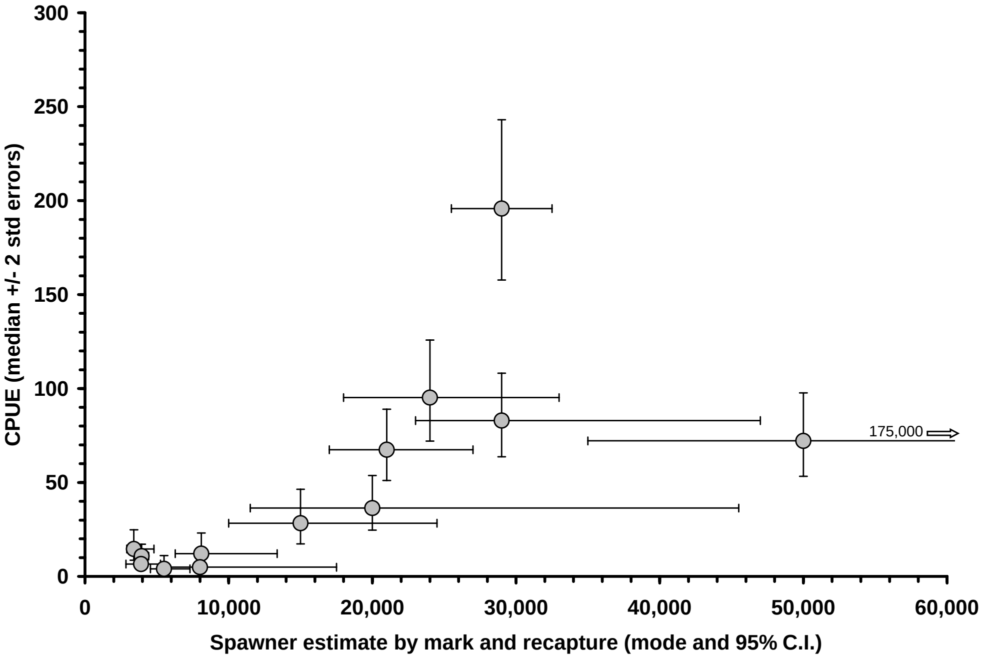


Figure 1.5. Association between spawner estimates from CPUE and mark-recapture for the same year. The CPUE estimate is based on catches during the mark-recapture experiment.

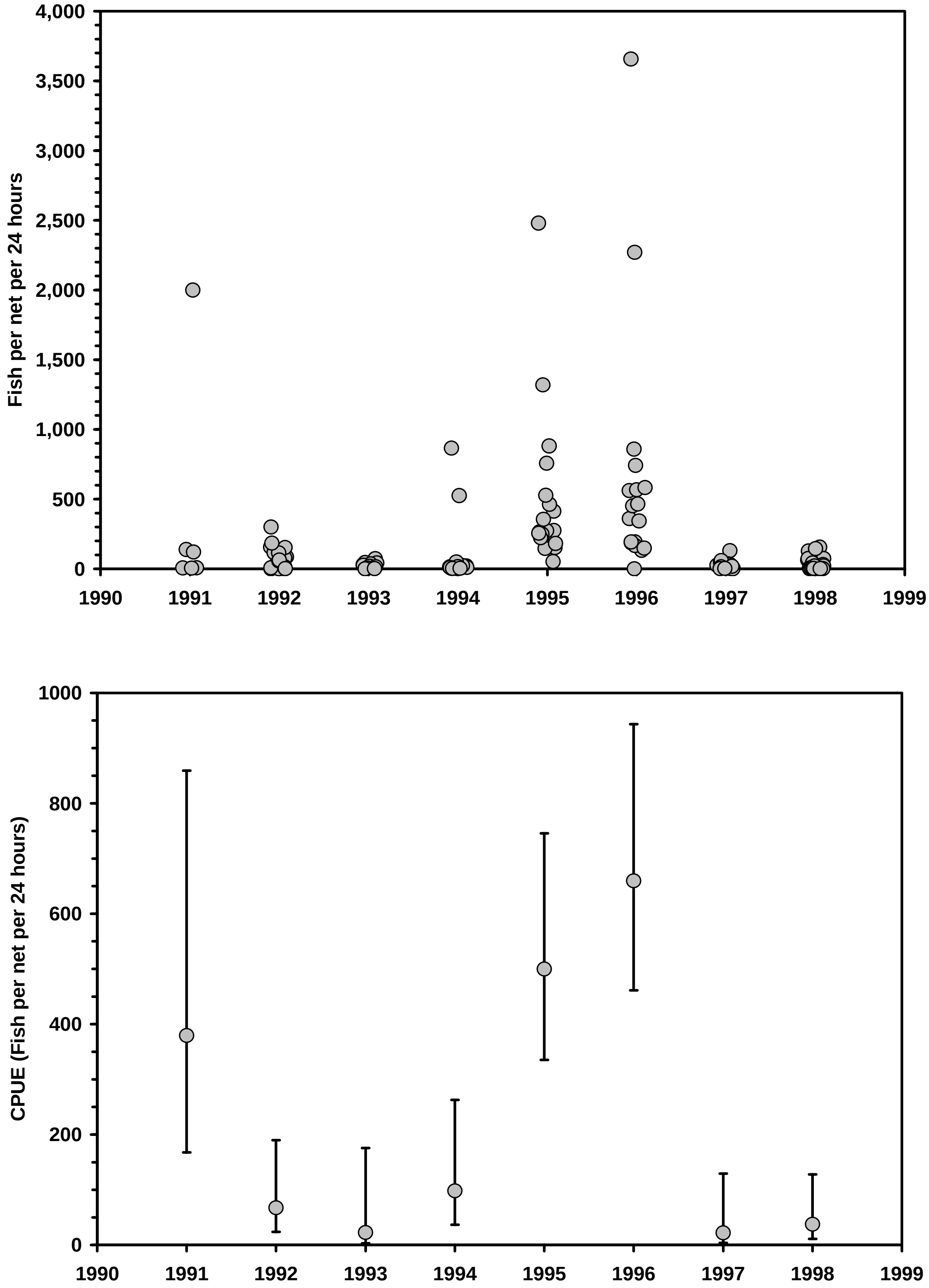


Figure 1.6. Fall abundance index of young-of-the-year striped bass as inferred from bycatch in the fall open water smelt fishery of the Miramichi River. The upper panel summarizes the observations by year (jittered by year to reduce overlap) and the lower panel illustrates the CPUE as derived from the General Linear Model analysis with only year as an explanatory variable.

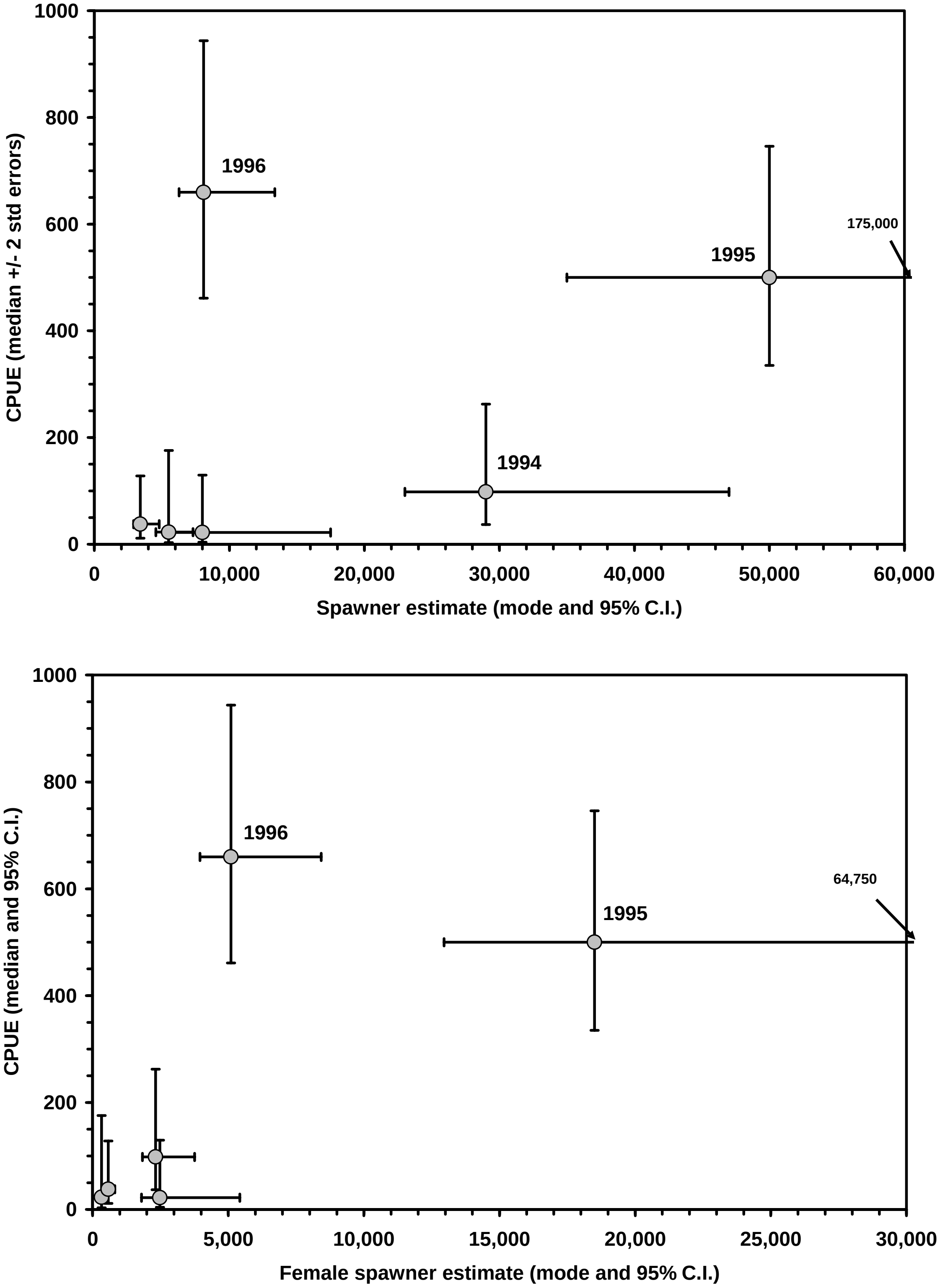


Figure 1.7. Association between the fall young-of-the-year abundance index (catch per net per 24 hours) and the estimated abundance of spawners (upper panel) and female spawners (lower panel) for the corresponding year.

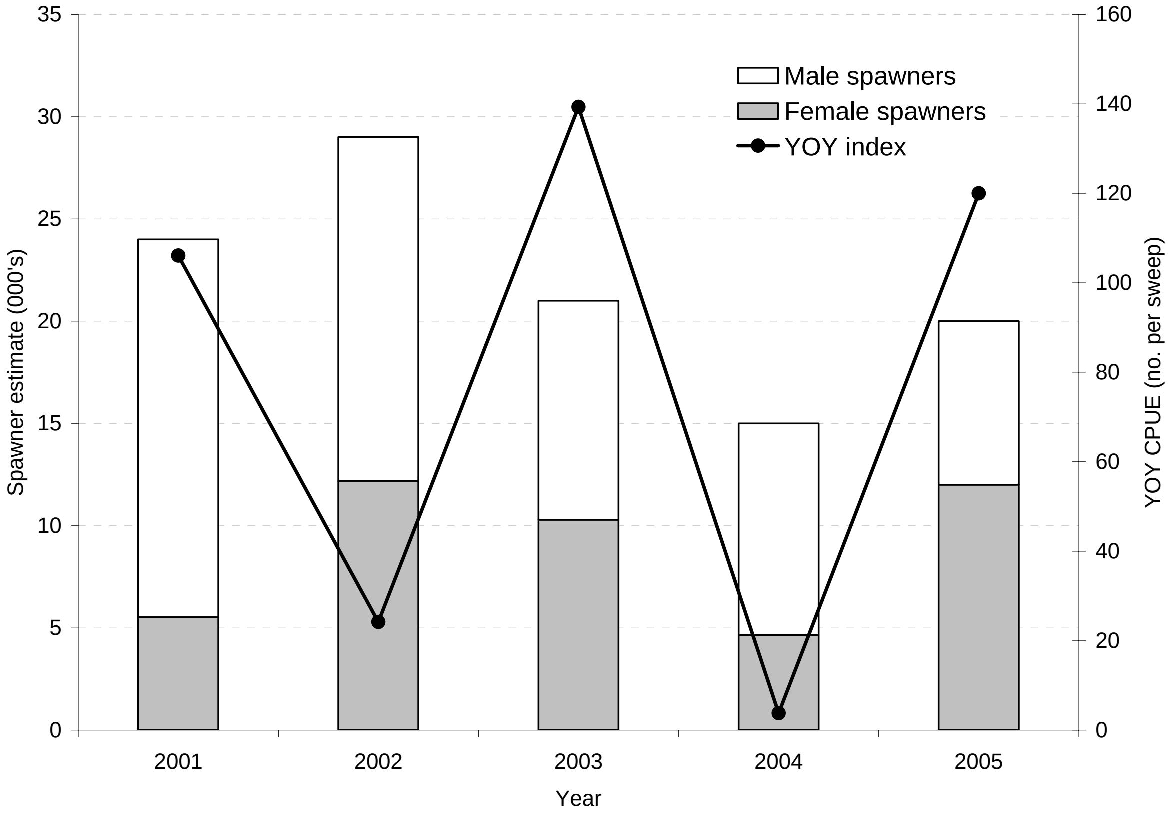


Figure 1.8. Relationship between YOY abundance derived from beach seine catches and spawner estimates based on mark recapture experiments.

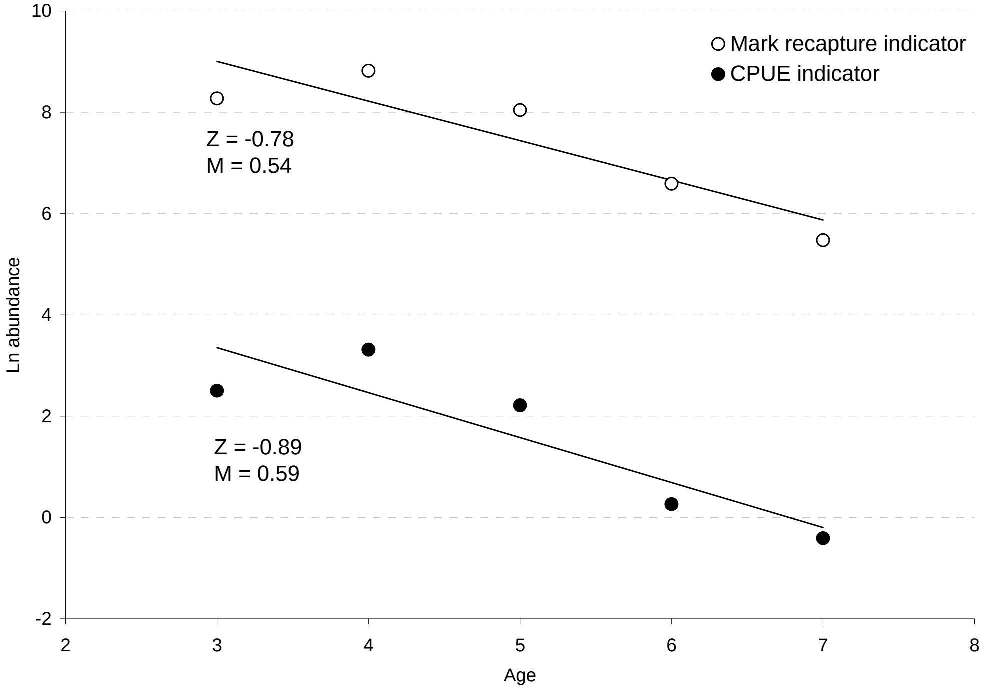


Figure 1.9. Average mortality of the southern Gulf striped bass using both the mark-recapture and CPUE spawner abundance indicators. Analysis includes only data collected from 1997 to 2005 after commercial fishing was closed.

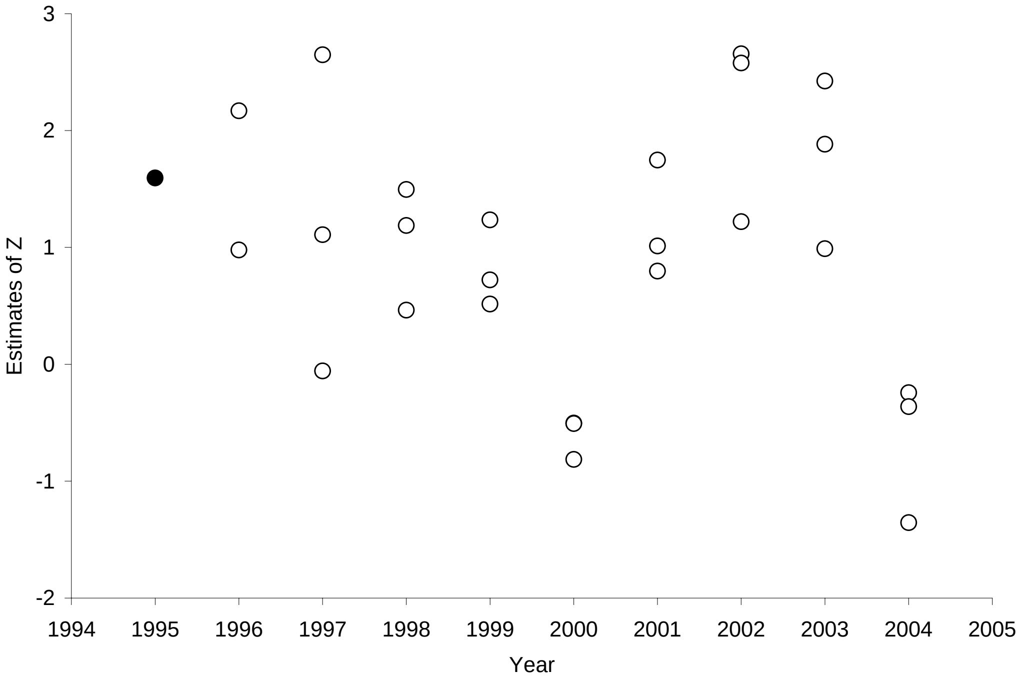
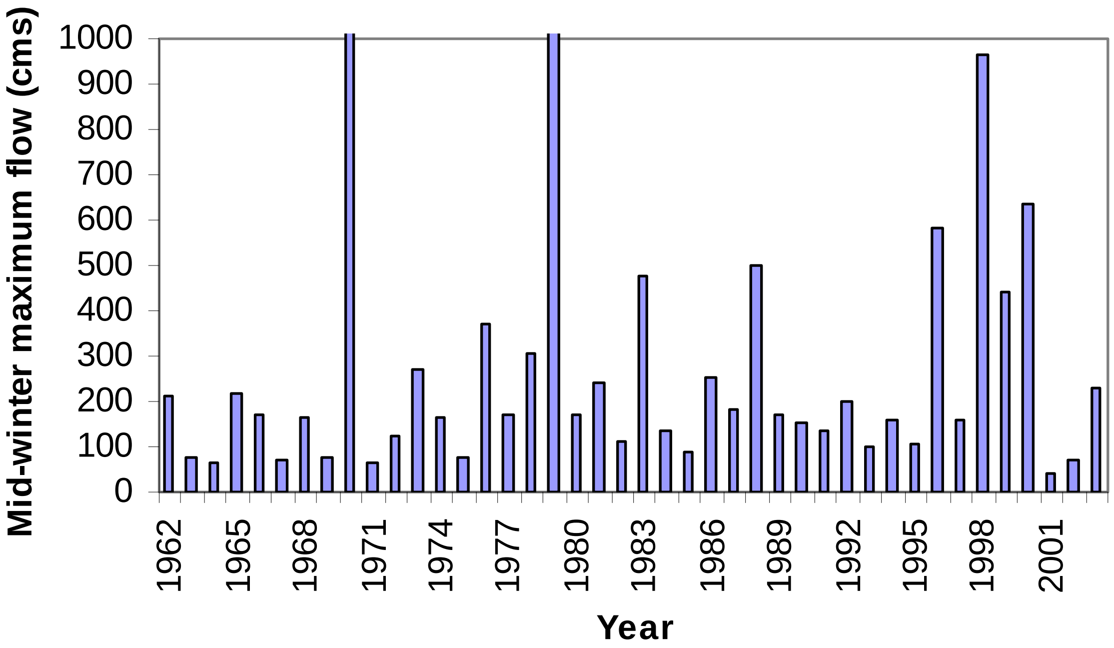


Figure 1.10. Individual Z estimates based on striped bass aged 4 to 6 only. Solid symbol indicates Z estimate when commercial harvesting was still permitted.



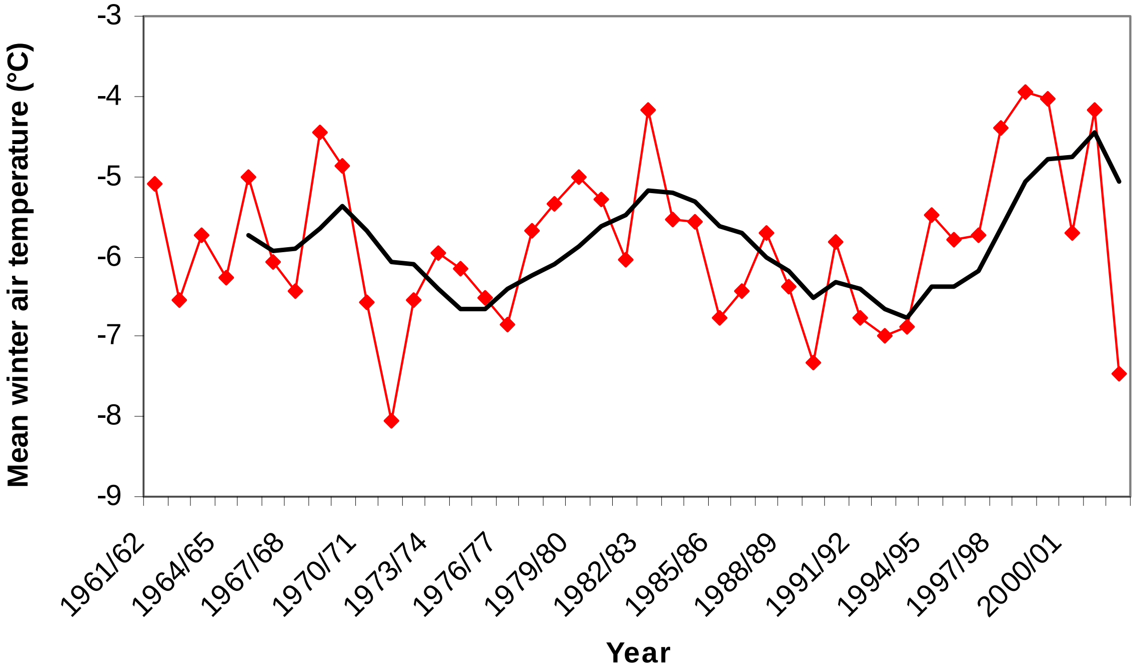


Figure 1.11. Maximum daily discharge in the winter months (Jan. to March) (upper panel) and mean winter air temperatures (Nov. to March) within the Miramichi area, 1961 to 2003 (lower panel). Solid line represents the 5 year running mean for winter air temperature.

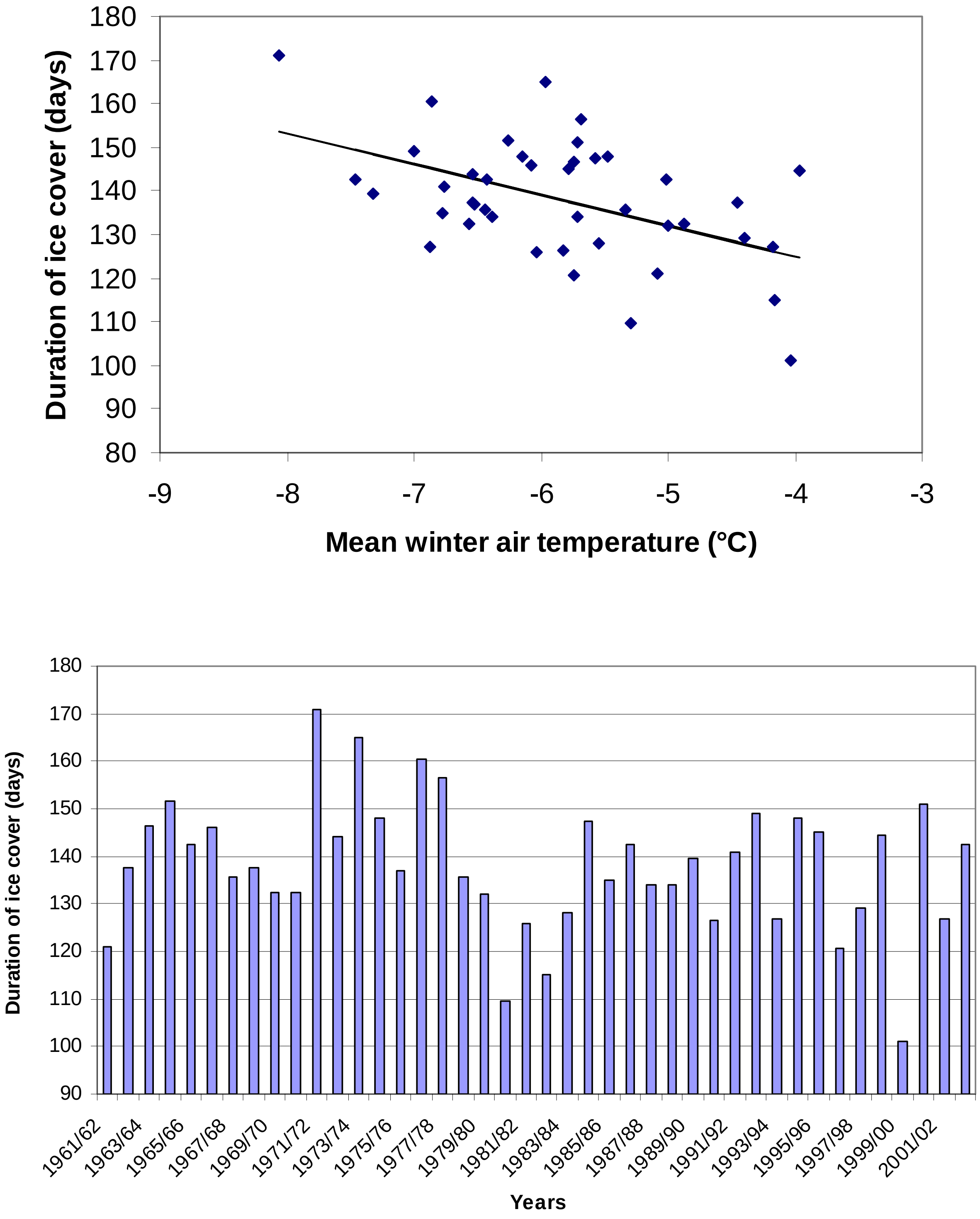


Figure 1.12. Mean air temperature in the winter (Nov. to March) versus duration of ice cover (days) (upper panel) and the duration of ice cover for the winters of 1961/62 to 2002/03 (lower panel) in the Miramichi area.

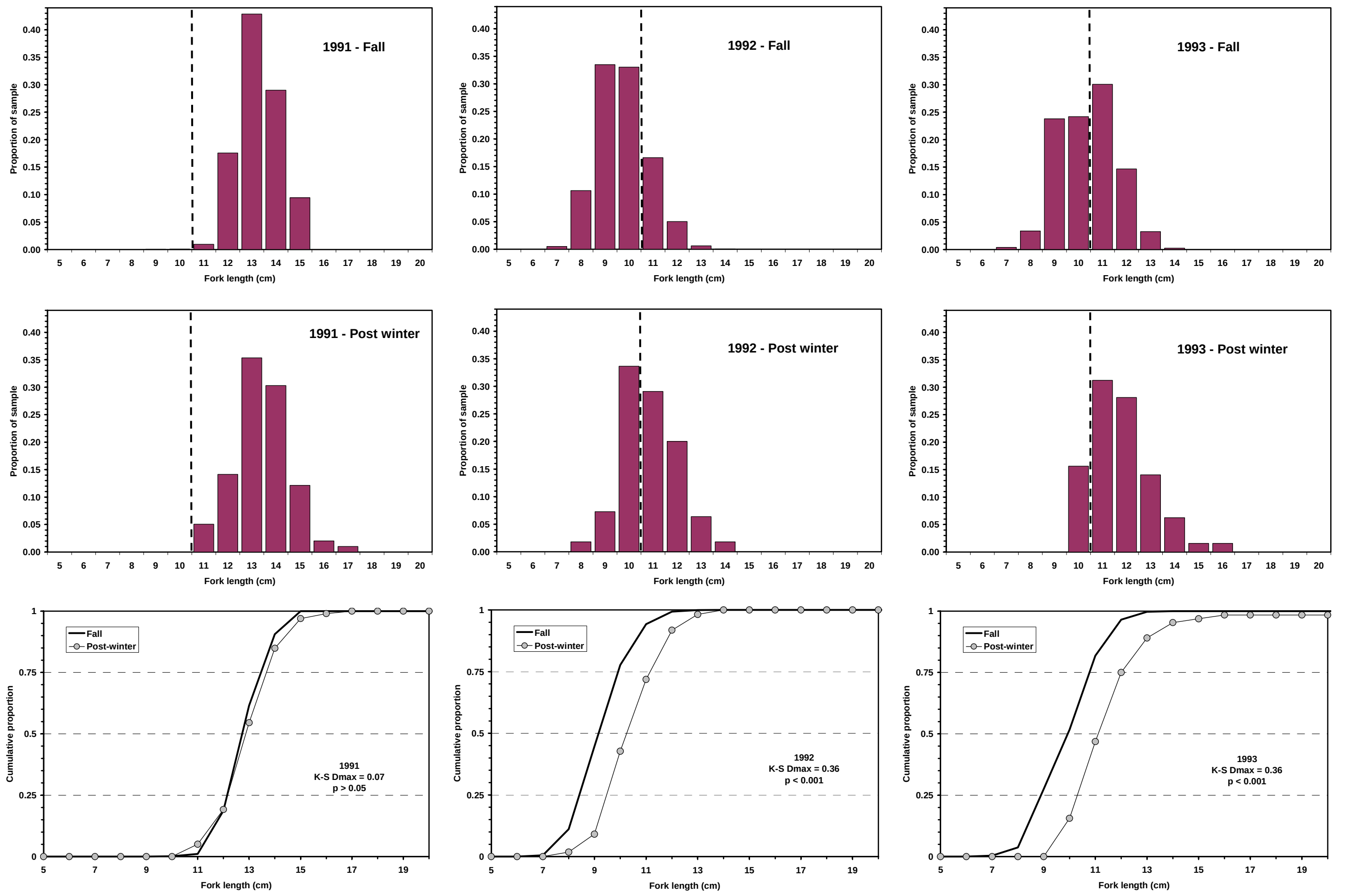


Figure 1.13. Length frequency distributions (4.5 to 5.4 = 5 cm group) of the 1991 to 1993 cohorts of juvenile striped bass in the fall open water smelt fishery of the Miramichi River (upper panels) and length frequency distributions of the survivors based on back-calculated size-at-age from the scales of striped bass sampled at age 2 years in 1993 to 1995 (middle panel) and cumulative distributions (lower panel). The vertical hatched lines were included for reference between the pre and post winter length distributions.

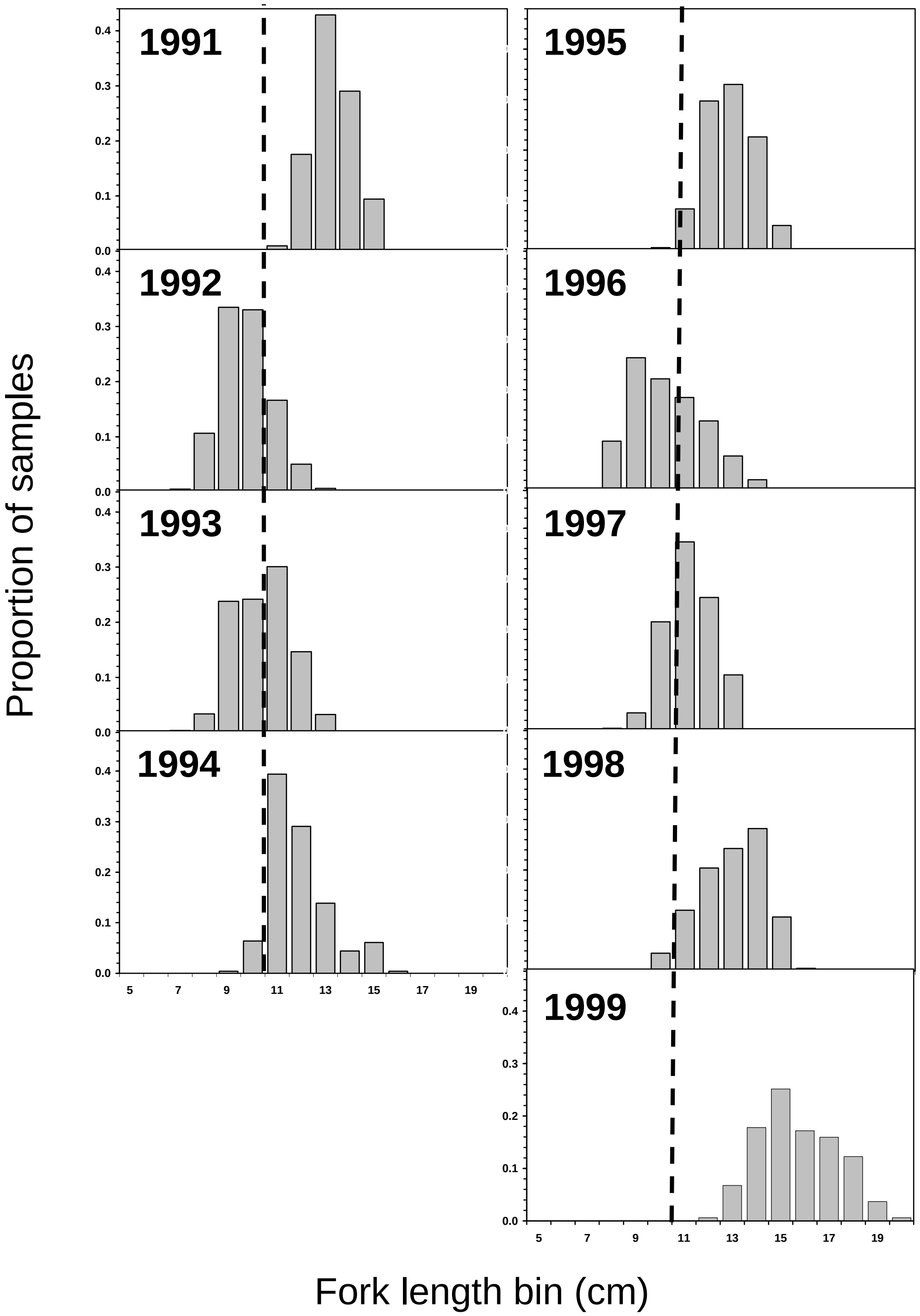


Figure 1.14. Length frequencies of young-of-the-year striped bass as sampled in the fall open-water smelt fishery of Miramichi Bay (1991 to 1998) and the Tabusintac estuary (1999). The vertical hatched lines were included to demonstrate the size variability among years.

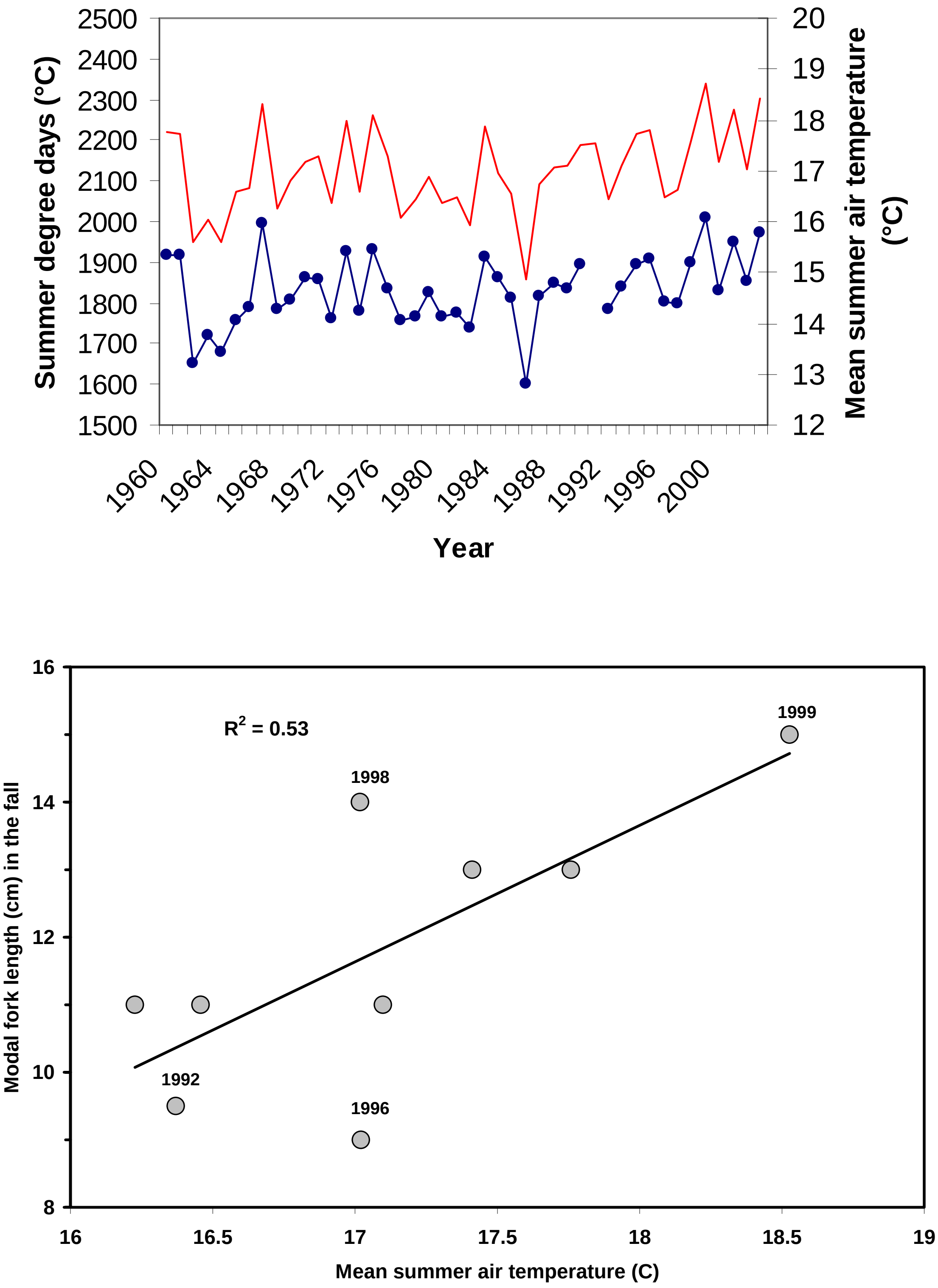


Figure 1.15. Degree days and mean summer air temperatures in Miramichi for 1960 to 2003 (upper panel) and association between modal length of YOY in the fall and mean summer air temperature for the 1991 to 1999 cohorts (bottom panel).

Figure 1.16. Place names of the southern Gulf of St. Lawrence indicating area of occupancy and locations where striped bass have been sampled.

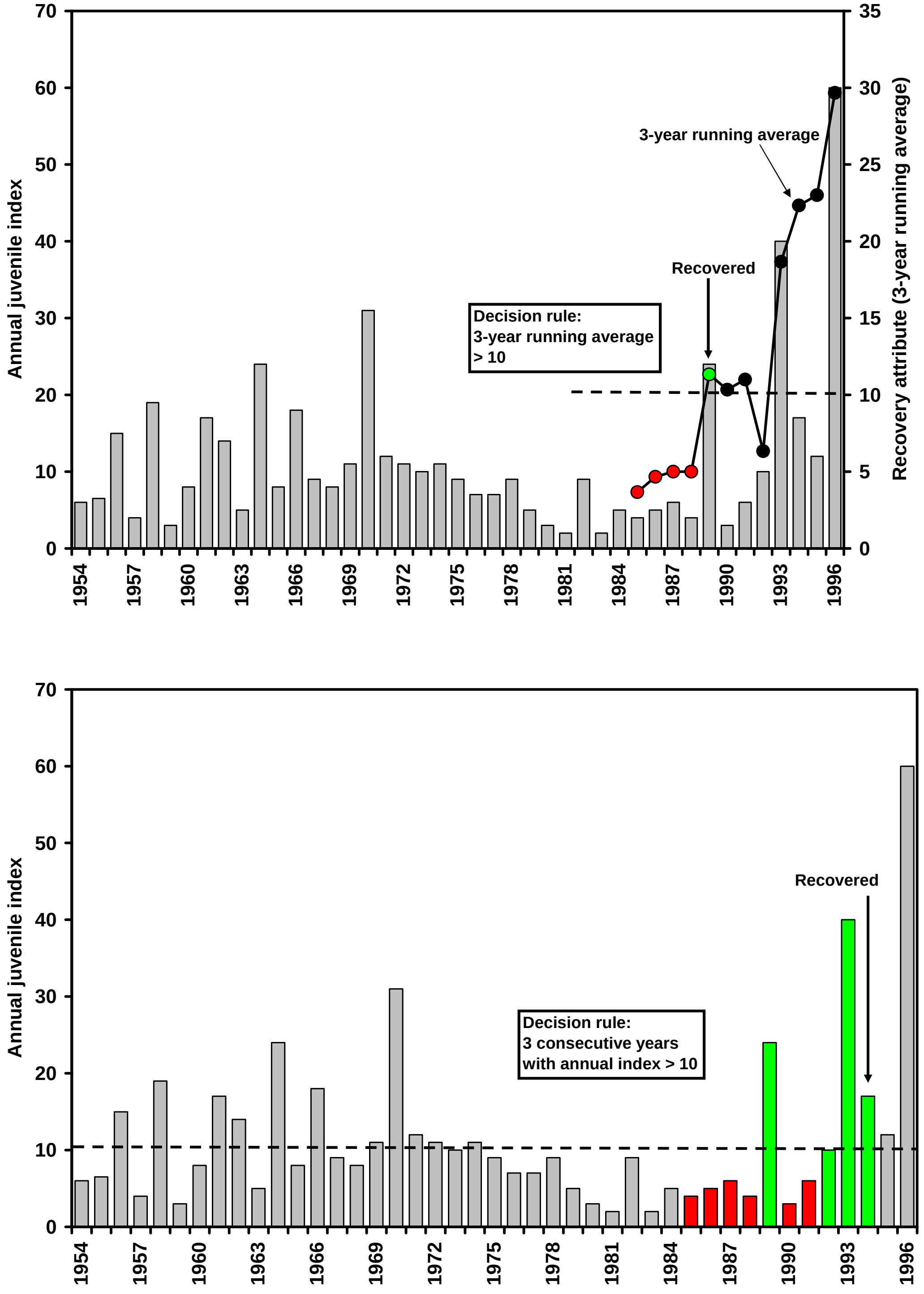


Figure 2.1. Illustrative examples of end point decision rules (or compliance rules) of stock recovery attributes and their application to the case of the Chesapeake Bay striped bass as described in Richards and Rago (1999). The upper panel illustrates the conclusion that the stock was recovered in 1989 based on the three-year running average of the annual index exceeding the long term average of the attribute. The lower panel illustrates an alternative end point decision rule which would have concluded that recovery occurred in 1994 after the annual index exceeded the recovery objective for three consecutive years.

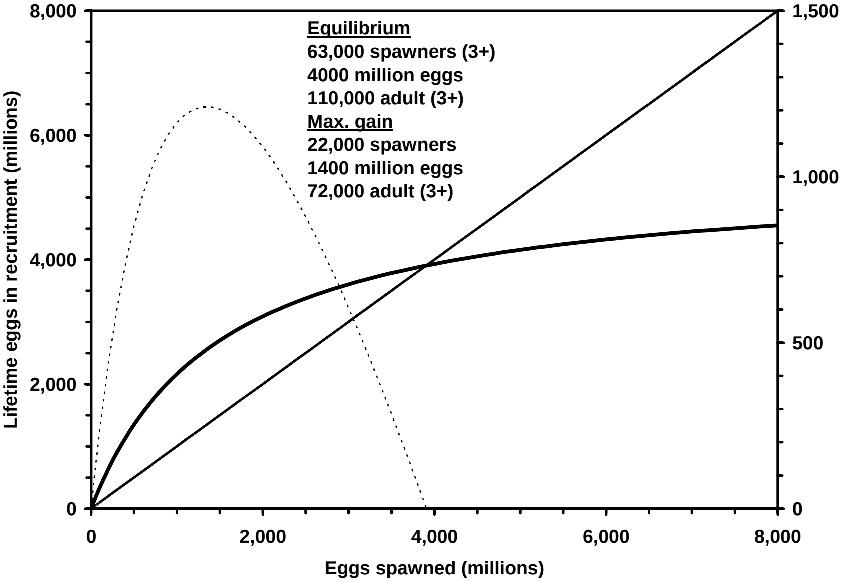


Figure 2.2. Beverton-Holt stock and recruitment relationship for striped bass eggs to young-of-the-year abundance in the fall based on density independent survival of 0.1% and young-of-the-year mean carrying capacity of 1.5 million fish. Also shown is the gain in eggs (lifetime egg production minus spawning eggs) line relative to eggs spawned.

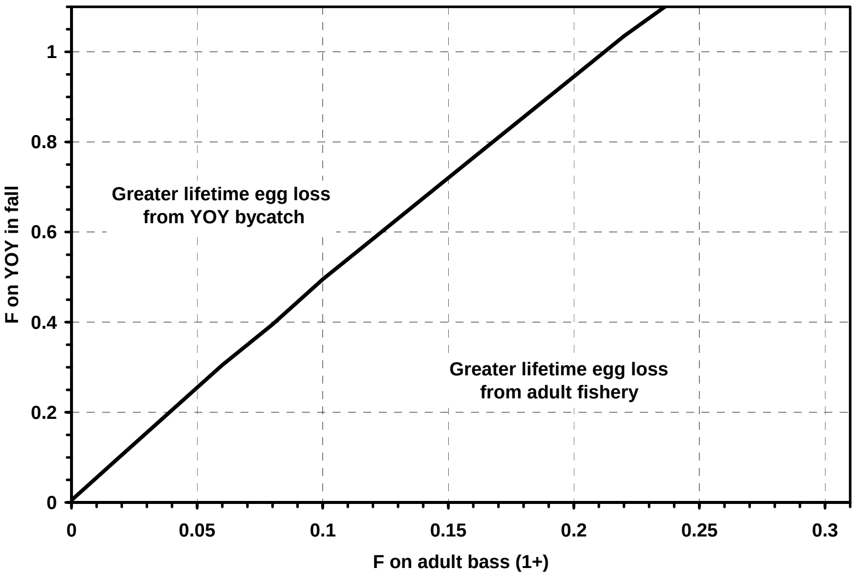
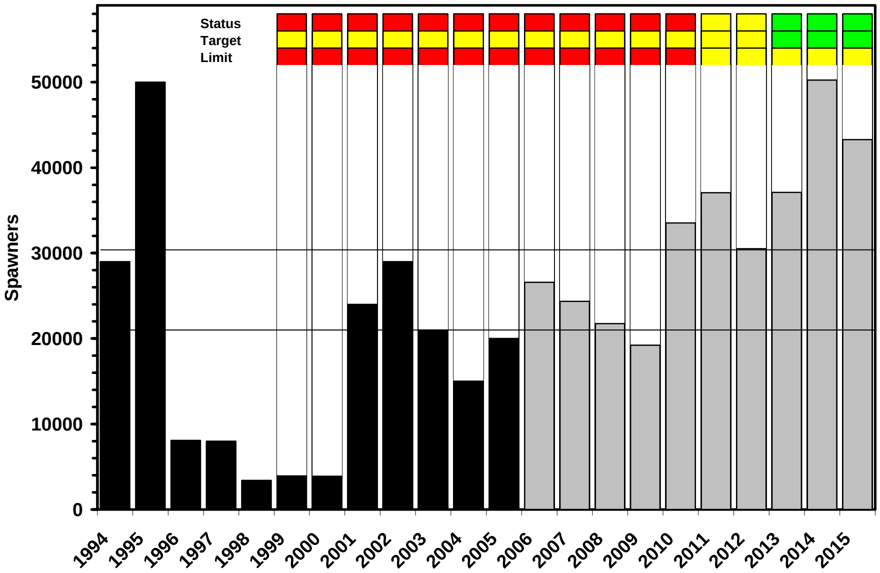


Figure 2.3. Lifetime egg loss equivalents of fishing YOY in the fall versus fishing bass age 1 and older. The diagonal line is the equivalence line for which lifetime egg loss is the same at the F described for YOY versus for age one and older bass. For YOY, overwinter survival (M) occurs over six months, of which one month occurs simultaneously with F. For age one and older bass, F is the same at all ages, occurs simultaneously with M, and all age groups 1 year and older are fully recruited to the fishery.



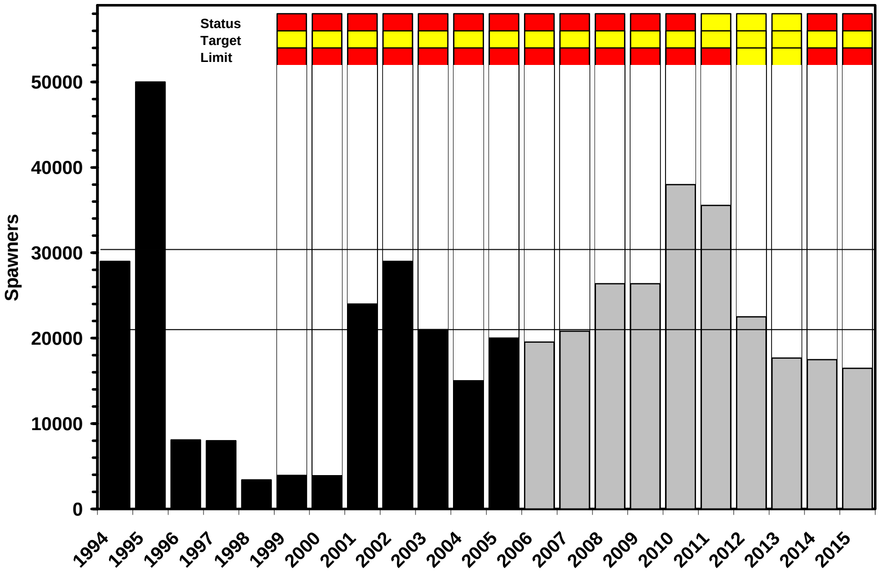


Figure 2.4. Application of traffic light summary to the spawner abundance attribute to illustrate status relative to the limit, the target and overall for striped bass from the southern Gulf. The graphs show two stochastic realizations under fishing on YOY (F=0.22) and adults (F = 0.044). The black bars are the modes of the estimated abundance of spawners based on mark and recapture experiments, 1994 to 2005. The grey bars are simulated values based on the simulations initiated using the spawner estimates for 1994 to 1996. The limit compliance rule is: red (0) spawners < 21,600 in more than one year out of six, yellow (1) otherwise. The target compliance rule is: green (2) if spawners > 31,200 spawners in 3 or more out of 6 consecutive years, yellow (1) otherwise. The status is evaluated as the product of the limit and target values.

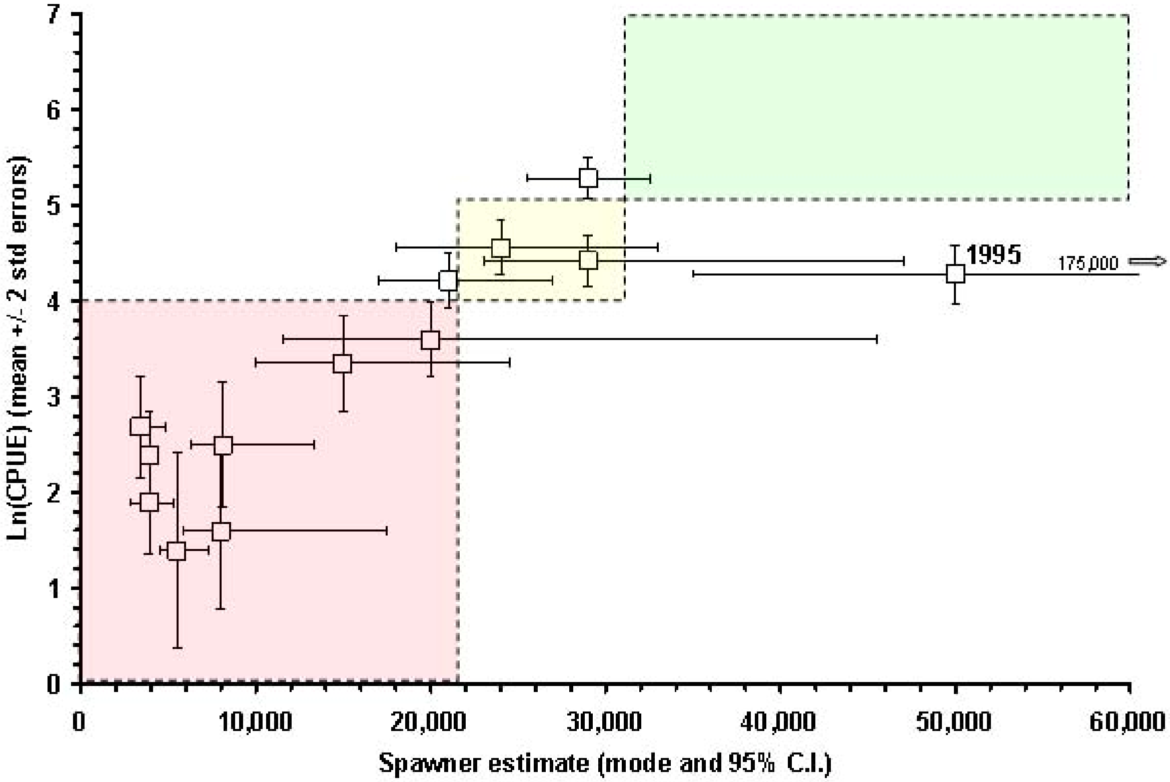


Figure 2.5. Association between spawner estimates from CPUE and mark-recapture experiments for the same year in the gaspereau fishery of the NW Miramichi. The CPUE estimate is based on catches during the mark-recapture experiment. The rectangles describe the corresponding red, yellow and green zones for defining the limit and target recovery objectives.

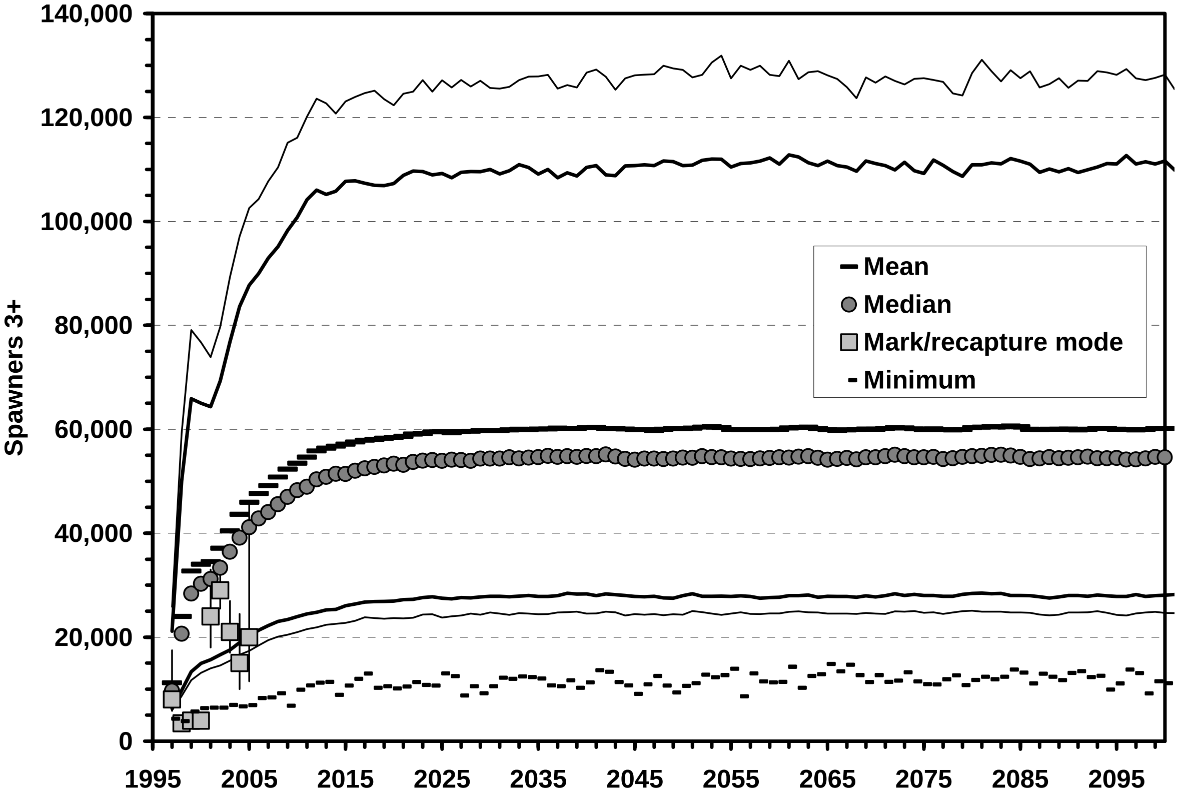


Figure 2.6. Trajectories of modeled spawner abundance of striped bass from the southern Gulf resulting from stochastic variation in survival in the absence of directed fisheries. Upper and lower lines represent 97.5 and 95th percentiles, 5 and 2.5th percentiles, respectively. Error bars around the mark and recapture modes are 95% confidence interval range for the spawner estimates, 1997 to 2005.

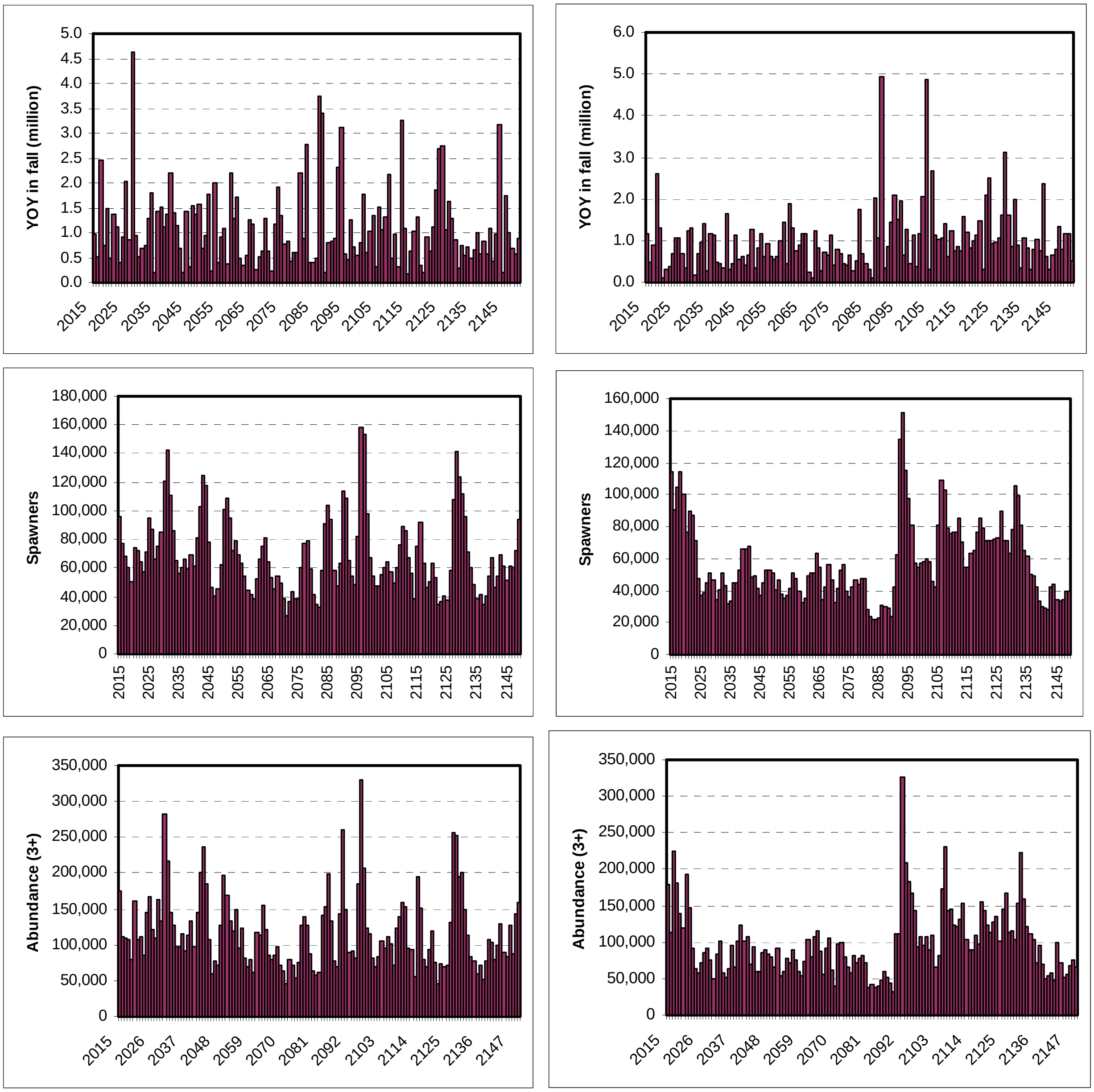


Figure 2.7. Examples of individual trajectories (left, right panels) of YOY abundance, spawner abundance and total adult abundance generated by the life history model with stochasticity in the absence of directed fisheries.

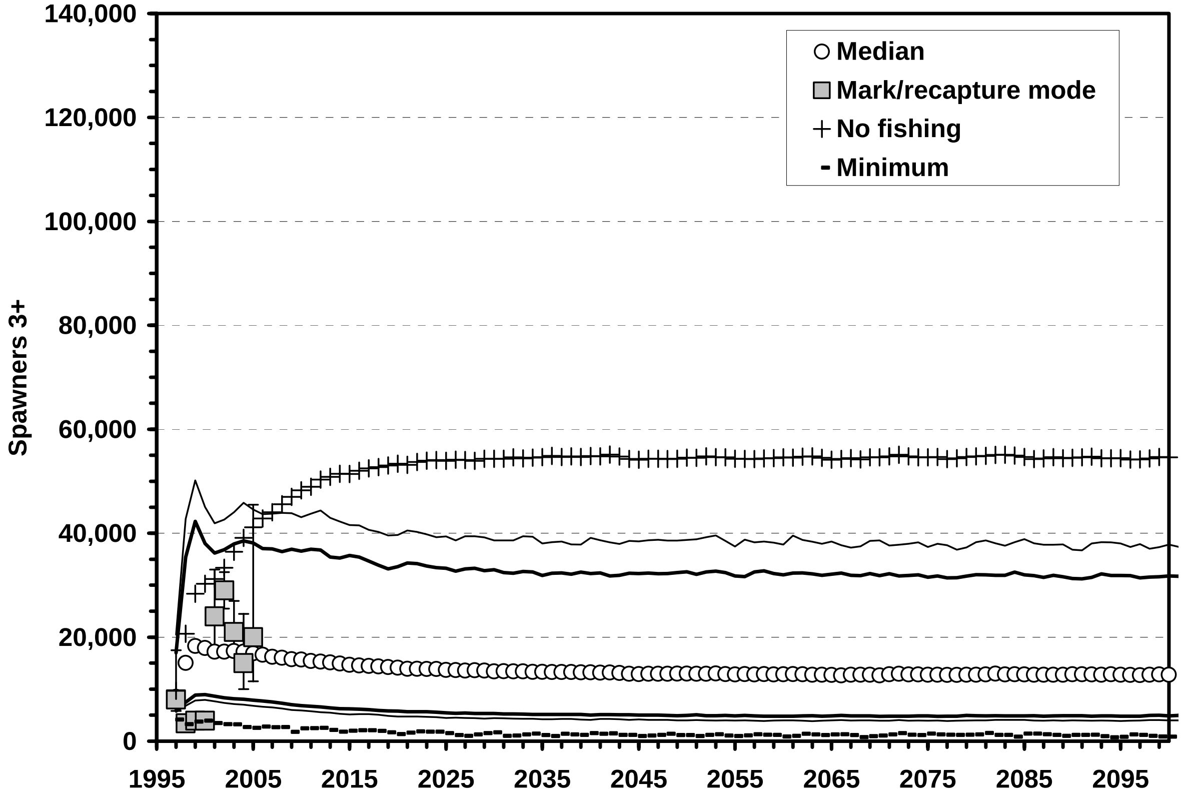


Figure 2.8. Trajectories of modeled spawner abundance of striped bass from the southern Gulf resulting from stochastic variation in survival with bycatch of young-of-the-year at a rate of F = 0.1 and fishing on adults age 2 years and older of F = 0.2. Upper and lower lines represent 97.5 and 95th percentiles, 5 and 2.5th percentiles, respectively. Error bars around the mark and recapture modes are 95% confidence interval range for the spawner estimates, 1997 to 2005.

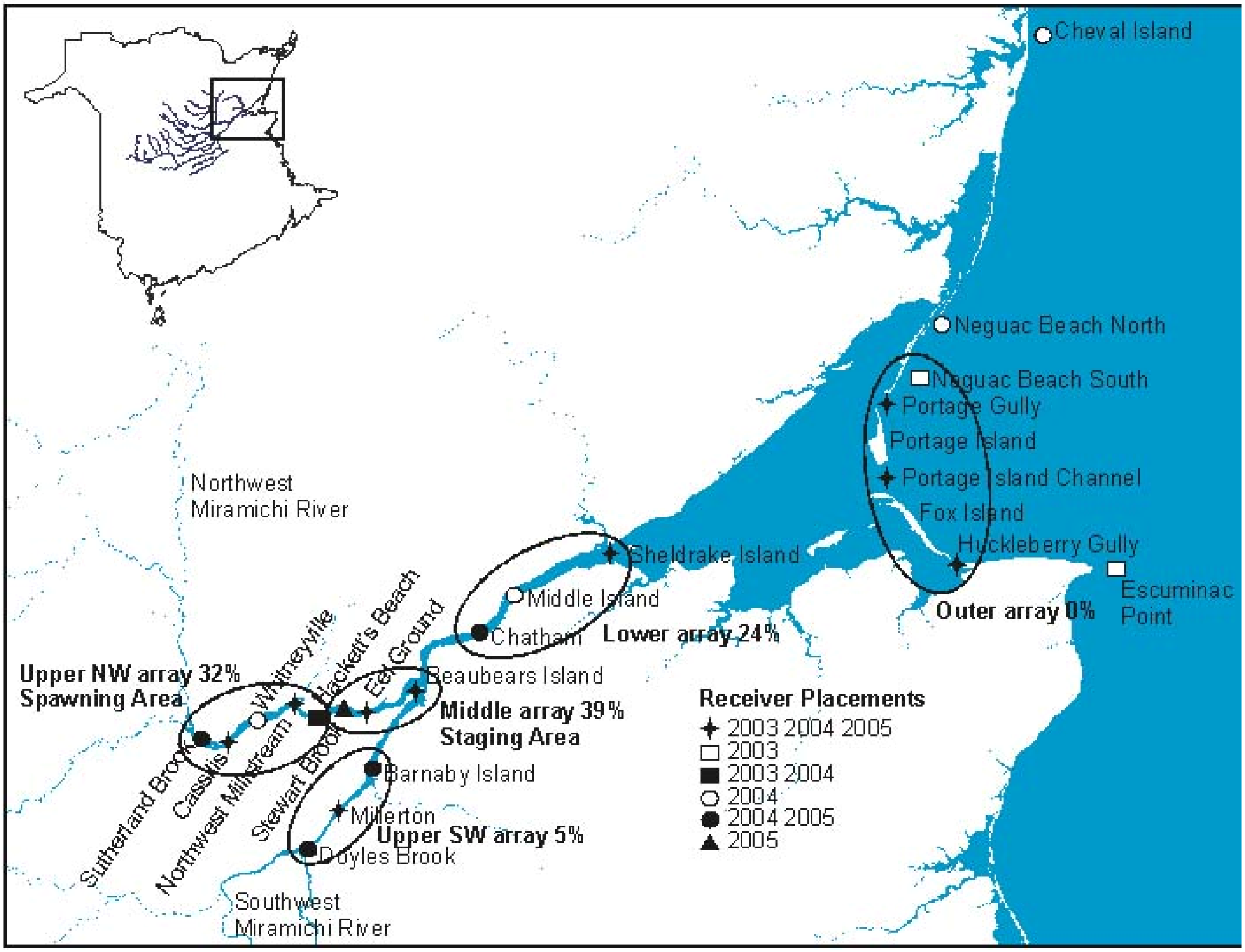


Figure 2.9. Preliminary analysis of the proportion of time (bolded text) occupied by implanted striped bass in the Miramichi system in 2004-05.

Appendix A. Remit for striped bass RPA held in Moncton on November 30-December 2, 2005.

Recovery Potential Assessment
Striped Bass Populations of Miramichi, Bay of Fundy and St. Lawrence Estuary

Meeting of the Gulf Regional Advisory Process

Gulf Fisheries Centre,
Miramichi Boardroom (6th floor)
Moncton, New Brunswick

November 30 – December 2, 2005

REMIT

Background

In November 2004, three designatable units of Striped Bass were considered by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). The Miramichi and Bay of Fundy DUs were both designated as Threatened (TH) while that of the St. Lawrence Estuary was designated as Extirpated (EX). These DUs are being considered for listing in Schedule 1 of the Canada's Species at Risk Act (SARA). For DUs designated and listed under SARA as EN or TH, activities that would harm the species would be prohibited and a recovery plan would be required. Until such a plan is available, section 73 (2) of SARA authorizes competent Ministers to permit otherwise prohibited activities affecting a listed wildlife species, any part of its critical habitat, or the residences of its individuals. These activities can only be authorized, 1) if the activity is scientific research relating to the conservation of the species and conducted by qualified persons, 2) or benefits the species 3) or is required to enhance its chances of survival in the wild, 4) or affecting the species is incidental to the carrying out of the activity.

Decisions made on permitting of incidental harm and in support of recovery planning need to be informed by the impact of human activities on the species, alternatives and mitigation measures to these and the potential for recovery. An evaluation framework, consisting of three phases (species status, scope for human induced harm and mitigation) has been established by DFO to allow determination of whether or not SARA incidental harm permits can be issued.

To inform decisions relating to listing of the Miramichi and Bay of Fundy Striped Bass DUs and their recovery planning, the meeting participants will review analyses prepared to meet the objectives stated below.

Objectives

For each Designable Unit (DU):

Phase I: Species Status

1. Evaluate present species trajectory
2. Evaluate present species status
3. Evaluate expected order of magnitude / target for recovery
4. Evaluate expected general time frame for recovery to the target
5. Evaluate Residence-Habitat Requirements

Phase II: Scope for Human – Induced Mortality

Évaluation du Potentiel de Rétablissement
Populations du bar rayé de la Miramichi, baie de Fundy et de l'estuaire du Saint Laurent

Processus Consultatif Régional de la région du Golfe

Centre des pêches du Golfe,
Salle Miramichi (6ième étage)
Moncton, Nouveau Brunswick

Du 30 novembre au 2 décembre 2005

MANDAT

Contexte

En novembre 2004, le Comité sur la situation des espèces en péril au Canada (COSEPAC) a considéré trois populations (unité désignable : UD) de bar rayé. Les populations de la Miramichi et de la Baie de Fundy ont été toutes deux désignées comme menacées (M) tandis que celle de l'estuaire du Saint Laurent a été désignée comme disparue du Canada (DC) par COSEPAC. Ces populations sont présentement considérées pour être ajoutées à la l'annexe 1 de la loi sur les espèces en péril (LEP) du Canada. Pour les populations désignées et inscrites sous la LEP comme menacées (M) ou en voie de disparition (VD), les activités qui nuiraient à l'espèce seraient interdites et un plan de rétablissement serait exigé. Jusqu'à ce qu'un tel plan ne soit disponible, la section 73 (2) de la LEP autorise des Ministres assignés de permettre des activités normalement interdites affectant une espèce inscrite, son habitat critique, ou les résidences de ses individus. Ces activités ne peuvent être autorisées que si les activités; 1) sont des travaux scientifiques visant à la conservation de l'espèce et sont conduites par des personnes qualifiées, 2) ou bénéficieront l'espèce 3) ou son nécessaires pour augmenter ses chances de survie en milieu naturel, 4) ou l'impact sur l'espèce est accidentel et le résultat d'activités fortuites.

La décision de permettre des dommages fortuits et du besoin d'un plan de rétablissement, doivent prendre en considération les impacts des activités humaines sur l'espèce, les alternatives et les mesures permettant d'atténuer ces impacts, ainsi que le potentiel de rétablissement. Une structure d'évaluation, consistant de trois phases (le statut d'espèce, la portée des activités humaines et impacts incités, ainsi que les mesures d'atténuation) ont été établis par DFO pour permettre de déterminer si vraiment des permissions de dommages fortuits peuvent peut être émis

Pour informer des décisions touchant à l'inscription du bar rayé de l'unité désignable de la Miramichi et de la Baie de Fundy et leur planification de rétablissement, les participants à la réunion passeront en revue des analyses préparées pour rencontrer les objectifs exposés ci-dessous.

Objectifs

Pour chaque unité désignable (UD)

Phase 1 : Statut de l'espèce

1. Évaluer la trajectoire de l'espèce (population).
2. Évaluer le statut de l'espèce.
3. Évaluer l'ampleur / la cible attendu pour le rétablissement.
4. Évaluer des délais généraux attendus pour le rétablissement à la cible.
5. Évaluer les exigences liées à la résidence de l'espèce.

Phase II Portée des activités humaines - Mortalité incitée

Appendix A continued. Remit for striped bass RPA held in Moncton on November 30-December 2, 2005.

6. Evaluate maximum human-induced mortality which the species can sustain and not jeopardize survival or recovery of the species
7. Document major potential sources of mortality/harm
8. For those factors NOT dismissed, quantify to the extent possible the amount of mortality or harm caused by each activity.
9. Aggregate total mortality / harm attributable to all human causes and contrast with that determined in task 5

*Phase III**: Mitigation and Alternatives***

To the extent possible,

10. Develop an inventory of all reasonable alternatives to the activities in task 7, but with potential for less impact. (e.g. different gear)
11. Develop an inventory of all feasible measures to minimize the impacts of activities in task 7
12. Document the expected harm after implementing mitigation measures as described and determine whether survival or recovery is in jeopardy after considering cumulative sources of impacts

Products

- For each DU, Canadian Science Advisory Secretariat (CSAS) Research Documents and a Recovery Assessment Report to address all objectives
- CSAS Proceedings of meeting
- For each DU, CSAS Research Document

List of requested and tentative participation

- National Headquater (NHQ) and Zonal DFO Science (requested)
- NHQ and Zonal DFO Fisheries Management (requested)
- First Nations
- Provinces NS, NB and Quebec (requested)
- NS, NB and Quebec harvesters
- Non Governmental Organizations

1. Évaluer le maximum de mortalité incitée par les activités humaines que l'espèce peut supporter tout en ne mettant pas en danger la survie ou le rétablissement de l'espèce.
2. Documenter les sources potentielles principales de mortalité et dommages.
3. Pour les sources ayant un impact, évaluez dans la mesure du possible l'impact et l'étendu de la mortalité ou du dommage causé par chaque activité.
4. Agréger la mortalité et les dommages attribuables aux activités humaines et les mettre dans le contexte des exigences identifiées à l'item 5.

*Phase III : **Réduction et alternatives***

Dans la mesure du possible,

5. Développer un inventaire de toutes les alternatives raisonnables aux activités dans l'item 7, mais avec le potentiel de minimiser l'impact. (Ex : différents engins de pêche).
6. Développer un inventaire de toutes les mesures réalisables permettant de réduire au minimum l'impact des activités dans l'item 7.
7. Documenter les dommages attendus après l'exécution de mesures de réduction/alternatives décrites. Déterminer si la survie ou le rétablissement sont menacés après considération du cumule des sources ayant des impacts.

Produits

- Pour chaque unité désignable, un document de recherche et un rapport d'évaluation du rétablissement seront produits afin d'adresser tous les objectifs identifiés. Ces documents seront publiés dans la série du secrétariat canadien de consultation scientifique (SCCS).
- Un compte rendu de la réunion
- Si possible, des documents de recherche du SCCS pour chaque unité désignable.

Participation requise et tentative

- Bureau National et les Sciences du MPO de différentes régions de l'atlantique (requis)
- Bureau National et Gestion de Pêche du MPO de différentes régions de l'atlantique (requis)
- Premières Nations
- Provinces de la N-É, du N-B et du Québec (requis)
- Industrie des pêches de la N-É, du N-B et du Québec
- Organisations non gouvernementales

Appendix B. Environmental conditions in the Miramichi.

INTRODUCTION

Hydrological events are important factors which can influence not only water resource availability but many fishery resources and their management. In particular, water availability and streamflow variability can affect stream biota at different life stages during the year. Striped bass is no exception, and environmental conditions can play an important role in its overall survival and growth. As a result and in order to increase our understanding of the influence of hydrometeorological events on striped bass habitat and population dynamics, an analysis was conducted within the Miramichi River basin to show historical environmental conditions.

The objective was to carry out hydrometeorological analyses within the Miramichi River basin and specifically to: a) investigate peak flow conditions in autumn and during winter, as well as annual flood events, b) determine overwintering conditions (e.g., duration of ice condition), c) determine average air temperature conditions during the summer as growth potential, and d) investigate hydrological conditions during spawning periods in the spring.

STUDY AREA

This study was conducted within the Miramichi river basin and two rivers were used for the analysis, the Southwest Miramichi River and the Northwest Miramichi River. The Southwest Miramichi River (station 01BO001) was used because it is the largest gauge basin within the Miramichi River and provides generalized hydrological information. The Northwest Miramichi River was used because striped bass are known to spawn there and therefore streamflow conditions within this river will better represent specific events of interest. Therefore, data from the Southwest Miramichi River was used for historical analysis and to analyze general environmental conditions while the Northwest Miramichi River data was used for specific event analysis related to spawning in the spring. The drainage basin of the Southwest Miramichi River at the hydrometric station is 5050 km^2 while the drainage basin area for the Northwest Miramichi River is 948 km^2 at the hydrometric station. Hydrometric data from 1962 to 2003 were obtained from the HYDAT CD-ROM version 2003 and more recent data 2004 and 2005 were obtained from Environment Canada.

METHODS

The analysis was carried out using historical hydrometric and weather data from the study area. Daily discharge data were also used to calculate annual floods and daily high flow events. Ice conditions in the Miramichi River were also obtained from hydrometric gauged data. Data on air temperature and precipitation were obtained from the Miramichi Airport.

For the study of flood data, each annual maximum daily discharge is established in relation to its cumulative frequency (*f*) using the Weibull plotting position formula (Chow et al. 1988):

[1]
$$f = \frac{m}{n+1}$$

where *m* refers to the rank of the annual maximum daily discharge in increasing order, and *n* is the number of years of record. For instance, the highest flood in 35 years of data has a value of *m* = 35 and *n* = 35. Therefore the frequency of such event is *f* = 35/36 = 0.972. Given the frequency (*f*) of an event it can be potted on a flood frequency paper where the position on the *x* axis is determined using the Gumbel reduced variable *y’*:

[2] $$y' = -\ln(-\ln(f))$$

where *f* is the cumulative frequency calculated by [1]. In the above case with 35 years of data (i.e. *f* = 0.972), the highest flood value has a *y'* value of 3.56 using [2]. This type of transformation was used for plotting annual floods due to the logarithmic nature of these events. Such a plotting transformation is referred to as a Gumbel paper frequency plot.

For the ice study, a *B* symbol indicator is included with the discharge data to identify that the discharge value had been corrected for periods when the hydrometric station was influenced by ice conditions. The presence of the *B* symbol was used as an index of ice conditions or ice cover. This ice condition index was observed within the Miramichi River using two approaches. The first approach estimated the duration of ice conditions in days and was obtained by the summation of all *B* indicators during the winter season. In the second approach, we identified both the beginning (first date with *B*) and end (last date with *B*) of ice condition in the river. If open water conditions are present in winter, the duration will be less than the difference between the beginning and the end of ice condition. Within the present study, only the ice duration was presented.

RESULTS

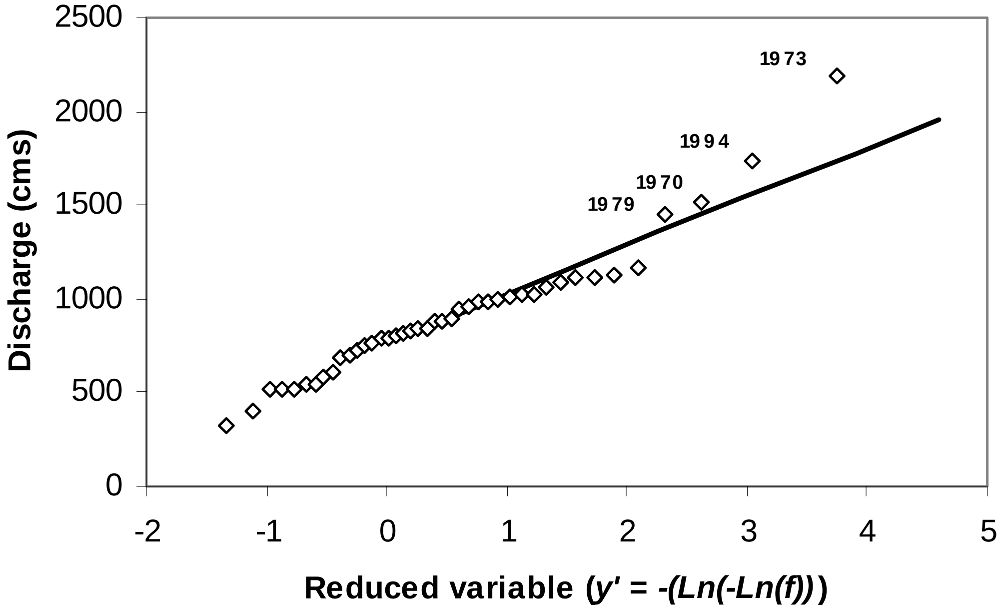
Flood events

The fitting of annual flood data for the Miramichi River was carried out using the 3 parameter lognormal distribution function to determine the frequency of events over the past 40 years (1962-2003). For this analysis data from the Southwest Miramichi River were used. Results showed good agreement between predicted frequencies and observed flood discharge, with the exception of very high flood events which exceeded calculated frequencies (Figure B-1). In fact, the four highest floods, which occurred in 1973, 1994, 1970 and 1979, all exceeded the predicted frequencies estimated using the 3 parameter lognormal distribution. It was also noted that among these four events, three occurred in the 1970s including the highest flood at 2190 m^3/s (1973) while the second highest and most recent high flood event occurred in 1994 at 1730 m^3/s.

Discharges as a function of recurrence interval are presented in Table B-1 for the Southwest Miramichi River. This table shows that the 2-year flood was estimated at 852 m^3/s (*y'* = 0.37 on Figure B-1) while the 50-year and 100-year flood events were estimated at 1780 m^3/s and 1957 m^3/s respectively (*y'* = 3.9 for 50-year and *y'* = 4.6 for 100-year flood; Figure B-1). It is clear from the fitted distribution function that the highest observed flood in the Miramichi River (i.e., at 2190 m^3/s in 1973), would exceed the recurrence interval of a 100 years. In fact, this event is more representative of a 1 in 240 year event based on the fitted distribution. The most recent high flood event was in 1994 at 1730 m^3/s and the recurrence interval of this event was estimated at 1 in 40 year based on data from Figure B-1.

Appendix Table B-1. Flood frequency analysis for the Miramichi River (using the Southwest Miramichi River data).

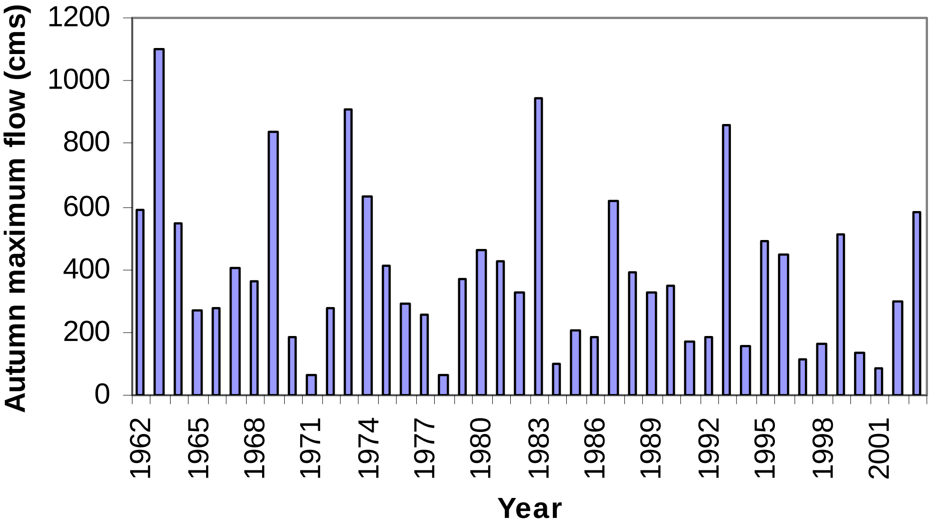
Recurrence interval (year)	Reduced variable (y')	Discharge (m^3/s)
2	0.37	852
5	1.50	1158
10	2.25	1355
20	2.97	1541
50	3.90	1780
100	4.60	1957



Appendix Figure B-1. Flood frequency analysis for the Miramichi River showing annual flood events between 1962 and 2003.

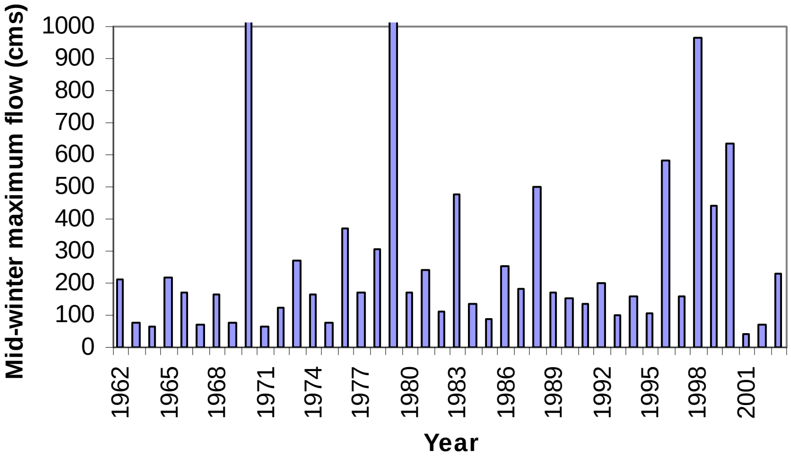
Autumn - winter peak flows

The second analysis consisted of looking at winter peak flows within the Miramichi River which may impact on overwintering survival conditions of striped bass. Data from the Southwest Miramichi River were used to better represent the basin wide condition within the Miramichi River. For this analysis, two separate periods were considered; autumn peak flows and mid-winter peak flows. For autumn peak flow, the month of November and December were selected while mid-winter peak flow consisted of analysing peak flows between January and March.



Appendix Figure B-2. Autumn peak flows (Nov-Dec) within the Miramichi River (using the Southwest Miramichi R. data) from 1962 to 2003.

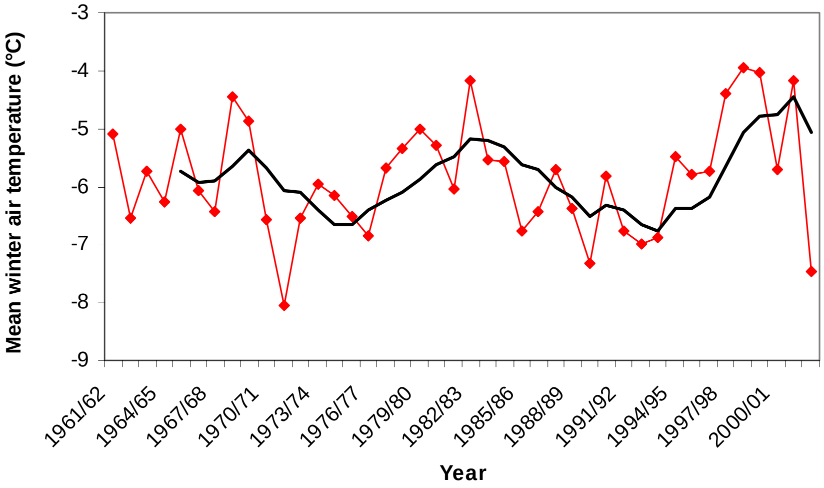
Peak flows in autumn showed historical values ranging from 65 m^3/s (1978) to a high value of 1100 m^3/s (1963) with a mean value of 390 m^3/s, which represented an average flow of approximately half of a 2-year flood (Table B-1). Recent years showed lower than average values with the exception of 1999 and 2003 which showed autumn peak flows of 514 m^3/s and 583 m^3/s respectively (Figure B-2). The second analysis of winter peak flow consisted of analysing mid-winter conditions for the period of January to the end of March. It should be pointed out that mid-winter conditions generally excludes the spring maximum discharge for the Miramichi River because the peak spring flow almost always occurs in April and May. Therefore, this period of mid-winter conditions will be reflective of mid-winter thaw period resulting from higher air temperature and rainfall events which may contribute to ice break-up and occasionally ice jams. From this time series, two peak flow values were observed to exceed 1000 m^3/s during mid-winter (Figure B-3). The highest value was observed in 1970 (Feb 5) at 1520 m^3/s and the second highest was observed in 1979 (1440 m^3/s; Mar 27). In recent years both high and low peak flows were observed. For instance, the lowest mid-winter peak flow value of the time series was observed in 2001 at 42 m^3/s. Conversely, a number of significant mid-winter peak flows were observed in the late 1990s and the maximum value was observed in 1998 at 964 m^3/s.



Appendix Figure B-3. Mid-winter peak flows (Jan-Mar) within the Miramichi River (using the Southwest Miramichi R. data) from 1962 to 2003.

Winter air temperatures
Following the analysis of peak flows on both an annual basis and during the winter period, mean winter air temperatures (from November to March) were investigated to help explain factors that could influence the overwintering survival of striped bass. Results of mean winter temperatures are shown in Figure B-4. Results show a significant level of variability in mean winter air temperature ranging from -8.1°C (1971-72) to -4.0°C (1998-99). The average mean winter temperature for the whole time series (between 1961 and 2003) was calculated at -5.8°C.

The long-term signal, represented by the 5 year running mean, was also shown in Figure B-4 and results show consecutive periods of warm and cold winter temperatures. For instance, a warm period was observed in mid-1980s while the warmest period of the time series was observed in 2001-02 mainly as a result of 4 warm winters starting in 1997-98 and extending into 2001-02 season.

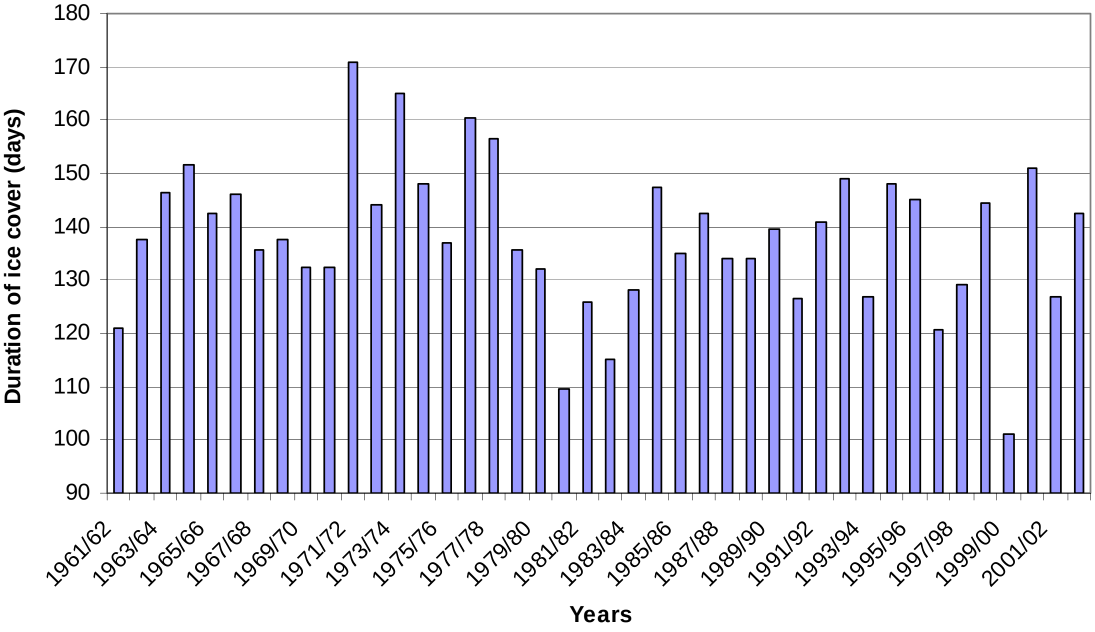


Appendix Figure B-4. Mean winter air temperature (Nov – March) within the Miramichi River (using the Miramichi Airport data) from 1962 to 2003 (square symbols). Solid line represents 5 year running mean.

Winter severity

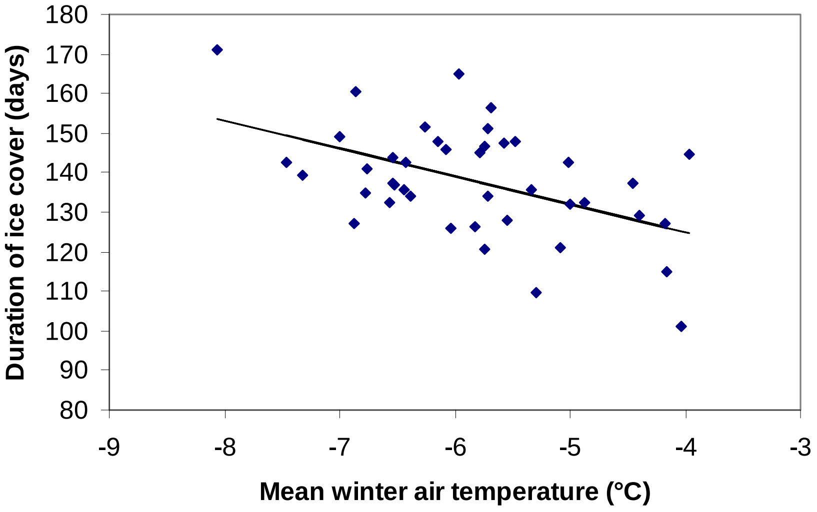
Winter conditions can be studied using peak flows and mean air temperature as indices; however, the ice conditions or the duration of ice cover can also be an important indicator of the severity of winter within the Miramichi River system. As such, ice condition within the Miramichi River was studied using hydrometric gauged data (using data from both the Southwest Miramichi and Northwest Miramichi rivers) based on the *B* indicator, which is used by Environment Canada when a station's water level is influenced by ice. Therefore, in this study ice influence will be assumed to be the same as ice cover. The two rivers were analyzed and average conditions among them were used to reflect ice conditions for the whole system. Data on the duration of ice cover are a good indicator of the severity of winters over the years and should provide valuable information on historical trends.

The duration of ice cover in the Miramichi River ranged from 101 days during the winter of 1999/00 to 171 days during the winter of 1971/72 (Figure B-5). The mean number of ice covered days in the river was calculated at 138 days (±14 days; std). Data indicate that the duration of ice cover in 1960s and 1970s were somewhat higher than in recent years. For instance, only one season experienced less than 130 days of ice cover in the 1960s while no seasons were observed with less than 130 days during 1970s. During the 1980s, four consecutive seasons experienced duration of ice cover less than 130 days (winters 1980-84). Similarly, during the 1990s, a total of five winter seasons experienced duration less than 130 days: 1990-91, 1993-94, 1996-97, 1997-98 and 1999-00. After the year 2000, the winter season of 2001-02 was observed to have less than 130 days of ice cover.



Appendix Figure B-5. Duration of ice cover within the Miramichi River from 1961 to 2003.

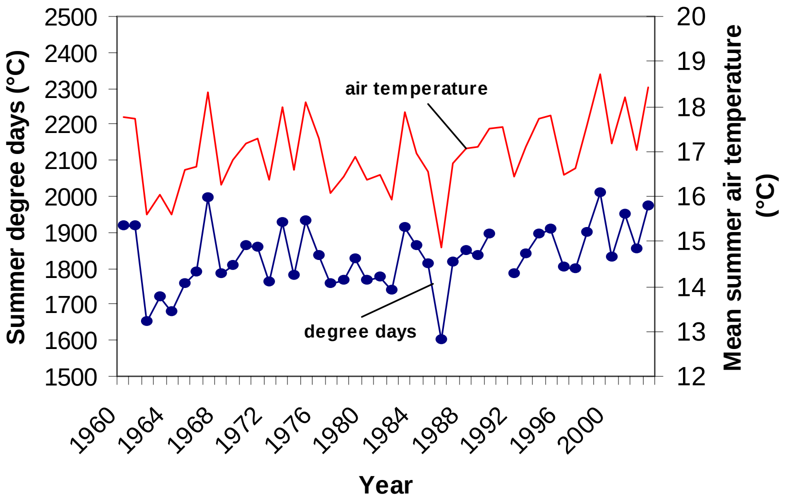
The severity of winter in terms of mean air temperature was significantly correlated with the duration of ice cover ($R^2 = 0.24$, $p<0.0001$; Figure B-6). Although the relationship shows some variability, it was observed to be significant and the regression shows that an increase in mean winter air temperature of 1°C will reduce the duration of ice cover by 7 days.



Appendix Figure B-6. Duration of ice cover related to the mean winter air temperature (°C) within the Miramichi River from 1961 to 2003.

Summer air temperature

Environmental conditions during the summer can also be important for the growth potential of striped bass. As such, we have investigated the mean summer air temperature from June to September as well as the total degree days during that period (Figure B-7). The mean summer temperature was calculated at 17.0°C; however, summer air temperature can vary between 14.9°C (1986) to a high value of 18.7 (1999). The years 2001 and 2003 experienced high summer temperatures at 18.2°C and 18.4°C respectively. When investigating trends within this time series a weak trend was detected for the whole period ($p < 0.05$) with an increase in temperature of approximately 0.19°C / decade; however recent years have shown higher increases in air temperature. For instance, the last 15 years (1989-2003) have shown a much more marked trend of 0.57°C / decade, although not significant ($p = 0.16$) due the variability.

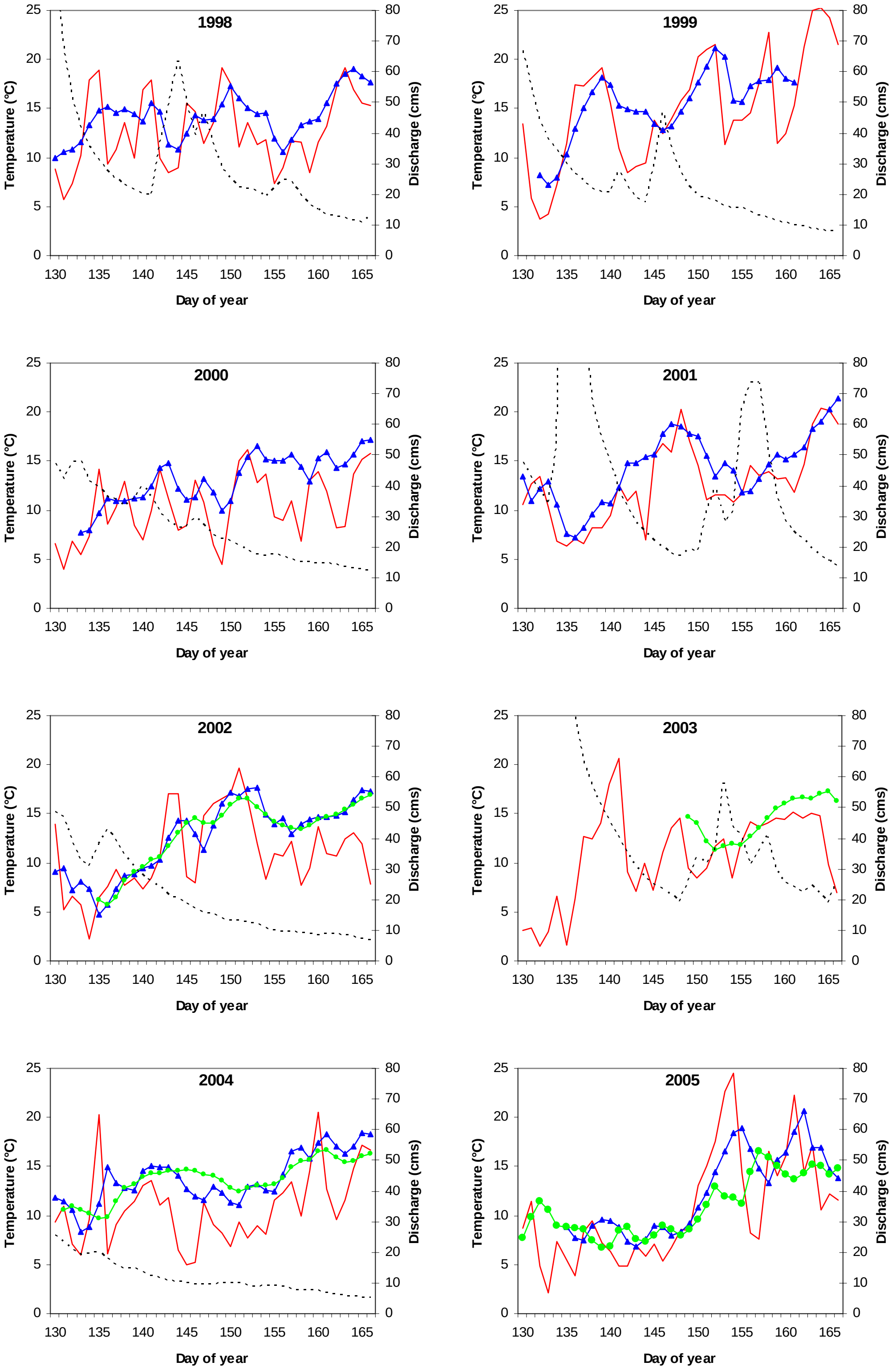


Appendix Figure B-7. Mean summer air temperature (June to September) and total degree days for the same period for the Miramichi River (Miramichi Airport data; 1960 to 2003).

Spring spawning conditions

The next analysis was conducted to provide a description of hydrological conditions during the striped bass spawning and hatching conditions (i.e., early spring conditions) as these activities could be impacted by river discharge, air and water temperature, and other abiotic factors. Figure B-8 shows the results of this analysis with parameters such as river discharge (Northwest Miramichi River), air temperature at the Miramichi Airport as well as water temperature data from the Cassilis Trap located on the Northwest Miramichi River.

Results show that discharge from the Northwest Miramichi generally decline from May 10 (day 130) to June 15 (day 166) whereas a few years have shown significant peak flows in May. In fact, a peak flow of 189 m^3/s was observed on May 15, 2001 (not shown on Figure B-8 because the discharge axis is limited to 80 m^3/s). Peak flows exceeding 80 m^3/s were also observed in mid-May 2003 and extending from May 10 (day 130) to May 15 (day 135) with a corresponding peak discharge of 97 m^3/s during that period. Discharge data were not shown in 2004 and 2005 because the data were not yet available. Other years showed relatively lower flows during the spring and this was observed in 2000 and 2002 where flows were below 20 m^3/s in late May and early June. The most complete water temperature time series was available from the Cassilis Trap and data were available from 1998 to 2005, with the exception of 2003. During the mid-May to early June period the water temperature is somewhat linked to air temperature; however water temperature shows much less variability with a significant lag effect. Also, when comparing Cassilis Trap water temperature data to those collected within the spawning ground of the Northwest Miramichi River, it can be observed that the Northwest Miramichi data are less variable than at Cassilis. Nonetheless, the two time series are showing relatively similar results.



Appendix Figure B-8. Spring environmental conditions on the Northwest Miramichi River including discharge (dashed line), water temperature (circles), Cassilis water temperature (triangles), and Miramichi Airport air temperature (solid line).

Appendix C. COSEWIC's quantitative criteria A to D for the designations of "Endangered" and "Threatened". Features which resulted in the proposed "Threatened" designation for southern Gulf striped bass are boxed and bolded.

	Endangered	Threatened
A. Declining Total Population Reduction in population size based on any of the following 4 options and specifying a-e as appropriate:		
	≥ 70 %	≥ 50 %
(1) population size reduction that is observed, estimated, inferred, or suspected in the past 10 years or 3 generations, whichever is longer, where the causes of the reduction are clearly reversible AND understood AND ceased, based on (and specifying) one or more of a-e below.		
	≥ 50 %	≥ 30 %
(2) population size reduction that is observed, estimated, inferred or suspected over the last 10 years or 3 generations, whichever is longer, where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible, based on (and specifying) one or more of of a-e below. (3) population size reduction that is projected or suspected to be met within in the next 10 years or 3 generations, whichever is longer (up to a maximum of 100 years), based on (and specifying) one or more of b-e below. (4) population size reduction that is observed, estimated, inferred, projected or suspected over any 10 year or 3 generation period, whichever is longer (up to a maximum of 100 years), where the time period includes both the past and the future, AND where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible, based on (and specifying) one or more of a-e below.		
a. direct observation b. an index of abundance appropriate for the taxon c. a decline in area of occupancy, extent of occurrence and/or quality of habitat d. actual or potential levels of exploitation e. the effects of introduced taxa, hybridisation, pathogens, pollutants, competitors or parasites		
B. Small Distribution, and Decline or Fluctuation		
1. Extent of occurrence Or	< 5,000 km^2	< 20,000 km^2
2. Area of occupancy	< 500 km^2	< 2,000 km^2
For either of the above, specify at least two of a-c:		
(a) either severely fragmented or known to exist at # locations	≤ 5	**≤ 10**
(b) continuing decline observed, inferred or projected in one or more of the following: i. extent of occurrence ii. area of occupancy iii. area, extent and/or quality of habitat iv. number of locations or populations v. number of mature individuals		
(c) extreme fluctuations in one or more of the following:	> 1 order of magnitude	**> 1 order of magnitude**
i. extent of occurrence ii. area of occupancy iii. number of locations or populations **iv. number of mature individuals**		

Appendix C continued. COSEWIC's quantitative criteria for the designations of "Endangered" and "Threatened". Features which resulted in the proposed "Threatened" designation for southern Gulf striped bass are boxed and bolded.

	Endangered	Threatened
C. Small Total Population Size and Decline Number of mature individuals and 1 of the following 2:	< 2,500	< 10,000
(1) an estimated continuing decline rate of at least:	20% in 5 years or 2 generations (up to a maximum of 100 years in the future)	10% in 10 years or 3 generations (up to a maximum of 100 years in the future)
(2) continuing decline, observed, projected, or inferred, in numbers of mature individuals and at least one of the following (a-b):		
(a) fragmentation-- population structure in the form of one of the following:	(i) no population estimated to contain >250 mature individuals	(i) no population estimated to contain >1,000 mature individuals
	(ii) at least 95 % of mature individuals in one population	(ii) all mature individuals are in one population
(b) extreme fluctuations in the number of mature individuals		
D. Very Small Population or Restricted Distribution (1) Number of mature individuals	< 250	< 1,000
Or		
(2) Applies only to threatened: Population with a very restricted area of occupancy (area of occupancy typically < 20 km^2) or number of locations (typically 5 or fewer) such that it is prone to the effects of human activities or stochastic events within a very short time period in an uncertain future, and thus is capable of becoming highly endangered or even extinct in a very short time period.		

Appendix D. Southern Gulf of St. Lawrence striped bass life history model.

The recovery objectives, potential, and time frame for recovery were examined using a discrete life history model (Table D-1). Mortality, fecundity, and stock and recruitment dynamics were modeled using general life history information of the species and observed or assumed values specific to the southern Gulf striped bass. The choice of parameter values in the model were governed by observations on characteristics of the population and balancing of life stage abundances. The characteristics of the southern Gulf population of particular interest included:

- relative age structure of the spawners
- sex ratio of spawners

Specific assumptions and functional relationships (Table D-1) are described below.

Egg production

A general fecundity relationship as summarized in Goodyear (1985) was used. Average weight at age data were taken from observations of bass from the southern Gulf (Fig. D-1). Total egg production was the product of the average weight at age, fecundity at weight relationship and number of female spawners at age.

There is no southern Gulf specific fecundity to weight relationship. Data presented in Paramore (1998) indicated that fecundity of Shubenacadie bass varied from 41,000 to 2.1 million eggs for bass ranging in length from 44.9 to 91.0 cm fork length. Goodyear (1985) presented fecundity at weight data for striped bass which translates to about 83,000 eggs per kg (Fig. D-2). Based on the observed mean length at age of bass from the Miramichi and the weight to length relationship, fecundity of an age 4 female bass averaged 83,000 eggs whereas fecundity of age 10 years and older (average weight 6 kg) averaged 600,000 eggs.

Egg to YOY functional relationship

The combination of high fecundity and iteroparity of striped bass are indicative of a species for which mortality in the early stages is high. Year-class variability in striped bass has been observed to be high and largely determined during the egg and larval stages and influenced by environmental factors (see references within Richards and Rago 1999). Increased juvenile production is not guaranteed by increased spawning stock but the chances of producing a strong year class are improved at high spawner abundances. We assumed that there is a density dependent compensatory function between eggs spawned and production of young-of-the-year (YOY) in the first summer (Goodyear 1985). We modeled this dynamic as a Beverton-Holt function (Hilborn and Walters 1992; Myers et al. 1995) and set the parameters based on survival values in the early stages reported in the literature and on reasonable abundance levels of young-of-the-year bass in the fall of the year.

Instantaneous daily rates of mortality ($M\ d^{-1}$) between the egg and the 8 mm larval stage have been estimated to vary between 0.11 and 0.34 with survival after 20 days varying between 0.03% and 11% (Rutherford et al. 1997). We chose a conservative rate of 0.1% for this population at the northern limit of the species distribution. Although there are no measures of absolute abundance of YOY in the fall, the abundance is assumed to be in the order of a few million fish on average with several million individuals possible for strong year classes. Estimates of bycatch in the fall open water smelt fishery of the Miramichi have been over half a million fish in an exceptional year (Bradford et al. 1997b).

Overwinter mortality of YOY

We assumed an instantaneous M of 1.5 (survival = 0.22) for YOY in the first winter (6 months). Overwinter mortality is expected to be high for this northern population. Like adults, juveniles do not feed in the winter and no food items have been found in stomachs of juvenile bass sampled from the open water smelt fishery in November at low water temperatures (pers. obs.). The period of fasting likely extends from late October to late April in most years (see Natural Mortality section above). There is empirical evidence that small bodied striped bass have a lower fitness than large bodied juveniles during the first winter. Some juvenile bass have been found frozen in surface ice in the Miramichi. Variations in quantity of optimal habitat in the winter has been suggested as a possible factor contributing to variations in recruitment of the Hudson River striped bass population (Hurst and Conover 1998).

Survival of age 1 and older bass

We assumed similar proportional survival for male and female bass, although the sex ratio at age data suggest that there may be differential survivals for males and females (age 8 and older fish are predominantly female). Mortality rate of age 1 bass is not known but assumed to be less than that of YOY but higher than age 2 and older bass. Instantaneous mortality rates for age 2 and older were assumed to be 0.6, an average value below the Z values estimated for spawners in the southern Gulf for 1994 to 2005 which are still subjected to some losses from fishing (see Phase I). The high mortality rate for the southern Gulf is consistent with the relative rarity of striped bass older than 10 years of age in the southern Gulf. For the eastern U.S. stocks, M is usually assumed to be 0.15 to 0.2 but these stocks have many fish older than 10 years and they do not undergo the same fasting and overwintering conditions of the southern Gulf fish (Richards and Rago 1999).

Maturity schedules

We assumed different maturity schedules for male and female bass, with male bass maturing at younger ages than female bass (Fig. D-1). To account for the observed sex ratios on the spawning grounds (biased towards males), we modeled female recruitment to the spawning grounds to a maximum of 75% of mature fish for age 5 and older.

Stochasticity

The annual variability in the abundance at age was incorporated as variation around the mean survival of the form $e^{\left(R\sigma-0.5\sigma^2\right)}$, where R is a random normal deviate and σ is the standard deviation of the natural log transformed deviations of year class survival. For the egg to YOY survival, we borrowed the standard deviation of Goodyear (1985) representing the variation in year class strength of the Maryland stock (σ = 0.72). For overwinter survival of YOY, we assumed σ = 0.2 and for all other age groups, σ = 0.1. The resultant survivals are log normal and had a range of 0.09 to 6.69 of the mean for egg to YOY, 0.5 to 1.8 of the mean for age 1, and 0.7 to 1.3 of the mean for age 2 and older fish. Variation in mortality was assumed to be similar for bass age one and older but the value varied annually. Variability in fork length at age was modeled assuming a triangular distribution bounded by the minimum and maximum observed length at age with the peak at the average length (Fig. D-1). The draws were independent across age but similar for all years in each run.

Initial values for the simulations

The model was initiated using the abundance of male and female spawners at age three and older for the years 1994 to 1996. The point estimates of the mode of the spawner estimates at age for the years 1994 to 2005 were retained. The number of eggs produced, YOY in the summer, abundance of spawners and total abundance of age 3 and older bass were simulated for the years 2005 to 2149.

Simulations were run using CrystalBall@, an add-in for Excel.

Appendix Table D-1. Life history model and functions.

$$Eggs_j = \sum_{i=3}^{20} 83177 * Weight_i * Spawners_{i,f,j}$$

where Eggs$_j$ = total eggs spawned in year j
Weight$_i$= length(cm)$_i^{2.993}$ * 0.0000137 for age i
Length$_i$ = length at age i, triangular distribution, range minimum and maximum, peak at mean length
Spawners$_{i,f,j}$ = abundance of female spawners of age i in May year j

$$YOYFall_j = \frac{\alpha * Eggs_j * \gamma}{1 + \frac{\alpha * Eggs_j * \gamma}{YOY_{cap}}} * e^{\left(R_j \sigma - 0.5 * \sigma^2\right)}$$

where YOYFall$_j$ = abundance of young-of-the-year bass in the fall in year j
α = mean density independent mortality, 0.1%
YOY$_{cap}$ = mean asymptotic abundance of YOY in the fall
γ = 1 for unexceptional event with probability A, uniform (0,1), 0.5 otherwise
R$_j$ = normal random deviate for year j
σ = inter year class variability

$$N_{1j+1} = \exp^{-(M+F_0)} YOYFall_j * e^{\left(R_j \sigma - 0.5\sigma^2\right)}$$

where N$_{1j+1}$ = abundance of one year old in May, year j+1
M = overwinter mortality, (M = 1.5)
F$_0$ = instantaneous fishing rate on YOY in fall and winter fisheries

$$N_{2,s,j+2} = \exp^{-(M+F_1)} N_{1,j+1} * 0.5 * e^{\left(R_j \sigma - 0.5\sigma^2\right)}$$

where N$_{2,s,j+2}$ = abundance of two year old bass at s (male, female) in May, year j+2
M = instantaneous mortality in year k (j+1 to j+2) (M = 1)
F$_1$ = instantaneous fishing rate on one year old bass during j+1 to j+2

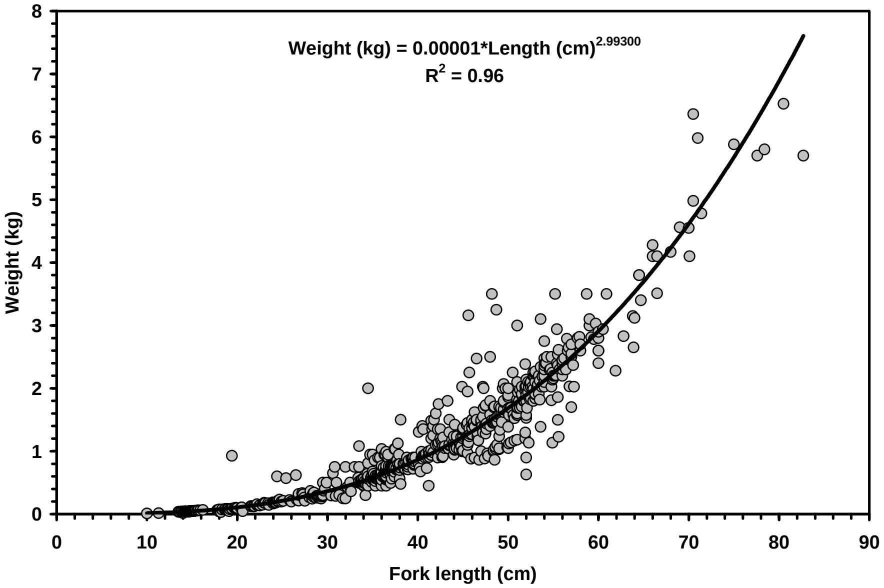
$$N_{i,s,j+i} = \exp^{-(M+F_{si})} N_{i-1,s,j+i-1} * e^{\left(R_j \sigma - 0.5\sigma^2\right)}$$

where N$_{i,s,j+i}$ = abundance of bass age i (i = 3 to 20) and sex s in May, year j+i
s = male, female
M = instantaneous mortality of sex s in year k (j+i-1 to j+i), (M = 0.5)
F$_i$ = instantaneous fishing rate on bass age i during j+i-1 to j+i

$$Spawner_{i,s,j} = N_{i,s,j} * Mat_{i,s}$$

where Spawner$_{i,s,j}$ = abundance on the spawning ground of age i, sex s, in May, year j
Mat$_{i,s}$ = proportion of bass age i and sex s on the spawning grounds

Appendix Figure D-1. Life history characteristics used in the striped bass in the life history model.



Fork length of striped bass from the Miramichi (1994 to 2005)

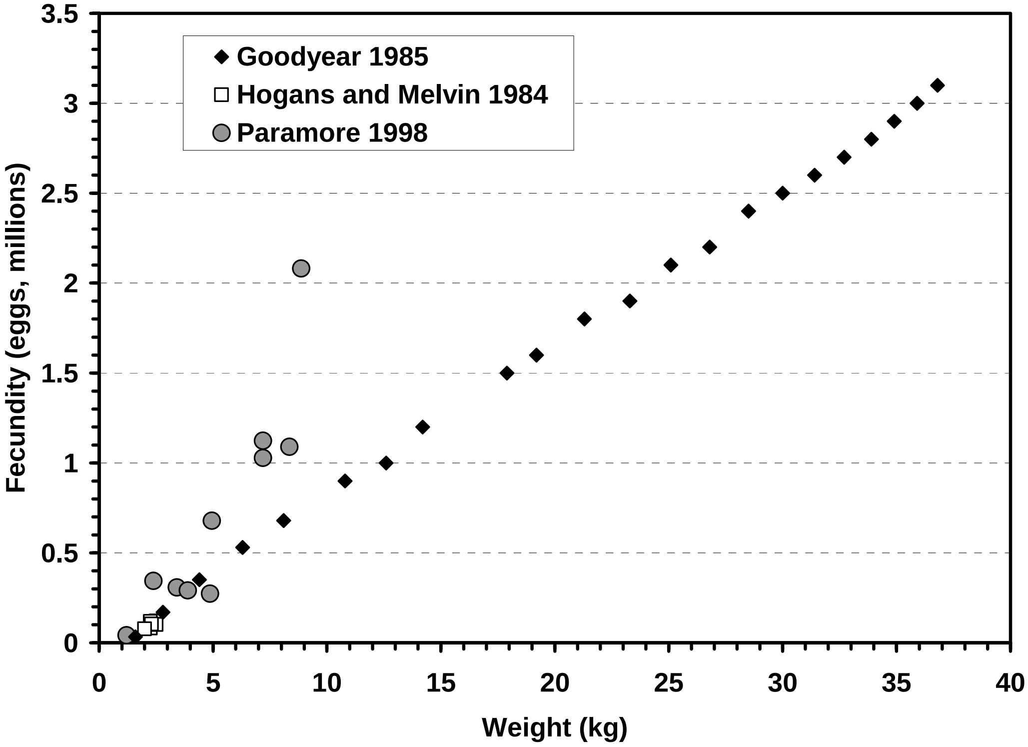
Age (years)	3	4	5	6	7	8	9	10	11	12	13
Mean	40.3	46.7	52.8	58.5	63.7	68.0	74.2	76.4	78.8	85.5	80.9
Min	27.0	29.0	35.7	44.8	52.3	51.5	63.2	72.4	70.2	81.7	80.9
Max	51.2	57.5	65.8	72.6	72.4	82.0	80.5	82.9	86.1	88.5	80.9
N	2812	2338	1073	300	76	37	18	4	6	4	1

In the population model, the mean, minimum and maximum lengths for ages 12 and older were assumed to be the average (81.4), minimum (70.2) and maximum (88.5) lengths observed for age 11 to 13 years.

Maturity schedule (proportion of age group on the spawning grounds)

Age (years)	3	4	5	6 and older
Male	0.5	0.95	1	1
Female	0.1	0.5	0.75	0.75

Appendix Figure D-2. Fecundity (eggs per female) versus weight (kg) of striped bass as reported by Goodyear (1985), Hogans and Melvin (1984) and Paramore (1998).



Fecundity (eggs per female fish) = weight (kg) * 83177 eggs per kg

Appendix E. Potential fishing effort and gear for gaspereau in eastern NB and PEI (upper panel) and Gulf NS (lower panel) (DFO 2000a; DFO 2001a; DFO 2002a).

Location	Fishers	Trapnets	Gillnets (fathoms)
Eastern New Brunswick			
Caraquet Bay	2	4	
Waterways and bays of Lamèque and Miscou Islands	1		150
Saint-Simon Bay	1	4	
Pokemouche River above the railway bridge at Inkerman	6	60	
Big Tracadie River	9	40	
Little Trcadie River	1	1	
Tabusintac River	1	2	
Napan River	4	5	
Miramichi River	13	17	
Northwest Miramichi River	6	12	
Black River	8	14	
Eel River	9	13	
Bay du Vin River	9	11	
French River	1	1	
Portage River	1	1	
Miramichi Bay	4	4	
Richibucto River	20	50	
Richibucto River	1		450
St. Charles (Aldouane) River	3	8	
Kouchibouguac River within the National Park	8	11	
Kouchibouguac River outside the National Park	5	15	
Buctouche River	5	15	
Little Buctouche River	5	10	
Cocagne River and Bay	1	2	
Shediac River	8	20	
Aboujagane River	2	5	
Scoudouc River	2	7	
No licence condition	2	2	
Eastern NB Total	**138**	**334**	**600**

Prince Edward Island	
*anywhere in the province - commercial license	23
**anywhere in the province - bait license	923
PEI Total	946

* gear includes: dipnets, gillnets, and trapnets, or any combination of the three

** gear includes: dipnets, gillnets, and trapnets

Gulf Nova Scotia Statistical District (Fishing area)	Commercial				Bait
	Inland	Coastal / estuary		*Mackerel	Coastal
	Weirs	Trapnets	Gillnets	Trapnets	Gillnets
01 Bay St. Lawrence					42
02 Pleasant Bay - Broad Cove Marsh	50				40
03 Inverness – Creignish					19
13 Aulds Cove – Arisaig	6		20	12	54
12 Lismore - Pictou Landing	4	4		1	18
11 Pictou Harbour - Logans Point	2	3	3		20
10 Barrachois		1			4
46 Malagash – Wallace		2	6		8
45 Pugwash – Linden		5	1		11
Total	**62**	**15**	**30**	**13**	**216**

*Only includes those commercial coastal mackerel trap nets that have allowances for gaspereau retention in-season

Appendix F. Potential fishing effort and gear for rainbow smelt in eastern NB and PEI (upper panel) and Gulf NS (lower panel) (DFO 2000b; DFO 2001b; DFO 2002b).

Location	Fishers	Box nets	Bag nets	Gillnets (fathoms)
Restigouche River above the Van Horne Bridge at Campbellton	18	69	5	
Chaleur Bay	17	49		750
Chaleur Bay, east of the ferry wharf at Dalhousie	1			75
Bathurst Harbour	7	41		225
Caraquet Bay	47	198		405
Saint-Simon Bay	65	265		555
Pokesudie Island	47	198		405
Pokemouche River between the Inkerman bridge and Route 113 bridge	3	14		300
Pokemouche River above the Landry office bridge	12	34		150
Shippagan Bay	18	120		300
Petite Lamèque Bay	17	118		300
Lamèque Bay	17	118		300
Miscou Bay	5	26		
Miscou Harbour	12	85		
Gloucester County	112	3		25,955
Tracadie Bay	12	123	1	
Big Tracadie River	24	163		150
Little Tracadie River	21	19	1	150
Tabusintac Bay, Tabusintac River	33	125	4	420
Neguac Bay	38	317	2	375
Miramichi Bay	130	1,321		315
Miramichi River	25	164		150
Napan River	1	31		
Bay du Vin River	2	17		
Black River	5	25		
Kouchibouguacis River, in Kouchibouguac National Park	12	47		1,095
Kouchibouguac Bay, in Kouchibouguac National Park	5	32		
Kouchibouguac River, in Kouchibouguac National Park	11	53	1	
Kouchibouguac River, outside of Kouchibouguac National Park	8	20	2	15
St. Louis Bay	2	15		
Richibouctou River	44	227	2	3,415
Richibouctou Harbour	2	12		330
Baie du Village, Richibouctou	4	6		500
Bouctouche River	5	10	2	
Bouctouche Bay	19	130		675
Cocagne River	13	26	1	1,747
Cocagne Bay	11	47		2,060
St. Charles River (Aldouane)	4	12		450
Shediac Bay	14	73		1,575
Shediac River	3	19		675
Aboujagane River	4	27		680
Shemogue	24	76	34	
Northumberland Strait, adjacent to Kent County	51	170		2,515
Northumberland Strait, adjacent to Westmorland County	29	93	9	1,125
Murray Corner Wharf	1	1		
Cape Spear	1	2		
Gaspereau River	3	2	5	
Baie Verte	1	2		
No fishing area indicated	2	7		
Eastern NB Total	**962**	**4,752**	**69**	**48,137**
Prince Edward Island				
*anywhere in the province - commercial license	359			
**anywhere in the province - recreational license	100			
***anywhere in the province - Lennox Island First Nation license	1			
PEI Total	**460**			

* gear includes: gillnets, trapnets, and bag nets or any combination of the three

** gear includes: gillnet only

*** gear includes: 10 gillnets and 2 trapnets

Appendix F continued. Potential fishing effort and gear for rainbow smelt in eastern NB and PEI (upper panel) and Gulf NS (lower panel) (DFO 2000b; DFO 2001b; DFO 2002b).

Gulf NS Statistical District	**Recreational**	**Commercial**		
(Fishing area)	**Gillnets**	**Gillnets**	**Bag Nets**	**Box / Trap Nets**
01 Bay St. Lawrence				
02 Pleasant Bay - Broad Cove Marsh	3	22		
03 Inverness - Creignish	4	8		
14 Aulds Cove		3		
13 Aulds Cove - Arisaig	23	33		6
12 Lismore - Pictou Landing	12	14	1	3
11 Pictou Harbour - Logans Point	9	50	6	11
10 Barrachois				4
46 Malagash – Wallace	4	5	2	9
45 Pugwash – Linden		5	15	13
Total	**55**	**136**	**24**	**46**

Note: Some recreational gillnet licenses overlap more than one statistical area and some are valid for all tidal waters of Nova Scotia that border on the Northumberland Strait, however they have been grouped according to the statistical districts that they are mostly fished in.

Appendix G. Potential fishing effort and gear for American eel in eastern NB and PEI (upper panel) and Gulf NS (lower panel) (DFO 2000c; DFO 2001c; DFO 2002c).

Location	Licenses	Trapnets	Fyke nets	Hooks	Weirs	Box nets
Eel River	1	2				
Pokesudie Island	1	1				
Waters off Pokesudie Island	1		2			
Pokesudie Island, east coast	1	1				
Pokesudie Island, small channel	1	2				
Saint-Simon Bay, below the wharf	1	14				
Pokemouche River, above the Landry Office River Bridge	8	5	382			
Pokemouche River, above the railway bridge at Inkerman to the bridge over the Pokemouche River at Landry Office	1		12			
Miscou Island Bay	4		22			
Miscou Island Bay	2	7				
Miscou Island Bay, excluding Miscou Harbour	1	2				
Lamèque Island Bay	1		2			
Shippagan Harbour	1		1			
Lamèque Island Bay, excluding Miscou Harbour	1	18				
Little Tracadie River	2		21			
Big Tracadie River	14	383	20	100		
Little and Big Tracadie River	1	75				
Little Tracadie River, including Tracadie Bay	1	5				
Tabusintac Bay and River	3	126				
Tbusintac River	4	123				
Tabusintac Bay	5	29				
Tabusintac Bay and Portage River	2	20				
Portage River	1	2				
Neguac Bay	26	151				
Neguac and Miramichi bays	2	7	2			
Miramichi Bay	8	50				
Miramichi River	5		68			
Miramichi Bay and River	3	48				
Miramichi Napan, Northwest and Southwest Miramichi rivers	1		80			
Black River and Napan Bay	1		20			
Miramichi Bay and Black River	1	13				
Miramichi Bay and Eel River	1	20				
Bay du Vin River	3	17				
French River	2	11				
Black River	1	21				
Kouchibouguac River, within the park boundaries	4			1,100		
Kouchibouguac River, within the park boundaries	6	38				

Appendix G continued. Potential fishing effort and gear for American eel in eastern NB and PEI (upper panel) and Gulf NS (lower panel) (DFO 2000c; DFO 2001c; DFO 2002c).

Location	Licenses	Trapnets	Fyke nets	Hooks	Weirs	Box nets
Kouchibouguac River and Kouchibouguac Bay, within the park boundaries	2	24				
Northumberland Strait, off Kent County	2	3				
Northumberland Strait, off Kent County	1				4	
Kouchibouguacis River, Kent County, inside and outside the park boundaries, to bridge on route 134	1			400		
Kouchibouguacis River, Kent County, inside and outside the park boundaries	1	2		300		
Kouchibouguacis River outside the park boundaries	1	2				
Kouchibouguac River	1	1				
Richibuctou Bay and River	1			500		
Richibuctou River	9	79				
Richibuctou River	1		7			
Richibuctou River	2			1,100		
Richibuctou Harbour, outside the park boundaries	1			850		
Richibuctou River outside the park boundaries	4			2,875		
Richibuctou and St. Charles rivers, outside the park boundaries	1			700		
Buctouche Bay	2		18			
Buctouche Bay	1	1				
Buctouche Bay	1				2	
Buctouche River	2	9				
Buctouche River, above the bridge on Route 11	1	20				
Buctouche and Cocagne Bays	1				3	
Buctouche and Cocagne Bays	1				7	
Northumberland Strait, Kent County, including Buctouche Bay	1				4	
Cocagne River	1				4	
Cocagne River	1	24				
Cocagne River, above the Route 11 bridge	2	6				
Cocagne Bay	3				11	
Cocagne Bay	1		5			
Cocagne Bay	3	7				
Cocagne Bay	1					2
Saint-Charles River (Aldouane River)	1	2				
Shediac Bay and River	1	31	4			
Shediac Bay	4	22				
Aboujagane River	1	4				
Shemogue Harbour	2	2				
Little Shemogue Harbour	1	2				
Northumberland Strait, along Westmorland County	2	3				
No conditions	1	8				
No conditions	1		4			
Eastern NB Total	**186**	**1,443**	**670**	**7,925**	**35**	**2**
Prince Edward Island						
anywhere in the province - commercial spear license	508					
anywhere in the province - commercial trapnet/fyke net license	127					
anywhere in the province - commercial spear and trapnet/fyke net license	120					
anywhere in the province - Lennox Island First Nation communal commercial s	50					
anywhere in the province - Abegweit First Nation communal commercial spear	23					
anywhere in the province - Native Council of PEI communal commercial spear	20					
*anywhere in the province - Lennox Island First Nation communal 25,000 lbs.						
PEI Total	**848**					

* gear includes: spears and 10 trapnets/fyke nets

Appendix G continued. Potential fishing effort and gear for American eel in eastern NB and PEI (upper panel) and Gulf NS (lower panel) (DFO 2000c; DFO 2001c; DFO 2002c).

Gulf NS Statistical district (Fishing area)	**Commercial**			**Recreational**	
	Trapnets	**Pots**	**Spears**	**Pots**	**Fykes**
01 Bay St. Lawrence		1			
02 Pleasant Bay – Broad Cove Marsh	25	5			
03 Inverness – Creignish	3	3			
14 Aulds Cove		3		1	
13 Aulds Cove – Arisaig	24	8	24	4	
12 Lismore - Pictou Landing	6	3	9		
11 Pictou Harbour – Logans Point	3	20	4	1	
10 Barrachois		3	3		
46 Malagash – Wallace	1	8		1	
45 Pugwash – Linden	6	12			
Total	**68**	**66**	**40**	**6**	**0**

Note: Some fishers have more than one fishing gear on their eel licence.
The total number of commercial eel fishers in the GNS area is 116.

Fisheries and Oceans Canada | Pêches et Océans Canada
Ecosystems and Oceans Science | Sciences des écosystèmes et des océans

Canadian Science Advisory Secretariat
Gulf Region | Science Advisory Report 2021/018

REFERENCE POINTS FOR STRIPED BASS (*MORONE SAXATILIS*) FOR THE SOUTHERN GULF OF ST. LAWRENCE POPULATION

*Striped Bass (*Morone saxatilis*)*
Image courtesy of New York State Department of Environmental Conservation

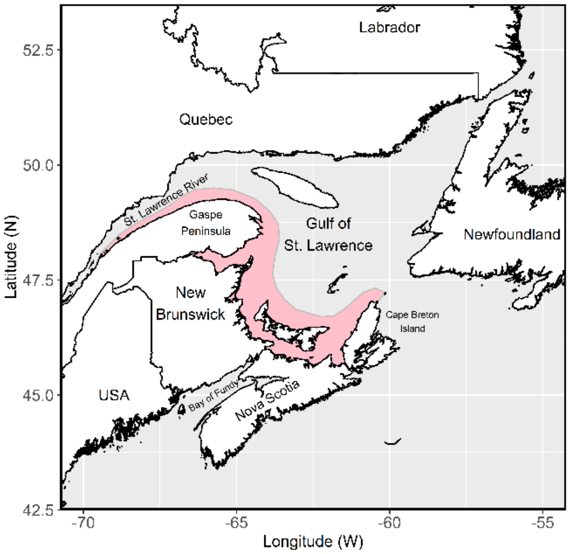


Figure 1. Distribution (coloured polygon) of the southern Gulf of St. Lawrence population of Striped Bass in eastern Canada.

Context:

*The Striped Bass (*Morone saxatilis*) population of the southern Gulf of St. Lawrence had declined to less than 5,000 spawners in the late 1990s. Due to conservation concerns, the commercial fishery closed in 1996, followed by the closure of the recreational fisheries and the suspension of Indigenous Food, Social, and Ceremonial (FSC) fisheries allocations for Striped Bass in 2000. The Striped Bass population of the southern Gulf, located at the northern distribution of the species range in eastern North America, is widely distributed in estuaries and coastal waters from the northern tip of Cape Breton Island, Nova Scotia to the north shore of the Gaspe Peninsula, Quebec. DFO Gulf Region Fisheries and Aquaculture Management requested advice on reference points that conform to the Precautionary Approach (PA) to guide further management decisions for the developing Striped Bass fisheries. DFO management also requested a review of the approaches and potential reference points for Striped Bass that take account of interactions of Striped Bass and other species of fisheries value. For purposes of assessment and development of fisheries reference points, the southern Gulf Striped Bass population consists of the Gulf of St. Lawrence region extending to the north shore of the Gaspe Peninsula in Quebec (Figure 1).*

A science peer review meeting was conducted November 23-25, 2020 (virtually) in Moncton, New Brunswick. Participants (23 in total) at the science review were from DFO Science, DFO Fisheries and Aquaculture Management, the province of Quebec (Ministère des Forêts, de la Faune et des Parcs), the

June 2021 (Erratum: December 2022)

State of Maine Department of Marine Resources, an Indigenous organization from PEI, and the COSEWIC co-chairs and report authors from academia.

SUMMARY

- Striped Bass (*Morone saxatilis*) of the southern Gulf of St. Lawrence (from western Cape Breton Island NS to the north shore of the Gaspe Peninsula QC) is considered to be a single biological unit for the purpose of this assessment and the derivation of reference points.
- This Striped Bass population has been annually assessed since 1994 and its life-history characteristics, including size-at-age, length-weight relationship, and sex ratio of spawners are well defined.
- An age structured population model with an underlying Beverton-Holt stock and recruitment relationship (from eggs to age-0) was used to model the population dynamics of Striped Bass. The model uses a combination of estimated and assumed life history parameters, most of which are specific to this population. Seven models were evaluated; the two preferred models are presented in this report.
- Monitoring and stock assessment data of the spawners in the Northwest Miramichi River, the annually predictable and confirmed spawning location that produces recruitment, are considered representative of Striped Bass in the southern Gulf and appropriate for deriving reference points.
- The near monotonic increasing trajectory of the population abundance from its low point in the late 1990s at less than 5,000 spawners to the highest abundance in the late 2010s of over 300 thousand spawners provides limited information to unequivocally define the maximum potential population size. A conclusion from the population modelling with the available data is that the maximum abundance of this population has not yet been realized.
- Reference values are defined in terms of eggs and converted to equivalent numbers of spawners to aid in interpretation.
- The number of eggs that equate to 80% Bmsy (80% of the spawning stock biomass that produces maximum sustainable yield) is proposed as the Upper Stock Reference (USR). There is no model consensus for the USR value. An Upper Stock Reference value of 54.3 billion eggs, equivalent to 720 thousand spawners, is the lowest value of two models retained (versus 1,2[1] million spawners for the other model).
- The number of eggs that result in half of Beverton-Holt carrying capacity is proposed as the Limit Reference Point (LRP). There is no model consensus for the LRP value; 17.3 billion or 30.0 billion eggs depending on the model, equivalent to 330 to 560 thousand spawners. Based on the trajectory of this population over the relatively short period of assessment, maintaining spawners above 330 thousand fish should be sufficient to avoid serious harm.
- The status is presented in terms of estimated eggs from spawners and perspectives on status are model dependent. The highest estimated spawner abundance of approximately one million fish in 2017 was approximately at the USR or in the cautious zone depending on

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[1] Erratum: December 2022, corrected the value to correspond to the table in the body of the report

the model. Otherwise, the status was either below the LRP in all years except 2017, or below the LRP until 2015 and in the cautious zone since 2016.

- Fisheries management actions were responsive to the decline and rebuilding of the Striped Bass population, beginning with the closure of all directed fisheries for Striped Bass in 2000, followed twelve years later with the re-opening of the Indigenous FSC fisheries in 2012, and the retention recreational fisheries in 2013. This increased fisheries access occurred as the stock abundance was on an increasing trajectory of abundance, moving into the cautious zone.
- Prey of Striped Bass in May and June in the Miramichi River include Rainbow Smelt, gaspereau, occasionally Atlantic Salmon smolts, as well as several other fish and invertebrate species. Alternate reference levels to address the multiple species concerns related primarily to predation by Striped Bass on these prey species cannot be determined at this time based on the available information.
- The most important assessment and management uncertainty is the lack of comprehensive recreational fisheries catch and harvest data. In absence of such data, assessments of the dynamics and robustness of this population to fishing and environmental variation cannot be provided.

INTRODUCTION

Striped Bass (*Morone saxatilis* Walbaum, 1792) is widely distributed throughout the estuaries and coastal waters of the southern Gulf of St. Lawrence (southern Gulf), from the northern tip of Cape Breton Island (NS) in the east to the north shore of the Gaspe Peninsula (Quebec) in the west. The population in the southern Gulf of St. Lawrence is considered to be the most northern spawning population of the species distribution (Figure 1).

Genetic analyses and conventional tagging studies have indicated that this population is geographically isolated within the southern Gulf and estuary of the St. Lawrence. Striped Bass juveniles (age-0) originating from the Miramichi River were used in a re-introduction program in the St. Lawrence River beginning in the late 1990s. Successful spawning and recruitment from this program has been confirmed (DFO 2017; L'Italien et al. 2020). Tracking studies of acoustically-tagged Striped Bass from the St. Lawrence River group and from the southern Gulf of St. Lawrence group, as well as differences in elemental composition of the otoliths of bass spawned in the Miramichi River and in the St. Lawrence River, show a general geographic isolation of the two groups. The St. Lawrence progeny are generally restricted to the St. Lawrence River itself (at least to date), whereas the fish originating from the Miramichi have a broader distribution that extends into the estuary of the St. Lawrence and to the lower north shore of the St. Lawrence River (Valiquette et al. 2017, 2018). An extraordinary expansion of Striped Bass into previously undocumented areas along the lower north shore of the St. Lawrence River and into southern Labrador occurred in 2017 (DFO 2018; Valiquette et al. 2018) and the potential range of the southern Gulf Striped Bass population is now considered to occasionally extend into those northern areas.

The Striped Bass population of the southern Gulf of St. Lawrence had declined to less than 5,000 spawners in the late 1990s. Due to conservation concerns, the commercial fishery was closed in 1996, followed by the closure of the recreational fishery and the suspension of Indigenous Food, Social, and Ceremonial (FSC) fishery allocations for Striped Bass in 2000. The estimated abundance of Striped Bass spawners subsequently increased to over 200 thousand spawners in 2011, followed by a peak abundance estimated at over 900,000 spawners in 2017 (DFO 2018). Accordingly, Indigenous FSC fisheries were reinstated in 2012 and allocations of Striped Bass to indigenous groups have gradually increased since then. The recreational fishery reopened in 2013 with increasing annual access. A pilot Indigenous commercial fishery was conducted in 2018 to 2020.

In response to the trend of increasing abundance of Striped Bass and with requests for additional fisheries access to southern Gulf Striped Bass, Fisheries and Oceans Canada (DFO) Gulf Ecosystems and Fisheries Management Branch requested the development of reference points that conform to the Precautionary Approach (PA) to guide future management decisions for Striped Bass fisheries. DFO Fisheries Management also requested a review of approaches and potential reference points for Striped Bass that take account of interactions between Striped Bass and other species of fisheries value.

The specific objectives of the science peer review and advice provided in this report include:

- A review of the available information on the abundance and biological characteristics (size-at-age, mortality rate estimates, size structure) of the Striped Bass population of the southern Gulf of St. Lawrence relevant for the definition of reference points;
- A review of candidate fishery reference points for Striped Bass and estimates of these based on the available information from the southern Gulf population;
- A review of the consequences of fishery management measures on the derivation of fishery reference point values;
- Options for incorporating species interaction considerations in the definition of reference points for Striped Bass; and
- Consideration of the uncertainties in the definition of the reference points and management approaches for Striped Bass.

Species Biology and Distribution

General descriptions of Striped Bass biology and life history are available in COSEWIC (2012) and summary information for the southern Gulf population is available in Douglas et al. (2003).

- Striped Bass is a relatively long-lived iteroparous spawner. Maximum age estimated from otoliths along the eastern seaboard of the US is 31 years. In the southern Gulf population, maximum age from scale interpretations is 15 years and maximum fork length in sampling records is 116 cm, although there are incidental reports of catches of larger Striped Bass in this region.
- The Northwest Miramichi River estuary is the only confirmed spawning location that is annually predictable in time and space and that has produced annual recruitment in the southern Gulf of St. Lawrence. In the last few years, opportunistic sampling has confirmed the presence of Striped Bass eggs and larvae in the Southwest Miramichi and the Tabusintac River, both geographically proximate to the Northwest Miramichi however the extent to which the spawning in these areas contributes to recruitment to the southern Gulf population has yet to be determined.
- Spawning occurs in late May to early June in the upper estuary at the upper extent of the salt wedge within tidal waters of the Northwest Miramichi River and the eggs and milt are broadcast simultaneously into the water column. The eggs float freely, are generally neutrally buoyant in slightly saline water, and hatch after a few days depending on water temperature.

- The larvae feed on planktonic organisms and move to the near shore shallow areas of the rivers shortly after the onset of exogenous feeding.
- Young-of-the-year (YOY) Striped Bass gradually migrate downstream to Miramichi Bay in the summer and diffuse in a northwest and easterly direction from the Miramichi (Robinson et al. 2004) with a coastal distribution of young of the year by the first autumn extending at least from Miscou Island (NB) in the north to Pictou (NS) in the east (Douglas and Chaput 2011).
- Post-spawned adults return to marine waters and undertake coastal feeding migrations through the summer and autumn.
- Striped Bass is a generalist feeder with shifts in prey composition occurring with age and size. Larger bass are known piscivores, and consume a wide range of invertebrate and vertebrate prey, including a number of anadromous species of fisheries interest (Rainbow Smelt, gaspereau, Atlantic Salmon smolts).
- The southern Gulf of St. Lawrence population cannot tolerate sub-zero water temperatures and therefore seek water temperatures above zero in the upper areas of estuaries in the southern Gulf to overwinter.
- In its most recent assessment, COSEWIC assessed the status of the population as Special Concern (COSEWIC 2012). The Government of Canada decided not to add the Striped Bass population of the southern Gulf of St. Lawrence to Schedule 1 of the *Species at Risk Act* in March 2013.

Fisheries

Striped Bass has been exploited in numerous fisheries of the southern Gulf of St. Lawrence for over a century of records. Many regulatory changes have occurred in the Striped Bass fisheries that impact these fishing activities and recorded harvests. The most important changes occurred in the mid-1990s in response to concerns about low population abundances.

In 1996, an amendment to the Maritime Provinces Fisheries Regulations eliminated the authorization for the retention of Striped Bass bycatch in commercial fishing gears for gaspereau, Rainbow Smelt, American Shad, and American Eel, effectively closing the commercial fishery. By 2000, all legal Striped Bass fisheries, including recreational fisheries were closed and allocations in Indigenous Peoples FSC fisheries were suspended. Following on the rebuilding of the Striped Bass abundance in the late 2000s, Indigenous FSC fisheries allocations were reinstated in 2012, the recreational fishery was reopened in 2013, and a pilot Indigenous commercial fishery in the Miramichi River was licenced in 2018 to 2020. Striped Bass originating from the southern Gulf is also exploited in the recreational fisheries along the south and north shores of the Gaspe Peninsula, in fishing waters managed by the province of Quebec. Fisheries management measures for the recreational Striped Bass fishery in Quebec that, for the most part, paralleled the fisheries management measures in DFO Gulf Region were introduced in 2013.

In addition to a season, daily bag and possession limits, and gear restrictions, short-term closures to directed recreational fisheries lasting 5 to 9 days in the spawning area of the Northwest Miramichi have also been imposed since 2017 to preclude harm to spawning fish.

Fisheries catches and harvests

Complete fishery catch data for Striped Bass in the southern Gulf of St. Lawrence are lacking. Historically, fisheries statistics included only commercial harvests, exclusive of recreational and

Indigenous peoples fisheries harvests. First records of bass landings in fisheries date to 1868 but these data have not been compiled into a single report. The reported landings of Striped Bass from the southern Gulf of St. Lawrence for the period 1917 to 1988, compiled by LeBlanc and Chaput (1991), show a maximum catch for that time period of 61.4 t reported in 1917. There were no recorded landings for the years 1935 to 1967. Peak recorded landings in the second period of records after 1967 were 47.8 t in 1981 with 15.25 t recorded in the last year (1996) of authorized commercial landings (Douglas et al. 2006).

There are no compiled reports of catches and harvests of Striped Bass in the Indigenous FSC fisheries in the southern Gulf.

Since the re-opening of the recreational fisheries in 2013, partial catch data from the recreational fishery for some geographic areas of the southern Gulf and in some years have been collated but they are incomplete. There is no licence requirement to fish recreationally in tidal and marine waters, hence the number of anglers is unknown. The recreational fishery occurs from pleasure boats and from shore, in estuaries and along the coast, from wharves, public beaches, etc. along a broad geographic area and the potential number of anglers is very large.

ASSESSMENT

Monitoring and stock assessment data of the Striped Bass spawners in the Northwest Miramichi River, the annually predictable and confirmed spawning location that produces recruitment, for the period 1994 to 2019 are considered representative of Striped Bass in the southern Gulf and used in the modelling of population dynamics and in the derivation of reference points. The monitoring programs also provide information on biological characteristics of the southern Gulf Striped Bass population, including length-at-age, weight-at-age, maturity-at-age, and proportion female-at-age on the spawning grounds. The biological characteristics information is used in an age-structured, population model to estimate stock and recruitment parameters and associated age-specific mortality rates at age.

Using the biological characteristics and the parameter estimates from the population dynamics modelling, equilibrium modelling is then used to derive candidate Limit Reference Points (LRP), Upper Stock Reference (USR) levels, and removal rate references that would conform with the PA.

Abundance and Abundance-At-Age

Since 1994, monitoring of the bycatch in the commercial gaspereau trapnets of the Miramichi River has been the principal source of information for the estimation of the Striped Bass spawning population of the southern Gulf of St. Lawrence (DFO 2020). Estimated abundances of spawners in the Northwest Miramichi were at or under 5,000 spawners (median) during 1996 to 2000 (DFO 2020). Abundance increased to between 16,000 and 26,000 during 2001 to 2006 and again to between 50,000 and 100,000 fish during 2007 to 2010. Abundances of 150 thousand to 300 thousand were estimated during 2011 to 2016 with a peak abundance in 2017 at just under one million fish (Figure 2). Striped Bass spawner abundance in 2018 and 2019 was estimated to have fallen back to approximately 300 thousand spawners.

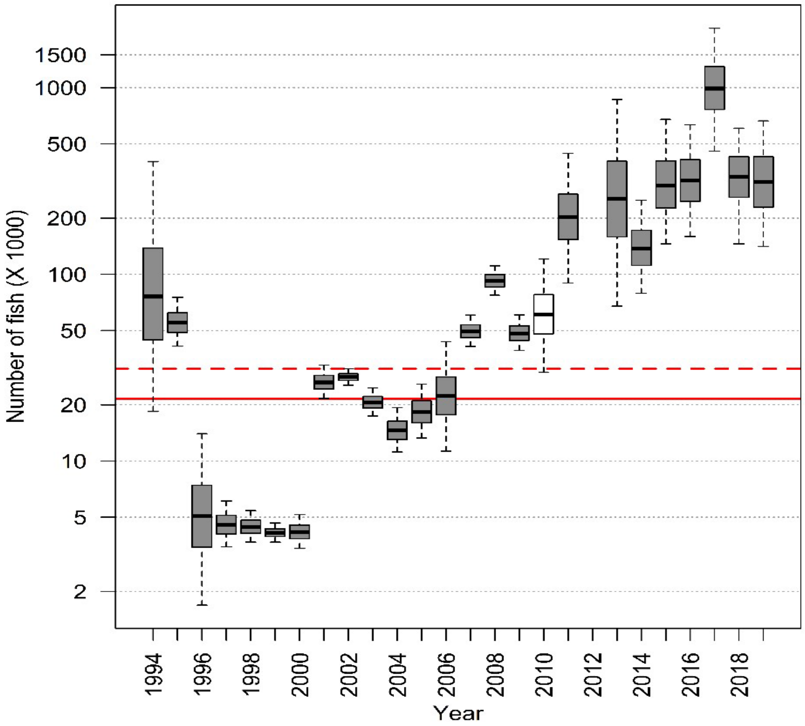

Figure 2. Estimated abundance of adult Striped Bass spawners in the Northwest Miramichi estuary between 1994 and 2019. The estimates are shown on a logarithmic scale for visibility of the full range of abundance values over the time series. The estimate for 2010 (unshaded interquartile box) is considered to be an underestimate due to the earlier timing of the spawning events (Douglas and Chaput 2011). There is no estimate for 2012 because spawning was very early and Striped Bass left the sampling area prior to monitoring activities (DFO 2013). Box plots are interpreted as follows: dash is the median, boxes are the interquartile range, and the vertical dashes are the 5th to 95th percentile ranges. The solid and dashed horizontal lines show the limit and target recovery objectives, respectively, defined in the Recovery Potential Assessment in support of the Species at Risk Act listing decision process (DFO 2006). The figure is reproduced from data in DFO (2020).

Scale samples from Striped Bass spawners sampled in May and June were interpreted for age and a von Bertalanffy growth model was used to characterize the fork length-at-age relationship. Scale sampling and age interpretations are not available for all assessment years, nor are there sufficient samples of older and larger fish in any year to adequately estimate their relative abundances. The predicted sizes-at-age (Table 1) from the growth model were used to derive an age-length key which was applied to the annual length distributions of the spawners to estimate the annual abundance-at-age of spawners.

A length-weight relationship, for sexes combined, was estimated using whole weight (kg) and fork length (cm) data obtained from sacrificed samples of spawners in the Northwest Miramichi River during May and June, 2013 to 2015. This relationship was used to derive mean weight-at-age values of spawners (Table 1).

There are no data with which to directly estimate the age or size at 50% maturity because no representative sampling of bass for age and maturation assessment is available. The maturation schedule of male and female bass was assumed to differ, with males first maturing at age 3 years and female bass first maturing at age 4 years, and all bass being mature by age 6 years (Table 1; Douglas et al. 2006). The assumed maturation schedule and the resulting proportion female at age of the spawners are supported by observations of the sex ratio at length from sacrificed samples of May and June in the Northwest Miramichi. There is a low proportion female for bass ranging from 33 to 48 cm, roughly equivalent to age 3, and an increasing proportion of females in the size range of age 4 bass with the proportion of females levelling off at around 0.5 for size ranges of bass aged 5 and older (Table 1).

There is no southern Gulf specific fecundity to weight relationship for Striped Bass. For modelling purposes, a value of 83,000 eggs per kg was used, based on estimates from other populations of Striped Bass.

Table 1. Biological characteristics by age of Striped Bass from the southern Gulf of St. Lawrence. These characteristics are used in the modelling of population dynamics and in the equilibrium modelling to derive reference points.

Age (years)	Predicted mean fork length (cm)	Predicted mean weight (kg)	Assumed proportion mature		Estimated proportion female at age of spawners
			Male	Female	
1	17.5	0.06	0	0	0
2	29.0	0.29	0	0	0
3	38.5	0.68	0.5	0.1	0.17
4	46.7	1.20	0.9	0.5	0.36
5	53.6	1.82	1.0	0.9	0.47
6	59.4	2.47	1.0	1.0	0.50
7	64.4	3.17	1.0	1.0	0.50
8	68.6	3.81	1.0	1.0	0.50
9	71.9	4.40	1.0	1.0	0.50
10	75.0	5.00	1.0	1.0	0.50
11	77.6	5.54	1.0	1.0	0.50
12	79.4	5.95	1.0	1.0	0.50
13	81.4	6.40	1.0	1.0	0.50
14	82.8	6.75	1.0	1.0	0.50
15	84.2	7.06	1.0	1.0	0.50

Mortality

Mortality-at-age is assumed to be similar for male and female Striped Bass.

Mortality-at-age is a parameter that is estimated in the population model. Inferences of mortality-at-age for the younger age groups (ages 0 to 3 years) cannot be made based on the data used in the model hence other methods are used to estimate it. Estimates of natural mortality (M) to be used as priors in the population modelling for age – 0 (overwinter survival), and ages 1 to 2 were derived using the empirical relationship published in Gislason et al. (2010) that relates instantaneous natural mortality rate to von Bertalanffy growth characteristics of the species (Table 2).

Acoustic tagging and tracking programs of Striped Bass conducted in 2003, 2004, 2008, 2009, and 2013 to 2017 provide independent data to estimate annual mortality (converse survival) rates of adult (age-4+) Striped Bass to the Miramichi River. Sequential detections of tagged bass from acoustic receivers in the Miramichi River are used. It was assumed that fish detected in the Miramichi one year would be expected to return to the Miramichi the following year, and the ratio of numbers of animals detected over the two periods provides an estimate of survival rate. These survival rate estimates would include both natural and fishing mortality because these fish would have been vulnerable to legal and illegal fisheries over those years. Estimates of instantaneous mortality rates (Z) were 0.41 (median; survival = 0.66) during the period 2003 to 2009 and Z = 0.22 (median; survival = 0.80) for the period 2014 to 2018 (Figure 3). It is not possible to partition the natural mortality rates from fishing mortality rates with these data however considering that fishery removals would have in part contributed to the estimated mortalities, natural mortality of adult sized (> 47 cm) Striped Bass should therefore be less than 0.2 (Figure 3).

Table 2. Predicted natural mortality (M) at age of Striped Bass based on the fitted von Bertalanffy growth characteristics and the empirical relationship of M to growth characteristics from Gislason et al. (2010).

Age	Mid-season mean size (mm) ($L_{a,t}$ to $L_{a+1,\,t+1}$)	Predicted M	Predicted survival (S = $\exp^{-M}$)
0	135 (110 to 160)	1.97	0.14
1	232 (175 to 290)	0.82	0.44
2	337 (290 to 385)	0.45	0.64
3	426 (385 to 467	0.31	0.73

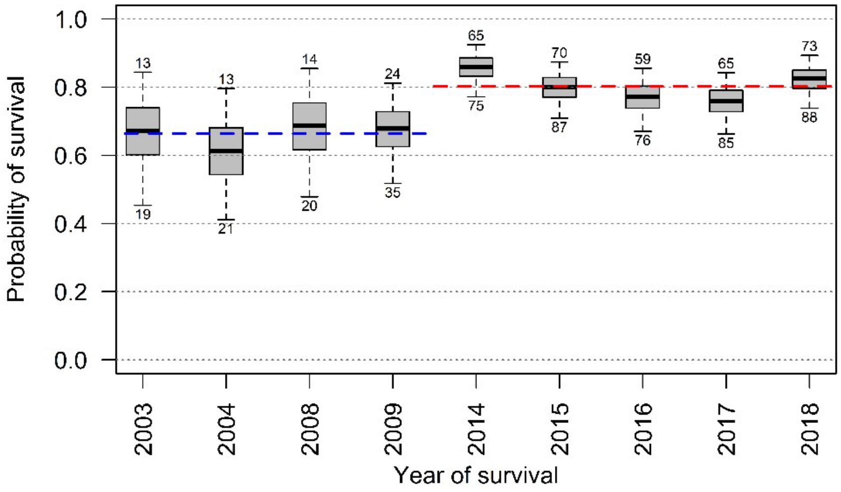

Figure 3. Posterior distributions of the estimated probabilities of survival (S) of acoustically tagged Striped Bass based on annual sequential detections in the Miramichi, pooled over size groups, tag types and release locations. The horizontal dashed lines represent the median annual survival probabilities for the 2003 to 2009 period and the 2014 to 2018 time period, respectively. The inferred year of survival represents the calendar year (e.g. 2017 is the survival over the period between winter 2016/17 and winter 2017/18). Boxplots show the 2.5 to 97.5 percentile ranges as whiskers, the interquartile range as the rectangle, and the median as the internal dash. The numbers shown in each panel for each boxplot are the numbers of fish detected (above) and the number of tags available (below) used in the estimation of the survival rates.

Population Modelling

An age structured population model with an underlying stock and recruitment relationship (from eggs to age-0 or to age-3 dependent on choice of model) was used to model the population dynamics of Striped Bass. The life cycle population dynamics incorporate the estimated and/or assumed life history characteristics of the Striped Bass population of the southern Gulf. Fork length-at-age, weight-at-age, and mortality-at-age are assumed to be similar for male and female Striped Bass. The beginning of the year is the spawning period, mid-May to mid-June, corresponding to the assessment period.

A series of age-structured life cycle models with differing assumptions and parameters to be estimated were examined (models 1 to 6). Some life history characteristics (mean weight-at-age, proportion female-at-age of spawners, eggs per kg of spawner) were set at fixed values in all model variants. For the other life history parameters (Beverton-Holt stock and recruitment parameters, mortality-at-age, proportion of recruits that are spawners), prior distributions were used. As there are no catch data for the Striped Bass fisheries of the southern Gulf that could be used in the population model, only total mortality (natural and fishing mortality) at age is estimated in the model. Time varying parameters were not considered in the models.

The time series of total abundance of spawners and estimated abundances at age for the period 1996 to 2019 were used (Figure 2). The data series begins in 1996 because prior to 1996, there was an active harvest of Striped Bass on the spawning grounds in the gaspereau fishery that was removing fish concurrent with the assessment program; the assessed population estimates for 1994 and 1995 are considered to be potential spawners rather than realized spawners. The same situation may apply since 2013 concurrent with the reopening of the Indigenous FSC fisheries and recreational fisheries, however, the harvest of Striped Bass during the assessment period (mid-May to mid-June) for those years is considered to be substantially less than what occurred prior to 1996.

Estimates of key life history and population dynamics parameters from the population model were used to derive Maximum Sustainable Yield (MSY) and other reference points. Estimates of natural mortality (M) at age of the age groups that are potentially exposed to directed fishing are required. Since the population model estimates total mortality (Z) these values are not used; rather information from acoustic tagging studies was used to define a reasonable value for natural mortality (M = 0.2) at ages 4 years and older. Model estimates of mortality-at-ages 0 to 3 were considered synonymous with natural mortality for those age groups.

Population modelling results

The time series of increasing abundance of spawners for the Striped Bass population during 1996 to 2019 follows a one way trajectory and the observations provide limited information to clearly define the population dynamics (Figure 2). Although, from a theoretical perspective, compensatory density dependence has been demonstrated to be necessary for population regulation (and hence surplus production to support a fishery) to occur, the data are insufficient to adequately characterize the strength of this relationship for this population. There is insufficient evidence to unequivocally conclude or reject the assumption of a density-dependent compensatory stock and recruitment relationship for this population. Model variants 1 to 3 were dismissed because of poor fits and models 4 to 6 are discussed below.

A priori, a density dependent Beverton-Holt stock and recruitment function with density dependence occurring between eggs and age 0 summer abundance is assumed (models 4 and 5). The spawning / nursery habitat and food base for the larvae and post-metamorphosis juveniles are constrained to a relatively small tidal area in the Northwest Miramichi, with the carrying capacity limit at the early juvenile (age – 0, summer) phase. Juveniles in their first summer gradually emigrate from the spawning area of the Northwest Miramichi and spread through other estuaries and coastal areas of the southern Gulf by their first autumn.

Model 6, with the stock and recruitment dynamics expressed from eggs to age 3, provided equally good fit to the observations compared to models that considered eggs to age 0 recruitment (models 4 and 5). However, the estimated carrying capacity at age 3 from model 6 is approximately nine times and four times higher than the scaled carrying capacity to age 3 for models 4 and model 5, respectively. Although, in-of-themselves, the data do not sufficiently preclude model 6, the limit reference points from this model are not consistent with the growth of this population. Specifically, the observation that the population has recovered rapidly from levels lower than the estimated limit reference points from Model 6 indicate that irreparable harm has not occurred at abundance levels lower than estimated LRP's from this model.

In terms of model diagnostics, the differences between model 4 and model 5 are small and insufficient to choose one model over the other (Table 3; Figure 4). Model 4 estimated survival rates at ages 0, 1, and 2 whereas model 5 estimated the cumulative survival from age 0 to 3. There were no observations of abundance at those age groups to objectively estimate those mortality rates.

All other parameters of the models including total mortality for ages 3 to 15+ and the proportion of recruits that become spawners had similar estimated values.

Table 3. Summary (median; 5th to 95th percentiles range) of posterior estimates of the stock and recruitment parameters and predicted abundances from models 4 and 5 with a Beverton-Holt stock and recruitment function from eggs to age-0. The equilibrium abundance estimates are based on the equilibrium model with life history parameters from the specific population model fits, assuming no fishing.

Population dynamics descriptor	Model 4 (BH-eggs to age-0)	Model 5 (BH-eggs to age-0)
Survival eggs to age 0 (α; slope at the origin)	5.34 E-4 (3.53 E-4 to 8.27 E-4)	2.28 E-4 (1.32 E-4 to 4.02 E-4)
Survival age 0 to 3	0.0631 (0.0449 to 0.0869)	0.163 (0.103 to 0.249)
Survival eggs to age 3 (in absence of density dependence)	3.34 E-5 (2.45 E-5 to 4.76 E-5)	3.65 E-5 (2.51 E-5 to 5.65 E-5)
Asymptotic abundance (K; Beverton-Holt model) Age 0 (millions)	9.10 (6.25 to 12.46)	6.80 (4.06 to 10.27)
Asymptotic abundance (K; Beverton-Holt model) Age 3 recruitment (thousands)	566 (383 to 834)	1,074 (640 to 1,799)
Equilibrium abundance from modelling Age 0 (millions)	7.37 (4.94 to 10.22)	5.23 (2.87 to 8.38)
Equilibrium abundance from modelling Age 3 recruitment (thousands)	456 (314 to 685)	824 (444 to 1,466)

Model 4 estimates a higher survival rate at the origin and a higher carrying capacity to age-0, however, the carrying capacity at age 3 is lower for model 4 compared to model 5 due to the lower cumulative survival from age 0 to age-3 inferred from Model 4 relative to Model 5. The lower carrying capacity at age 3 and the higher survival rate at the origin from Model 4 will in turn result in lower reference values for MSY and other reference points compared to Model 5 (Table 3; Figure 4).

For the reasons expressed above, models 4 and 5 are carried forward as the preferred models. Estimates of MSY and candidate reference values are presented for both models 4 and 5.

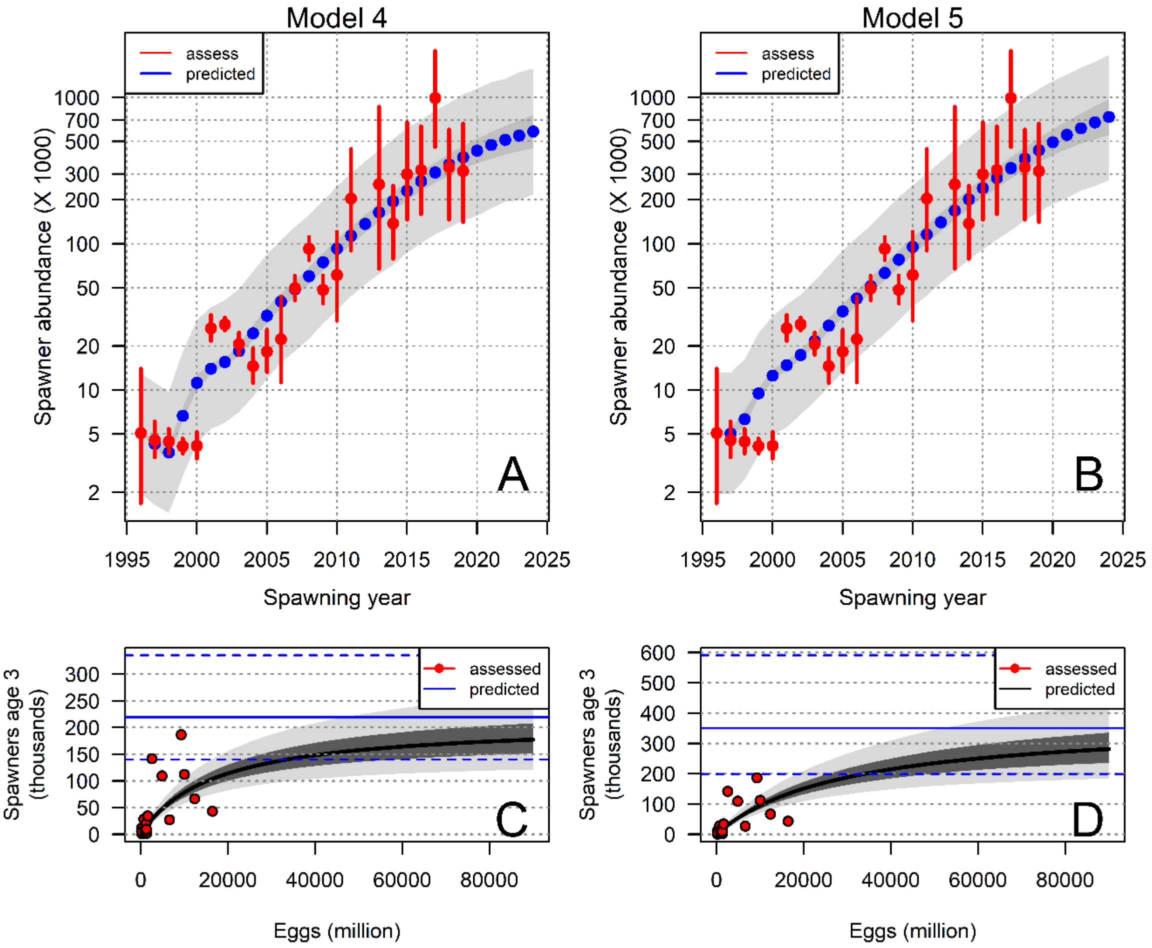

Figure 4. Observed and predicted total spawners of Striped Bass from the southern Gulf of St. Lawrence (upper row; A, B) and the stock and recruitment predicted abundance of spawners at age 3 years old (lower row; C, D) based on Model 4 (left panels A and C) and Model 5 (right panels B and D). In the upper row of panels, the assessed abundances are shown as red symbols for the median with 5th to 95th percentiles ranges as red vertical lines. The blue symbols are the predicted abundances, the darker grey shading is the 5th to 95th percentile range of mean predicted abundance and the light grey shading represents the 5th to 95th percentile range of the predicted spawner abundance accounting for the full process uncertainty. Note the y-axis abundance is shown on the log scale. In the lower panel, the assessed abundance of 3-year old spawners is shown as red symbols and the predicted median line with 25th to 75th and 5th to 95th percentile intervals are dark and light grey shading, respectively. The upper (blue) solid horizontal line (median) and the dashed horizontal lines (5th to 9th percentile range) are the Beverton-Holt asymptotic abundance (K).

Equilibrium Modelling to Define Reference Points

A forward projecting equilibrium approach, which simulates population abundance trajectories based on estimated and fixed life history parameters, is used to compare abundance, age structure, and fisheries yields at different levels of fishery exploitation as the population approaches its equilibrium abundance. The equilibrium model uses the same life cycle equations as in the estimation model with modifications to the catch equation to consider fisheries management strategies. Reference values based on MSY and spawner-potential-per-recruit (SPR) are examined with this model.

MSY is derived by searching over a range of fully-recruited fishing mortality rate (F) for the fishing rate (Fmsy) that results in maximum yield (in weight). Biomass at MSY (Bmsy), spawner abundance (number of fish) at Bmsy, catch (Cmsy; in number and weight), and age structure of the catch and of the spawners at MSY are extracted from the simulation outcomes.

SPR is presented as the percentage of the spawner potential (in terms of biomass or egg production) produced by an individual recruit throughout its life; this contribution decreases with increasing fishing mortality. SPR is presented as a percentage of the spawning potential which remains after fishing relative to a population that is not fished.

Estimates of natural mortality (M) at age are required for equilibrium modelling. For ages 0 to 3, the estimates of total mortality, as equivalent to natural mortality, from the population models are used because these age groups are not considered to be exploited in directed fisheries. For ages 4 years and older, the population model estimates are for total mortality (Z). Inferences on the maximum level of natural mortality expected for the southern Gulf of St. Lawrence population were obtained from the analysis of survival rates of acoustically tagged Striped Bass. For purposes of equilibrium modelling and to define reference points, comparisons of two assumptions for M were considered:

- Assuming M = Z, based on mortality rates derived from the population model for ages 3 to 15+;
- M at age 3 based on Z estimates from the population model and M = 0.20 for ages 4 to 15+ inferred from acoustic tagging information.

MSY and SPR reference points are context specific. The reference point values depend not only on the parameter estimates of the population dynamics (survival, proportion recruits to spawners) but also on the fisheries management scenarios, particularly those that have size restrictions for harvest retentions The size limits, combined with the size distributions at age, define the partial recruitment at age to the fishery and hence the proportion of the total annual losses at age attributed to fishing.

Management strategies based on size limits are examined with the model with respect to how these modify derived reference points (Table 4). Fishery selectivity at age (s_a) to fully-recruited F is determined using the predicted fork length distribution at age from the von Bertalanffy model relative to a defined management strategy based on fork length.

Table 4. Example management strategies based on size limits that were examined in the context of defining fishery reference points for Striped Bass.

Retention regulations	Minimum size (fork length, cm)	Maximum size (fork length, cm)	Comment
No size limits	na (30)	na (150)	For purposes of modelling, a minimum size of 30 cm was assumed as the smallest fish that would be retained. A maximum size of 150 cm was set that exceeds the expected size of any fish.
Slot size	47	61	As per recreational fisheries plan of 2016 to 2020.
Maximum size only	na (30)	65	For purposes of modelling, a minimum size of 30 cm was assumed as the smallest fish that would be retained.

MSY and SPR reference values

MSY and SPR reference values derived from equilibrium modelling are dependent upon the assumptions of natural mortality (Table 5). As expected, equilibrium abundances, abundance at Bmsy, and potential realized catch at Fmsy are higher when natural mortality is assumed to be lower for ages 4 to 15+ (Table 5, right column). In contrast, Fmsy (minimally) and fishing rates at 50% SPR and 30% SPR are higher when natural mortality is higher.

Table 5. MSY and SPR reference levels (median; 5th to 95th percentile range) derived from the equilibrium modelling based on life history parameters and population dynamics parameters from model 5 for the two scenario values of M. The results are specific to the management strategy without any size limit for retention and no accounting for catch and release mortality.

Reference values from Model 5	M = Z from modelling	M informed from acoustic tagging
Equilibrium abundance ages 3 to 15+ at F = 0		
Equilibrium total biomass (biomass, t)	4,140 (2,120 to 11,450)	13,980 (8,040 to 24,710)
Equilibrium total abundance (number, thousands)	2,320 (1,380 to 4,340)	4,700 (2,800 to 8,060)
Equilibrium spawners (biomass, t)	2,810 (1,430 to 8,100)	10,340 (5,400 to 19,410)
Equilibrium spawners (number, thousands)	1,360 (800 to 2,620)	3,110 (1,760 to 5,610)
Equilibrium spawners (eggs, millions)	104,300 (51,300 to 317,300)	413,900 (214,100 to 783,600)
MSY references (ages 3 to 15+) at F = Fmsy		
Equilibrium total abundance (biomass; t)	1,620 (890 to 3,600)	4,610 (2,680 to 8,000)
Equilibrium total abundance (number, thousands)	1,230 (740 to 2,230)	2,430 (1,460 to 4,130)
Equilibrium spawners (biomass, t)	1,010 (550 to 2,350)	3,200 (1,770 to 5,830)
Equilibrium spawners (number, thousands)	660 (390 to 1,240)	1,450 (850 to 2,550)
Equilibrium spawners (eggs, millions)	34,560 (18,190 to 85,230)	121,680 (65,990 to 224,330)
Fishing rate and yield at msy		
Fmsy (fully recruited F)	0.18 (0.12 to 0.23)	0.17 (0.15 to 0.19)
Catch at msy (biomass, t)	210 (130 to 380)	650 (370 to 1140)
Catch at msy (number, thousands)	160 (100 to 270)	340 (190 to 590)
SPR fully recruited F (ages 3 to 15+)		
F at 50%SPR	0.19 (0.14 to 0.27)	0.12 (0.11 to 0.13)
F at 30%SPR	0.39 (0.28 to 0.53)	0.24 (0.22 to 0.27)

Of the two retained models (4 and 5) with the stock and recruitment dynamic modelled from eggs to age 0, the MSY and SPR reference values are higher for Model 5 compared to Model 4 (Figure 5). Based on M for ages 4+ inferred from acoustic tagging observations, Bmsy from Model 5 is approximately twice as high as that from Model 4. Fmsy estimates of F = 0.17 are similar between models resulting in higher catch at msy (Cmsy) values, by a factor of two, from Model 5 compared to Model 4 (Figure 5).

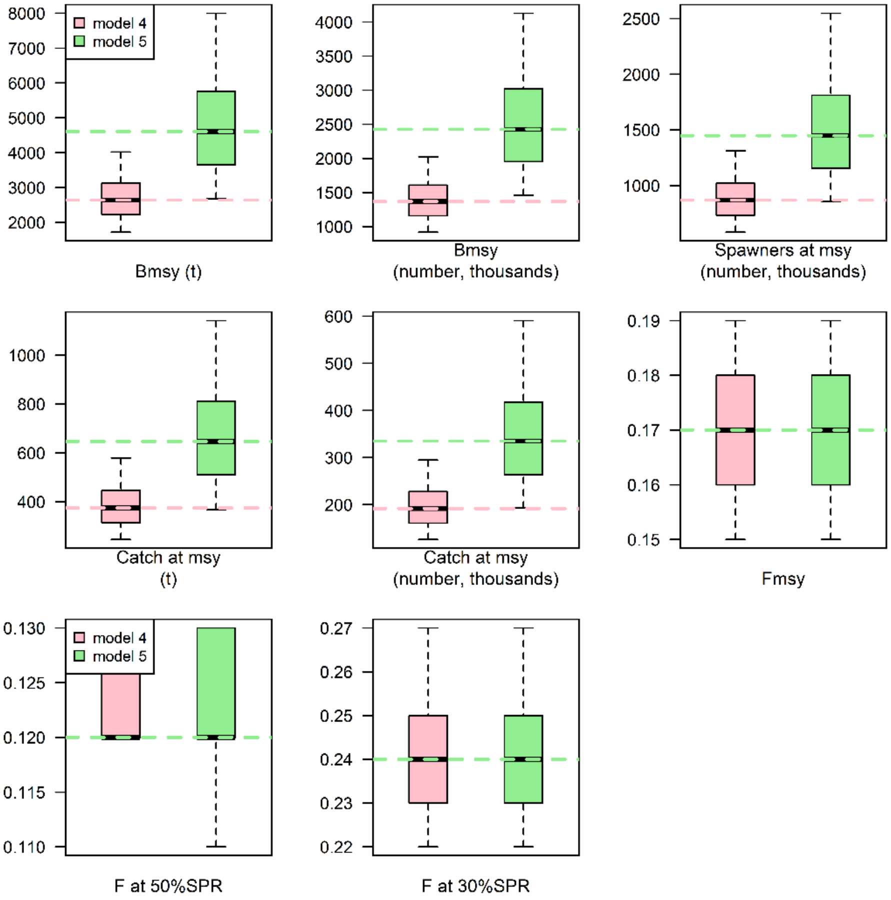

Figure 5. Comparison of MSY and SPR reference levels from Model 4 and Model 5 for scenarios with M informed by observations and for the default fishing strategy with no size limit and excluding catch and release mortality. The boxplot summaries are interpreted as follows: vertical dashed lines encompass the 5th to 95th percentile range, the boxes encompass the interquartile range, and the internal dash and dashed horizontal lines are the medians.

Fishing strategies (Table 4) have consequences on the MSY references when these are expressed in terms of numbers of fish because fishing changes the age structure of the population relative to the unfished condition (Table 6). A fishing strategy that maximizes yield in weight differs from one that maximizes yield in number. However, the consequences of fishing strategy on reference point values are small relative to the differences resulting from uncertainties in the underlying population dynamics (model 4 versus model 5).

Table 6. Comparison of estimated MSY reference values for different fishing strategies conditioned by size limits. The equilibrium simulations were run based on life history characteristics from Model 5 and assuming M for ages 4 to 15+ based on acoustic tagging observations. There is no accounting for catch and release mortality in these scenarios. Summary statistics shown are the median with the 5th to 95th percentile range. The results are similar for Model 4, in terms of direction of effects.

Reference values for Model 5	No size restrictions (slot = 30 to 150)	Slot size (47 to 61 cm FL)	Maximum size limit (30 to 65 cm FL)
Total abundance (biomass, t)	4,610 (2,680 to 8,000)	3,720 (2,210 to 6,450)	3,800 (2,250 to 6,630)
Total abundance (number, thousands)	2,430 (1,460 to 4,130)	2,060 (1,250 to 3,520)	1,990 (1,200 to 3,390)
Spawners (biomass, t)	3,200 (1,770 to 5,830)	2,550 (1,460 to 4,540)	2,610 (1,480 to 4,700)
Spawners (number, thousands)	1,450 (850 to 2,550)	1,180 (720 to 2,040)	1,140 (690 to 1,970)
Catch at msy (weight, t)	650 (370 to 1,140)	530 (300 to 940)	490 (280 to 850)
Catch at msy (number, thousands)	340 (190 to 590)	360 (210 to 640)	400 (230 to 700)

Candidate Reference Points

Striped Bass is a species of Indigenous FSC, recreational, and commercial fisheries value. Accordingly, the candidate reference points examined are based on concepts of MSY and spawner abundances to maintain a defined level of recruitment. A number of candidate reference points, based on those discussed in literature and policy (Mace 1994; DFO 2009) were examined.

Fishing strategies may have consequences on reference points because fishing changes the age structure of the population relative to the unfished condition. For purposes of defining values for the reference points, a fishing strategy that has no size restrictions for retention and that excluded catch and release mortality is used.

Reference points are presented in units of total eggs, as well as equivalences in number and biomass of spawners. Spawners are the component of the overall population of Striped Bass aged 3 years and older that are on the spawning grounds of the Northwest Miramichi at the time of the assessment during May and early June. This abundance is less than the total population of Striped Bass of those ages, as some of these are not mature while others are not on the spawning grounds during the period of monitoring and assessment.

Upper Stock Reference

Under DFO's PA policy, the USR point defines the boundary between the Cautious and the Healthy zones. The USR is the stock level threshold below which removals must be progressively reduced in order to avoid reaching the LRP and must be set at an appropriate distance above the LRP to provide sufficient opportunity for the management system to recognize a declining stock status and sufficient time for management actions to have effect. The USR is determined by productivity objectives for the stock, broader biological considerations, and social and economic objectives for the fishery (DFO 2009). An Upper Stock Reference point that differs with fishing strategy is consistent with policy of the PA as the Upper Stock Reference can reflect socio-economic considerations.

Candidate upper stock reference points examined include:

- Eggs (spawner abundance) at 80% Bmsy and

- Eggs (spawner abundance) at equilibrium when the stock is fished at F corresponding to 50% of the SPR.

Of these candidates, the eggs (spawner abundance) at 80% Bmsy is preferred to avoid the arbitrary selection of the appropriate level of SPR (here 50%) (Table 7).

Table 7. Upper Stock Reference points for Striped Bass of the southern Gulf of St. Lawrence based on population dynamics parameters of two models, assuming M from acoustic tagging observations. Summary statistics shown are the median with the 5th to 95th percentile range.

Upper Stock Reference (80%Bmsy)	Model 4	Model 5
Eggs (millions)	54,300 (33,700 to 86,400)	91,320 (49,990 to 168,040)
Spawner biomass (t)	1,460 (920 to 2,290)	2450 (1360 à 4450)[2]
Spawner number (thousands)	720 (480 to 1,090)	1210 (710 à 2110)[2]
Eggs per spawner	75,400 (65,600 to 85,000)	75,670 (64,820 to 86,000)
Mean age of spawners	5.28 (4.94 to 5.61)	5.28 (4.91 to 5.64)

Limit Reference Point (LRP)

Under DFO's PA policy, the LRP defines the boundary between the Critical and the Cautious zones. The LRP represents the stock status below which serious harm is occurring to the stock. At this stock status level, there may also be resultant impacts to the ecosystem, associated species and a long-term loss of fishing opportunities. Candidate limit reference points examined include:

- Lowest abundance (eggs) that resulted in rebuilding of the stock (Brecover);
- Abundance (eggs) corresponding to 40% Bmsy;
- Eggs (or spawner number, spawner biomass) for half saturation (50% of Beverton-Holt K); and
- Eggs (or spawner number, spawner biomass) that result in 50% of recruitment at the unfished equilibrium population size based on Beverton-Holt stock and recruitment relationship and life history characteristics.

These options differ in their underlying assumptions and behaviour. Based on the PA policy (DFO 2009), the LRP should be determined by biological considerations and as such invariant to fisheries exploitation strategies. The 40% Bmsy reference point is not invariant, however, Brecover (although not entirely, based on fishing strategies of the past) and eggs for half saturation, or half equilibrium abundance are such points).

Brecover is not considered to be an appropriate LRP for this Striped Bass population. The lowest abundance that resulted in the rebuilding of the stock provides context about how the population has responded in the past. The lowest historical spawner abundance that did not prevent rebuilding of the population is equal to the low abundances during 1996 to 2000 with a

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[2] Erratum: December 2022, corrected a transcription error from the model outputs

mean value of 4,300 fish spawners. There was a near monotonic increase from these low abundances to over 300 thousand spawners in less than 20 years. Estimated equilibrium abundance of ages 3 to 15+ in the absence of fishing is projected to be 3 to 5 million fish, 2 to 3 million spawners, dependent on the model. A Brecover value of 4,500 fish represents only 0.1% to 0.2% of the projected unfished abundance (synonymous with B0), and substantially less than a commonly discussed threshold biomass value of 20% B0.

Total eggs for half saturation or for half equilibrium abundance can be invariant to fisheries management strategy if the recruitment stage being maximized is not subject to fishing mortality and if the spawning stock is expressed in terms of eggs. The eggs for half saturation is based on the capacity of the habitat (environment) to produce recruits and does not depend on life history parameter values (e.g. natural mortality rates, age-at-maturity) for age classes older that the age at recruitment (age 3). As such, if the cumulative effects of natural and fishing mortality reduce abundance to the LRP, there is a “hard stop” and human-induced mortality is reduced to the lowest possible level. In contrast, the abundance (eggs) corresponding to 40% Bmsy and the eggs that result in 50% of the equilibrium abundance are both dependent on the life history parameter values for older fish. As such, if the natural mortality increases, both the equilibrium and the LRP would both decrease to lower and lower levels as natural mortality increases.

The eggs that result in half carrying capacity is proposed as the LRP (Table 8). Equivalent values in terms of biomass and number of fish are provided; the conversion from eggs to biomass or number of fish accounts for the changes in age structure of the population resulting from fishing. Overall, fishing has the effect of reducing the average age and average weight of the spawners resulting in a reduction in the population level eggs per spawner.

Table 8. Limit Reference Points for Striped Bass of the southern Gulf of St. Lawrence based on population dynamics parameters of two models. Summary statistics shown are the median with the 5th to 95th percentile range.

Limit Reference Point	Units	Model 4	Model 5
Brecover	Eggs (millions)	200	
	Spawners (biomass, t)	6.5	
	Spawners (number, thousands	4.5	
40%Bmsy	Eggs (millions)	24,500 (15,400 to 38,500)	40,580 (22,430 to 74,480)
	Spawners (biomass, t)	700 (450 to 1,080)	1160 (650 à 2090)[3]
	Spawners (number, thousands)	420 (280 to 630)	700 (410 to 1220)[3]
	Eggs per spawner	58,000 (50,700 to 65,200)	58,030 (50,080 to 65,850)
	Mean age of spawners	4.65 (4.41 to 4.90)	4.66 (4.39 to 4.92)
Half saturation Bev Holt	Eggs (millions)	17,300 (11,300 to 26,500)	29,950 (17,450 to 54,180)
	Spawners (biomass, t)	510 (340 to 760)	870 (520 to 1,560)
	Spawners (number, thousands	330 (220 to 490)	560 (350 to 980)

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[3] Erratum: December 2022, corrected a transcription error from the model outputs

Limit Reference Point	Units	Model 4	Model 5
	Eggs per spawner	52,400 (46,400 to 59,100)	53,250 (46,240 to 60,640)
	Mean age of spawners	4.46 (4.26 to 4.69)	4.49 (4.25 to 4.74)
Half equilibrium abundance	Eggs (millions)	15,200 (10,000 to 23,000)	26,160 (15,420 to 47,040)
	Spawners (biomass, t)	450 (300 to 670)	770 (460 to 1,360)
	Spawners (number, thousands)	300 (210 to 440)	510 (310 to 880)
	Eggs per spawner	50,800 (44,900 to 57,200)	51,470 (44,590 to 58,450)
	Mean age of spawners	4.41 (4.21 to 4.62)	4.43 (4.20 to 4.67)

Fishing Removal Rate

The fishing rate reference points considered are:

- Fmsy;
- F corresponding to 30% SPR as a maximum fishing rate; and
- F corresponding to 50% SPR as a target fishing rate.

Fmsy is proposed as the removal rate reference (Table 9). Fmsy values when presented as fully-recruited F values are dependent on the fisheries management strategy. For clarity, the Fmsy values are also presented in terms of exploitation rate, expressed as the ratio of catch (number) to total abundance of fish ages 3 to 15+. The lowest overall exploitation rate is realized for a fishing strategy without size limits. Exploitation rates at Fmsy for three fishing strategies examined are at or less than assumed natural mortality rate of M = 0.2 (S = 0.82).

Table 9. Removal rate reference for Striped Bass of the southern Gulf of St. Lawrence based on population dynamics parameters of two models. Summary statistics shown are the median with the 5th to 95th percentile range.

Removal rate reference	Model 4	Model 5
Fmsy (fully recruited fishing rate)	0.17 (0.15 to 0.19)	0.17 (0.15 to 0.19)
Exploitation rate	0.14 (0.13 to 0.16)	0.14 (0.12 to 0.16)

Stock Status Perspective Based on Reference Points

The stock status relative to these model derived reference points, over the period of assessment 1994 to 2019 is shown in Figure 6. The status is presented in terms of estimated eggs, on the same unit as the reference points. Perspectives on status are model dependent. The estimated spawner abundance has been approximately at the USR only once (in 2017) based on Model 4 whereas the abundance in 2017 was in the cautious zone based on Model 5. Dependent on the model, the spawner abundances were either below the LRP in all years except 2017 (model 5) or below the LRP until 2015 and in the cautious zone since 2016 (Model 4) (Figure 6).

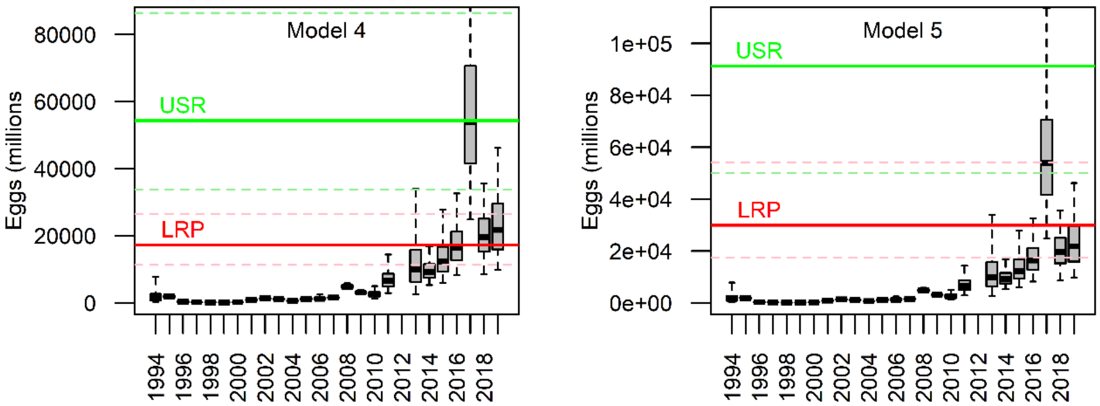

Figure 6. Assessed abundance of eggs in spawners (boxplots; eggs in millions) and status relative to the USR (upper green horizontal line) and the LRP (lower red horizontal line) candidate references from Model 4 (left panel) and Model 5 (right panel) for Striped Bass from the southern Gulf of St. Lawrence, 1994 to 2019. For Model 4 and Model 5, the USR corresponds to the median estimate of eggs at 80% Bmsy and the LRP corresponds to the median estimate of eggs that result in 50% of Beverton-Holt K (half saturation). The dashed red lines and green lines are the 5th to 95th percentile ranges of the LRP and USR respectively. Note the 95th percentile line of the USR and the 95th percentile point of eggs in 2017 are off scale in both panels.

Multi-Species Interactions

The policy to support rebuilding plans under the PA framework for stocks that are in the critical zone indicates that rebuilding objectives of a fish stock that has the potential to negatively impact the status of another species or stock (by example, rebuilding a predator species results in decline of a prey species) need to consider a balanced approach to ensure neither is depleted to a point of serious harm (DFO 2019). It is also indicated that it is not possible to simultaneously achieve yields corresponding to MSY predicted from single-species assessments for a system of multiple, interacting species (DFO 2019).

The reference points discussed in the previous section are based on single species management approaches for the purpose of maximizing yield and avoiding serious harm specific to Striped Bass. Modifying the single species reference points to account for interactions requires evidence of cause and effect consequences of Striped Bass on other species.

Striped Bass is a large bodied and generalist feeder on a variety of fish and invertebrates, with prey composition dependent upon the predator size (larger bass eat more fish), the time of year, and the foraging habitat. Striped Bass can switch among prey types based on availability and there are ample opportunities for Striped Bass to feed on diadromous species when these fish are migrating into rivers to spawn or out of rivers post-spawning and to feeding areas at sea.

Concerns have been expressed by several fisheries users that the rebuilding of the Striped Bass population in the southern Gulf has contributed to declines in abundances and their catches of Atlantic Salmon (*Salmo salar*), gaspereau (*Alosa pseudoharengus, A. aestivalis*), Rainbow Smelt (*Osmerus mordax*) and other species. The interaction is by Striped Bass predation on these potential prey species. The body sizes of gaspereau and Rainbow Smelt adult spawners and juveniles, as well as the body size of seaward migrating Atlantic Salmon smolts, are within the range of prey size for adult Striped Bass.

Gaspereau and Rainbow Smelt are important (occurrence) prey identified in Striped Bass stomachs sampled in May and June in the Miramichi River (DFO 2016). Recorded commercial landings of gaspereau and Rainbow Smelt from the NB districts of DFO Gulf Region for the period 1990 to 2018 show a steep decline beginning in 2005 (Figure 7). Commercial fisheries landings are generally not proportional to abundance, unless the proportion harvested is the same over time, which is almost never the case. The annual variations and declines in the landings are also likely due to factors such as changes in effort, changes in the number of active licences, and some differences in sales to buyers versus local sales (for bait) over time.

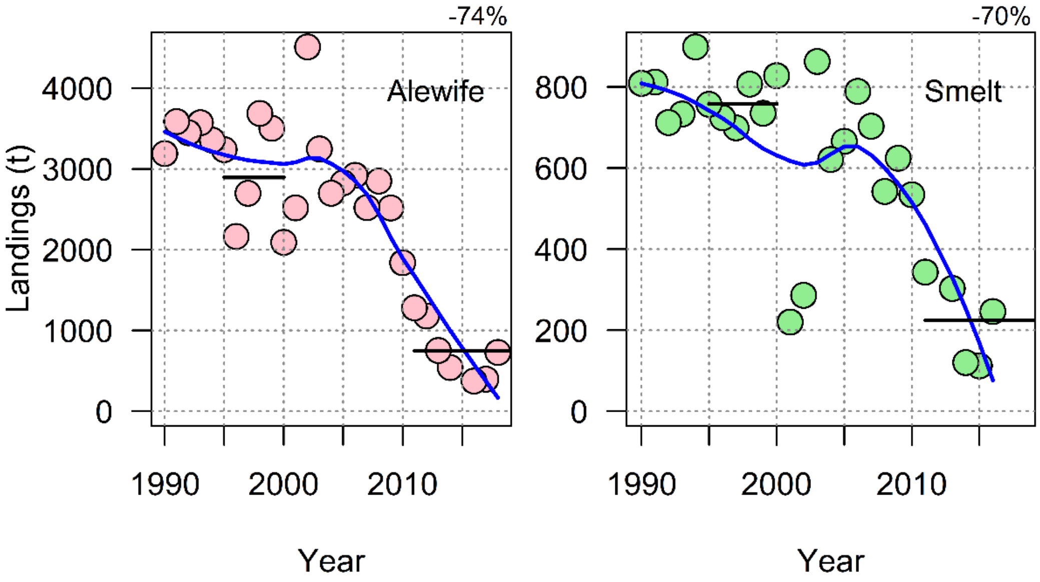

Figure 7. Recorded landings (t) of gaspereau (left panel; includes Alewife and Blueback Herring) and Rainbow Smelt (right panel) from the province of NB districts in DFO Gulf Region, 1990 to 2018. Some data are missing due to confidentiality restrictions. The blue line in each plot is a LOESS smoother using a span value of 0.8. The mean landings for the periods 1995 to 2000 and 2011 to 2018 are shown as black horizontal lines and the percent change of the 2011 to 2018 period relative to the 1995 to 2000 period is shown in the top right above each panel.

Fishery independent indices of abundance, based on total annual catches of four diadromous species, are also available from index estuary trapnets operated by DFO Science in the Northwest Miramichi (since 1998) and the Southwest Miramichi (since 1994) rivers (Figure 8). The index trapnets have been installed at the same location and monitored using similar procedures and protocols over the entire time series of operation.

Gaspereau and Atlantic Salmon indices declined at the facilities in both rivers, with the most important decline in the gaspereau index of the Southwest Miramichi (Figure 8). Collectively, causal Striped Bass predation and commercial fisheries would be expected to be most important in the Northwest Miramichi, however the decline in gaspereau indices was more important in the Southwest Miramichi.

Large increases in Striped Bass were noted in the catches in both the Northwest Miramichi and Southwest Miramichi; large increases in the Northwest Miramichi would be expected given the large increase in the population size of stock that spawns in the Northwest Miramichi. The abundance indices of American Shad (*Alosa sapidissima*) have increased at both facilities, with a larger increase in the Southwest Miramichi in which there is a recognized shad spawning area (Figure 8).

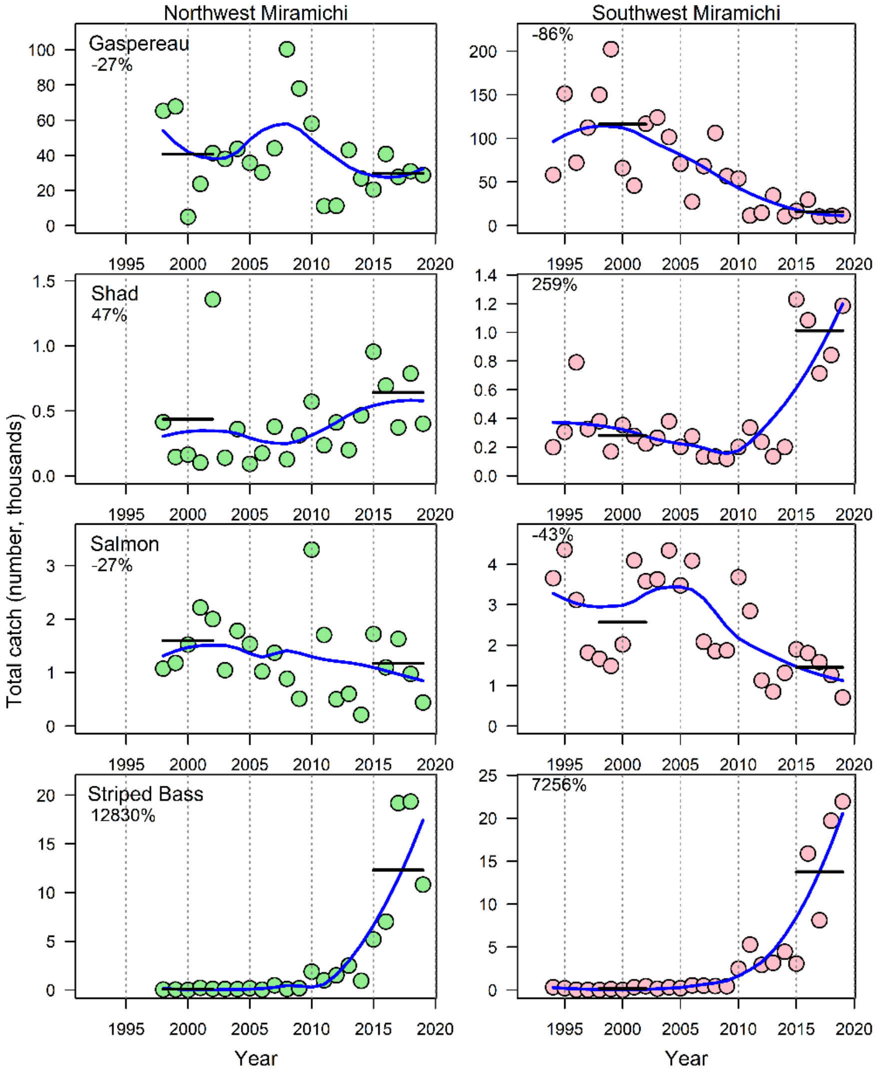

Figure 8. Summary of total catches (number, thousands) of diadromous fish species (Gaspereau = Alosa pseudoharengus and A. aestiavalis, top row; Shad = A. sapidissima, second row; Salmon = Salmo salar (adults), third row; Striped Bass = Morone saxatilis, fourth row) at the DFO index estuary trapnets in the Northwest Miramichi (left column) and the Southwest Miramichi (right column), 1994 (1998 for Northwest Miramichi) to 2019. Total catches are not corrected for dates of operation which can vary between years and between trapnets. The blue line in each plot is a loess smoother using a span value of 0.8. The mean catches for the periods 1998 to 2012 and 2015 to 2019 are shown as black horizontal lines and the percent change of the 2015 to 2019 period relative to the 1998 to 2002 period is shown in the top left corner of each panel.

Atlantic Salmon specific interactions

The most likely interaction between Atlantic Salmon and Striped Bass is expected during the seaward outmigration phase of Atlantic Salmon smolts. Atlantic Salmon smolts are of suitable body size range for most adult Striped Bass and the smolt migration timing and corridor from freshwater to the sea has smolts from the Northwest Miramichi River in particular migrating through the spawning area and the staging areas of Striped Bass at approximately the same time as Striped Bass are aggregating and spawning in the Northwest Miramichi.

There are direct observations of predation by Striped Bass on smolts based on stomach samples collected in May and June in the Miramichi River (DFO 2016). Indirect evidence of predation is provided from several studies using acoustic tags placed in Atlantic Salmon smolts with inferences of predation events based on movement patterns (Daniels et al. 2018), changes in identification codes of tags signaling a predation event (Daniels et al. 2019), and from changes in estimated survival rates in the early phase of migration through Miramichi Bay (Chaput et al. 2018).

A long term acoustic tagging and tracking study, conducted by the Atlantic Salmon Federation (ASF) since 2003 in four rivers in the southern Gulf of St. Lawrence, estimated that survival rates of “tagged smolts” through Chaleur Bay (Restigouche, Cascapedia rivers) were relatively high (67% to 95%), and with no change over time in contrast to the survival rates of “tagged smolts” through Miramichi Bay which were lower (28% to 82%) and showed a decline in survival beginning in 2010 (Chaput et al. 2018). The differences in apparent survival rates in two neighbouring coastal embayments have been hypothesized to be in part related to differences in predation pressure on migrating smolts from Striped Bass present in the Miramichi Bay during the smolt migration period but not in Chaleur Bay.

There is a negative relationship between Striped Bass abundance estimates and the estimated survival rates of acoustically tagged smolts (Figure 9). In both rivers, the lowest survival rates from head of tide to bay exit were estimated in the recent period (2013 to 2016) when the estimated abundance of Striped Bass was greater than 100 thousand spawners.

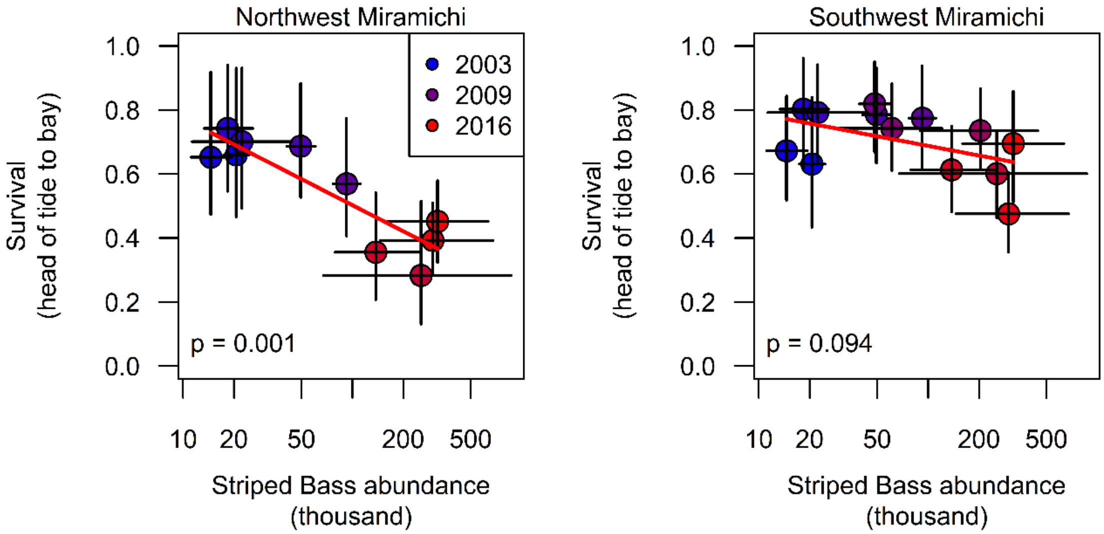

Figure 9. Association between the estimated survival rates from head of tide to bay exit of acoustically tagged smolts (Northwest Miramichi left panel, Southwest Miramichi right panel; data from Chaput et al. 2018) and the estimated spawner abundance of Striped Bass (log scale) in the Miramichi River, 2003 to 2016. For both the survival rates and spawner abundance values, the symbol is the median and the black lines are the respective 5th to 95th percentile range of the estimates. The linear relationship (red line) and the corresponding p-value of the slope of the regression = 0 is shown in the lower left corner of each panel.

Population level effects of predation by Striped Bass on Atlantic Salmon smolts were examined using annual indices of juvenile salmon abundance and the estimated returns of one-sea-winter and two-sea-winter first time spawning (maiden) salmon adults. Estimated relative survival rates in the first year at sea for the smolt migration years 1994 to 2018 show wide variation for both the Southwest and Northwest Miramichi river returns (Figure 10). Plotted against corresponding Striped Bass spawner abundances for the year of smolt migration (and the year of potential predation by bass), there is an apparent decline in relative survival rates of smolts from the Southwest Miramichi, especially for the 2006 to 2018 migration years (the highest relative survival rates were estimated for the 2009 smolt migration year) associated with increasing Striped Bass abundance (Figure 10). However, low relative survival rates for the Southwest Miramichi were estimated in the late 1990s when Striped Bass abundances were low. The relationship between relative survival rates and Striped Bass spawner abundances is not statistically significant for the Northwest Miramichi smolts (Figure 10).

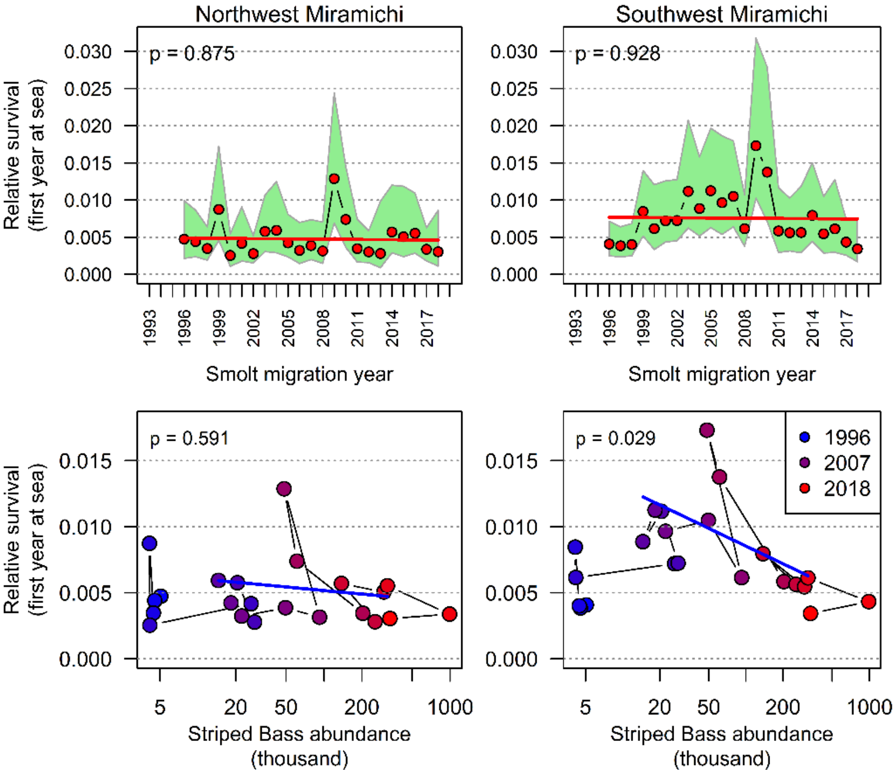

Figure 10. Posterior distributions of the relative survival rates during the first year at sea for smolts migrating from the Northwest (left column) and the Southwest (right column) Miramichi rivers for the smolt migration years 1996 to 2018 (top row). The bottom row shows the relative survival rates during the first year at sea plotted against the estimated (log scale) Striped Bass spawner abundances in the Miramichi River for the corresponding smolt and Striped Bass spawning years 1996 to 2018. The solid blue line is the linear regression of relative survival rates to log of Striped Bass abundances for the 2003 to 2016 years corresponding to the acoustic tagged smolt survival time series of the Miramichi River (see Figure 9).

Conclusions on species interactions

There is contradictory evidence of reductions in examined anadromous fish abundance indicators associated with increased abundance of Striped Bass in the southern Gulf. For gaspereau, recorded commercial landings have greatly declined in Gulf NB portion, since 2005.

Catches from estuarine index trapnets also show declines, beginning in 2005, but less severe than indicated by the commercial landings. Commercial landings of Rainbow Smelt have also greatly declined in the Gulf NB portion since 2005 whereas American Shad indicators from the estuary trapnets have increased. For Atlantic Salmon, there is direct evidence of predation by Striped Bass on Atlantic Salmon smolts. Studies using acoustic tags placed in salmon smolts have inferred predation events and changes in survival rates during the early phase of migration through Miramichi Bay, with the lowest survival rates estimated for the years when Striped Bass spawner abundances exceeded approximately 100 thousand spawners. Population level effects are contradictory between the two branches of the Miramichi, with relative survival rates for the Southwest Miramichi showing a negative association with Striped Bass abundance indices for the years 2003 to 2016 but not for the Northwest Miramichi.

Management Considerations

Fisheries management actions were responsive to the decline and rebuilding of the Striped Bass population beginning with the closure of all directed fisheries for Striped Bass in 2000, followed twelve years later with the re-opening of the Indigenous FSC fisheries in 2012, and the retention recreational fisheries in 2013. The re-opening of the Indigenous fishery occurred following the conclusion that the Striped Bass population in 2011 had first achieved both the limit and target recovery objectives, at a median abundance of 200 thousand spawners and a 5th percentile value of 90 thousand spawners (DFO 2013). A cautious recreational fisheries strategy (two short retention seasons, 1 fish per day, slot size limit of 55 to 65 cm TL) was chosen in 2013. Further increases in abundance in 2015, to a median estimate of 300 thousand spawners, resulted in an extended retention period in the recreational fishery for 2016. The largest change in the recreational fishery access occurred in 2018 with an authorization to retain 3 fish per day, following on the exceptional abundance estimate in 2017 of just under 1 million spawners. The pilot commercial fishery was also first authorized in 2018. This increased fisheries access occurred as the stock abundance was on a trajectory of increasing abundance from the critical zone to the cautious zone as defined in this assessment.

New and alternative fisheries access requests could be anticipated when the assessed abundance of Striped Bass surpasses the USR and be situated in the healthy zone. The fisheries exploitation potential on this species is high. During the spawning aggregations Striped Bass are captured in large numbers in gaspereau trapnets in the Miramichi with catch rates (fish per trapnet per day) that can exceed several thousand fish per net haul (DFO 2020). Striped Bass are also reportedly captured in high numbers in gaspereau fishery trapnets in other estuaries of DFO Gulf New Brunswick. Following on the expanded distribution of bass to the north shore of the St. Lawrence and Labrador in 2017, important Striped Bass harvests were reported from this northern area (DFO 2018) and the presence and harvest of Striped Bass that remained from the 2017 emigration from the southern Gulf continue to be reported from this northern area.

The recreational fishery is increasing in popularity throughout the Gulf of St. Lawrence, including westward to the north shore of the Gaspe Peninsula (Quebec). The current recreational fisheries management plan that provides a three fish daily and possession limit is the highest allocation of any jurisdiction in eastern North America. There is a one fish daily and possession limit in the DFO Maritimes Region management area as well as in the eastern US, with size limits dependent on region.

Slot size limits have been in place for the southern Gulf recreational fisheries since 2013. The slot size minimum length is intended to reduce the exploitation on younger fish until they have had an opportunity to spawn once whereas the maximum length of the slot is intended to protect older fish of higher fecundity and to maintain a spawning pool to guard against successive year classes of poor recruitment. Slot size measures are considered to be superior to other size limit strategies as a compromise regulation for achieving competing objectives of different users. The use of size limits in fisheries results in catch and release of fish that are outside the size window for retention. The recreational fishery for Striped Bass in the southern Gulf has a large component of catch and release, in part due to the mandatory slot size restrictions for retention but also associated with the fishing practices of individual anglers that favour a lot of angling activity without intent to retain.

There are very limited catch data from the recreational fishery and the consequences of retentions as well as catch and release mortality on the productivity of the population are unknown. A catch and release mortality rate of 9% is assumed in the coastwide Striped Bass assessment of the US but the catch and release mortality rate has been shown to depend upon the fishing gear, water temperature, maturity state and angler practices (NEFSC 2019). In the southern Gulf, there is a large angler presence and quantities of Striped Bass are caught and released on pre-spawning and spawning aggregations of Striped Bass in May and June in the Miramichi River. There may be non-mortality consequences to individual fish (releasing of milt and eggs when fish are handled and released, disruption of spawning behaviour) and to the spawning population of angling activities and catches at that time of year. Since 2017, short term closures of all recreational fishing in sections of the Northwest Miramichi of 5 to 9 days depending on the year have been implemented when spawning activities have been reported to DFO Fisheries Management. Such closures would reduce some of the acute and chronic consequences of recreational fishing on spawners.

Sources of Uncertainty

The Striped Bass population of the southern Gulf of St. Lawrence has been monitored annually since 1994. The estimated number of spawners is assessed using catch rates from the commercial gaspereau fishery in the Northwest Miramichi. There are large uncertainties in the annual estimates (coefficient of variation range 6% to 91%) however the near monotonically increasing abundance from approximately 5,000 spawners in the late 1990s to over 300 thousand since 2016 is confirmed from the catches by month and overall at the DFO index trapnets in the Miramichi.

Life history characteristics and population dynamics parameters required for population modelling are known with varying degrees of uncertainty. The length-weight relationship and the fork length-at-age, based on interpretations of scales, is well described from sampling data. The oldest age determined from scales for this population is 15 years. Scales are reported to underestimate the age of bass older than 8 years. Bass are relatively slow growing after age 8 to 10 years (fork lengths greater than 65 cm for this population), at less than 2 cm fork length per year. An age-length key, derived from samples obtained over years and analyzed using a von Bertalanffy growth model, is used to translate the size distribution of spawners to an age distribution. The bias introduced from this underestimation of age using scales is to overestimate the growth rate based on the model and by using the age-length key to underestimate the abundance of older fish in the population. The use of a plus group at age 15 in the age-length key addresses in part the underestimation of older fish, but older fish would remain underrepresented in the population overall, and bass at younger ages overrepresented to some degree. Collectively, this would result in overestimation of survival of the younger ages and underestimation of survival of the older ages.

Differences in growth rate, size-at-age, and weight-at-length between male and female bass are also reported in literature, aspects which are not considered in the population model that aggregates male and female abundances and their population dynamics. Maturity schedules are assumed based on assumptions from other studies and limited observations that males mature earlier than females; the resultant proportion female at age of the spawners is supported by observations from intensive sacrifice sampling of Striped Bass on the spawning grounds in recent years. Fecundity at age is not known for this population and a fixed value of eggs per kg was borrowed from other studies. The scale of the uncertainties and the bias these may introduce in the modelling of the population dynamics were not quantified. The uncertainties are considered to be small relative to the estimation of other population dynamics parameters (stock and recruitment parameters, mortality) from the model.

Mortality rates of young age classes, including overwinter mortality of age 0 and mortality-at-ages 1 and 2 years, are not known. Mortality rates of the younger age groups are expected to be high, based on general inverse relationships of size-at-age and mortality. The Striped Bass population of the southern Gulf is also at the northern limit of the species distribution and environmental conditions that modify the size of the young-of-the-year going into their first winter and the overwintering conditions through which Striped Bass fast, are important contributing factors that result in unpredictable survival rates among cohorts. Uncertainties in the mortality rates of younger ages have consequences in population models that assume a stock and recruitment relationship from eggs to age 0 in the first summer. There are no observations of relative abundance of these age groups; observations of spawners begin at age 3. Due to the lack of data at those younger ages, two models with differing equations for mortality rates from age 0 to age 3 are carried forward in the derivation of reference points.

Despite the closure of the directed fisheries for Striped Bass, fishing mortality mostly associated with illegal fisheries was considered to have continued. The directed recreational retention fisheries subsequently reopened in 2013. In the absence of fisheries harvests, the estimates of mortality from the population modelling are considered to be total mortality. Natural mortality (M) rates are required for calculating maximum sustainable yield. Acoustic tagging and tracking data from 2003 to 2018 provide estimates of total mortality of larger (> 40 cm) Striped Bass. The total instantaneous mortality (Z) estimate for the years 2014 to 2018 was 0.22 (median). Fish which overwintered in areas other than Miramichi would have been considered a mortality if there were no detections in subsequent years in the Miramichi. With this in mind, and the fact that some mortalities would have been associated with fishing, there is high certainty that the instantaneous natural mortality is very likely no higher than 0.2, the value ultimately used in the equilibrium modelling to define reference points.

It was assumed that there is a density dependent compensatory function between eggs spawned and production of young-of-the-year in the first summer. Other studies have reported that inter-year class variability in Striped Bass is high, largely determined during the egg and larval stages, and influenced by environmental factors. The population models used also consider the recruitment dynamic from eggs to young of the year as a stationary process thus ignoring the non-stationary variation in survival associated with auto-correlated variations in environmental conditions. The consequences of environmental variation begin at the egg and larval stage, and carryover into variable conditions that affect growth during their first summer with subsequent consequences associated with size biased survival of larger bodied young of the year during the first winter. A population model that ignores these non-stationary events that affect survival will not adequately characterize the variations in cohort strength that are otherwise assumed to be determined by spawner abundance and temporally independent stochastic variability. Some of these dynamics could be incorporated in the equilibrium modelling as stochastic and probabilistically determined events that change the probability of survival of a cohort in order to assess the consequences of these events on the derivation of the reference points; until these analyses are completed, the consequences are not known.

Based on the available observations, the stock and recruitment dynamic is adequately described by a proportional function or Beverton-Holt stock and recruitment function. The near monotonic increasing trajectory of the population abundance from its low point in the late 1990s to the highest abundance in the late 2010s provides limited information to unequivocally define the unfished population size. The conclusion from population modelling with the available data is that the maximum abundance for this population has not yet been realized. The recruitment from the 2017 to 2019 spawner abundances have not been assessed with 3-year olds from the 2017 spawning first available for assessment in 2020, and the other year classes in 2021 and 2022.

There is compelling evidence that the Northwest Miramichi River is the major spawning area for the Striped Bass population of the southern Gulf of St. Lawrence and the assessment data and population model assume that this is the only area that produces recruitment. Recently, eggs and larvae of Striped Bass have been sampled from geographically proximate tidal areas to the Northwest Miramichi during a period of high Striped Bass spawner abundance. Observations of spawning activities outside the Northwest Miramichi would be expected as the overall spawner abundance increases. The establishment of new spawning areas is possible as evidenced from the results of the restoration program of the St. Lawrence River. The consequence to population modelling of not considering other spawning areas depends upon whether there are exchanges of recruitment and spawners between the spawning areas. If there are exchanges, then the carrying capacity would currently be underestimated although density independent survival rates from eggs to age 0 in summer would likely not be as this is a characteristic specific to the spawning location. Currently, the assessment in the Northwest Miramichi is the only and best available information on both the spawners and recruitment of Striped Bass in the southern Gulf.

The most important uncertainty in understanding the population dynamics of the Striped Bass population of the southern Gulf of St. Lawrence is the near total absence of fisheries catches and harvest data. In the absence of catch and harvest data from all the fisheries, the best that could be done is to track the response of the population abundances to variations in fisheries management strategies. The variations in abundance could not be partitioned into components related to fishing which leaves the dynamic and robustness of this population to fishing and environmental variations unknown.

There is contradictory evidence of reductions in examined anadromous fish abundance indicators associated with increased abundance of Striped Bass in the southern Gulf. Correlation analyses are a first step in examining the potential interactions but they do not demonstrate cause and effect. A carefully designed ecological experiment with long-term monitoring would be required to resolve the question of these species interactions.

CONCLUSIONS AND ADVICE

The Recovery Potential Assessment (RPA) of the southern Gulf Striped Bass population conducted in 2006 proposed abundance recovery objectives intended to guide management actions that would promote recovery of the population (DFO 2006; Douglas et al. 2006). The RPA recovery objectives were never intended to be reference points that conformed to the PA. At the time of the RPA, there were twelve years of spawner abundance estimates available with the maximum median estimate of 28,000 fish in the early 2000s, that followed on the very low abundances of the late 1990s. The RPA objectives were first exceeded in 2011; the median spawner abundance in that year had been estimated at 200 thousand spawners (DFO 2013). The available series of assessed abundances to 2019 and the population modelling to define reference points provide a much higher potential population size than was derived during the RPA with data available to 2006. The RPA recovery objectives are not appropriate for managing fisheries on this population of Striped Bass.

An Upper Stock Reference (USR) point conditional on fishing strategy is consistent with the Precautionary Approach (PA) policy; the USR could reflect socio-economic considerations. The 80% Bmsy definition has been most frequently used in fisheries management and is proposed as the USR. There is no model consensus for the USR value. An Upper Stock Reference value equivalent to 720 thousand spawners, based on Model 4, was only surpassed in 2017. The carrying capacity of the Striped Bass southern Gulf population is very uncertain. The equilibrium modelled abundances of age 3 and older Striped Bass at Bmsy are 1 to 2 million fish, 0.7 to 1.2 million spawners, depending on the model. Potential removals when the stock is at Bmsy are in the range of 200 to 400 thousand fish annually.

Based on the PA policy, the LRP should be determined by biological considerations and preferably invariant to fisheries exploitation strategies. Brecover, the lowest historical spawner abundance that did not prevent rebuilding of the population, is not considered an appropriate LRP for this Striped Bass population. Eggs for half saturation (half the Beverton-Holt carrying capacity value) is proposed for the LRP; equivalent values in units of spawner number and spawner biomass are also provided. There is no model consensus for the LRP value. Based on the trajectory of this population over the relatively short period of assessment, maintaining spawners above 330 thousand fish (median of the LRP equivalent value from Model 4) should be sufficient to avoid serious harm.

A USR value of minimally 720 thousand spawners may underestimate the production potential however full exploitation to rates equivalent to Fmsy and potential removals at MSY (Cmsy) would likely only be considered once the trajectory of the population had placed the abundance in the healthy zone. A re-assessment of population dynamics with additional observations could be undertaken at that time to determine the appropriateness of the defined USR and LRP.

The recent fisheries management history is informative of the management decision making process in response to increased abundance. Fisheries access was responsive to the rebuilding of the Striped Bass population beginning initially with the re-opening of the Indigenous FSC fisheries in 2012, the retention recreational fisheries in 2013, and a pilot commercial fishery in 2018. Fisheries were gradually reopened and access increased as the spawner abundances progressed from levels that were in the proposed critical zone, increasing to the LRP and eventually to the cautious zone by 2019.

The exceptional 2017 value of approximately one million spawners and the decline in 2018 and 2019 to just over 300 thousand spawners provides a cautionary note on variations in size of the stock under new population dynamics conditions (extensive migration of Striped Bass beyond its historic distribution range with associated mortalities) and increasing fisheries exploitation.

In the absence of any monitoring of recreational catches and harvests, it is not possible to provide fisheries management advice in terms of total allowable catches nor can the status of the population be assessed relative to directed fisheries losses (retention and catch and release mortality). More importantly, the absence of catch and harvest data from all the fisheries precludes understanding the causes of variations in assessed spawner abundances of Striped Bass which leaves the dynamic and robustness of this population to fishing and environmental variations uncertain.

Stomach content analyses provide direct evidence of predation by Striped Bass on Rainbow Smelt, gaspereau and Atlantic Salmon smolts in the Miramichi River. There is contradictory evidence that the reductions in examined anadromous fish abundance indicators were correlated with increased predation by Striped Bass in the southern Gulf. It is not clear that reducing Striped Bass spawner abundances to lower levels would improve any of the indices of the examined anadromous species including landings of gaspereau and Rainbow Smelt in the commercial fisheries and indices of acoustic tagged smolt survival rates in the Miramichi. Alternate reference levels to address the multiple species concerns cannot be defined based on the available information. Setting a management objective for Striped Bass that is less than the defined LRP would reduce the potential yields of the Striped Bass directed fishery and be non-compliant with the PA policy (DFO 2009).

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SOURCES OF INFORMATION

This Science Advisory Report is from the regional advisory meeting of November 23-25, 2020 on Reference points that conform to the Precautionary Approach for the Striped Bass population of the southern Gulf of St. Lawrence. Additional publications from this process will be posted on the Fisheries and Oceans Canada (DFO) Science Advisory Schedule as they become available.

Chaput, G., Carr, J., Daniels, J., Tinker, S., Jonsen, I., and Whoriskey, F. 2018. Atlantic Salmon (*Salmo salar*) smolt and early post-smolt migration and survival inferred from multi-year and multi-stock acoustic telemetry studies in the Gulf of St. Lawrence, northwest Atlantic. ICES J. Mar. Sci. 76: 1107-1121.

COSEWIC. 2012. COSEWIC assessment and status report on the Striped Bass *Morone saxatilis* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. iv + 82 pp.

Daniels, J., Chaput, G., and Carr, J. 2018. Estimating consumption rate of Atlantic salmon smolts (*Salmo salar*) by striped bass (*Morone saxatilis*) in the Miramichi River estuary using acoustic telemetry. Can. J. Fish. Aquat. Sci. 75: 1811–1822.

Daniels, J., Sutton, S., Webber, D., and Carr, J. 2019. Extent of predation bias present in migration survival and timing of Atlantic salmon smolt (Salmo salar) as suggested by a novel acoustic tag. Anim. Biotelemetry 7:16.

DFO. 2006. Recovery Assessment Report for the St. Lawrence Estuary, Southern Gulf of St. Lawrence and Bay of Fundy Striped Bass (*Morone saxatilis*) Populations. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2006/053.

DFO. 2009. A fishery decision-making framework incorporating the Precautionary Approach.

DFO. 2013. Update to 2012 on spawner abundance and biological characteristics for striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Sci. Resp. 2013/010.

DFO. 2016. Spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 2015. DFO Can. Sci. Advis. Sec. Sci. Resp. 2016/017.

DFO. 2017. Information in Support of Critical Habitat Identification for Striped Bass (Morone saxatilis) of the St. Lawrence River. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2017/001.

DFO. 2018. Spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence in 2017. DFO Can. Sci. Advis. Sec. Sci. Resp. 2018/016.

DFO. 2019. Guidance for the development of rebuilding plans under the Precautionary Approach Framework: Growing stocks out of the critical zone.

DFO. 2020. Update of spawner abundance and biological characteristics of Striped Bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence to 2019. DFO Can. Sci. Advis. Sec. Sci. Resp. 2020/009.

Douglas, S.G. and Chaput, G. 2011. Assessment and status of Striped Bass (*Morone saxatilis*) in the Southern Gulf of St. Lawrence, 2006 to 2010. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/097. iv + 22 p.

Douglas, S.G., Bradford, R.G., and Chaput, G. 2003. Assessment of striped bass (*Morone saxatilis*) in the Maritime Provinces in the context of species at risk. DFO Can. Sci. Advis. Sec. Res. Doc. 2003/008: iii + 49 p.

Douglas, S.G., Chaput, G., and Caissie, D. 2006. Assessment of status and recovery potential for striped bass (*Morone saxatilis*) in the southern Gulf of St. Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2006/041: viii + 95 p.

Gislason, H., Daan, N., Rice, J.C., and Pope, J.G. 2010. Size, growth, temperature and the natural mortality of marine fish. Fish and Fisheries 11: 149–158.

LeBlanc, C.H., and Chaput, G.J. 1991. Landings of estuarine fishes in the Gulf of St. Lawrence 1917 – 1988. Can. Data Rep. Fish. Aquat. Sci. 842: viii + 101 p.

L'Italien, L., Mainguy, J., et Valiquette, E. 2020. Dynamique et habitats de reproduction de la population réintroduite de bars rayés l) dans le fleuve Saint-Laurent, ministère des Forêts, de la Faune et des Parcs, Québec, xvi + 123 p.

Mace, P.M. 1994. Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. Canadian Journal of Fisheries and Aquatic Sciences 51:110-122.

Northeast Fisheries Science Center (NEFSC). 2019. 66th Northeast Regional Stock Assessment Workshop (66th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 19-08; 1170 p.

Robinson, M., Courtenay, S., Benfey, T., Maceda, L., and Wirgin, I. 2004. Origin and movements of young-of-the-year striped bass in the southern Gulf of St. Lawrence, New Brunswick. Trans. Amer. Fish. Soc. 133: 412-426.

Valiquette, E., Harvey, V., and Pelletier, A.-M. 2017. Information update on the Identification, Description and Spatio-Temporal Use of Striped Bass (Morone Saxatilis) Habitat in the St. Lawrence River, Quebec. DFO Can. Sci. Advis. Sec. Res. Doc. 2017/005. ix + 55 p.

Valiquette, E., M. Legault, J. Mainguy, V. Bujold et A.-M. Pelletier. 2018. *Répartition du bar rayé au Québec – mise à jour des connaissances,* Ministère des Forêts, de la Faune et des Parcs, Québec, v + 17 p.

THIS REPORT IS AVAILABLE FROM THE:

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ISSN 1919-5087
ISBN 978-0-660-38600-3 Cat. No. Fs70-6/2021-018E-PDF

Correct Citation for this Publication:

DFO. 2021. Reference points for Striped Bass (*Morone saxatilis*) for the southern Gulf of St. Lawrence population. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2021/018. (Erratum: December 2022).

Aussi disponible en français :

MPO. 2021. Points de référence pour la population de bar rayé (Morone saxatilis) *du sud du golfe du Saint-Laurent. Secr. can. de consult. Sci. du MPO. Avis sci. 2021/018. (Erratum : décembre 2022).*

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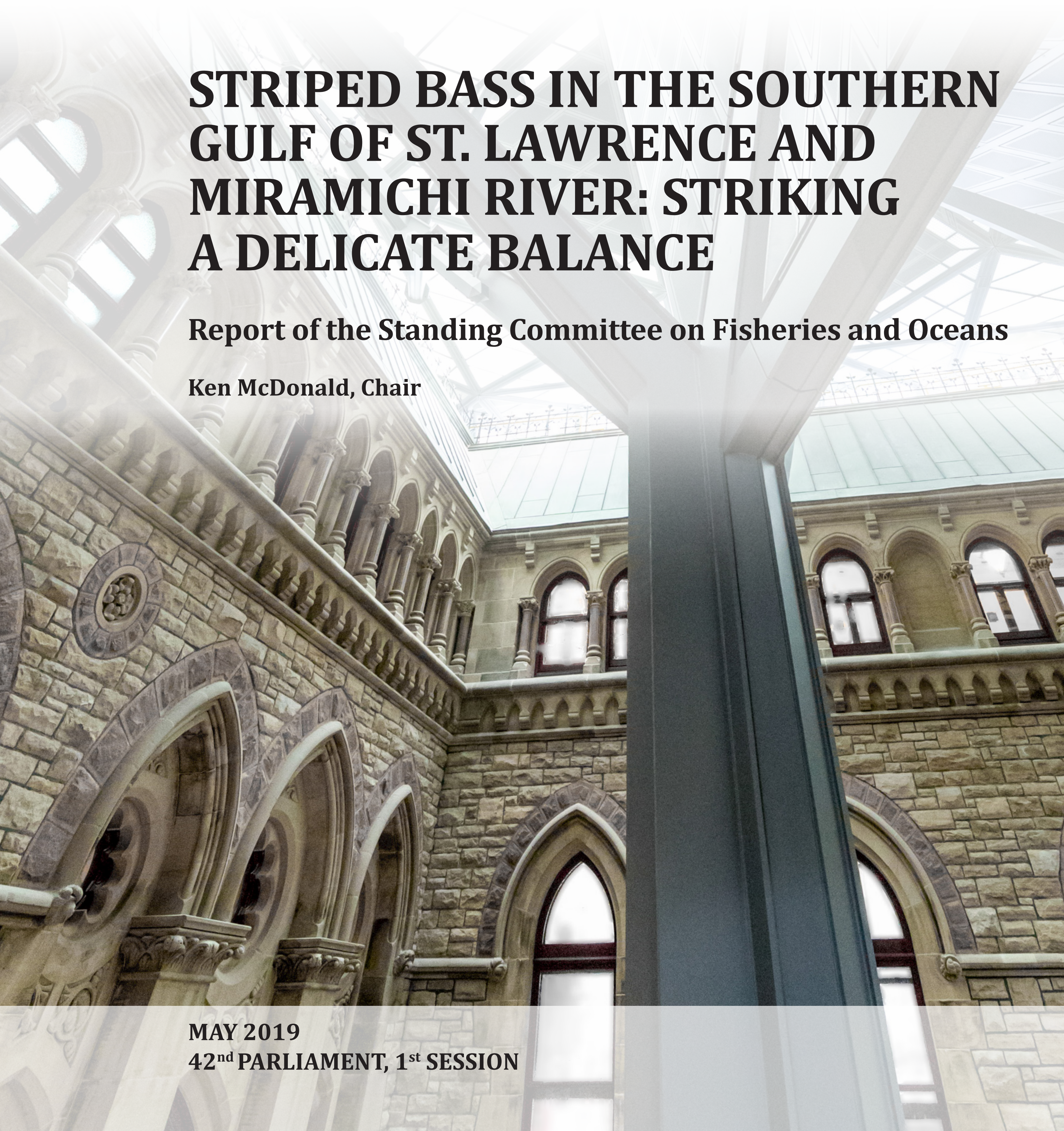

STRIPED BASS IN THE SOUTHERN GULF OF ST. LAWRENCE AND MIRAMICHI RIVER: STRIKING A DELICATE BALANCE

Report of the Standing Committee on Fisheries and Oceans

Ken McDonald, Chair

MAY 2019
42nd PARLIAMENT, 1st SESSION

Published under the authority of the Speaker of the House of Commons

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NOTICE TO READER

Reports from committee presented to the House of Commons

Presenting a report to the House is the way a committee makes public its findings and recommendations on a particular topic. Substantive reports on a subject-matter study usually contain a synopsis of the testimony heard, the recommendations made by the committee, as well as the reasons for those recommendations.

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THE STANDING COMMITTEE ON FISHERIES AND OCEANS

has the honour to present its

TWENTY-THIRD REPORT

Pursuant to its mandate under Standing Order 108(2), the Committee has studied the impact of the rapid increase of the striped bass in the Miramichi river and the Gulf of St. Lawrence and has agreed to report the following:

TABLE OF CONTENTS

LIST OF RECOMMENDATIONS

As a result of their deliberations committees may make recommendations which they include in their reports for the consideration of the House of Commons or the Government. Recommendations related to this study are listed below.

Recommendation 1

That Fisheries and Oceans Canada establish upper and lower limit reference point thresholds for striped bass and adjust them if necessary based on justifiable scientific evidence. 11

Recommendation 2

That Fisheries and Oceans Canada's restoration framework prioritize the long-term balance of fish species in the Southern Gulf of Saint Lawrence and Miramichi River. 12

Recommendation 3

That a mechanism for controlling the striped bass population include a First Nation commercial and food, socal and ceremonial fishery, and recreational public food catch and possession limits. 12

Recommendation 4

That Fisheries and Oceans Canada remove the maximum length restriction of 65 cm for the striped bass retention fishery, unless required for protection of spawners and population management. 13

Recommendation 5

That Fisheries and Oceans Canada invest more resources to obtain better data on the striped bass numbers and life cycle as well as how it is affected by climate change. 13

STRIPED BASS IN THE SOUTHERN GULF OF ST. LAWRENCE AND MIRAMICHI RIVER: STRIKING A DELICATE BALANCE

INTRODUCTION

The recovery of the striped bass (*Morone saxatilis*) in the Southern Gulf of St. Lawrence and Miramichi River from critically low population levels in the 1990s to its current abundant state has been termed a "good news story" by Fisheries and Oceans Canada (DFO).[1] The increase of the striped bass population has led to the creation of a successful recreational fishing industry and has helped re-establish a First Nation commercial fishery, which advances reconciliation. However, the rapid population increase of striped bass has also raised concerns of ecosystem imbalances and further strains on struggling wild Atlantic salmon (*Salmo salar*) populations.

In this context, on 6 November 2018, the House of Commons Standing Committee on Fisheries and Oceans (the Committee) adopted a motion to:

> …undertake a study on the impact of the rapid increase of the Striped Bass in the Miramichi River and the Gulf of St. Lawrence, and how and when reference points are determined which may trigger interventions on this and other predators affecting other species and marine life; that this study be comprised of no less than three meetings; and that the committee report its findings with recommendations back to the House.[2]

The Committee heard from all witnesses that a balance must be struck between managing a healthy striped bass population and ensuring that other species, in particular wild Atlantic salmon, are protected from excessive predation. As Deborah Norton of the Miramichi Watershed Management Committee noted: "Striped bass is not a bad fish. It just has to eat."[3]

The Committee held two public meetings on 25 February 2019 and 1 April 2019, during which it heard testimony from commercial and recreational fishing organizations,

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[1] Serge Doucet, Regional Director General, Gulf Region, Fisheries and Oceans Canada, *Evidence*, 1 April 2019.

[2] House of Commons, Standing Committee on Fisheries and Oceans [FOPO], *Minutes of Proceedings*, 6 November 2018.

[3] Deborah Norton, President, Miramichi Watershed Management Committee Inc., *Evidence*, 1 April 2019.

salmon conservation groups, the Miramichi Watershed Management Committee, and Gulf Region DFO officials.

The members of the Committee would like to extend their sincere thanks to all the witnesses who participated in this study. The Committee is pleased to present the results of its study in this report, along with recommendations based on the evidence it heard.

BACKGROUND

Figure 1—Striped Bass (Morone saxatilis)

Illustration: Jeffrey Domm

Source: Fisheries and Oceans Canada, *Striped Bass (Southern Gulf of St. Lawrence Population)*, 19 December 2016.

Striped bass is an anadromous fish (i.e., adult fish live in the sea and migrate into fresh water to spawn) that is found throughout the coastal areas of eastern North America from northern Florida to the St. Lawrence River. The largest populations of striped bass exist in the centre of their range in the Chesapeake Bay in Maryland and in New York's Hudson River. In Canada, there are three distinct populations of striped bass: the St. Lawrence River, the Southern Gulf of St. Lawrence (SGSL) and the Bay of Fundy. The St. Lawrence River population was considered extirpated (i.e. locally extinct), but with efforts being undertaken to reintroduce the population, it is now listed as endangered by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). The Bay of Fundy population is largely confined to spawning in the Shubenacadie River in Nova Scotia and the Saint John River in New Brunswick.[4]

The SGSL population (which was the focus of this study) spawns exclusively in the Northwest Miramichi River in New Brunswick, usually in early to mid-June. Mark Hambrook of the Miramichi Salmon Association informed the Committee that a possible second spawning location was found on the Southwest Miramichi River.[5] As shown in Figure 2, the SGSL population of striped bass is a coastal species which largely limits itself to a 10-km band off the coast between the Gaspé Peninsula and Cape Breton Island, as well as off the coast of Prince Edward Island.[6] Witnesses informed the Committee that the striped bass was also found "prospecting" in Labrador.[7]

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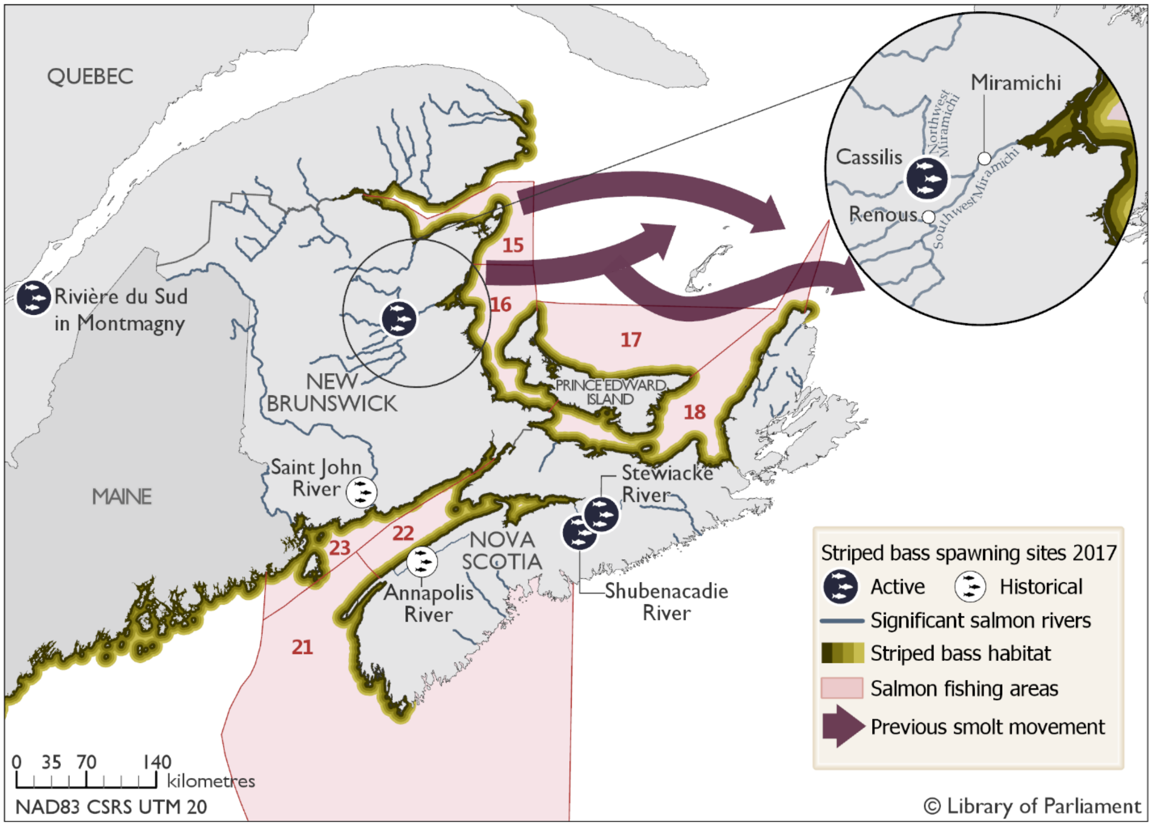
[4] Committee on the Status of Endangered Wildlife in Canada, *COSEWIC Assessment and Status Report on the Striped Bass (*Morone saxatilis*)*, 2012.

[5] Mark Hambrook, President, Miramichi Salmon Association Inc., *Evidence*, 1 April 2019.

[6] Committee on the Status of Endangered Wildlife in Canada, *COSEWIC Assessment and Status Report on the Striped Bass (*Morone saxatilis*)*, 2012.

[7] "It would seem that they collectively decided to go out and prospect to see if there were other places to live." Doug Bliss, Regional Director, Science, Gulf Region, Fisheries and Oceans Canada, *Evidence*, 1 April 2019.

Figure 2—Striped Bass Habitat and Salmon Fishing Areas



Source: Map prepared by Library of Parliament, Ottawa, 2019, using data from Natural Resources Canada (NRCan), *Atlas of Canada National Scale Data 1:5M Series*, "Boundary Polygons," Ottawa, NRCan, 2013; Committee on the Status of Endangered Wildlife in Canada (COSEWIC), *Assessment and Status Report on the Atlantic Salmon in Canada*, Ottawa, Species at Risk Public Registry, 2011; COSEWIC, *Assessment and Status Report on the Striped Bass Morone saxatilis in Canada*, Ottawa, Species at Risk Public Registry, 2012; *Administrative boundaries in Canada – CanVec Series*, "Administrative features," 1:1M, Ottawa, NRCan, 2018; *Lakes, rivers and glaciers in Canada – CanVec Series, "*Hydrographic features*,"* 1:1M, Ottawa, NRCan, 2018; Statistics Canada, *Drainage Regions of Canada*, 2017; Fisheries and Oceans Canada (DFO), *Spawner Abundance and Biological Characteristics of Striped Bass (Morone saxatilis) in the Southern Gulf of St. Lawrence in 2017*, DFO Canadian Science Advisory Secretariat Science Response 2018/016; Atlantic Fishery Regulations, 1985, SOR/86-21; DFO, *Aquatic Species at Risk*, accessed January 2019. The following software was used: Esri, ArcGIS Pro, version 2.1.0. Contains information licensed under Open Government Licence – Canada and Statistics Canada Open Licence Agreement.

The SGSL striped bass population has historically been exploited by commercial and recreational fishers, as well as by First Nations. According to COSEWIC, the spawning population decreased to fewer than 5,000 individuals in the late 1990s, due to factors which include: climatic constraints, overfishing, illegal fishing, bycatch and the presence of contaminants.[8] These threats led to the closure of the commercial fishery in 1996, and the closure of the recreational, and Aboriginal food, social and ceremonial (FSC) fisheries in 2000.[9] The FSC fishery was reopened in 2013.[10] In July 2018, the first commercial striped bass fishery since 1996 was opened, with the Eel Ground First Nation being licensed to fish for 25,000 individual fish in October 2018.[11]

In November 2004, COSEWIC assessed the population as "threatened," and reclassified it in 2012 as a species of "special concern."[12] According to Mark Hambrook, the striped bass remains on the COSEWIC list solely because there is only a single confirmed spawning location.[13]

The 2007 Recovery Potential Assessment conducted by DFO during the *Species at Risk Act* (SARA) listing decision process after the initial COSEWIC assessment, proposed a recovery target of 21,600 spawners within five of the next six consecutive years, and thereafter at least 31,200 spawners over the next three of six consecutive years to consider opening a directed fishery with catch parameters.[14]

After conducting consultations in 2013, as required under SARA, the Governor in Council declined to list the SGSL striped bass population as a species at risk under the Act, largely due to "significant socio-economic impacts on communities" that would follow closures of other coastal fisheries as a result. The Governor in Council pointed specifically to negative impacts on the rainbow smelt, gaspereau and American Eel fisheries.[15] The Governor in Council instead pointed to other measures to protect the population, including:

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[8] Committee on the Status of Endangered Wildlife in Canada, *COSEWIC Assessment and Status Report on the Striped Bass (*Morone saxatilis*)*, 2012.

[9] Committee on the Status of Endangered Wildlife in Canada, *COSEWIC Assessment and Status Report on the Striped Bass (*Morone saxatilis*)*, 2012.

[10] Fisheries and Oceans Canada [DFO], *Striped Bass (Southern Gulf of St. Lawrence Population)*, 19 December 2016.

[11] Hadeel Ibrahim, "First striped bass commercial fishery in 20 years goes ahead on Miramichi," *CBC News*, 3 September 2018.

[12] COSEWIC defines "threatened" as "a wildlife species that is likely to become endangered if nothing is done to reverse the factors leading to its extirpation or extinction," and "special concern" as "a wildlife species that may become threatened or endangered because of a combination of biological characteristics and identified threats." See, COSEWIC, *COSEWIC wildlife species status categories and definitions*.

[13] Mark Hambrook, President, Miramichi Salmon Association Inc., *Evidence*, 1 April 2019.

[14] DFO, Canadian Science Advisory Secretariat [CSAS], *Spawner Abundance and Biological Characteristics of Striped Bass (*Morone Saxatilis*) in the Southern Gulf of St. Lawrence in 2017*, March 2018; and DFO, CSAS, *Recovery Potential Assessment for the St. Lawrence Estuary, Southern Gulf of St. Lawrence and Bay of Fundy Striped Bass (*Morone saxatilis*) Populations*, May 2007.

[15] *List of Wildlife Species at Risk (Decisions Not to Add Certain Species) Order*, SI/2013-27, 7 March 2013, in *Canada Gazette*, Part II, Vol. 147, No. 7, 27 March 2013.

- increasing enforcement patrols;
- closing the spawning grounds of the Northwest Miramichi River to anglers annually between 1 May and 30 June;
- training harvesters on how to effectively handle and release striped bass caught as bycatch; and
- conducting further scientific studies to better understand the SGSL striped bass population and its distribution.[16]

In March 2018, DFO released findings that showed the SGSL striped bass population had increased to over 300,000 spawners in 2016, and that "the median of the estimated spawner abundance in 2017 was 994,000." However, it should be noted that although the median abundance was used, there was a large variance between the maximum and minimum abundance measures; varying between a statistically significant lower value of 486,600 (5th percentile) to the higher value of 2,063,000 (95th percentile).[17]

In March 2019, DFO released its updated spawner abundance findings, that showed that the 2018 striped bass spawner abundance declined to a median of 333,000 spawners. Again, there was a large variance between the maximum and minimum abundance estimates; varying between a statistically significant lower value of 154,000 (5th percentile) to the higher value of 623,000 (95th percentile). Figure 3 shows that since 2006, the number of spawners exceeded the 31,200 target set by DFO during more than the three of six years set forth as the benchmark for reopening the fishery, in addition to the 21,600 recovery target.[18]

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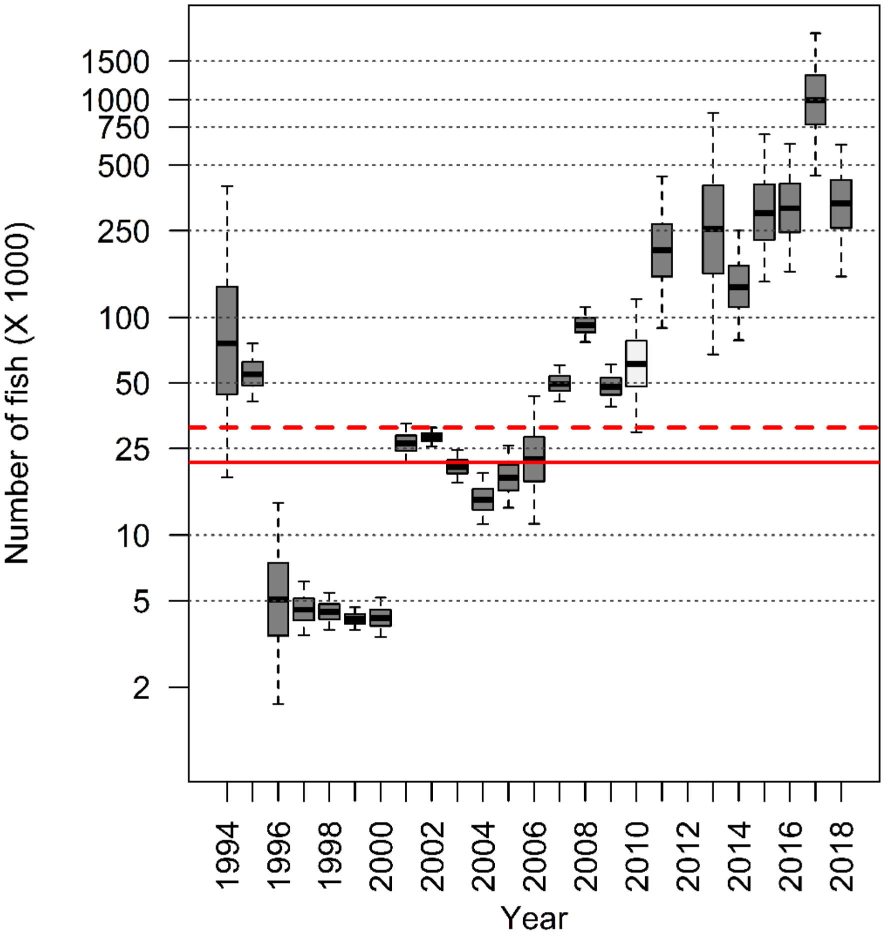
[16] *List of Wildlife Species at Risk (Decisions Not to Add Certain Species) Order*, SI/2013-27, 7 March 2013, in *Canada Gazette*, Part II, Vol. 147, No. 7, 27 March 2013.

[17] DFO, CSAS, *Spawner Abundance and Biological Characteristics of Striped Bass (*Morone Saxatilis*) in the Southern Gulf of St. Lawrence in 2017*, March 2018.

[18] DFO, CSAS, *Update of Spawner Abundance and Biological Characteristics of Striped Bass (Morone Saxatilis) in the Southern Gulf of St. Lawrence to 2018*, March 2019.

Figure 3—Estimated Abundances of Adult Striped Bass Spawners in the Northwest Miramichi Estuary Between 1994 and 2018[19]



Source: Fisheries and Oceans Canada, Canadian Science Advisory Secretariat, *Update of Spawner Abundance and Biological Characteristics of Striped Bass (Morone Saxatilis) in the Southern Gulf of St. Lawrence to 2018*, March 2019.

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[19] Box plots are interpreted as follows: dash is the median, boxes are the interquartile range, and the vertical dashes are the 5th to 95th percentile ranges. The solid and dashed horizontal lines show the recovery objectives defined in the Recovery Potential Assessment in support of the Species at Risk Act listing decision process.

WHAT THE COMMITTEE HEARD

Throughout the Committee's study, witnesses repeatedly raised a number of themes, in particular, the need:

- for evidence-based decision-making with respect to setting population reference points;[20]
- for a nimble, meaningful and engaged consultation process and co-management strategy for the species;
- for increased First Nation input and participation in the co-management of the species and the development of a viable commercial striped bass fishery;
- to properly protect the iconic wild Atlantic salmon from predators, including (but not limited to) the striped bass; and
- for further follow through and participation by DFO in response to stakeholder concerns, including First Nation communities, recreational and commercial fishers and this Committee.

Reference Points and Evidence-based Decision Making

The Committee heard different proposals for establishing an upper reference point for the striped bass population in the SGSL, but all emphasized that decisions on the issue should be based on the empirical data and evidence collected in the ecosystem.

Martin Mallet of the Maritime Fishermen's Union (MFU) noted that while there is a perception that a growing striped bass population is acting as a strain on the lobster fishery, he is waiting for "DFO data on that fact to try to dispel these myths."[21] According to Mr. Mallet, given the fluctuating populations of striped bass over the past few years, the "fragility of the stock" requires further ecological research.[22]

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[20] DFO defines a "limit reference point" as "the boundary between the cautious and critical zones. When a fish stock level falls below this point, there is a high probability that its productivity will be so impaired that serious harm will occur. The limit reference point is established based on the best available scientific information." See DFO, *A Fishery Decision-Making Framework Incorporating the Precautionary Approach*.

[21] Martin Mallet, Executive Director, Maritime Fishermen's Union, *Evidence*, 25 February 2019.

[22] Maritime Fishermen's Union, *Brief*, 25 February 2019.

Witnesses who represent the recreational catch and release Atlantic salmon fishery on the Miramichi River called on DFO to establish an upper limit reference point, but disagreed on what target should be the reference point. John Bagnall of the New Brunswick Salmon Council believed that the striped bass population should be brought down to less than 100,000 spawners.[23] He noted that the target of 300,000 spawners is far too high. Deborah Norton emphasized that she was not a biologist, but that developing an upper reference point is a priority.[24]

Bill Taylor, representing the Atlantic Salmon Federation (ASF), a conservation group, proposed both an upper and lower reference point, with an upper limit of 300,000 spawners and a lower limit of 31,200 spawners, representing the recovery target set by DFO in 2007.[25] Jeff Wilson, representing the Miramichi Striper Cup, however noted that 300,000 spawners should be the absolute minimum reference point.[26]

When DFO officials appeared before the Committee they confirmed that its proposed science plan is:

> …to provide biological reference points for the species; to examine or re-examine recurring questions about the species; such as striped bass diet measurements and assessing whether other spawning areas exist; and finally, to undertake focused studies to understand the environmental stresses and ecosystem dynamics influencing this and many other species.[27]

DFO did not, however, indicate what the reference points would be.

Even though opinions vary as to what should be used as the upper limit reference point for a sustainable striped bass population, the Committee recommends:

Recommendation 1

That Fisheries and Oceans Canada establish upper and lower limit reference point thresholds for striped bass and adjust them if necessary based on justifiable scientific evidence.

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[23] John Bagnall, Chair, Fisheries Committee, New Brunswick Salmon Council, *Evidence*, 1 April 2019.

[24] Deborah Norton, President, Miramichi Watershed Management Committee Inc., *Evidence*, 1 April 2019.

[25] Bill Taylor, President and Chief Executive Officer, Atlantic Salmon Federation, *Evidence*, 1 April 2019.

[26] Jeff Wilson, Co-host and Founder, Miramichi Striper Cup, *Evidence*, 25 February 2019.

[27] Doug Bliss, Regional Director, Science, Gulf Region, Fisheries and Oceans Canada, *Evidence*, 1 April 2019.

Recommendation 2

That Fisheries and Oceans Canada's restoration framework prioritize the long-term balance of fish species in the Southern Gulf of Saint Lawrence and Miramichi River.

In addition to reference points, methods of controlling the striped bass population in the SGSL include the First Nation commercial fishery and recreational bag limits. While Martin Mallet insisted that there has not been enough study on the population to justify a commercial striped bass fishery, witnesses generally approved of the Eel Ground First Nation's participation in the commercial fishery. Jeff Wilson, who was otherwise sceptical of controlling striped bass populations conceded that there is the notion of "First Nation's first right; they have to have a reasonable number."[28]

Jeff Wilson also called for lowering the recreational bag limit to one striped bass per day, and instead encourage a catch and release fishery, which he estimated has a positive economic impact of $3.8 million over the six-day period of the Striper Cup.[29] Bill Taylor however, was encouraged by DFO's 2018 increase in the daily bag limit.[30]

Slot limits were also discussed, with Jeff Wilson stating:

> In the Miramichi you have to keep a fish between 55 centimetres and 65 centimetres. That slot limit protects the big spawners. You're not allowed to kill big fish, that big female.… You're not going to kill your most fertile cow and keep all the little ones; you're just not going to do that. It's exactly the same with a fishery.[31]

By contrast, Bill Taylor called on DFO to eliminate the slot limit in the Eel Ground First Nation's commercial fishery and eliminate the upper slot size for the recreational fishery.[32]

Recommendation 3

That a mechanism for controlling the striped bass population include a First Nation commercial and food, socal and ceremonial fishery, and recreational public food catch and possession limits.

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[28] Jeff Wilson, Co-host and Founder, Miramichi Striper Cup, *Evidence*, 25 February 2019.

[29] Jeff Wilson, Co-host and Founder, Miramichi Striper Cup, *Evidence*, 25 February 2019.

[30] Bill Taylor, President and Chief Executive Officer, Atlantic Salmon Federation, *Evidence*, 1 April 2019.

[31] Jeff Wilson, Co-host and Founder, Miramichi Striper Cup, *Evidence*, 25 February 2019.

[32] Bill Taylor, President and Chief Executive Officer, Atlantic Salmon Federation, *Evidence*, 1 April 2019.

Recommendation 4

That Fisheries and Oceans Canada remove the maximum length restriction of 65 cm for the striped bass retention fishery, unless required for protection of spawners and population management.

When DFO officials appeared before the Committee, they pointed to several ongoing studies, some in collaboration with stakeholders such as the ASF, to determine population numbers of striped bass in the Miramichi River, as well as the consumption rate of salmon smolts by striped bass. DFO explained that it is adequately resourced, but that conducting scientific research "takes a village," stating:

> We're now working in broad collaboration with experts in the academic community and other communities to bring the resources to bear—mostly the intellectual capacity—to be able to tackle these questions.[33]

While some organizations such as the ASF believe that DFO adequately funds research on the species, other witnesses including Chief George Ginnish of the Eel Ground First Nation noted that his First Nation has been requesting funding for an Indigenous knowledge study as they continue to develop their commercial fishery.[34]

The Committee believes that more research is required in order fully understand the striped bass population of the SGSL to provide local residents, stakeholders and the Eel Ground First Nation with complete information and to determine future trends. The Committee also calls on DFO to continue to make the results of its studies available to Canadians. Therefore, the Committee makes the following recommendations:

Recommendation 5

That Fisheries and Oceans Canada invest more resources to obtain better data on the striped bass numbers and life cycle as well as how it is affected by climate change.

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[33] Doug Bliss, Regional Director, Science, Gulf Region, Fisheries and Oceans Canada, *Evidence*, 1 April 2019.

[34] Chief George Ginnish, Chief Executive Officer, North Shore Mi'gmaq District Council, Eel Ground First Nation, *Evidence*, 1 April 2019.

Recommendation 6

That Fisheries and Oceans Canada commit to transparent and timely publication of all research and data related to striped bass, including rationale for decisions made on striped bass management and other decisions regarding the Miramichi ecosystem, and that this information is made readily available to the public, and explained to all local stakeholders.

Consultation and Co-Management

The Committee heard calls for DFO to exercise a nimble and responsive approach to the management of the Miramichi River ecosystem, as it relates to the striped bass population. This approach should entail greater meaningful consultation with stakeholders, including First Nations, to facilitate the co-management of the ecosystem.

While some organizations, such as the Miramichi Striper Cup, have cooperative relationships with DFO, the Committee heard that too often, local and traditional knowledge has been ignored by DFO in their decision-making process. Chief George Ginnish expressed frustration with the lack of inclusion of Indigenous traditional knowledge in the management process noting:

> We would say that absolutely traditional knowledge is a requirement of management. We've been preaching co-management to DFO for many years. We're always promised to be part of that decision-making process. That hasn't happened.[35]

Jeff Wilson, who noted that he had a generally cooperative relationship with DFO, nevertheless explained that information on striped bass is insufficient, and that not enough information is shared with the advisory committees that he sits on.[36] Martin Mallet agreed but added that “a multi-stakeholder approach is important, but better listening to the science and better science should be number one.”[37]

Deborah Norton, whose organization has a memorandum of understanding with DFO and the Government of New Brunswick to co-manage the watershed, reiterated that she would still like to see greater participation in the process. She explained that at consultation sessions with DFO, organizations have only three minutes to present their positions, which she noted was less than the time allotted to opening statements at a parliamentary

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[35] Chief George Ginnish, Chief Executive Officer, North Shore Mi’gmaq District Council, Eel Ground First Nation, *Evidence*, 1 April 2019.

[36] Jeff Wilson, Co-host and Founder, Miramichi Striper Cup, *Evidence*, 25 February 2019.

[37] Martin Mallet, Executive Director, Maritime Fishermen’s Union, *Evidence*, 25 February 2019.

committee.[38] John Bagnall explained that his organization has proposed the development of a fisheries liaison committee to facilitate communication and consultation.[39]

DFO reiterated its own commitment to working with local communities to manage aquatic resources, with Serge Doucet, the Regional Director General for the Gulf Region stating that:

> Entire communities are built around those industries and they expect us at DFO to help protect and manage the resource. To that effect, one-third of our workforce is dedicated to science. Our scientists work in labs, conduct surveys in the field or do research on various species, marine protected areas or species at risk. Ongoing consultation and engagement with our partners from fishing communities, industry and first nations allow us to make the right decisions based on scientific data and facts.[40]

DFO also reiterated that consultations happen with at least a few weeks' notice and that the results of its recreational advisory councils are always published online. Mark Hambrook explained however, that when he used to work for DFO in Prince Edward Island, he communicated with local community fish and game clubs at least once a week, but now he only receives calls from the Gulf Region recreational fisheries coordinator every few months.[41]

Given the concerns raised by local stakeholders, including recreational fishers, watershed managers and the Eel Ground First Nation, the Committee recommends:

Recommendation 7

That traditional knowledge of First Nations, local stakeholders and stewards be part of the science-based decision-making through meaningful consultations, and that these consultations happen in their communities.

Recommendation 8

That Fisheries and Oceans Canada work to improve the timeliness of public notices regarding recreational angling of striped bass to ensure stakeholders and anglers are aware of the regulations as early as possible prior to the season opening.

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[38] Deborah Norton, President, Miramichi Watershed Management Committee Inc., *Evidence*, 1 April 2019.

[39] John Bagnall, Chair, Fisheries Committee, New Brunswick Salmon Council, *Evidence*, 1 April 2019.

[40] Serge Doucet, Regional Director General, Gulf Region, Fisheries and Oceans Canada, *Evidence*, 1 April 2019.

[41] Mark Hambrook, President, Miramichi Salmon Association Inc., *Evidence*, 1 April 2019.

First Nations Participation

The Committee heard from Chief George Ginnish about how his community is being excluded from the benefits provided by the increase in the striped bass population in the Miramichi River. He explained that when the striped bass population fell in the 1990s, the Eel Ground First Nation was asked to stop its FSC salmon fishery, and that the community complied voluntarily. He also shared how his community is among the poorest communities in New Brunswick, whose population lacks access to traditional foods such as striped bass and salmon. He stated that Eel Ground First Nation residents are “able to access … the equivalent [of] one tablespoon per day when we look at all the moose and fish we're able to access as a community.”[42]

The lack of access to salmon caused by the need to recover the striped bass population also led to decreased access to economic opportunities, opportunities that presented themselves with the Supreme Court of Canada’s decision in *R. v. Marshall*, which recognized a treaty right to fish commercially for a “moderate livelihood.”[43] Chief Ginnish explained that closure of the salmon fishery led to a loss of “millions of dollars of investments that [were] occurring through *Marshall* decision agreements.”[44]

Chief Ginnish expressed frustration with DFO’s treaty implementation process as well as the Government of New Brunswick’s engagement on the file. He explained that while the striped bass population increased, his First Nation was still forced to wait to receive a commercial licence, and only be allowed to retain 2,000 striped bass as part of the FSC fishery. He explained that:

> In New Brunswick, we have a trilateral treaty implementation table. That process has been ongoing for 12 years, and DFO has just come to the table within the last couple of years. We're very frustrated with the approach. It seems to be, “Let's drag this out; let's delay it; let's not really deal with the concerns of the people whose livelihood is that river.”[45]

In 2018, the first Indigenous commercial striped bass fishery was opened, with a total allowable catch (TAC) of 50,000 fish to be divided between a spring and fall fishing season. The TAC was allocated by DFO, with Chief Ginnish noting that he was not

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[42] Chief George Ginnish, Chief Executive Officer, North Shore Mi’gmaq District Council, Eel Ground First Nation, *Evidence*, 1 April 2019.

[43] *R. v. Marshall*, [1999] 3 S.C.R. 456 at para. 7.

[44] Chief George Ginnish, Chief Executive Officer, North Shore Mi’gmaq District Council, Eel Ground First Nation, *Evidence*, 1 April 2019.

[45] Chief George Ginnish, Chief Executive Officer, North Shore Mi’gmaq District Council, Eel Ground First Nation, *Evidence*, 1 April 2019.

"feeling the love" from DFO with respect to implementation of the principles of free, prior and informed consent. The licence for the first 25,000 fish arrived too late, and then combined with an early winter, the First Nation did not have much of an opportunity to exploit the TAC. Chief Ginnish called on DFO to allow the 50,000 TAC each spring to be exploited throughout the season.[46] The MFU however, recommended that DFO postpone the launch of a commercial striped bass fishery pending further study on the species.[47] The Committee recognizes the socio-economic importance that the striped bass fishery represents for the Eel Ground First Nation and recommends:

Recommendation 9

That Fisheries and Oceans Canada prioritize First Nations food, social and ceremonial fishery when determining the striped bass catch.

Recommendation 10

That the First Nation commercial fishery be given a full annual allotment of striped bass at the beginning of the spring fishing season.

Recommendation 11

That Fisheries and Oceans Canada ensure licences are granted on time to the First Nations to ensure they can legally operate a commercial fishery each year.

Predator Control and Protection of Wild Atlantic Salmon

The Committee heard that the increased striped bass population has led to pressure on the recovery of wild Atlantic salmon, particularly the survival of salmon smolts in the Miramichi estuary.

Bill Taylor of the ASF cited an ASF peer reviewed smolt tracking study over the last 14 years, which found that on the Restigouche and Cascapédia Rivers, smolt survival was relatively stable, ranging between 70% and 95%, while on the Miramichi River, the survival rate plummeted from around 70% to 8% between 2010 and 2017, which coincides with the increase in the striped bass populations.[48] Another ASF study, using

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[46] Chief George Ginnish, Chief Executive Officer, North Shore Mi'gmaq District Council, Eel Ground First Nation, *Evidence*, 1 April 2019.

[47] Maritime Fishermen's Union, *Brief*, 25 February 2019.

[48] Bill Taylor, President and Chief Executive Officer, Atlantic Salmon Federation, *Evidence*, 1 April 2019.

acoustic telemetry, however, puts the predation rate of smolts by striped bass in the Miramichi River at between 2% and 20%.[49]

All witnesses agreed that striped bass are a carnivorous fish and an opportunistic predator, but the level of concern regarding predation on wild Atlantic salmon varied. The MFU stated that the striped bass is not a concern for commercial fishers in the Gulf region but recommended that DFO continue to study the effect of striped bass population increases on smelt and alewife stocks.[50] Bill Taylor of the ASF expressed deep concern for wild Atlantic salmon however, noting that:

> Salmon populations throughout the north Atlantic face challenges, but predation by striped bass is by far the biggest threat right now. There are general declines, but if you look at the Miramichi compared with the Restigouche and Gaspé rivers, the north shore rivers or the Labrador and Newfoundland rivers, there is a general decline, but nowhere is that decline more pronounced than on the Miramichi.[51]

In January 2017, the Committee released a unanimous report entitled *Wild Atlantic Salmon in Eastern Canada* and made two recommendations directly related to striped bass predation, notably that DFO "allow a significant increase in the harvest of striped bass by the recreational fishery by lengthening the retention season and increasing catch limits, where striped bass populations warrant it," and "investigate the opportunity for a First Nations striped bass commercial fishery."[52]

While DFO states that both recommendations were implemented, the Committee heard that stakeholders disagree. Deborah Norton stated that "to my knowledge none of the recommendations have gone anywhere."[53] Chief George Ginnish asserted that "good recommendations haven't been implemented."[54] Bill Taylor remarked that there has been "some action on some of the recommendations, but far too few," especially with respect to the striped bass. He went on to state that:

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[49] Jason Daniels et al., "Estimating consumption rate of Atlantic salmon smolts (*Salmo salar*) by striped bass (*Morone saxatilis*) in the Miramichi estuary using acoustic telemetry," *Canadian Journal of Fisheries and Aquatic Science*, Vol. 75, 1811-1822, 2018.

[50] Maritime Fishermen's Union, *Brief*, 25 February 2019.

[51] Bill Taylor, President and Chief Executive Officer, Atlantic Salmon Federation, *Evidence*, 1 April 2019.

[52] FOPO, *Wild Atlantic Salmon in Eastern Canada*, Fifth Report, 1st Session, 42nd Parliament, January 2017.

[53] Deborah Norton, President, Miramichi Watershed Management Committee Inc., *Evidence*, 1 April 2019.

[54] Chief George Ginnish, Chief Executive Officer, North Shore Mi'gmaq District Council, Eel Ground First Nation, *Evidence*, 1 April 2019.

> If the Department of Fisheries and Oceans had the resources and the will to implement all 19 recommendations, that would go a long way to at least slowing the salmon's decline and hopefully beginning the recovery process.[55]

The Committee agrees with the witnesses who appeared before it and calls on DFO to fully implement the recommendations set out in its report on wild Atlantic salmon. Unfortunately, the Minister of Fisheries, Oceans and the Canadian Coast Guard was unavailable to appear before it on this study but the Committee looks forward to engaging with the Minister going forward with respect to the Government's responses to its recommendations.

While the increased striped bass population was the focus of the Committee's study, concerns were also expressed about other aquatic predators including the smallmouth bass, an invasive species in Miramichi Lake, which threatens to enter the Miramichi River.[56] The Committee recognizes that the introduction of smallmouth bass would be catastrophic for salmon parr and encourages DFO to approve the use of rotenone, a broad spectrum pesticide, to safely eradicate the species from Miramichi Lake. The Committee further reiterates its previous recommendation that DFO support a grey seal harvest program to reduce the threat of predation by grey seals on wild Atlantic salmon in the SGSL and Miramichi estuary.[57]

Recommendation 12

That other predators of Atlantic salmon be managed; specifically, the eradication of the invasive smallmouth bass from Miramichi Lake using the rotenone pesticide, and a sustainable harvest of the grey seal.

CONCLUSIONS

During its study, the Committee heard from a wide range of witnesses representing diverse interests and viewpoints. All witnesses agreed, however, that an ecosystem approach represents the way forward in addressing the challenges and opportunities presented by the recovered striped bass population in the SGSL and Miramichi River.

The Committee heard that DFO should engage affected communities to develop approaches that recognize the socio-economic importance of balancing a healthy striped bass population and the recovery of the iconic wild Atlantic salmon. Working with

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[55] Bill Taylor, President and Chief Executive Officer, Atlantic Salmon Federation, *Evidence*, 1 April 2019.

[56] Mark Hambrook, President, Miramichi Salmon Association Inc., *Evidence*, 1 April 2019.

[57] FOPO, *Wild Atlantic Salmon in Eastern Canada*, Fifth Report, 1st Session, 42nd Parliament, January 2017.

conservation groups, commercial and recreational fishers, and the Eel Ground First Nation represents a unique opportunity for DFO to promote economic development and tourism, rebuild native fish species and advance reconciliation. The Committee looks forward to working with DFO to achieve these goals.

APPENDIX A
LIST OF WITNESSES

The following table lists the witnesses who appeared before the Committee at its meetings related to this report. Transcripts of all public meetings related to this report are available on the Committee's webpage for this study.

Organizations and Individuals	Date	Meeting
Maritime Fishermen's Union	2019/02/25	133
Martin Mallet, Executive Director		
Miramichi Striper Cup	2019/02/25	133
Jeff Wilson, Co-host and Founder		
Atlantic Salmon Federation	2019/04/01	137
Bill Taylor, President and Chief Executive Officer		
Department of Fisheries and Oceans	2019/04/01	137
Doug Bliss, Regional Director Science, Gulf Region		
Serge Doucet, Regional Director General Gulf Region		
Eel Ground First Nation	2019/04/01	137
Chief George H. Ginnish, Chief Executive Officer North Shore Mi'gmaq District Council		
Miramichi Salmon Association Inc.	2019/04/01	137
Mark Hambrook, President		
Miramichi Watershed Management Committee Inc.	2019/04/01	137
Deborah Norton, President		
New Brunswick Salmon Council	2019/04/01	137
John Bagnall, Chair Fisheries Committee		
John Pugh, President		

APPENDIX B
LIST OF BRIEFS

The following is an alphabetical list of organizations and individuals who submitted briefs to the Committee related to this report. For more information, please consult the Committee’s webpage for this study.

Maritime Fishermen's Union

New Brunswick Salmon Council

REQUEST FOR GOVERNMENT RESPONSE

Pursuant to Standing Order 109, the Committee requests that the government table a comprehensive response to this Report.

A copy of the relevant *Minutes of Proceedings* (Meetings Nos. 133, 137, 142 and 146) is tabled.

Respectfully submitted,

Ken McDonald
Chair

Jonsson, N., B. Jonsson, And L.P. Hansen. 1998. The relative role of density-dependent and density-independent survival in the life cycle of Atlantic salmon Salmo salar. Journal of Animal Ecology 0887\56\ 640_651_ 0887 British Ecological Society 640.

1. Density-dependent factors appeared important for the survival of juvenile Atlantic salmon in the River Imsa whilst density-independent factors were more important for the older fish at sea. In fresh water, density dependence was indicated by a stock-recruitment relationship with increasing loss-rates from eggs to smolts and from eggs to adults as egg density increased. 73% of the loss-rates were explained by variation in egg density. At sea density independence was indicated by the lack of a significant relationship between loss-rates and smolt densities.

5. Total adult biomass (adults caught at sea and in rivers) and the returning adults to the River Imsa in mass or energy were correlated with the size of the smolt cohort from which they originated. Yearly total adult biomass ranged between 139 and 2600 kg per 10,000 m^2 when the number of smolts ranged from 286 to 1640 respectively. The biomass of adults returning to the River Imsa was between 48 and 503 kg produced from between 561 and 0510 smolts.

Gibson, A.J.F. 2006. Population Regulation in Eastern Canadian Atlantic salmon (Salmo salar) populations. Research Document 2006/016

In the marine environment, density dependence was potentially detected in three of the 15 return-rate data series for salmon maturing after one winter at sea, but was not detected in any of the nine return-rate data series for fish maturing after two winters at sea.

Density Dependence in the Marine Environment

Density dependence in Atlantic salmon populations in the marine environment is relatively unstudied. Here, we apply the approach used above to the smolt-to-adult returns data from 15 populations (Table 5) to determine whether density dependence can be detected in the marine environment. Three models (density independent, Beverton- Holt and Ricker) were fit to data for the smolt-to-1SW return data and smolt-to-2SW return data individually. As such, the statistical comparisons do not distinguish between survival rates and age-at-maturity when testing for density dependence.

Results

Of the 15 smolt-to-1SW comparisons (Figure 5), density dependence was potentially detected in three populations: Campbellton, NE Trepassey, and St. Jean. However, both the Campbellton and NE Trepassey models produced biologically impossible parameter estimates (Table 6) from the Beverton-Holt model, although not so from the Ricker. For the other 12 populations, the density-dependent models produced infinite estimates of the carrying capacity in five cases, such that the fits were virtually identical to the density-independent models, a result that strengthens conclusions about the lack of density dependence in the marine environment for these populations.

Density dependence was not detected in any of the nine smolt-to-2SW returns (Figure 6). Similar to the 1SW results, five of the nine density-dependent models produced infinite estimates of the carrying capacity (Table 7), again strong evidence against density dependence in the marine environment

As outlined by Jonsson and Jonsson (2004), most salmon fisheries theory assumes that the mortality of salmon in the ocean is density-independent, a rationale based on the idea that the population density is far below the assumed carrying capacity for salmon in that habitat. However, other density-dependent effects are possible, such as density-dependent predation on migrating smolt in estuaries or adults prior to upstream migration for spawning. Beverton's (1995) concentration hypothesis states that the potential for density dependence should be greatest when organisms are most concentrated, which is potentially during migration near the mouth of the river for salmon in the marine environment.

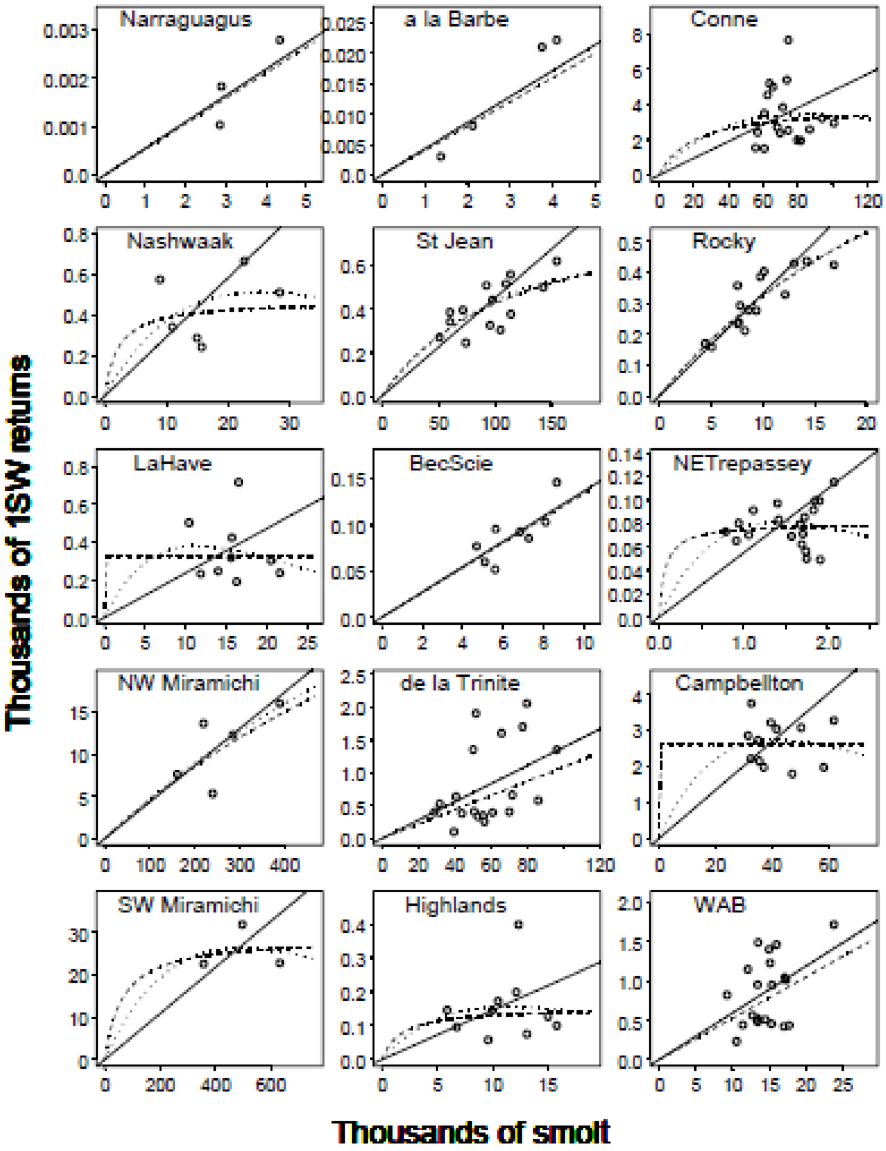

Figure 5. Observed (points) and predicted (lines) densities of Atlantic salmon obtained by fitting three models to the smolt-to-1SW spawner data. The data are the observed abundance or density within a cohort by age. The solid line is a one-parameter model that shows the fit obtained based on the assumption that survival is density independent. The dashed and dotted lines show the fits obtained from two-parameter Beverton-Holt and Ricker models respectively. Parameter estimates and statistical comparisons of the fits are provided in Table 6.

From Brian Dempson, pers. comm.

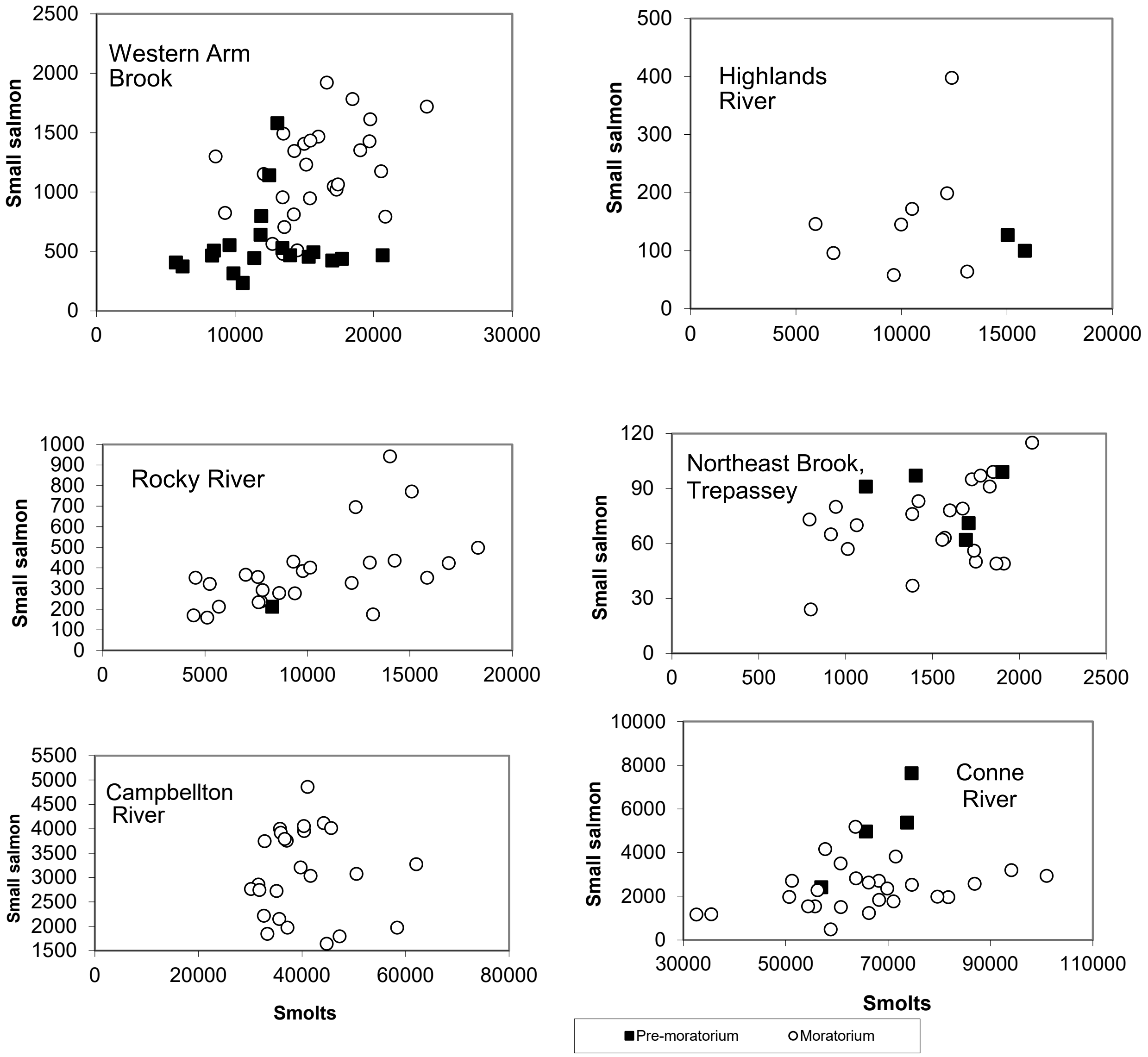

Figure xx. Relationship between smolt production and subsequent return of adult small salmon from various Newfoundland rivers. Moratorium years (1992 to 2018) are shown separately