NOVA SCOTIA AQUACULTURE REVIEW BOARD

IN THE MATTER OF: *Fisheries and Coastal Resources Act*, SNS 1996, c 25

- and -

IN THE MATTER OF: An Application by Kelly Cove Salmon Ltd. for a boundary amendment to Marine Finfish Licence and Lease AQ#1039

Affidavit of Andrew Swanson, PhD affirmed on May 3, 2021

I, Andrew Swanson of Halifax, Nova Scotia, affirm and give evidence as follows.

1. I am the Vice President of Research and Development at Cooke Aquaculture Inc, the parent company of the Applicant in this proceeding, Kelly Cove Salmon Ltd, which is the Canadian Farming Division of Cooke Aquaculture Inc. ("**Kelly Cove Salmon**").

2. I have personal knowledge of the evidence affirmed in this affidavit except where otherwise stated to be based on information and belief.

3. I state, in this affidavit, the source of any information that is not based on my own personal knowledge, and I state my belief of the source.

Rebuttal Report

4. I have reviewed the report for the Nova Scotia Aquaculture Review Board prepared by Jonathan Carr, MSc and Stephen Sutton, PhD attached as Exhibit A to the affidavit of Jonathan W. Carr affirmed April 23, 2021 and filed in this proceeding (the "**Report**").

5. My response to the concerns contained in the Report as well as Gregory Heming's concerns raised in his application for intervenor status is attached as **Exhibit A**.

6. My CV was previously filed in this proceeding and is located at Exhibit E of Jeffery Nickerson's affidavit affirmed on April 26, 2021. I affirm that my CV is true and accurate.

AFFIRMED before me virtually in Halifax, Nova Scotia, on May 3, 2021.	
Sara D. Nicholson A Barrister of the Nova Scotia Supreme Court	**Andrew Swanson, PhD**

Boundary Amendment Application
AQ#1039

This is Exhibit A referred to in the Affidavit of Andrew Swanson, PhD, affirmed before me virtually on May 3, 2021.

Sara D. Nicholson
A Barrister of the Nova Scotia Supreme Court

Report for the Nova Scotia Aquaculture Review Board

With respect to the Application by Kelly Cove Salmon Ltd. for a boundary amendment to Marine Finfish Licence and Lease AQ#1039

Andrew Swanson, PhD of Cooke Aquaculture Inc

May 3, 2021

<u>Background</u>

1. We appreciate the opportunity to respond to the written concerns raised in the report for the Aquaculture Review Board prepared by Jonathan Carr, MSc and Stephen Sutton, PhD attached as Exhibit A to the affidavit of Jonathan W. Carr affirmed on April 23, 2021 and filed in this proceeding.

2. Mr. Carr and Dr. Sutton are employees of the Atlantic Salmon Federation ("**ASF**"). Their report is put forward on behalf of the intervenor Gregory Heming and relates to possible impacts on wild Atlantic salmon resulting from the application by Kelly Cove Salmon Ltd ("**KCS**") for a boundary amendment to marine finfish licence and lease AQ #1039 in the Annapolis Basin, Digby County.

3. Successful salmon farming requires both local permissions and more broadly, societal acceptance of scientifically justified benefits and risks. Given that this application process is assessing the permission of a boundary change simply to reflect the true location of the existing and otherwise compliant salmon farm, permitted to operate for the past 18 years, we have reasonably restricted our comments to the specific concerns regarding KCS' Rattling Beach farm, AQ #1039 (the "**Farm**").

4. We focused and responded to the following comments:

   (a) What impacts, if any, the Farm has had, or may pose to future wild Atlantic salmon recovery efforts?

   (b) What mitigation efforts has KCS taken, or could take, to assist local wild Atlantic salmon recovery efforts if the Farm is permitted to continue operations?

   (c) How "*any changes to the water in the Annapolis Basin will affect plant life and bird life in the marsh and in the rewilded area*."

Impacts

5. We acknowledge that no human activity, especially food production, is free of a "footprint" on the surrounding environment. Importantly, footprints often represent inefficiencies in the use of limited resources, typically all having real value. Within the highly regulated Atlantic Canada aquaculture industry, significant monetary penalties are also triggered for environmental non-compliance of scientifically based and accepted thresholds and limits. As such, successful companies like KCS, naturally reduce unnecessary inefficiencies by continuously monitoring compliance, routinely refining best practices, and by deploying leading aquaculture technologies.

6. Through a multitude of assessed parameters, informed and constantly updated by unbiased scientific research, Fisheries and Oceans Canada ("**DFO**") and Nova Scotia Department of Fisheries and Aquaculture ("**NSDFA**") management, scientists (including DFO, NSDFA and universities) and stakeholders, the KCS Farm (AQ#1039 Rattling Beach) has operated for 18 years, without any detected adverse effect to local wild Atlantic salmon, or to any recovery effort.

7. Further, using the best science available, and backed by nearly 50 years of global research into aquaculture's environmental impacts, including the studies cited by Mr. Carr and Dr. Sutton, we are of the opinion that there is no definitive proof of any lasting harm by modern aquaculture, much less in the Annapolis Basin.

8. Mr. Carr and Dr. Sutton identified some weak correlations of historic aquaculture methods with the decline in the health of wild salmon but, as centuries of standardized scientific dictums and methodology has shown, correlation is never proof of causality. Furthermore, in the absence of such proof, despite 50 years of directed examination of this question, scientific norms always default to the null hypothesis, which is no association, or more specifically, there is no proof of lasting adverse effect of salmon aquaculture upon wild Atlantic salmon in the Annapolis Basin created by this Farm's past or future operation.

9. We are not aware of any scientific study of modern aquaculture practices that proves a specific activity, or indirect consequence of operations that has had an negative effect on population dynamics of wild Atlantic salmon, or on-going efforts to recover salmon populations, locally in the Annapolis Basin, regionally in Bay of Fundy, or globally. In contrast, highly relevant to the primary concerns of Mr. Carr and Dr. Sutton concerning the status of the wild Atlantic salmon in this area is the current lack of good quality salmon habitats in the area. Although historically the region supported strong salmon populations, the adjoining Annapolis River and Bear River watersheds have been highly impacted through centuries of silting and runoff from forestry and intensive agricultural activities, acid rain, commercial fishery dredging, as well as substantial dams and sewer outfalls associated with significant human settlements along former salmon habitats.

10. DFO stock status reports that pre-date the KCS Farm indicated the dire condition of this stock. Importantly to the question poised, according to the DFO Science Stock Status Report D3-12 (1998) (**Tab 1**), the commercial fishery of salmon in the area, due to very low numbers, was closed in 1985. Due to continuing salmon declines, recreational and First Nations fisheries were eventually closed as well in 1990. The massive tidal power dam on the Annapolis River has clearly had and continues to have a tangible adverse effect on all anadromous fish. The recent 2019 DFO report by Gibson, Fulton & Harper titled *Fish mortality and its population-level impacts at the Annapolis tidal hydroelectric*

generating station, Annapolis Royal, Nova Scotia: a review of existing scientific literature (**Tab 2**) reported that the barrage and turbines presented an "extreme risk" to wild Atlantic salmon and all species of migrating fish into the Annapolis river.

11. This determination of wild salmon vulnerability occurred many years prior to the establishment of the Farm. The significant salmon declines and closures at best overlap the very early days of unscaled salmon aquaculture along the New Brunswick portion of the Bay of Fundy. The presence of the Farm clearly did not cause past salmon declines, and is unlikely to represent a hindrance to successful restoration of wild Atlantic salmon in the area. The restoration of wild Atlantic salmon in the area is controlled by far more important negative variables related to on-going agricultural and forestry habitat disturbances of remaining salmon breeding areas, as well as the tidal energy barrage facility.

<u>Mitigation Efforts and Going Forward</u>

12. As corporate stewards of the environment, and personally, as scientists, naturalists and recreational anglers, we share the concerns of Mr. Carr and Dr. Sutton with respect to the continuing losses of wild Atlantic salmon, despite decades of effort to identify the root cause(s) of massive losses, and to promote recovery of Atlantic Canadian salmon populations. This is a complex problem with no easy solutions yet identified. We applaud the perseverance of individuals and non-government organizations ("**NGOs**") such as the ASF to advocate and constructively contribute to the future of our regions remaining salmon rivers.

13. To this end, Cooke Aquaculture Inc ("**CAI**"), the parent company of KCS, is heavily engaged in multiple areas of marine research, including sponsorship of the Natural Sciences and Engineering Research Council-Cooke Industrial Research Chair in Sustainable Aquaculture held by Jon Grant, PhD at Dalhousie University. In addition, we support research programs at the Atlantic Veterinary College and have access to world-class epidemiologists to address fish health. We are at the forefront of modern aquaculture embodied by precision fish farming via the use of the latest ocean sensor technology, actively partnering in the development of these sensors.

14. Moreover, for the above reasons, we have partnered with First Nations, other NGO salmon conservation groups, universities, and governments, dedicating significant corporate resources and more than five years effort to use our skills and technologies to contribute to several Atlantic salmon recovery project rivers within Atlantic Canada (e.g. Big Salmon River, NB; Petitcodiac Watershed, NB; Miramichi River, NB; and Medway River, NS) and the Northeastern United States (e.g. East Branch Penobscot, ME). Given the remarkable and encouraging early successes observed in subsequent wild salmon returns, we are very keen to share this breakthrough, and offer this novel approach more broadly towards other highlighted salmon rivers of mutual concern.

15. With respect to the escape concerns from the Farm, the use of sterile fish is an approach we have utilized in some of our marine farms. It may represent an option for farms in certain circumstances. The use of sterile fish for this Farm may be counter-productive to the intents of Mr. Carr and Dr. Sutton. Triploid fish (i.e. sterile fish) are generally more prone to diseases and pests, require more medications and handling, and are not as

productive. Interesting, several NGOs, including the ASF in the past, have actively campaigned against and opposed aquaculture companies using this approach. Perhaps this strategy has evolved within their organization.

16. The most effective approach to preserving wild Atlantic salmon is through strong preventive measures including state-of-the-art net technology and rigorous farm health management plans. We take our role as environmental stewards seriously – the farming of healthy salmon depends on it.

17. Historically at the Farm, we have noted (and reported as required) a minor hole in one marine net, likely caused by a seal observed near the marine cage. This was reported to NSDFA as it was a potential escape event. The hole was immediately repaired and the subsequent harvest number did not detect any loss of Saint John River strain salmon and thus, presented no risk of a potential introgression with any remaining wild Annapolis Basin salmon. All marine nets in use at the Farm have been replaced with newer generation materials nets like Sapphire Ultra Core which has performed to specifications and has contributed to a zero-escape track record.

18. Sea lice in the Annapolis Basin farms have happily been remarkably low over the past 18 years, with only one single occasion necessitating the use of provincially and Canadian Food Inspection Agency ("**CFIA**") approved infeed sea lice treatment. As such, infeed treatments and our non-chemical sea lice treatments, while used routinely in adjacent New Brunswick farms, are not required at this time at the Rattling Beach Farm.

19. The KCS lumpfish based cleaner fish program is prophylactic and removes any sea lice from the marine cages continuously. As mentioned earlier, sea lice are not a problem at this Farm or elsewhere in the province. The introduction of lumpfish into the Farm, subject to regulatory approval, could provide a control for any early establishment of sea lice populations.

20. Although limited wild Atlantic salmon likely remain in the Annapolis Basin, the proposal by Mr. Carr and Dr. Sutton for monitoring of wild salmon by KCS for proximity impacts by potential farm parasites or diseases is legally prohibited. However, given that healthy salmon are the most productive, all KCS salmon are monitored routinely for presence of disease and pests, and in compliance with provincial codes, maintained disease and pest-free, thus presenting a minimal and acceptable risk to wild salmon.

21. There are regions of Atlantic Canada (for example, Labrador) thousands of kilometers from the nearest salmon farm that are often described as heavily infested with sea lice. Sea lice are a natural consequence of salmon abundance, and during historic periods of much higher Eastern Canada wild salmon numbers, sea lice numbers were likely similarly abundant.

22. In short, we maintain there is negligible risk of past or future impacts to wild Atlantic salmon posed by the application by KCS for a boundary amendment to marine finfish licence and lease AQ #1039 in the Annapolis Basin, Digby County.

Effect of the Farm on Mr. Heming's proposal to re-wild a portion of his property

23. Mr. Heming's property has been identified as PID 5098934, civic address 3852 Granville Road Port Royal, Nova Scotia. It is located approximately 14.46 kilometres from the Rattling Beach Farm in a straight line, not "2.5 km directly downstream from the aquaculture lease site #1039" as he represented in his application to become an intervenor.

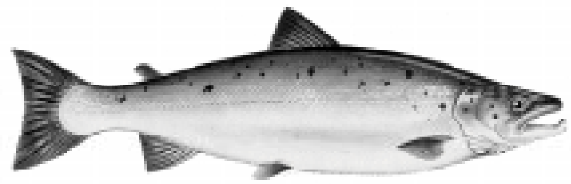
24. By water, the shore of his property is further distant than 14.46 km from the Farm. Any water flowing proximal to our Farm would become highly mixed during its turbulent passage on the upstream flood tide, blending daily with billions upon billions of liters of seawater from the Bay of Fundy, adjoining rivers, and the entire ~144 km^2 (35,583 acres) of the Annapolis Basin.

25. Mr. Heming's property abuts only a minute fraction of the Basin's total perimeter and thus, due to massive dilutions occurring daily, could not possibly be influenced by our Farm. Although not apparent from available imagery, any existing salt marsh grasses and/or rocky tidal communities on his property naturally require daily enrichment of tidal nutrients to be healthy and sustained. Given the incredible dilution effects influencing the Basin, any specific contribution of dissolved nutrients by a distant salmon farm would likely be undetectable, using any available modern technologies.

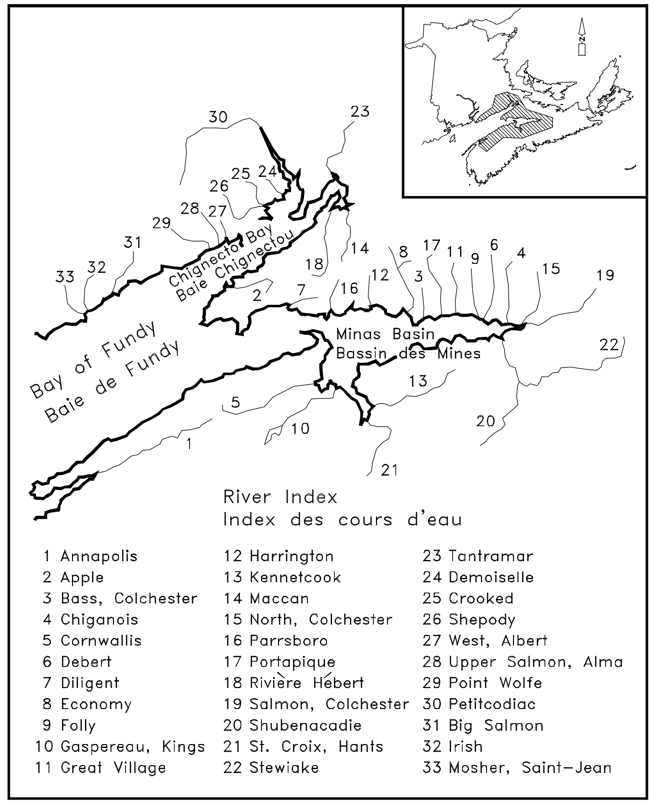
26. For instance, recently CAI sponsored research conducted by Dr. Grant, and other globally prominent co-authors (Howarth et al, 2019) (**Tab 3**) to use a very sensitive isotope signature methodology to determine the spatial zones of influence of our salmon farms in Liverpool Bay, Nova Scotia. Although quite specific to that area's environment and currents, the Liverpool Bay farm's influence, accumulated over 3 months, was shown to be limited to a maximum of 2.4km from the marine cages. It is therefore highly unlikely that the Farm over 14km away would have any effect whatsoever on Mr. Heming's property, salt marsh or any associated and reliant bird populations.

27. All upland vegetation, including the native trees and wild plant species Mr. Heming mentions, are severely damaged by any salt water contact, so this cannot possibly be a relevant concern.

28. In summary, upstream influences from massive on-going agricultural activity from cattle and plant crop farms along the Annapolis Valley, as well as interruptions of natural water flow caused by the NS Power's tidal barrage are certainly the far more significant impacts to the water quality adjacent to the shore portion of Mr. Heming's property than any conceivable influence from the distant Rattling Beach Farm.



Atlantic Salmon
Inner Bay of Fundy
SFA 22 & part of SFA 23



Background

Atlantic salmon (Salmo salar) utilize both freshwater and the ocean for their life cycle. Salmon spawn in freshwater, grow to smolts in two to three years, migrate to the sea, mature and return to their natal river to complete the cycle. Variations in duration spent at these stages occur among stocks and generations. Salmon of Inner Bay of Fundy usually enter rivers in the fall of the year, have a high proportion that return to spawn after one winter at sea, are not generally known to migrate to the North Atlantic Ocean, and have high survival between consecutive spawnings. Inner Bay of Fundy rivers share similarities in geography, biology and probably marine distribution. Inner Bay of Fundy stocks inhabit twenty-six rivers in Salmon Fishing Area (SFA) 22, Nova Scotia, and ten rivers in SFA 23, east of the Saint John River, New Brunswick. Two rivers, the Big Salmon River and the Stewiacke River account for more than half of the current production of salmon in inner Bay of Fundy rivers.

Two stocks, Annapolis and Gaspereau, are situated in SFA 22 but are different from the Inner Bay of Fundy stocks; they have a significant 2-sea-winter salmon component and migrate to the northwest Atlantic.

The Inner Bay of Fundy stocks have been in decline since 1986. Conservation requirements based on 2.4 eggs m^{-2} have not been met in any inner Bay of Fundy river since 1989. These rivers have been closed to all fishing for salmon since 1990.

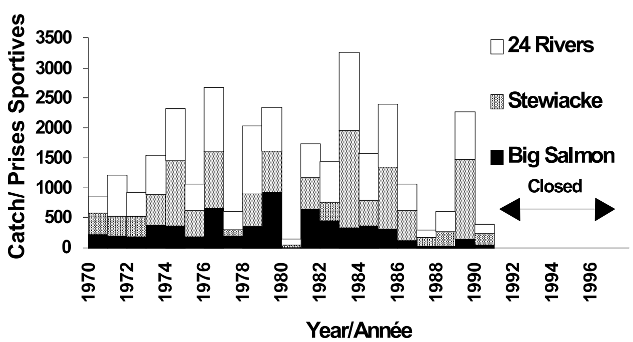
Summary

- All salmon fisheries have been closed since 1990.
- Juvenile salmon abundance has remained low since 1991.
- At least four years of recovery are needed before a fishery can be considered.

The Fishery

Inner Bay of Fundy rivers historically supported commercial and recreational salmon fisheries. Annual landings by commercial fisheries in the inner Bay of Fundy averaged 1,061 salmon during 1970-1984. The fishery was closed in 1985 and all licenses have since been retired.

Average annual recreational catches for 25 of 33 Inner Bay of Fundy rivers were 1,462 small salmon (< 63.0 cm) and 597 large salmon (≥ 63.0 cm) for 1970-1990. There have been no recreational or aboriginal fisheries since 1990, with the exception of the Gaspereau River which was opened to a catch-and-release angling fishery in 1997. Both the Gaspereau and the Annapolis are situated in SFA 22 but neither river has a stock typical of Inner Bay of Fundy stocks.



Resource Status

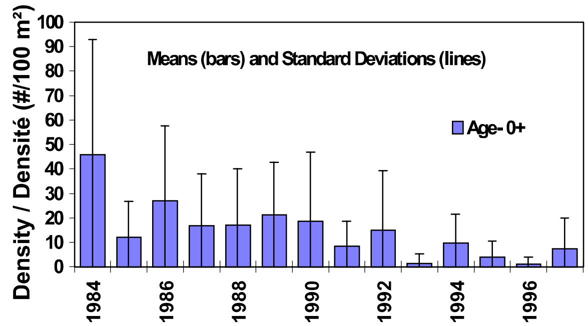
Inner Bay of Fundy Stocks

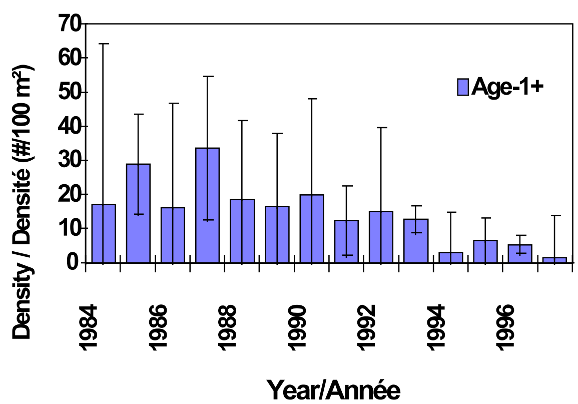
In general, conservation requirements have not been met since 1989 in any Inner Bay of Fundy river and current stock levels are very low. Rivers of Inner Bay of Fundy remain closed for conservation reasons until at least four years of good juvenile densities are measured. The status of the Inner Bay of Fundy salmon stocks is largely assessed on the basis of the performance of the Stewiacke and Big Salmon River stocks. This year's assessment is based on juvenile salmon abundances in the Stewiacke River and Big Salmon River.

Stewiacke River:

Densities of age-0^+ and age-1^+ salmon parr in the Stewiacke River have declined significantly since 1990 and are currently very low.

RivièreStewiacke/Stewiacke River



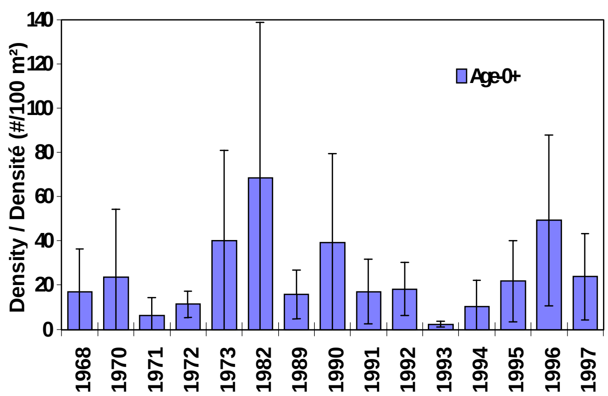


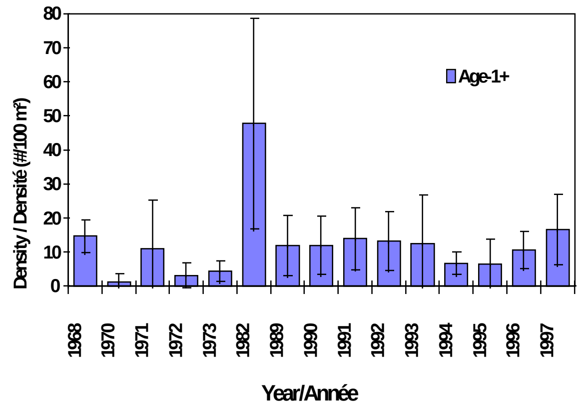
The stocking of hatchery smolts has been ineffective for enhancing the Stewiacke salmon stock. Their failure to return in significant numbers is further evidence that the Inner Bay of Fundy stocks are experiencing low marine survival.

Big Salmon River: Approximately 50 small and large salmon were estimated from visual counts to have returned to Big Salmon River in 1997 (New Brunswick Department of Natural Resources and Energy). This return corresponds to <10% of the conservation requirement of 700 salmon.

Juvenile densities in the Big Salmon River do not show the same dramatic decrease noted in the Stewiacke River. The most notable difference between the rivers is the higher density of age-0^+ parr in the Big Salmon River in 1995 and 1996. This is assumed to result from releases into the river of mature adult salmon of native stock origin grown in sea cages. More than 200 adult salmon were released in each of 1994 and 1995 by the Big Salmon River Association and the New Brunswick Department of Natural Resources and Energy. The spawning success of these cage-grown adults was confirmed by the presence of age-0^+ parr above a natural obstruction to passage where a small number of the cage grown adults were stocked in 1994.

Rivière Big Salmon/Big Salmon River





Other Inner Bay of Fundy Rivers: Electrofishing at nine sites in six other Inner Bay of Fundy rivers in 1997 confirmed that juvenile densities were low throughout the area. Densities of age-0^+ parr were low and densities of older parr were below historic values. Trends in densities in these six rivers follow the pattern observed in the Stewiacke River.

<u>Non-Inner Bay of Fundy Stocks</u>

Only the Annapolis and Gaspereau rivers stocks fall under this grouping. Information to assess these stocks are limited to data acquired during collections of broodstock for the hatchery program and counting of adult salmon in the fishway in the Gaspereau River.

Annapolis River: Seining operations carried out during the past several years to collect broodstock for the hatchery program indicate stock abundance is low.

Gaspereau River: Total returns of salmon through the fishway at White Rock were 98 fish in 1997. This return, comprising 35% hatchery fish, equaled 71% of the conservation requirement for the river.

Outlook

Salmon returns to all Inner Bay of Fundy rivers are extremely low as a result of abnormally low sea survival by 9 of the last 11 smolt classes (the exceptions being the 1988 and 1990 smolt classes). Low parr abundances indicate that there is little chance for these stocks to recover within the next four years.

Neither the Gaspereau River nor the Annapolis River salmon stock are expected to meet conservation requirements in 1998 because of low marine survival that these and other distant migrating stocks are experiencing. Expectations of returns equal or greater than conservation requirements are low even with hatchery returns included. There is little chance that the Annapolis River stock returns will exceed conservation requirements within the next ten years. Fish passage constraints, river acidification and agricultural practices also affect this stock.

Management Considerations

Marine survival continues to be low and there is no indication when conditions may change. As well, juvenile densities are low which will delay recovery if marine survival improves. No exploitation should be considered until stocks rebuild. Recovery is expected to take at least a generation (4-5 years) under conditions of improved marine survivals and escapements.

Inner Bay of Fundy stock levels are reduced to the point that action should be considered to hedge against their extirpation.

References

Amiro, P.G. and E. M. Jefferson. MS 1998. Status of Atlantic salmon in Salmon Fishing Areas 22 and 23 for 1997, with emphasis on inner Bay of Fundy stocks. DFO CSAS Res. Doc. 98/40.

For more information

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Fish Mortality and its Population-Level Impacts at the Annapolis Tidal Hydroelectric Generating Station, Annapolis Royal, Nova Scotia: a Review of Existing Scientific Literature

A. Jamie F. Gibson, Samantha J. Fulton and Danni Harper

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2019

Canadian Technical Report of Fisheries and Aquatic Sciences 3305

Fisheries and Oceans Canada / Pêches et Océans Canada

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Canadian Technical Report of Fisheries and Aquatic Sciences

Technical reports contain scientific and technical information that contributes to existing knowledge but which is not normally appropriate for primary literature. Technical reports are directed primarily toward a worldwide audience and have an international distribution. No restriction is placed on subject matter and the series reflects the broad interests and policies of Fisheries and Oceans Canada, namely, fisheries and aquatic sciences.

Technical reports may be cited as full publications. The correct citation appears above the abstract of each report. Each report is abstracted in the data base *Aquatic Sciences and Fisheries Abstracts*.

Technical reports are produced regionally but are numbered nationally. Requests for individual reports will be filled by the issuing establishment listed on the front cover and title page.

Numbers 1-456 in this series were issued as Technical Reports of the Fisheries Research Board of Canada. Numbers 457-714 were issued as Department of the Environment, Fisheries and Marine Service, Research and Development Directorate Technical Reports. Numbers 715-924 were issued as Department of Fisheries and Environment, Fisheries and Marine Service Technical Reports. The current series name was changed with report number 925.

Rapport technique canadien des sciences halieutiques et aquatiques

Les rapports techniques contiennent des renseignements scientifiques et techniques qui constituent une contribution aux connaissances actuelles, mais qui ne sont pas normalement appropriés pour la publication dans un journal scientifique. Les rapports techniques sont destinés essentiellement à un public international et ils sont distribués à cet échelon. II n'y a aucune restriction quant au sujet; de fait, la série reflète la vaste gamme des intérêts et des politiques de Pêches et Océans Canada, c'est-à-dire les sciences halieutiques et aquatiques.

Les rapports techniques peuvent être cités comme des publications à part entière. Le titre exact figure au-dessus du résumé de chaque rapport. Les rapports techniques sont résumés dans la base de données *Résumés des sciences aquatiques et halieutiques.*

Les rapports techniques sont produits à l'échelon régional, mais numérotés à l'échelon national. Les demandes de rapports seront satisfaites par l'établissement auteur dont le nom figure sur la couverture et la page du titre.

Les numéros 1 à 456 de cette série ont été publiés à titre de Rapports techniques de l'Office des recherches sur les pêcheries du Canada. Les numéros 457 à 714 sont parus à titre de Rapports techniques de la Direction générale de la recherche et du développement, Service des pêches et de la mer, ministère de l'Environnement. Les numéros 715 à 924 ont été publiés à titre de Rapports techniques du Service des pêches et de la mer, ministère des Pêches et de l'Environnement. Le nom actuel de la série a été établi lors de la parution du numéro 925.

Canadian Technical Report of
Fisheries and Aquatic Sciences 3305

2019

Fish Mortality and its Population-Level Impacts at the Annapolis Tidal Hydroelectric Generating Station, Annapolis Royal, Nova Scotia: a Review of Existing Scientific Literature

By

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ABSTRACT

Gibson, A.J.F., Fulton, S.J., and Harper, D. 2019. Fish mortality and its population-level impacts at the Annapolis Tidal Hydroelectric Generating Station, Annapolis Royal, Nova Scotia: a review of existing scientific literature. Can. Tech. Rep. Fish. Aquat. Sci. 3305: vi + 90 p.

The purpose of this document is to provide a summary and evaluation of the scientific literature pertaining to the mortality of fish at the Annapolis Tidal Generating Station (Annapolis TiGS) and the population-level impacts associated with this mortality. Information is available about fish passage, turbine mortality rates, population-level changes and diversion systems for several species at the Annapolis TiGS, although the information is dated and contains a high level of uncertainty. The most recent field study at the TiGS identified during this review occurred in 1999. While there is a diverse fish community present at the Annapolis TiGS, most studies have focused on *Alosa spp.* and the majority of information for other species is limited to small fish amenable to capture in ichthyoplankton nets.

There is evidence that mortality of fish occurs at the Annapolis TiGS. This evidence includes: anecdotal reports of dead fish (with injuries consistent with turbine passage) in the vicinity of the Annapolis TiGS, observations by SCUBA divers of dead fish in the turbine tailrace, and in-situ research about survival of fish passing through the turbine. There is limited information for estimating population-level impacts of the Annapolis TiGS for any species. Data gaps include: the proportion of populations that would be expected to encounter the turbine, rates of fishway usage, survival of fish of all life stages that pass through the turbine, the expected number of passes through the turbine, and rates of other human-induced mortality. Methods for studying fish behavior and survival have improved significantly since the studies were undertaken at the Annapolis TiGS. For species and life stages large enough to carry acoustic tags, advances with this technology afford the opportunity to address many of the data gaps. For species and life stages that are too small to carry acoustic tags, research to better understand and improve capture and handling methods, as well as capture efficiency, would lead to a better understanding of the impacts of the Annapolis TiGS.

RÉSUMÉ

Gibson, A.J.F., Fulton, S.J., and Harper, D. 2019. Fish mortality and its population-level impacts at the Annapolis Tidal Hydroelectric Generating Station, Annapolis Royal, Nova Scotia: a review of existing scientific literature. Can. Tech. Rep. Fish. Aquat. Sci. 3305: vi + 90 p.

L'objet de ce document est de fournir un sommaire et une évaluation de la documentation scientifique sur la mortalité des poissons à la centrale marémotrice d'Annapolis (centrale d'Annapolis) et les répercussions de cette mortalité à l'échelle de la population. On dispose de renseignements sur le passage des poissons, les taux de mortalité provoqués par la turbine, les changements à l'échelle de la population et les systèmes de dérivation pour plusieurs espèces à la centrale d'Annapolis, bien que les renseignements soient datés et comportent un niveau élevé d'incertitude. La plus récente étude sur le terrain réalisée à la centrale et relevée au cours de cet examen remonte à 1999. Bien qu'il existe une communauté de poissons diversifiée à la centrale d'Annapolis, la plupart des études se sont concentrées sur *Alosa spp.* et la majorité des renseignements sur les autres espèces se limitent aux petits poissons pouvant être capturés dans des filets à ichthyoplancton.

Il existe de la preuve de la mortalité de poissons à la centrale d'Annapolis, notamment des rapports anecdotiques de poissons morts (avec des blessures compatibles avec le passage dans la turbine) à proximité de la centrale d'Annapolis, des observations de poissons morts dans le canal de fuite de la turbine par des plongeurs et des recherches sur place sur la survie des poissons franchissant la turbine. On dispose de peu de renseignements pour estimer les répercussions de la centrale marémotrice d'Annapolis sur la population d'une espèce donnée. Les données manquantes comprennent : la proportion des populations susceptible d'arriver à la turbine, les taux d'utilisation des passes migratoires, la survie des poissons, à tous les stades biologiques, qui franchissent la turbine, le nombre prévu de passages dans la turbine, et les taux de mortalité d'origine anthropique. Les méthodes d'étude du comportement et de la survie des poissons se sont considérablement améliorées depuis celles qui ont été entreprises à la centrale d'Annapolis. Pour les espèces et les stades biologiques suffisamment grands pour porter des étiquettes acoustiques, les progrès réalisés grâce à cette technologie permettent de combler bon nombre des lacunes dans les données. Pour les espèces et les stades biologiques qui sont trop petits pour porter des étiquettes acoustiques, des recherches visant à mieux comprendre et à améliorer les méthodes de capture et de manipulation, ainsi que l'efficacité de la capture, permettraient de mieux comprendre les répercussions de la centrale d'Annapolis.

INTRODUCTION

The Annapolis River estuary is home to Canada's first tidal hydroelectric generating station, located in Annapolis Royal, Nova Scotia, Canada. The Annapolis Tidal Generating Station (Annapolis TiGS) was constructed to test the feasibility of using the STRAFLO^{TM} turbine in marine environments in anticipation of larger scale tidal generating development in the upper Bay of Fundy and has been in operation since 1985. Several scientific studies have been undertaken to provide information about fish-related impacts of the Annapolis TiGS, including fishway utilization studies, fish mortality studies and studies of changes in life- history characteristics of fish that use this estuary. This review of the scientific literature about these impacts was undertaken to provide information to inform regulatory decisions under Canada's Fisheries Act as well as prohibitions under Canada's Species at Risk Act. In this document, the focus is on the mortality of fish associated with passage at the Annapolis TiGS and its direct impacts on the affected population. Indirect effects such as changes in habitat quality and distribution, or prey abundance resulting from the construction and operation of the station are not addressed in this review. Additionally, changes to the Annapolis River and its estuary resulting from the construction of the causeway between Granville Ferry and Annapolis Royal where the Annapolis TiGS is located, which occurred in the early 1960's, are outside the scope of this review. Specifically, the following eight terms of references (TOR) are addressed in this document:

1. Provide a description of the Annapolis TiGS including its size and capacity, its operation, fish passage facilities, etc., sufficient as background to the literature review.
2. Provide a description of the search effort to identify the relevant literature. Provide a list of publications and identify any that were not included in the review.
3. Describe the fish community in the vicinity of the TiGS. To the extent known, include information about their abundance, life history, life stages present in the vicinity of the TiGS, their susceptibility to turbine mortality, their resiliency to increased mortality rates, whether the fish are migratory, resident or transient, the months they are present in the vicinity of the TiGS, whether the Annapolis River and/or estuary supports a population of the species, their status, and whether the species directly or indirectly supports aboriginal, commercial or recreational fisheries.
4. Provide a summary and critical review of scientific literature pertaining to fish passage utilization for species identified under TOR 3.
5. Provide a summary and critical review of scientific literature pertaining to fish mortality studies undertaken at the TiGS. Include other evidence of mortality, as appropriate. Include specific mortality rates if available. Describe whether the studies fully characterize the population-level annual mortality rates (e.g. immediate versus delayed mortality; effects of multiple passes, etc.).
6. Provide a summary of scientific literature pertaining to population-level changes (such as changes to population size and/or structure) that have occurred coincident to the construction and operation of the TiGS. To the extent possible, determine if they are consistent with the mortality rates identified under TOR 5.
7. Provide a summary and critical review of fish diversion studies undertaken at the Annapolis TiGS. Include estimates of their effectiveness for the species identified under TOR 3.
8. For key species identified in TOR 3, provide recommendations for further research to address any information gaps identified in TORs 1-7.

REVIEW

TOR 1: DESCRIPTION OF THE ANNAPOLIS TIDAL GENERATING STATION

Facility Description

The Annapolis TiGS is located on the Annapolis River estuary in Annapolis Royal, Nova Scotia (Figure 1). In the early 1960's, a barrage was built between Granville Ferry and Annapolis Royal as a water control structure to allow for the marshland upriver to be used for agriculture without the risk of tidal flooding. The barrage allowed for the construction of a highway between the two sides of the estuary and consisted of a dam on the Granville Ferry side, sluice gates on the Annapolis Royal side, and Hog's Island in between. This construction transformed the estuary upriver of the causeway from a vertically homogenous estuary with about a 10 m tidal range (similar to many around the Bay of Fundy) to a highly stratified salt wedge estuary with a tidal range around 0.5 m (Daborn et al. 1979).

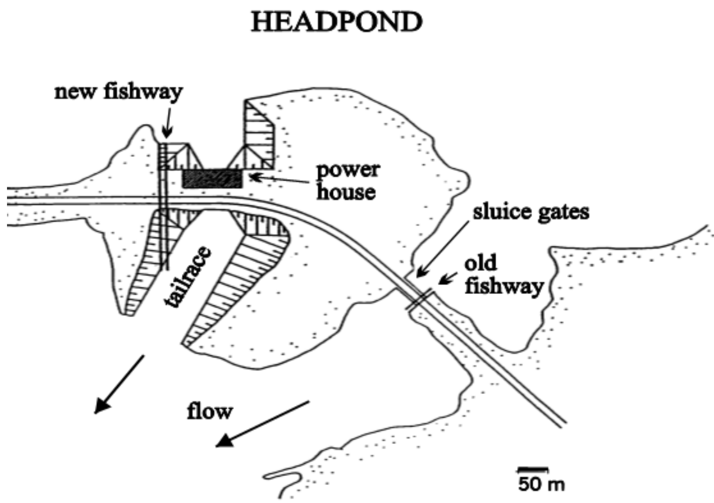

Figure 1. The Annapolis Tidal Generating Station in Annapolis Royal, Nova Scotia (from Gibson 1996b, adapted from Ruggles and Stokesbury 1990).

The Annapolis TiGS was constructed at the causeway and has been in operation since 1985 (Dadswell et al. 1986, Baker and Daborn 1990). Its operation increased the tidal range upriver of the causeway to about one meter (Daborn et al. 1979), although the estuary upstream of the causeway remains highly stratified, particularly in its upper reaches (Gibson and Daborn 1993).

The Annapolis TiGS was designed as a prototype to test the STRAFLO™ turbine in anticipation of proposed, large-scale hydroelectric development in the upper Bay of Fundy. The unit is a low-head propeller turbine that generates only when water is flowing seaward (normal operating head range: 1.4 - 6.8 m). It has a 7.6 m runner diameter and rotates at a rated speed of 50 rpm. Output at a head of 5.5 m is 17.8 MW, with a corresponding discharge of 378 m^3/s (Table 1). The turbine has not been in operation continuously since 1985. Periodically, the turbine has been taken offline for extended periods of time for maintenance or for other reasons (e.g. when a whale was present in the headpond).For this reason and others, although annual generation of the turbine was predicted to be 50 GW hrs (Douma and Stewart 1981), its realized production has been less.

Table 1. Turbine description and predicted generation capacity of the Annapolis Tidal Generating Station. Modified from Douma and Stewart (1981).

Characteristic	Value
Diameter of runner	7.6 m
Number of blades	4
Number of wicket gates	18
Normal operating head range	approx. 1.4-6.8 m
Maximum operating head	7.1 m
Rated operating head	5.5 m
Output at rated head	17.8 MW
Maximum output	19.8 MW
Discharge at rated head	378 m^3/s
Efficiency for full load at rated head	89.1%
Rated speed	50 rev/min

Description of Fishways

Two fishways have been constructed to augment fish passage at the Annapolis TiGS. The “old fishway” was built during the original barrage construction in the early 1960’s and is essentially a sluice gate, four meters wide (Table 2), that is kept open throughout the tidal cycle. It is located next to the sluice gates used to fill the headpond about 300 m from the turbine forebay (Figure 1). The “new fishway”, which was constructed at the same time as the generating station, runs between the turbine forebay and the tailrace. It is 3 m wide (Table 2) and is located about 12 m from the turbine intake. It is an unlit square culvert that runs under the highway. In 2002, the new fishway was extended on the seaward side to ensure that it discharges into the water in the tailrace at all times when water is flowing seaward. Water depth in both fishways varies between 1.5 m and 2.5 m depending on the stage of the tide. Flows through the two fishways are 42.7 m^3/s and 10.1 m^3/s for the old and new fishways respectively, for a 0.3 m head (Stokesbury and Dadswell 1991).

Depending on the stage of the tide, water may follow in either direction through the fishways providing for both landward and seaward fish passage. However, the majority of fish moving landward likely do so via the sluice gates when they are open to fill the headpond.

Table 2. Characteristics of the fishways at the Annapolis Tidal Generating Station.

	New fishway	**Old fishway**
Location[1]	Adjacent turbine entrance	Adjacent sluice gates
Width[1, 2]	3 m	3 m
Height[1, 2]	3.7 m	7.3 m
Length[1]	75 m	10 m
Water depth[3]	1.5 to 2.5 m	1.5 to 2.5 m
Max. flow at 0.3 m head[3]	10.1 m^3/s	42.7 m^3/s

[1]Douma and Stewart (1981); [2]Beak (1991); [3]Stokesbury and Dadswell (1991)

Operation

The Annapolis TiGS utilizes the water level differential between the headpond and the estuary downstream in order to generate power. This head differential is created by the difference in the tidal range upstream and downstream of the causeway, which is controlled at the causeway.

During most tidal cycles, the headpond is filled primarily through the three sluice gates located just south of the generating station (Figure 1), although water also flows landward though the new fishway during filling (Figure 1). The sluice gates are closed when the desired water level in the headpond is reached. In the past during low neap tides, the turbine tube was opened to allow the headpond to fill more completely, but this has not been standard practice for the last 10 to 15 years.

On the ebb tide, once the water level downstream of the turbine drops below the water level in the headpond, water begins to flow seaward initially only through the two fishways. Generation begins once the water level downstream of the causeway drops to about 1.5 m below the headpond level, and continues through the ebb and flood tides until the water level differential is again less than 1.5 m. Water continues to flow seaward through the two fishways until the water level downstream of the causeway reaches the level in the headpond. At this point the sluice gates are opened to fill the headpond for the next generation cycle. During a typical tidal cycle, the unit generates for about 5.5 hours.

TOR 2: SEARCH EFFORT

There are two parts to the search undertaken for this review. First, this review is focused on studies undertaken at the Annapolis TiGS that directly pertain to turbine mortality, fishway utilization, fish diversion or population-level effects. In total, 24 documents describing the results of 12 studies (including a previous review) were identified in the search for documents directly related to these themes (Table 3). The majority of the documents were provided by two researchers who studied fish passage at the Annapolis TiGS and include primary literature, field and technical reports and graduate student theses. Additional searches were undertaken using the Fisheries and Oceans Canada Library (https://science- libraries.canada.ca/eng/fisheries-oceans/) and using Google Scholar (https://scholar.google.ca/) to ensure no studies were overlooked. The primary search terms were “Annapolis Royal OR Annapolis River”, with additional searches carried out using “Fish*”, “passage”, “mortalit*”, “population*” and “diversion” Or “deterrent” to narrow down to specific articles.

The second part of the search effort pertains to TOR 3. After developing a species list from the information obtained in the first component of the search, information regarding the life history, resiliency, population structure and status of each species was compiled. Sources of information included the primary literature, books, research theses, CSAS documents, NOAA reports, COSEWIC reports and online databases (e.g. FISHBASE). Studies about fish in the Annapolis River and estuary that do not pertain specifically to survival of fish at the Annapolis TiGS were included as appropriate for evaluating the effects of mortality at this site. These studies are incorporated into TOR 3 and its supporting Appendix.

Table 3. Fish passage, mortality, population impact, and fish diversion related research at the Annapolis Tidal Generating Station. Alosa spp. *include Alewife (A. pseudoharengus) Blueback Herring (A. aestivalis) and American Shad (A. sapidissima).*

Study number	Year	Study description	Field study	Focus species	TOR 4 Fishway utilization	TOR 5 Fish mortality	TOR 6 Population-level impacts	TOR 7 Fish diversion	Publications / reports
1.	1981 1982	Pre-operational shad assessment	Yes	American Shad			X		Melvin et al. 1985
2.	1985 1986	Turbine mortality and fish passage	Yes	Clupeidae, including *Alosa* spp.	X	X			Stokesbury 1985,1986, 1987; Stokesbury and Dadswell 1989; Stokesbury and Dadswell 1991
3.	1985-1986	Observations by SCUBA divers and others				X			Hogans and Melvin 1985; Hogans 1987; Dadswell and Rulifson 1994
4.	1985 1986	American shad turbine mortality	Yes	American Shad		X			Hogans and Melvin 1985; Hogans 1987
5.	1988 1989	Fish diversion	Yes	*Alosa* spp.				X	McKinley and Patrick 1988; McKinley and Kowalyk 1989
6.	1989	Juvenile *Alosa* study	Yes	*Alosa* spp.	X			X	Ruggles and Stokesbury 1990
7.	1989 1990	Post-operational shad assessment	Yes	American Shad			X		Dadswell and Themelis 1990a, 1990b
8.	1991	Review of fish mortality studies at Annapolis	No	*Alosa* spp.	X	X	X	X	BEAK 1991
9.	1993 1994	Juvenile *Alosa* study	Yes	*Alosa* spp.	X				Gibson and Daborn 1993, 1995b; Gibson 1996b
10.	1995 1996	Shad assessment	Yes	American Shad			X		Gibson and Daborn 1995a; Gibson 1996a
11.	1999	Fish diversion and mortality	Yes	*Alosa* spp. others	X	X		X	Gibson and Myers 2000, 2002a, 2002b
12.	2018	Population-level effects	No	American Shad, Striped Bass, Atlantic Sturgeon			X		Dadswell et al. 2018

TOR 3: DESCRIPTION OF THE FISH COMMUNITY IN THE VICINITY OF THE ANNAPOLIS TIGS.

Fish

The Annapolis River and estuary are home to a variety of marine, diadromous, and freshwater fishes. A summary of the distribution, fisheries, life history, biological characteristics and available information relevant to mortality and passage at the Annapolis TiGS for 40 marine and diadromous species observed in the Annapolis River system is provided in the Appendix. This list is not all inclusive: other species may be present intermittently or in low abundance, or were missed due to sampling gear selectivity, timing of sampling events, or the goals of the research projects.

When assessing the impact of a project or activity, a first question is “Impact on what?”. For the purposes of this review, the appropriate level for assessment is considered to be that of a “closed” population. A population of a specific species is considered closed if intrinsic factors such as growth, reproductive rates, carrying capacity, natural mortality, and mortality caused by human activities are the significant factors determining the population’s dynamics; while extrinsic factors such as immigration and emigration are minimal and can be ignored. This definition aligns closely with the goals of assessment, which are to determine whether mortality rates and abundances are within appropriate limits. Under this definition, more than one population of a particular species may be present in vicinity of the activity or project. Striped Bass are an example of such: historically individuals from both a local population and strays from the U.S. present in the vicinity of the Annapolis TiGS (Harris 1988). This definition of a population is appropriate for the majority of species found at the Annapolis TiGS, with the possible exception of American Eel.

A qualitative assessment of the expected impact of the Annapolis TiGS on fish populations was undertaken as part of this review. Population-level impacts of the Annapolis TiGS were categorized using the definitions set forth in “Guidance on assessing threats, ecological risk, and ecological impacts for species at risk” (DFO 2014a). Potential population-level impacts are categorized as extreme, high, medium, low, or unknown (Table 4) based on the effect that turbine mortality at the Annapolis TiGS would have on the size of the population as a whole.

Uncertainty in the population-level impact is categorized on a five point scale from very low to very high, based on the information available about population structure, life history and turbine mortality rates (Table 5). Very low uncertainty corresponds to a well-defined population with strong evidence for the effects of mortality, while very high uncertainty corresponds to a species with little to no information on population structure, life history or turbine mortality rates. Uncertainty in the population-level impact is not independent of the population-level impact itself. As population-level impact increases, uncertainty in the categorization often increases due to the increased quality and reliability of the data needed to make the conclusion, while a low population-level impact may only need one strong piece of evidence (e.g. lack of individuals which may encounter the turbine) to categorize the impact. Many of the species that are present around the Annapolis TiGS likely have a low population-level impact due a broad geographic range with only a small portion of the spawning stock encountering the Annapolis TiGS.

In addition to the mortality rate for fish passing through the turbine, there are several other factors that contribute to the impact on a population, including: the resiliency of the population to increased mortality rates; the proportion of the population in the vicinity of the turbine; the number of times that individuals move past the causeway; the proportion of the population that use the fishways; the timing of mortality relative to life history events such as growth, maturation, reproduction and density-dependent processes that regulate population size; and rates of mortality associated with other human activities. The available information on these factors along with uncertainty in the data for each factor contributes to the categorization of both the population-level effect and the uncertainty associated with it. The level of impact and uncertainty for each of the 40 fish species identified in this review is summarized in Figure 2. The rationale for the category selection can be found in Table 6, with additional information in the Appendix. Species may be indirectly affected by the Annapolis causeway through changes such as habitat alteration, but the focus in this review is restricted to the direct effects of turbine mortality.

Table 4. Categories for the population-level impact of turbine mortality at the Annapolis Tidal Generating Station. Modified from DFO (2014a).

Level of Impact	Definition
Extreme	Severe population decline (e.g. 71-100%) with the potential for extirpation
High	Substantial loss of population (31-70%)
Medium	Moderate loss of population (11-30%)
Low	Little change in population (1-10%)
Unknown	No prior knowledge, literature or data to guide the assessment of threat severity on population

Table 5. Categories for the uncertainty in the population-level impact of turbine mortality at the Annapolis Tidal Generating Station.

Uncertainty	Definition
Very High	Little to no information on population structure, life history or turbine mortality rates
High Moderate Low	↓ Increasing information quantity, data certainty, and reliability relating to population structure, life history and turbine mortality rates
Very Low	A well-defined population with strong evidence for the effects of mortality

In this review, the highest priority species for studying the impacts of turbine mortality are considered to be those for which a COSEWIC assessment resulted in a status designation of “extirpated”, “endangered”, “threatened” or “special concern”; those that are SARA-listed as “extirpated”, “threatened” or “endangered”; and those species that support commercial, recreational or aboriginal fisheries species for which the expected population-level impact is categorized to be greater than “low”. There are thirteen species that meet one of these criteria. These are: Alewife, American Eel, American Shad, Atlantic Menhaden, Atlantic Salmon, Atlantic Sturgeon, Atlantic Wolffish, Blueback Herring, Lumpfish, White Hake, Rainbow Smelt, Spiny Dogfish and Striped Bass. The Atlantic Wolfish is the only species captured in studies at the Annapolis TiGS that is listed under the Species at Risk Act, but decisions about whether or not to list are pending for some other species.

For several species, the potential population-level impact is categorized as low, but with high or very high uncertainty. For the majority of the populations with these categories, the major source of uncertainty is the population structure and whether a significant portion of the population would be exposed to the turbine. These species include: Atlantic Tomcod, Four-beard Rockling, Ninespine Stickleback, Northern Pipefish, Rock Gunnel, and Saint John River population of Atlantic Sturgeon. As discussed under TOR 8, further work to identify appropriate population units for these species would improve the assessment undertaken here.

Marine mammals and sea turtles

Two records of whales in the vicinity of the Annapolis TiGS were identified during this review. A Humpback Whale became trapped in the Annapolis River upstream of the Annapolis TiGS in 2004, but is presumed to have successfully navigated the Annapolis causeway and returned to the Annapolis Basin. The lone whale mortality event at the Annapolis TiGS was attributed to an immature Humpback Whale (possibly a Minke Whale) discovered near the head of tide in Bridgetown in 2007 during the spring thaw. Both occurrences of whales at the Annapolis TiGS appear to be isolated instances and the population-level impacts are low with low uncertainty.

Seals are often present at the Annapolis TiGS and appear to move freely between the tailrace and the headpond using the sluice gates (Personal communication Jamie Gibson). No documented accounts of turbine mortality on seals from the Annapolis TiGS were found during this review. Although seals have not been identified to species, the population structure and large population of the four potential species present in the Bay of Fundy suggest that the seals present at the Annapolis TiGS would represent only a small portion of the population regardless of the species.

No records of the presence of sea turtles near the Annapolis TiGS were found during this review.

		Uncertainty				
		Very Low	Low	Moderate	High	Very High
Level of Impact	Extreme				• Striped Bass (Annapolis) • Atlantic Salmon	
	High				• Atlantic Sturgeon (Annapolis)	
	Medium			• American Shad • Alewife • Blueback Herring	• Rainbow Smelt	• Atlantic Menhaden
	Low	• Bluefish • Flying Gurnard • Meek’s Halfbeak	• Atlantic Mackerel • Atlantic Wolffish • Pollock • Smooth Flounder • Spiny Dogfish • White Perch	• American Eel • American Sand Lance • Atlantic Herring • Atlantic Silverside • Blackspotted Stickleback • Butterfish • Cunner • Fourspine Stickleback • Hake • Longhorn Sculpin • Lumpfish • Sea Lamprey • Sea Raven • Striped Bass (Shubenacadie) • Striped Bass (other) • Threespine Stickleback • Windowpane • Winter Flounder • Wrymouth • Mummichog	• Atlantic Sturgeon (Saint John) • Four-beard Rockling • Ninespine Stickleback • Northern Pipefish • Rock Gunnel	• Atlantic Tomcod

Figure 2. Species matrix for the expected population-level impact and uncertainty for 40 species which encounter the Annapolis TiGS. Because fish from more than one population of the same species may encounter the turbine, Striped Bass is split into Annapolis River, Shubenacadie River, and U.S.A.-origin populations, and Atlantic Sturgeon is split into a presumed Annapolis River population and the population in Saint John River.

Table 6. Rationale for the impact and uncertainty categorizations for 40 species present at the Annapolis TiGS . Category definitions can be found in Table 4 (impact) and Table 5 (uncertainty).

Species	Impact level (Uncertainty)	Rationale for Impact Level	Rationale for Uncertainty Level
Alewife	Medium (Moderate)	Well defined population structure with all of the population likely to encounter the Annapolis TiGS; high productivity; estimated acute turbine mortality rates that could lead to a decline in population size greater than 10%.	Limited information about fishway usage; delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known.
American Eel	Low (Moderate)	Panmictic population structure such that only a small portion of the population is likely to encounter the turbine; estimated turbine mortality rates are low.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known; information about population structure is limited.
American Sand Lance	Low (Moderate)	Broad geographic distribution is suggestive that only small proportion of population may encounter the turbine.	Fishway usage, acute and delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known; population structure is not well understood.
American Shad	Medium (Moderate)	Well defined population structure with all of the population likely to encounter the Annapolis TiGS, high productivity; acute turbine mortality rate estimates that could lead to a decline in population size greater than 10%.	Limited information about fishway usage; delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known; cumulative effects of fishing and passage at the TiGS have not been studied.
Atlantic Herring	Low (Moderate)	Broad geographic distribution of population suggestive that only a portion of the population is likely to encounter the turbine; utilization of the new fishway; medium resiliency.	Fishway usage, delayed mortality and movement patterns in the vicinity of the turbine are not well known; limited information about population structure; cumulative effects of fishing and passage at the TiGS have not been studied.
Atlantic Mackerel	Low (Low)	The population has a broad geographic distribution with a high degree of mixing suggestive that only a small proportion of the population is likely to encounter the Annapolis TiGS; medium resiliency.	Distribution and population are structure well understood. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact.
Atlantic Menhaden	Medium (Very High)	Broad geographic distribution suggestive that only a small portion of the population is likely to encounter the turbine; medium resiliency.	Presence of juveniles in the Annapolis River is atypical and their usage of the river is not known.Limited information about population structure,turbine mortality rates and movement patterns in the vicinity of the turbine.

Species	Impact level (Uncertainty)	Rationale for Impact Level	Rationale for Uncertainty Level
Atlantic Salmon	Extreme (High)	Well defined population structure; low population size; currently very low maximum lifetime reproductive rate. All of the population expected to encounter the turbine.	Fishway utilization, acute and delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known; cumulative effects of mortality in other parts of their life cycle.
Atlantic Silverside	Low (Moderate)	Low acute turbine mortality rate estimate; high rate of fishway utilization; high abundance and high resiliency.	Uncertain population structure; delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not well understood.
Atlantic Sturgeon (Annapolis)	High (High)	Evidence of turbine mortality and very low population resiliency. All of a native population would be expected to encounter the turbine.	Existence of a population native to the Annapolis River is not known; fishway usage, acute and delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known.
Atlantic Sturgeon (Saint John River)	Low (High)	Well defined population structure; proportion of the population expected to encounter the turbine thought to be low.	Proportion of the population encountering the turbine, fishway usage, acute and delayed turbine mortality rates, and movement patterns in the vicinity of the turbine are not known.
Atlantic Tomcod	Low (Very High)	Broad geographic distribution suggestive that only small proportion of population is likely to encounter the turbine; medium resiliency.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not known (local population possible). Timing of studies does not overlap with their expected presence.
Atlantic Wolffish	Low (Low)	Low catch and broad geographic distribution suggestive that only small proportion of population is likely to encounter the turbine; habitat in the vicinity of the TiGS not typical of Atlantic Wolffish habitat.	Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact. Distribution and habitat preference are well understood.
Blackspotted Stickleback	Low (Moderate)	Very low turbine mortality rate estimate and high resiliency.	Limited information about fishway usage; delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known; little information about population structure (local population likely).
Blueback Herring	Medium (Moderate)	Well defined population structure with all of the population likely to encounter the Annapolis TiGS; high productivity; acute turbine mortality rate estimates that could lead to a decline in population size greater than 10%.	Limited information about fishway usage; delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known.

Species	Impact level (Uncertainty)	Rationale for Impact Level	Rationale for Uncertainty Level
Bluefish	Low (Very Low)	Given their range, presence in the Annapolis Basin is likely infrequent.	Range is well known. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact.
Butterfish	Low (Moderate)	Low acute turbine mortality rate estimates; broad geographic distribution suggestive that only small proportion of population is likely to encounter the turbine; high resiliency.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not well known. Population structure not well understood.
Cunner	Low (Moderate)	No evidence found of a population specific to the Annapolis Basin; medium resiliency; evidence that they can survive passage at the TiGS; preference for marine habitat.	Limited information about population structure (non-migratory with a small home range).
Flying Gurnard	Low (Very Low)	Given their distribution and habitat preference, presence in the Annapolis Basin is likely infrequent; high resiliency.	Habitat preferences and distribution are well known. The single specimen captured at the Annapolis TiGS in 1999 was a first Bay of Fundy record.
Fourbeard Rockling	Low (High)	Broad distribution in the Northwest Atlantic suggestive that only a small proportion of the population is likely to encounter the Annapolis TiGS; medium resiliency; single specimen recorded at the Annapolis TiGS in 1999.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not well understood.
Fourspine Stickleback	Low (Moderate)	Small size; low turbine mortality rate estimates for other stickleback species; high resiliency.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not well understood (location population likely).
Hake spp. (White, Silver, Red)	Low (Moderate)	Healthy commercial fisheries on the Scotian Shelf (Silver Hake); broad geographic distribution suggestive that only a small portion of their populations are likely to encounter the Annapolis TiGS; low acute turbine mortality rate estimates.	Individuals are not identified by species; information on population structure is limited; resiliency is variable among species.
Longhorn Sculpin	Low (Moderate)	High abundance throughout the Bay of Fundy suggestive that only a small proportion of their population is likely to encounter the Annapolis TiGS; high resiliency.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not known (local population possible).

Species	Impact level (Uncertainty)	Rationale for Impact Level	Rationale for Uncertainty Level
Lumpfish	Low (Moderate)	High abundance in the outer Bay of Fundy; broad geographic distribution suggestive that only small proportion of the population is likely to encounter the turbine.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not known (local population possible); low resiliency.
Meek's Halfbeak	Low (Very Low)	Given their range, presence in the Annapolis Basin is likely infrequent.	Range is well known. The single specimen captured at the Annapolis TiGS in 1999 was a first Canadian record.
Mummichog	Low (Moderate)	Small size; high abundance; population level tolerance to pollution, predation and fishing; medium resiliency.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not known (local population probable).
Ninespine Stickleback	Low (High)	Small size; low turbine mortality rate estimates for other stickleback species; medium resiliency.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not well understood (local population likely). Lower resiliency than other sticklebacks.
Northern Pipefish	Low (High)	Low estimates of acute turbine mortality rates; high resiliency.	Limited information about fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not known, but reproduction in the Annapolis River estuary is probable.
Pollock	Low (Low)	Large population size (Western Component); low numbers observed at the Annapolis TiGS; broad geographic distribution suggestive that only small proportion of population is likely to encounter the turbine.	Population size and structure are reasonably well known. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact.
Rainbow Smelt	Medium (High)	Population likely native to the Annapolis River suggestive that all of the population is likely to encounter the Annapolis TiGS; medium resiliency.	Limited information about fishway usage; delayed mortality rates and movement patterns in the vicinity of the turbine are not known.
Rock Gunnel	Low (High)	Broad geographic distribution suggestive that only small proportion of population is likely to encounter the turbine.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not known (local population possible).

Species	Impact level (Uncertainty)	Rationale for Impact Level	Rationale for Uncertainty Level
Sea Lamprey	Low (Moderate)	Very low acute turbine mortality rate estimates (0%); potential for panmictic population structure such that only a small portion of the population is likely to encounter the turbine.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not well understood (local population possible).
Sea Raven	Low (Moderate)	No recorded captures at the Annapolis TiGS; lack of preferred habitat within the Annapolis River system suggestive that only small proportion of population is likely to encounter the Annapolis TiGS.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure and life history are not well understood.
Smooth Flounder	Low (Low)	A single catch record; broad geographic distribution and preference for habitat not widely available in the Annapolis River system indicative that only small proportion of population is likely to encounter the turbine.	Distribution and habitat preference are well understood. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact.
Spiny Dogfish	Low (Low)	Broad geographic distribution and high abundance in the Bay of Fundy indicative that only a small portion of the population would encounter the Annapolis TiGS.	Distribution, abundance and habitat preferences are well understood. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact.
Striped Bass (Annapolis)	Extreme (High)	Well defined population structure; potentially extirpated; seasonally resident near the Annapolis TiGS; currently very low reproductive rate. All of the population expected to encounter the turbine.	Fishway usage, turbine mortality rates and movement patterns in the vicinity of the turbine are not known; cumulative effects with high mortality rates in other parts of their life cycle not understood.
Striped Bass (Shubenacadie)	Low (Moderate)	Low proportion of the population thought to encounter the turbine.	Fishway usage, turbine mortality rates and movement patterns in the vicinity of the turbine are not known; limited information about the proportion of population encountering Annapolis TiGS.
Striped Bass (U.S.A origin)	Low (Moderate)	Low proportion of the population thought to encounter the turbine.	Fishway usage, turbine mortality rates and movement patterns in the vicinity of the turbine are not known; limited information about the proportion of population encountering Annapolis TiGS.

Species	Impact level (Uncertainty)	Rationale for Impact Level	Rationale for Uncertainty Level
Threespine Stickleback	Low (Moderate)	Small size; low turbine mortality rate estimates for other stickleback species; high resiliency.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not well understood (local population likely).
White Perch	Low (Low)	Resident in the Annapolis River and estuary, but preference for lower salinity would limit encounters with the Annapolis TiGS.	Salinity preference is well understood. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact.
Windowpane	Low (Moderate)	Low acute turbine mortality rate estimates; broad geographic distribution suggestive that only small proportion of population is likely to encounter the Annapolis TiGS.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not well understood.
Winter Flounder	Low (Moderate)	Low acute turbine mortality rate estimates; broad geographic distribution suggestive that only small proportion of population is likely to encounter the Annapolis TiGS.	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not well understood.
Wrymouth	Low (Moderate)	Broad geographic distribution suggestive that only small proportion of population is likely to encounter the turbine	Fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Population structure is not known

TOR 4: FISHWAY UTILIZATION STUDIES

As described under TOR 1, fish moving downstream at the Annapolis TiGS have three options for passage at the causeway: the old fishway, the new fishway, or through the turbine. Upstream movement is almost entirely though the sluice gates. If the filling of the headpond is augmented by sluicing through the turbine, as was done in the past on neap tides, fish can move upstream through the turbine. However, the effects of this process have not been studied and this method of filling the headpond has not been carried out for more than a decade. When migrating downstream, the proportion of individuals utilizing each route is an important determinant of the population-level impacts of the station. To date, there have been four studies that provide information about downstream movement (fishway versus turbine usage) for *Alosa* species, three of which have information on other species. These are: the *Alosa* studies in 1985 and 1986, the juvenile *Alosa* study in 1989, the juvenile *Alosa* migration studies in 1993 and 1994, and the fish diversion studies in 1999. Survival of fish moving through the fishways has not been studied.

Alosa migration studies in 1985 and 1986 (Table 3: study number 2)

As part of the downstream migration studies undertaken at the Annapolis TiGS during 1985 and 1986, researchers monitored the downstream migration of juvenile *Alosa* through the Annapolis causeway during the late summer and fall (Stokesbury 1987, Stokesbury and Dadswell 1989). Four one-meter-diameter ichthyoplankton nets were used to conduct sampling, one net in each of the two fishways and two nets in the tailrace area. Sampling occurred between September 25 and October 30, 1985. All nets were suspended about one meter below the surface. In 1985, during comparison fishing (Stokesbury and Dadswell 1989), the majority (98%) of *Alosa* individuals were captured in nets deployed in the tailrace, and less than 2% of the total catch came from the two fishways (1,282 *Alosa* caught in the tailrace versus 17 caught in the fishways). Due to the low numbers caught in 1985, the fishways were not sampled in 1986.

The proportions caught at each of the three locations is not in of itself indicative of the proportion of fish utilizing each route of passage because the nets capture different proportions of the fish at each location. However, because the nets in the tailrace cover a much smaller portion of the cross-sectional area than do the nets in the fishways, the proportion caught in the fishways (2%) could be considered a maximum value if interpreted as an estimate of fishway usage (Stokesbury and Dadswell 1991, BEAK 1991). While there is no technical reason to reject this conclusion, the results differ from those of Gibson and Daborn (1993, 1995b) and Gibson and Myers (2000), who fished with modified ichthyoplankton nets that were also suspended near the surface (see below).

Juvenile Alosa study in 1989 (Table 3: study number 6)

Ruggles and Stokesbury (1990) monitored seaward fish movement at the Annapolis TiGS in the fall of 1989 using ichthyoplankton nets to capture fish. Nets were placed in the old fishway, the seaward flow of the sluice gates, and the tailrace of the turbine, but no sampling occurred in the new fishway. Over the course of six weeks of sampling, 2,209 individuals of 19 species were captured at the causeway; the majority of which were Atlantic Herring. Of the 374 juvenile *Alosa* captured, 95% were located in the tailrace, 5% in the old fishway and no individuals were captured in the seaward flow of the sluice gates. This study provides evidence that individuals move through both the old fishway and turbine, although difficulties calculating the proportion of flow sampled at the two sites and differences in fishing efficiency and fishing effort among locations preclude estimation of the proportion of individuals using each route. However, the results indicate the majority of fish captured in this study passed through the turbine when moving downstream.

Juvenile Alosa studies in 1993 and 1994 (Table 3: study number 9)

Gibson and Daborn (1993) monitored downstream fish passage at the Annapolis TiGS using modified icthyoplankton nets from late September to October, 1993. Fish were sampled at the old fishway, new fishway and tailrace to determine which route individuals travelled through the causeway. In total, 14 generation cycles were sampled in the tailrace, 13 were sampled in the old fishway and 16 in the new fishway. Although *Alosa* were the intended target species , 20 species were captured at the Annapolis causeway during the study. In total, 4,158 individuals were captured across the three sites. For the main species of interest, American Shad, Alewife, and Blueback Herring, 76.2% were caught in the tailrace, 23.3% in the new fishway and only 0.5% in the old fishway. Of the other species, mainly Atlantic Herring and Atlantic Silversides, 23.7% were captured in the tailrace, 61.0% in the new fishway, and 15.3% in the old fishway. Although the old fishway appears to play a greater role in fish passage for some non-*Alosa* species, the authors concluded that the old fishway is the least used pathway through the causeway for the majority of species.

Gibson and Daborn (1995b) monitored downstream fish passage at the Annapolis TiGS using the same modified ichthyoplankton nets from late July to mid-November, 1994. Sampling intensity was much higher than in previous studies, with 70 generation cycles sampled in the tailrace, 66 in the old fishway and 64 in the new fishway. Although *Alosa* were the intended target species of the study, a total of 82,194 individuals representing 26 species were captured. The majority of the fish were Atlantic Silversides (77,064 fish) most of which were captured in the new (79.0%) and old (18.3%) fishways. When Atlantic Silversides are not considered, 31.7% of fish were captured in the tailrace, 40.7% in the new fishway, and 27.6% in the old fishway. For many of the species, such as Atlantic Mackerel and Mummichog, there were too few individuals captured during the study to determine if there is a preferred route through the causeway. Of the 2,186 *Alosa* captured during the study, 870 were Alewife caught in the new fishway on a single sampling day. If this catch is removed from consideration, the majority of the *Alosa* catch was captured in the tailrace (79.8%), with 19.4% coming from the new fishway and 0.8% from the old fishway. Although *Alosa* did not tend to utilize the old fishway, Sea Lamprey, Stickleback spp., Pipefish, and Lumpfish were captured in the old fishway more frequently than either the new fishway or tailrace.

As reported in Gibson (1996b), in the 1994 study, attempts were made to determine the capture efficiency of the nets in the new fishway by fishing smaller nets in a grid to determine the distribution of fish passing through new fishway. Although subject to high uncertainty because of the use of smaller nets to infer the capture rate of larger nets, the results of these experiments suggest that the 0.75 m diameter net used in the study would catch about 6% of the fish passing through the new fishway.

With respect to *Alosa*, the results of the 1993 and 1994 studies differ from those obtained in 1985 and 1986. The nets used in the 1993 and 1994 studies were different from those used in 1985 and 1986. The 1993/94 nets included a cylindrical section with coarse mesh to reduce avoidance, and a collector with a minnow-trap-like entry to keep fish from escaping. The overall efficiency of these nets was higher, and the differences in the results of these studies might simply be a difference in the efficiency of the nets (Gibson and Daborn 1995b). In addition, fish hammers used to deter fish from passing through the turbine (see TOR 7) may have been in operation at least intermittently during the 1993 and 1994 studies (whether they were operating is not known), but not during 1985 and 1986.

With respect to estimating the proportion of fish moving downstream through the fishways, the 1993 and 1994 studies provide coarse information that is best interpreted qualitatively. The proportions of fish captured in the tailrace and two fishways are not corrected for the number of generation cycles that were sampled at each location and the proportions of the fish moving through each passage that are captured by the nets (net capture efficiencies) are not known. Additionally, capture efficiency would be expected to vary among species. Without more information on net efficiencies and coverage, the true proportion of individuals selecting each route remains unclear. However, given the large cross-sectional size of the tailrace relative to that of the fishways, these studies provide evidence that the majority of individuals move downstream through the turbine.

Fish diversion and mortality study in 1999 (Table 3: study number 11)

The objectives of Gibson and Myers (2000) were not to estimate fishway usage, but the results from their fish diversion study do provide information about fishway usage for some species. Monitoring methods used by Gibson and Myers (2000) were similar to those of Gibson and Daborn (1995b). As expected, the fish diversion study concluded that high frequency sound was not an effective diversion for non-*Alosa* species, so the results from the sampling at the new fishway, old fishway and tailrace provide information about how other species are using these three routes. Sampling at the three locations took place in the fall during 48 generation cycles, of which 48 cycles were monitored in the tailrace, 46 cycles in the new fishway, and 44 in the old fishway. Also similar to Gibson and Daborn (1995b), Atlantic Silversides dominated the catch and were caught mostly in the new (74.8%) and old (22.3%) fishways. Only 0.05% of Atlantic Herring were caught in the old fishway, and 47.3% of their catch was from the new fishway. Excluding *Alosa*, 5.3% of fish were caught in the tailrace, 72.6% in the new fishway and 22.0% in the old fishway. Excluding Atlantic Silverside as well as *Alosa*, changes the results to 31.4% of the catch in the tailrace, 49.7% in the new fishway and 18.9% in the old fishway.

Gibson and Myers (2000) attempted to estimate the capture efficiency of nets in the tailrace by releasing marked fish into the turbine. Of 4,170 fish released over three occasions, 10 were captured in the tailrace nets. They concluded that the two nets fished in the tailrace captured one out of every 335 fish that passed through the turbine (Gibson and Myers 2000). These authors then combined this estimate of capture efficiency for nets in the tailrace with the capture efficiency estimates reported by Gibson (1996b) from the 1994 study. Based on these values, and treating the old fishway as inconsequential, Gibson and Myers (2000) reported that in 1999, about 57% of the Atlantic Silversides used the new fishway. Their estimates of new fishway usage by *Alosa* ranged from 3.5% for Blueback Herring to 15.6% for Alewife (Gibson and Myers 2000). These results for *Alosa* are not comparable with the earlier studies because the fish diversion system being tested during this study was partially effective for Alewife, Blueback Herring and American Shad.

Summary of fishway utilization studies

To date, studies of fishway utilization have been able to report relative catches of fish from ichthyoplankton nets at the three possible routes through the Annapolis causeway indicating that, with the exception of Atlantic Silverside, the majority of individuals pass downstream through the turbine. Fishway usage is variable among species. Proportions of the total catch coming from the fishways versus the tailrace varied among studies, potentially due to differences in the efficiency of the nets being used or differences among years.

A limitation of the fishway utilization studies is that the studies are restricted to small fish that can be captured in nets deployed for monitoring. Larger fish, such as Striped Bass, adult

Atlantic Herring and Atlantic Mackerel, were known to be present during the study but were not captured in these nets. Fishway usage by these larger species and individuals is not known, although adult American Shad have been observed using the new fishway in early summer. A second limitation is that all monitoring has occurred during late summer and fall. No information is available about fishway utilization during other time periods.

The fishway utilization studies provide some rough information about the numbers of fish passing through the turbine. In 1994, Gibson and Daborn (1995b) caught just over 3,600 fish using two nets in the tailrace. Combined with the Gibson and Myers (2000) capture efficiency estimates, these results indicate that over 1.2 million fish passed through the turbine during the 70 tidal cycles that were monitored in 1994 between the end of July and mid-November.

Similarly, in 1999, roughly 1 million fish would be expected to have passed through the turbine on the 48 tidal cycles monitored between early September and Late October (roughly ½ of the tidal cycles in the study period). These estimates do not include larger fish that were not susceptible to capture in the one-meter-diameter nets.

TOR 5: FISH TURBINE MORTALITY STUDIES AT THE ANNAPOLIS TIGS

Turbine mortality studies can be loosely divided into two groups: those that use fish experimentally released into the turbine intake and those that use naturally-entrained fish captured in nets in the turbine tailrace (Gibson and Myers 2002a). Turbine mortality studies can be further divided into those that estimate mortality rates immediately after fish pass through the turbine (short-term or acute), and those that estimate rates that include mortality occurring over a longer period (delayed mortality), the latter of which is by far more relevant for determining the population-level impacts of the mortality but much harder to estimate. There have been three studies at the Annapolis TiGS that have reported turbine mortality rate estimates: the 1985-86 adult American Shad mortality rate study using experimentally released fish; the 1985-86 young-of-the-year clupeid study using naturally entrained fish; and the 1999 fish diversion and mortality study. In addition, there is documentation of mortality of fish at the Annapolis TiGS from diver and shoreline observations. All of these studies resulted in more than one publication (Table 3). Overall, the three studies provide evidence that there is mortality associated with turbine passage occurring at the Annapolis TiGS. However, for the majority of species the rates of mortality associated with turbine passage, particularly delayed mortality, are not well quantified.

When interpreting the turbine mortality rate estimates, it is important to remember that the turbine mortality rate alone does not determine the impact on the population. First, if some portion of the population moves past the causeway via the fishways, overall survival will be higher than what is implied by the turbine mortality rate. Second, if fish move back and forth past the causeway more than one time, they may pass through the turbine more than one time. In this case, the overall survival will be lower than that implied by the mortality rate for a single pass through the turbine. Additionally, many of the biases in turbine mortality rate studies lead to single-pass estimates that are biased high (Gibson and Myers 2002a).

Observational records (Table 3: study number 3)

Observations of dead fish below the Annapolis TiGS were documented during the 1985 and 1986 American Shad mortality studies (Hogans and Melvin 1985, Hogans 1987), and subsequently summarized and modified by Dadswell and Rulifson (1994). Dead fish were collected or observed along the shoreline, from surface waters, or on the bottom of the estuary by SCUBA divers. These reports include observations of larger fish (e.g. Atlantic Mackerel, flounder spp., Striped Bass, Atlantic Sturgeon) that were not captured using the ichthyoplankton nets used in some of the other studies. The majority of the dead fish were observed on the estuary bottom. Hogans and Melvin (1985) and Hogans (1987) provided information about the search effort, location and dates of the surveys, types of injuries, and the dead fish were observed by more than one individual. Dadswell and Rulifson (1994) summarized these data together with observations along shoreline and on the water surface, although the methods used to collect these additional data are unclear. Dadswell and Rulifson (1994) reported that 11 species and 4,428 specimens were observed during 1985-1986, about 90% of which were Atlantic Herring and Blueback Herring. Overall, these records provide credible qualitative evidence of fish mortality occurring at the Annapolis TiGS. However, the differences in the numbers of macerated fish observed during the 1985 and 1986 studies are suggestive that the much larger numbers observed in 1985 may not be the norm (BEAK 1991). These reports were not intended to provide mortality rate estimates or estimates of the number of fish killed during the study period and are best interpreted either qualitatively or as minimum values.

Turbine mortality rates for adult American Shad (Table 3: study number 4)

Hogans and Melvin (1985) was the first study to directly measure turbine mortality rates at the Annapolis TiGS. Hogans and Melvin (1985) and Hogans (1987) used experimentally released fish to estimate short-term (up to 5 hours post-release) mortality rates for adult American Shad. In May, June, and July of 1985, American Shad were tagged and released, and their passage through the turbine was monitored. Twenty-four fish were tagged using sonic transmitters inserted into their esophagus and released in four batches during the study period. Of the 24 fish tagged, three swam upstream away from the turbine, one had a defective tag and no signal was received after release, and 20 individuals travelled through the turbine. Fish were tracked for three hours after travelling through the turbine and were considered to have survived if they continued to move downstream. Fish were considered dead if there was no active swimming or non-current-related movement for the three hours after introduction into the turbine. Current movements were determined by introducing a tagged but dead individual into the turbine. Nine of the 20 fish to pass through the turbine were considered to have survived. To account for mortality associated with handling and tagging, 39 control fish were tagged with dummy tags and held in a holding pen. Eight of these 39 control fish died resulting in a control mortality rate of 20.5%. After accounting for handling mortality, the mean turbine mortality rate was estimated to be 46.3% (90% CI ± 34.7%).

As a follow up to the Hogans and Melvin (1985) study, Hogans (1987) employed the same general study design but improved upon many of the issues identified in the original study. Captured fish were either tagged with sonic transmitters and released into the area in front of the turbine or tagged with dummy transmitters and held as control fish. Fish that went through the turbine were tracked for five hours to determine their live/dead status. Fish were considered to be dead if there was no movement detected. Forty-one American Shad were tagged and released in five batches, of which 11 returned upstream and did not pass through the turbine and four transmitters were lost. Of the 26 fish that were known to have passed through the turbine, 20 survived and six died. Survival of control fish was greatly improved over the original study: 95.5% of control fish were alive after 24 hrs. After accounting for control mortality, the mean turbine mortality rate of the study was calculated to be 21.3% (90% CI: ±15.2%) for a single pass downstream through the turbine.

In a review of the Hogans and Melvin (1985) and Hogans (1987) studies, BEAK (1991) identified high control mortality rates and small sample size as the two main issues with the 1985 study, the first of which was largely corrected in 1986. The number of test fish and control fish were increased in 1986 and all individuals were post-spawners as opposed to the mix of pre- and post-spawning fish used by Hogans and Melvin (1985). Improvement in handling methods led to a lower control mortality rate and likely contributed to the lower estimated turbine mortality rate. For this reason, BEAK (1991) concluded that the estimate of the turbine mortality rate of 21.3%±15.2% for adult American Shad represents a more reliable estimate than the estimate from 1985. When discussing reasons for the differences in the estimates between 1985 and 1986, Hogans (1987) stated that mortality may have been decreased by better handling of test and control fish; this resulted in a better relationship between fish passage and mortality in the 1986 study.

As described under TOR 6, Gibson (1996a) provided a summary of instantaneous total mortality rate estimates for adult American Shad in the Annapolis River before and after the Annapolis TiGS came online. Comparing the average total mortality rate from pre-operational studies in 1981 and 1982 to the average instantaneous total mortality rate from the four post-operation years (1989, 1990, 1995, 1996) indicates an increase of 0.36 for males and 0.47 for females, which may be due to turbine mortality at the Annapolis TiGS or to some other factor. The turbine mortality rate estimate for adult American Shad of 21.3% (Hogan 1987) alone would be expected to increase the instantaneous total mortality rate by 0.24, a value less than the observed change in the population, but within the confidence interval provided for the estimate. As such, the turbine mortality rate estimate for adult American Shad is not inconsistent with the changes in the annual survival rates observed in this population. However, because a portion of the shad use the fishways and because there is also the potential for a shad to pass through the turbine on more than one occasion during their emigration from the river, the turbine mortality rate, in of itself, is not representative of the change in annual survival rates expected to occur as a result of the operation of this facility. Passage back upstream of the causeway by post-spawning shad that had passed downstream through the turbine was not observed during these tracking studies.

Turbine mortality rates for young-of-the-year clupeids in 1985 and 1986 (Table 3: study number 2)

During the summer and fall of 1985 and of 1986, a study was undertaken to estimate the survival of young-of-the-year (YOY) clupeids, including Alewife, Blueback Herring, American Shad, Atlantic Herring, and Menhaden, passing through the turbine at the Annapolis TiGS (Stokesbury and Dadswell 1991). Naturally-entrained fish were used in this study.

Ichthyoplankton nets, placed near the water surface in the tailrace, were used to capture fish that had passed through the turbine. Nets were set for five hours in 1985 and one, two or five hours in 1986. Fish removed from the nets were then autopsied to determine if there were pressure, mechanical, or shearing injuries. Autopsy results were aggregated for the five species listed above, resulting in a single mortality rate estimate for the species combined. The effect of the net was determined by capturing fish upstream and placing them in the tailrace nets for either 30 or 60 minutes. Mortality estimates were determined by classifying fish as either damaged or undamaged and after accounting for an estimated 18.6% net mortality rate, turbine mortality was estimated at 46.3%.

Several issues have been raised with respect to this study. BEAK (1991) questioned the use of damaged/undamaged criteria to determine the cause of death captured in the tailrace and also considered the level of control mortality in their experiments to be high relative to acceptable levels proposed by Ruggles et al. (1990). Gibson and Daborn (1993) pointed out that the nets used to capture fish caused injuries similar to the injuries used by Stokesbury and Dadswell (1991) to identify fish killed by turbine passage. For example, Stokesbury and Dadswell (1991) reported that 31.3% of fish captured in tailrace exhibited redeye, which they considered indicative of pressure injuries. However, fishing with similar nets, Gibson and Daborn (1993) found that 100% of Alewife and Blueback Herring, 82.4% of American Shad and 35.2% of Atlantic Herring captured in the new fishway (fish that had not gone through the turbine) also exhibited redeye. Gibson (1996b) questioned the duration of the control experiments relative to the length of time that nets were fished in the tailrace. To determine mortality rates caused by the nets, control fish were placed in the tailrace nets for 30 or 60 minutes. After 60 minutes, about 65% control fish were dead (estimated from Figure 2 in Stokesbury and Dadswell 1991). However, the nets used to sample fish in the tailrace were fished for an average of 5.2 hours in 1985 (Stokesbury 1987), while in 1986, 83.7% of the catch occurred when nets were set for 3.75 hours or longer (Stokesbury 1987). The implication is that the majority of specimens that were autopsied during the Stokesbury and Dadswell (1991) study were captured when nets were fished for most of the generation period and the length of the control experiments were short relative to the length of time that fish would be expected to be in the net (Gibson 1996b). As is the case with any study using naturally entrained fish, if live fish are able to avoid capture in the net, the resulting turbine mortality rate will be biased high. Additionally, the equation used by Stokesbury and Dadswell (1991) to calculate the turbine mortality rate is incorrect and does not appropriately correct for control mortality (in this specific instance the turbine mortality rate was slightly underestimated). Given that the majority of these issues would lead to overestimation of the acute turbine mortality rate, the estimate provided by Stokesbury and Dadswell (1991) is likely biased high.

The accuracy of the turbine mortality rate of 46.3% for YOY *Alosa* has also been questioned based on observed changes to the population. This rate would be expected to reduce the future population size by slightly more than this amount. BEAK (1991) highlighted that the very strong year class of age-5 fish in 1990 reported by Dadswell and Themelis (1990b) was not consistent with the 46.3% YOY turbine mortality estimate (this year class would have passed the Annapolis TiGS in 1985), and further questioned this estimate for this reason. However, American Shad do have high recruitment variability, and in addition, due to the sampling methods (see TOR 6), the relative strength of the 1985 year-class may not be that well determined. While this turbine mortality estimate and the strong year class are inconsistent with each other, recruitment variability and/or sampling issues may, at least in part, account for this discrepancy.

Acute turbine mortality rates for 12 species at the Annapolis TiGS (Table 3: study number 11)

Separating fish mortality caused by handling and capture techniques from that caused by passage through a turbine is a fundamental problem in turbine mortality studies. When estimating turbine mortality using naturally-entrained fish, control fish are used to correct for capture and handling mortality. Determination of an appropriate duration for the control experiments is problematic because the length of time that the entrained fish are in the net is unknown. To address this issue, Gibson and Myers (2000, 2002a) proposed that, by varying the duration of the net deployment, logistic regression methods could be used to estimate the survival of fish that had not spent time in the net. They demonstrated the method using data for naturally-entrained fish at the Annapolis TiGS that were captured in the tailrace using ichthyoplankton nets.

During their study, nets were fished for varying lengths of time ranging from 15 minutes to five hours. Each time the catch was removed from the net, individuals were identified to species and assigned a live or dead status. The resulting data were then modelled using logistic regression to determine the effect of time in net on the resulting mortality rate estimate. The intercept from the model is an estimate of the mortality rate for fish that have not spent time in the net, which is interpreted as an estimate of the acute turbine mortality rate. Twelve species were captured in sufficient numbers to estimate turbine mortality rates. These ranged from 0.0% for Sea Lamprey to 23.4% for American Shad (Table 7). The reason that the estimate for American Shad is much higher than for the other *Alosa* is not known, but the estimate does have a large confidence interval associated with it (similar to some other estimates). Delayed turbine mortality rate estimates were also produced for some species as part of this study (Gibson and Myers 2000), although due to low sample sizes and because all components of handling mortality were not accounted for, the authors did not consider the estimates to be reliable.

Table 7. Estimates of acute mortality for 12 species of fish at the Annapolis Tidal Generating Station (adapted from Gibson and Myers 2002a). YOY = young-of-the-year; C.I. = confidence interval.

	Mortality (%)		
Species	mean	95% C.I. lower limit	95% C.I. upper limit
YOY American Shad	23.4	6.1	58.8
YOY Blueback Herring	8.1	3.5	17.2
YOY Alewife	7.7	1.5	31.4
YOY Atlantic Herring	15.7	10.8	22.1
Sea Lamprey	0.0	n/a	n/a
Blackspotted Stickleback	<0.1	<0.1	5.6
Atlantic Silverside	2.2	1.1	4.1
Northern Pipefish	2.2	0.7	6.4
Butterfish	8.7	1.7	34.5
Winter Flounder	5.8	0.8	31.2
Windowpane	8.8	<0.1	59.4
Hake (spp.)	8.7	0.3	20.9

Gibson and Myers (2002a) provided an overview of the strengths and weaknesses of their method. First, when fish are abundant, estimation of acute mortality using naturally entrained fish captured with nets in the tailrace is a relatively easy and cost effective method of obtaining data. Handling of fish is reduced and the method can be applied in situations where test fish are not readily available and estimates can be obtained for several species simultaneously without an increase in the required effort. The method developed by Gibson and Myers (2002a) can be applied with only slight changes to the field methods when sampling fish with nets in tailraces. Two aspects of turbine mortality that were not addressed in their study are mortality caused by sources other than the length of time in the net and delayed mortality. As described by Gibson and Myers (2002a), mortality that is the result of the capture process can be separated into three components: the proportion of fish that die while entering the net (e.g. fish that are impinged or abraded against the net), the proportion of fish that die as a result of time in the net (e.g. due to crowding or suffocation), and proportion of fish that die while being removed from the net (e.g. as a result of handling). Their regression model estimates the mortality rate if the time in the net is zero. The resulting estimates will be biased high if mortality associated with other aspects of the capture process is high. When the resulting turbine mortality rate estimates are low, the effect of this bias must also be low. When the resulting mortality estimate is high, the estimate should not be interpreted as turbine mortality without quantification of mortality from other components of the capture process (Gibson and Myers 2002a). Gibson and Myers (2002a) proposed a method to address this issue using fish as controls, but this method has not been tested. With respect to delayed mortality, their modelling approach can also be applied if fish are held in pens for a period of time prior to evaluating their live/dead status. However, holding facilities and handling protocols must not be a significant source of mortality if delayed mortality is to be reliably estimated (Gibson and Myers 2002a).

Similar to the 1985 and 1986 studies, if live fish are able to avoid the nets, the resulting turbine mortality rate estimates from the Gibson and Myers (2002a) method would also be biased high. The modified nets used in 1999 were expected to be more efficient than the nets used in 1985 and 1986, but the capture efficiency of the nets for live fish relative to dead fish is not known for any of these studies.

Summary of fish mortality studies

Both the observations of dead fish downstream of the turbine, and the studies that provide estimates of turbine mortality rates, provide evidence that fish mortality has occurred at the Annapolis TiGS. However, the per-capita rates of mortality for fish passing through the turbine are not well known for any life stage of any species encountered at the Annapolis TiGS. Some of the estimates are likely inflated due to high control mortality (>10%) resulting in inaccurate turbine mortality estimates (Ruggles et al. 1990) and delayed mortality has not been sufficiently addressed in any of the studies to date. Turbine mortality rates are expected to be variable among species, as evidenced by the results of Gibson and Myers (2002a). Additionally, with the exception of adult American Shad (Hogans 1987), turbine mortality studies are limited to species amenable to capture in one-meter-diameter nets. As such, no information is available for larger fish species and life stages at the Annapolis TiGS.

The estimates of the turbine mortality rates from Gibson and Myers (2002a) for 12 species, and from Hogans (1987) for adult American Shad are the best available at the Annapolis TiGS, although neither fully account for the capture and handling processes, and neither address mortality that does not occur during or shortly after turbine passage. Because most of the methodological issues described above would lead to overestimation of the acute turbine mortality rate for a single pass, the 1985-1986 mortality estimate for YOY clupeids is likely biased high.

The turbine mortality rate estimates from all studies are for a single pass through the turbine. Some species, especially species that are resident in the vicinity of turbine, have the potential to make multiple passes through the turbine throughout the course of the year. In this situation, the total mortality rate resulting from turbine passage is a function of the turbine mortality rate and the expected number of passes through the turbine. The number of passes that fish would be expected to make through the turbine has not been studied for any species.

TOR 6: STUDIES EVALUATING POPULATION-LEVEL IMPACTS OF FISH MORTALITY AT THE ANNAPOLIS TIGS

Population-level changes coincident with the construction and operation of the Annapolis TiGS have only been monitored for one population, Annapolis River American Shad, but have been inferred for Striped Bass based on recreational fishing records, and for Atlantic Sturgeon based on observations of dead fish.

American Shad

Studies of the American Shad population in the Annapolis River were completed by Melvin et al. (1985) in 1981 and 1982 to establish a pre-operational baseline of the population before the Annapolis TiGS was constructed. Post-operational studies were undertaken in 1989 and 1990 (Dadswell and Themelis 1990a, 1990b), about one generation (c. six years) after the turbine came online, and in 1995 and 1996 (Gibson and Daborn 1995a, Gibson 1996a), about two generations after the turbine came online. Gibson and Daborn (1995a) and Gibson (1996a) provided a comparison of life history characteristics before and after the construction of the Annapolis TiGS; these results are also summarized in Dadswell et al. (2018).

All three studies (1981/82, 1989/90, and 1995/96) recorded, mostly by sex, the mean and maximum observed length, the mean and maximum observed weight, the mean and maximum observed age, the percentage of repeat spawners, the mean age at maturity, vonBertalanffy growth rate parameters, and the instantaneous total mortality rate. Comparisons between the pre-operational study and post-operation studies indicate that there has been a decline in the mean size, mean age, maximum age, age at a maturity, and percent of repeat spawners, and an overall increase in the total mortality rate (Table 8).

Table 8. Comparison of selected biological characteristics of the Annapolis River American Shad population from assessments before (1981 and 1982) and after (1989, 1990, 1995, and 1996) the Annapolis Tidal Generating Station began operation. Values are not corrected for sampling selectivity. (adapted from Gibson and Daborn 1995a and Gibson 1996a). Numbers in brackets are one standard deviation.

Biological Characteristic	**Year**					
	1981	1982	1989	1990	1995	1996
Sample Size	309	195	416	620	534	831
Sex ratio (F:M)	0.86	1.82	1.5	1.6	2.9	1.4
Males:						
Mean length (mm)	459 (38.0)	463 (40.5)	450 (25.8)	442 (24.9)	412.7 (31.7)	414.3 (28.7)
Max. length (mm)	577	530	544	544	490	510
Mean age (y)	6.68	6.98	6.61 (1.06)	5.76 (1.14)	5.97 (1.11)	5.56 (1.03)
Max. age (y)	12	12	10	9	9	8
Repeat spawners (%)	76.5	80.0	87.9	82.5	53.0	55.3
Instantaneous total mortality rate	0.36	0.26	0.81	0.89	0.47	0.52
Females:						
Mean length (mm)	506 (42.8)	527 (37.6)	484 (26.9)	479 (25.1)	447.2 (28.8)	459.8 (30.4)
Max. length (mm)	602	605	587	554	525	540
Mean age (y)	7.34	8.28	6.87 (1.02)	6.03 (1.03)	6.24 (1.10)	6.32 (1.22)
Max. age (y)	12	13	11	9	10	10
Repeat spawners (%)	72.2	93.6	87.4	59.8	53.7	53.1
Instantaneous total mortality rate	0.25	0.22	0.85	0.93	0.56	0.49

Comparisons of these assessments must be undertaken with caution for several reasons. First, fish were captured with gillnets that are known to be selective and nets of the same size were not fished in all years. Second, *Alosa* runs are highly structured (Gibson et al. 2017, Chaput and Atkinson 2001) with older, repeat-spawning fish returning to the river earlier than younger, first time spawning fish. Therefore, the timing and location of sample collections have the potential to influence the summary statistics for the population characteristics depending on the timing of the run in that year. Third, the life history characteristics presented correspond to a snapshot in a particular year, and the resulting biological characteristics can therefore be influenced by recruitment variability. For example, an increase or decrease in the mean age could result from a strong recruitment event, rather than a change in the mortality rate. Fourth, the calculation of the instantaneous total mortality rate from the numbers-at-age was undertaken using log-scale least squares, a method that is known to be more biased than some other methods and its use is not recommended (Smith et al. 2012, Millar 2014, Gibson et al. 2017). Further, two assumptions are required for the estimation of the mortality rate from age-structured data collected in a single year to be valid: variability in recruitment rates needs to be small enough to have negligible effects and mortality rates need to be more or less constant through time. Neither assumption is valid for American Shad.

Using a simulation model, Gibson et al. (2017) showed that mortality estimates obtained from these methods are highly variable and biased for a closely related *Alosa* species and recommended that they not be used for *Alosa* species. Finally, if there were other things that changed during the fifteen years over which these assessments occurred that effected the population, these would incorrectly be attributed to the effects of the turbine. Changes in environmental conditions, natural mortality or fishing mortality would be expected to lead to changes in the biological characteristics.

Despite the limitations above, the studies still indicate that there have been changes to the population structure. Most notable is the change in the maximum age observed for both male and female American Shad between the pre- and post-operational assessments. The number of American Shad sampled in the post-operational assessments was higher, and in 1995 and 1996 occurred throughout the run and in spawning areas well upriver. As such, if older American Shad were present in the population those years they likely would have been observed. This change, which is indicative of higher mortality rates for adult fish, lends credibility to the decrease in size and mean age observed in the post-operational assessments.

While the changes in the total instantaneous mortality rates (which may not be accurately estimated) are not inconsistent with the 1986 turbine mortality rate for adult American Shad, it is not clear that the changes can be fully attributed to the turbine. Comparison of the average instantaneous total mortality rate from pre- operational studies in 1981 and 1982 to the average instantaneous total mortality rate from the four post-operation years (1989, 1990, 1995, 1996) indicates an increase of 0.36 for males and 0.47 for females, which may be due to turbine mortality at the Annapolis TiGS or other factors (e.g. changes in the fisheries). The turbine mortality rate for adult American Shad of 21.3% would be expected to increase the instantaneous total mortality rate by 0.24, a value less than the observed change in the population. Additionally, as discussed under TOR 5, the turbine mortality estimate may not be representative of the actual mortality rate of fish passing the causeway because adult American Shad have been observed using the new fishway and the potential exists for a fish to pass through the turbine more than one time. As such, the turbine mortality rate and the change in the total mortality rate are not directly comparable because they are metrics of different phenomena. In summary, while the changes observed in the American Shad population are roughly consistent with turbine mortality estimate for adult shad, for the reasons above it would be inappropriate to attribute the observed population-level changes solely to the construction and operation of the Annapolis TiGS based on the existing information. This said, an increase in mortality rates for adult American Shad would be expected to lead to changes in the biological characteristics of the population in the direction observed in these studies.

Striped Bass

Information for assessing the population-level effects of the Annapolis TiGS on the Annapolis River Striped Bass population is limited to information on length, weight, and catch and effort data from creel surveys, questionnaires, fishing contests, and private angling records; data which may not be representative of the native Annapolis River Striped Bass population. Dadswell et al. (2018) combined these records to provide a data set which contains information from the early 1970's through to 2008, grouped into pre- and post-operational assessments. Pre-operational records from all sources indicate the majority of the catches were comprised of large fish >4.0 kg with the proportion of >4.0 kg fish decreasing after construction of the Annapolis TiGS. A more detailed analysis of the private angling records, show that in addition to a decrease in the proportion of >4.0 kg fish, the overall CPUE and average size of fish has also declined. Dadswell et al. (2018) attribute the loss of large Striped Bass to size-selective turbine mortality: large individuals are more likely to be struck by turbine blades, selectively removing large individuals from the population.

There are two important factors that need to be considered when evaluating the population-level impacts of the Annapolis TiGS on Striped Bass. First, some of the changes to the Annapolis River Striped Bass population occurred before the construction of the turbine. Second, the assemblage of Striped Bass in the Annapolis Basin consists of Striped Bass from more than one population, although the actual composition is not known and likely has varied through time.

Prior to the construction of the Annapolis TiGS, there was concern with the egg survival of Striped Bass in the Annapolis River. While eggs were obtained from the Annapolis River in several years between 1976 (Williams et al. 1984) and 1994 (Jessop 1995), the last larvae captured was in 1976 (Williams et al. 1984). Seine surveys in 1976 (Williams et al. 1984), 1980 (Jessop 1983), 1985 (Stokesbury 1985), 1986 (Stokesbury 1986), 1992 (Jessop 1995), 1993 (Gibson and Daborn 1993), 1994 (Gibson and Daborn 1995b), 2009 ,2010, 2014 and 2015 (Clean Annapolis River Project, unpublished data) have failed to capture any young-of-the-year Striped Bass. Together, these results provide evidence that Striped Bass have not been successfully reproducing in the Annapolis River since at least the mid-1970's. COSEWIC (2012c) concluded that the Annapolis River Striped Bass population is extirpated, in part based on this evidence.

Creel surveys in 1976 indicated an increase in the size of fish which is consistent with reproductive failure as it is expected to lead to increase in the average size of fish until the last fish are gone (Jessop and Doubleday 1976). However, a creel survey undertaken after the installation of the turbine identified smaller, younger fish than earlier creel surveys (e.g. Harris 1988). The presence of these younger fish in conjunction with the evidence of reproductive failure is suggestive that these fish may be from other populations. Both genetic and tagging studies have shown that Striped Bass undergo coastal migrations and forage in estuaries other than those at the mouths of their natal rivers (Waldman et al. 1990, Wirgin et al. 1995). Striped bass tagged elsewhere, including USA rivers, have been captured in the Annapolis River estuary (Waldman et al. 1990), and Striped Bass tagged in the Annapolis River have been recaptured in other areas (Harris 1988). In both cases, the actual origin of the fish is not known and the number of recaptured fish is small. The relative abundance of these natal and non-natal fish in the Annapolis River estuary would be expected to vary depending on many factors including the abundance decline of the Annapolis River population and the abundance of fish in other populations. Based on this information, the Annapolis TiGS is not the only factor that would contribute to the documented changes to the size distribution of Striped Bass caught in the recreational fishery in the Annapolis River. The present composition of the Striped Bass assemblage in the Annapolis River is not known.

This said, there is evidence that the Annapolis TiGS kills Striped Bass. Turbine mortality would be expected to hasten the decline in abundance of a native population, and also has the potential to limit population growth if the river is recolonized. However, the paucity of information about the rates of mortality of Striped Bass moving past the causeway precludes an evaluation of the population-level effect of the Annapolis TiGS on the Annapolis River Striped Bass population.

Atlantic Sturgeon

Information about the effects of the TiGS on Atlantic Sturgeon in the Annapolis River is comprised of 21 records of mortality attributed by Dadswell et al. (2018) to turbine passage at the Annapolis TiGS from 1985 to 2018. Maturity was determined for 11 of the 21 mortalities and consisted of four ripe females, three spent females, two ripening males, one spent male and one immature animal (corrected from Dadswell et al. 2018 – Personal communication Michael Dadswell).

The major question regarding the impacts of the Annapolis TiGS on Atlantic Sturgeon is whether or not there was, and still is, a population native to the Annapolis River. There is a paucity of information about this species in this river, and similar to Striped Bass, sturgeon encountering the Annapolis TiGS may be part of a native population or else may be transient fish from other populations. Atlantic Sturgeon are known to have a large home range and their foraging habitat includes foraging habitat in non-native rivers (e.g. Savoy and Pacileo 2003) .

Records of Atlantic Sturgeon in the Annapolis River are few. One publication, Dadswell et al. (2018), states there are no records of an Atlantic Sturgeon population native to the Annapolis River estuary pre-operation of the generating station. However, historically they were fished commercially further out in the estuary, and Huntsman (1922) states: “They [Atlantic Sturgeon] are known also in the Annapolis river, which they are said to ascend as far as Middleton.” Although neither of these records definitely demonstrate that there is (or was) a population native to the Annapolis River, they strengthen the argument that there may have been.

Dadswell et al. (2018) attributes the presence of ripe and spend individuals as evidence of a spawning population due to the fidelity Atlantic Sturgeon have to their natal river, a population that was not known prior to the construction of the turbine. However, particularly given their large home range, the origin of these fish collected near the Annapolis TiGS is not known. Given the small sample size (11 fish for which maturity was determined), it is not clear if these fish are sufficient as evidence of a spawning population. Young-of-the-year sturgeon (indicative of local reproduction) have not been captured in the Annapolis River, although there have not been any surveys designed specifically for this species. The presence of a native population at this time is not definitively known.

Overall, a population-level impact cannot be determined from this information. If the animals are fish from the Saint John River population or some other populations, the effect on those populations is likely quite low. Dadswell et al. (2017) estimated that the size of the Atlantic Sturgeon population in the Saint John River is similar to its size over 100 years ago and currently supports a small fishery compliant with a non-detrimental finding by the Convention on International Trade in Endangered Species. However, if there is a population native to the Annapolis River, the effects of turbine mortality on a native population would be expected to be higher as all animals in the population would be expected to move past the causeway several times throughout their lives, potentially more than one time during the year.

Summary of population studies

For American Shad, comparison of population characteristics between studies conducted pre-operation and post-operation of the Annapolis TiGS indicate a decrease in the oldest captured age, the overall length, and percentage of repeat spawners. While comparison of the studies is undertaken with some uncertainty, the overall patterns in many of the population characteristics indicate that there have been changes in the Annapolis River American Shad population since the Annapolis TiGS has been in operation.

With respect to Striped Bass, there is evidence that the Annapolis TiGS kills fish. However, the Annapolis River population is considered extirpated by COSEWIC, with spawning failure (failure of eggs to hatch and/or very young fish to survive) over several decades as evidence for their extirpation. Turbine mortality at the Annapolis TiGS would have been expected to hasten the decline in abundance, although the magnitude of this effect is not known. If Striped Bass are able to recolonize the river, the effect that the Annapolis TiGS would have on the recolonization rate and the population's ability to persist is not known. As discussed under TOR 3, the effect of turbine mortality on other Striped Bass populations is thought to be low (with moderate uncertainty) due to the small proportion of these other populations thought to utilize habitat in the vicinity of the Annapolis TiGS.

The major question regarding Atlantic Sturgeon is whether or not there is a population native to the Annapolis River. If the Atlantic Sturgeon in the vicinity of the TiGS are primarily fish from other populations (e.g. the Saint John River population), the population-level impacts are likely low, whereas if there is a population natal to the Annapolis River, the impacts would be expected to be greater.

TOR 7: FISH DIVERSION STUDIES AT THE ANNAPOLIS TIGS

Fish diversion systems can be separated into two categories: physical and behavioral (Popper and Carlson 1998). Due to small size of many fish species and life stages, as well the large volume of water moving through the Annapolis TiGS physical barriers have been considered impractical. To date, there have been two studies investigating the application of behavioral barriers to fish movement at the Annapolis TiGS fish diversion studies of adult and juvenile *Alosa* in 1988 and 1989, and the 1999 fish diversion of young-of-the-year *Alosa*. Currently, fish diversion systems are not in operation at the Annapolis TiGS.

Fish diversion studies in 1988 and 1989 (Table 3: study number 5)

McKinley and Patrick (1988) tested the effectiveness of sound and light fish diversion technologies at the Annapolis TiGS during June-July and September-October of 1987. The first phase of the study focused on adult *Alosa* with the second phase focusing on juveniles. Two devices were evaluated for their effectiveness as an attractant with the intent of increasing the usage of the fishways. The first device, known as a fish drone, emitted bursts of sound, while the second device used a filtered mercury vapour light to attract fish. Preliminary results from phase one using the fish drone indicated that the background noise from the existing infrastructure was too great to make the fish drone an effective attractant and it was subsequently dropped from further assessment. The third technology tested, a “fish hammer”, also employed sound, but as a deterrent away from the turbine instead of an attractant. The “fish hammer” uses repetitive high energy low frequency sound as a deterrent. The effectiveness of the devices was tested by using fixed hydro-acoustic surveys to determine if fish activity increased or decreased.

The hydro-acoustic surveys were supplemented with gillnetting to determine species, although with limited success. American Shad, Alewife and Blueback Herring were the most common species identified in the gillnet sampling during phase one, while only Alewife and Atlantic Menhaden were identified during phase two. Because the hydro-acoustic surveys are only able to determine relative changes in fish activity, the results of the fish diversion study are not species specific. The fish distribution portion of the study determined that the majority of fish activity occurred at night and therefore all experiments were run during the night time pre-generation and generation period.

Overall, McKinley and Patrick (1988) determined that the filtered mercury vapour lights did not elicit a strong behavioural response attracting adult fish (phase one) or juveniles (phase two) towards the fishway. There was a minimal increase in activity 3–13 m in front of the light source, and the authors propose that, due to limited light penetration into the water, a longer period of time may be needed for fish to adjust to the light levels. This was not tested. The fish hammers were determined to be the most effective device, indicating a decrease in fish activity in an area more than 15 m from the hammer. The response was greatest for adult fish, with only minimal avoidance indicated for juvenile fish, which the authors propose may be due to a lower fish density during phase two. What constitutes an effective device is only loosely defined in the study, without associated quantitative values.

As a follow up study, McKinley and Kowalyk (1989) examined the effectiveness of a sonic barrier at repelling adult (phase one) and juvenile (phase two) *Alosa* from the turbine intake at the Annapolis TiGS. Similar to the previous study, hydro-acoustics were used to determine the amount of fish activity in the turbine forebay and sluiceway areas at the Annapolis TiGS. During phase one (June-July 1988), four fish hammers were placed equidistant across the turbine intake approximately three meters in front of the turbine wall at a depth of 0.75 meters. Each of the four fish hammers emited a high-energy, low-frequency sound at a rate of 20 pulses per minute. All experiments were run during the evening or night when fish activity was highest and encompassed the time two hours pre generation to two hours into generation. Effectiveness of the barrier was determined by alternating five minute periods of hammers on to a control period where the hammers were off. An effectiveness index was then calculated using the difference between the number of echoes detected when the hammers were on versus off.

During phase two, three fish hammers were used instead of the four used during phase one. The authors report a minimum 70% effectiveness index for adult fish in phase one and an effectiveness index ranging from 48% to 66% for juveniles in phase two. These reported values are likely overestimating the effectiveness of the device, as they do not consider the motivational state of the fish. *Alosa* caught during the study are migrating downstream and not residents. Given the short time frame of the test (five minute periods), the reduction in detections may reflect a startle response by the fish, but it is unclear if this translates to increased fishway usage. Gill nets were used to determine the species composition near the generating station during phase one of the study. Greater than 50% of the individuals caught were either Alewife or American Shad, with the remaining catch including Cunner, Atlantic Mackerel, Tomcod, flounder spp., Striped Bass, Blueback Herring, and White Hake. Larval tow nets were deployed during phase two but were unsuccessful. Overall the study concludes that the use of a sonic fish protection scheme was successful in reducing the amount of fish activity near the turbine intakes for both adult and juvenile fish.

McKinley and Kowalyk (1989) provided a more in depth assessment into the effectiveness of using fish hammers to deter fish than the previous study of McKinley and Patrick (1988). The design of the study emphasized comparing short periods of the fish protection scheme on, to control periods where it was off. This provides a good comparison, as all other effects (e.g. time of year, environmental effects, etc.) are held constant, but how this translates to long term effectiveness is unclear. Acclimation to the fish hammers which may limit the usefulness of the device and the potential for the sound to act as an attractant for some species was not addressed in this study. Additionally, the hydro-acoustic transducers were generally detecting small numbers of fish, but at high densities of fish they are unable to distinguish individuals. This limits the ability of the study to quantify the effectiveness of the device at high fish densities. Effectiveness estimates were calculated without consideration of sampling errors. In both these studies, the hydroacoustic monitoring did not allow for species to be differentiated, and which species were responding remains unclear. Overall the authors provide evidence that there is, at least on the short term, a behavioural response of the fish to sound but the effect on the number of fish travelling through the turbine versus the fishways was not determined in this study. Although effectiveness was not well quantified in this study, the fish hammers remained in use at the Annapolis TiGS for several years after the initial study.

Fish diversion studies in 1999 (study number 11)

Gibson and Myers (2002b) tested the effectiveness of an ultrasound fish diversion system to reduce fish passage through the turbine at the Annapolis TiGS. Young-of-the-year *Alosa* were the target organisms in this study, but the effectiveness of the diversion systems was also tested for other species present in the estuary, albeit with little expectation of success (most fish cannot detect high frequency sound). The diversion system used was a band-limited, random noise signal with most of the energy focused between 122 and 128 kHz. The signal was projected into the turbine forebay on randomly selected generation cycles. Effectiveness was assessed by monitoring fish passage in the tailrace and the two fishways, between September 7th and October 12, 1999. Sampling occurred during 28 generation cycles with the fish diversion system turned off, and 20 generation cycles with the fish diversion system turned on. Catch was then modeled as a function of the on/off status of the fish diversion system controlling for environmental effects. Effectiveness of the system was defined as the ratio of the mean number of fish captured with the diversion system turned on to the mean number of fish captured with the diversion system turned off. Results were reported for fish passing through the fishways and through the tailrace. Unlike the previous study, this method of estimating effectiveness allowed for species-specific results. Gibson and Myers (2002b) estimated that the diversion system reduced the rates of passage of American Shad and Alewife by 42% and 48%, respectively.

When the diversion system was turned on, the catch in the new fishway increased by 3.6 times for American Shad, and by 4.1 times for Alewife. Results for Blueback Herring were less clear: catches in the tailrace decreased by 49% when the diversion system was on, but only at low abundance. The system was not effective for the other species encountered during the study.

Gibson and Myers (2002b) provided species specific estimates of effectiveness with measurement error. The design of the study allowed for direct measurement of both turbine and fishway passage rates, thereby evaluating whether fish deterred from passing through the turbine could find the fishways. As described by the authors, the study provided a real-world example of testing a fish diversion system, and the effectiveness of the system could likely be improved by modifying the sound field based on the results of this study. The system was not tested for adult fish, and the reduction in effectiveness at higher abundance for Blueback Herring raises questions about its overall utility. Further testing would therefore be required before using it in the long term.

Summary of Fish Diversion Studies

Two studies at the Annapolis TiGS that have examined the effectiveness of four fish diversion technologies for reducing the passage of fish through the turbine: a fish drone and filtered mercury vapour lights to attract fish to the fishway; and fish hammers and ultrasound to deter them away from the turbine intake. McKinley and Patrick (1988) determined that a fish drone and filtered mercury vapour lights were not effective in attracting fish towards the fishway. McKinley and Kowalyk (1989) suggest fish hammers may be effective at deterring fish away from the turbine but several issues such as the potential for acclimation, species identification and a need for an improved measure of effectiveness would need to be addressed before this technology could be recommended.

Gibson and Myers (2002b) suggest that the use of an ultrasound fish diversion at the Annapolis TiGS shows promise for reducing turbine passage and increasing fishway usage for *Alosa*. Further study into the placement of the system and longer term effects would be useful to determine if a sound based diversion system provides a significant benefit at the Annapolis TiGS. Other technologies would be needed for the non-*Alosa* species. As described by Popper and Carlson (1998), the effectiveness of a behavioral guidance system depends on the time of year, the time of day, the age of the fish, the flow field, environmental conditions, and the "motivational state" of the fish, and as shown by Gibson and Myers (2002b) the abundance of fish. These factors all need to be considered when developing and testing behavioral guidance systems.

TOR 8: RESEARCH RECOMMENDATIONS

Fisheries and Oceans Canada has developed a sustainable fisheries framework that provides the basis for ensuring that Canadian fisheries support conservation and sustainable use of fisheries resources. As part of this framework, Fisheries and Oceans Canada (DFO) has adopted a fishery decision-making framework incorporating the precautionary approach (DFO 2006) that is to be used where decisions regarding commercial, recreational, or subsistence controls on harvest and all other removals are required. The primary components of the precautionary approach framework are reference points and stock status zones (healthy, cautious and critical) intended to guide decision-making. For each stock, a removal reference rate is defined relative to the stock status zones as the maximum acceptable removal rate from all types of fishing and other human activities. For species found in the vicinity of the Annapolis TiGS, identification of reference mortality rates associated with passage at the Annapolis TiGS would ensure that mortality rates are consistent with DFO's sustainable fisheries framework and could help guide regulatory decisions under the Fisheries Act and Species at Risk Act. It would also help guide decisions about future research requirements, because determining whether a population-level impact is below a reference level is a very different question than determining the magnitude of the population-level impact itself. In this context, reference mortality rates would aid in prioritization of specific questions and would determine the resolution required from experiments undertaken to determine whether fish passage survival objectives are being met.

More generally, information about the fish community is dated. Most of the data collected about fish community and turbine mortality at the Annapolis TiGS were collected in the 1980's and 1990's, with the most recent field study occurring in 1999. Population abundances and species assemblages may be different today than they were when these studies were conducted. Additionally, most of the information about the fish community was collected during the late summer and fall, with little to no information about the community during winter and spring. Updating this information is best done concurrently while addressing other research questions.

With respect to evaluating population-level impacts for fish at the Annapolis TiGS, the state of knowledge for the highest priority species identified in TOR 3 is provided in Table 9. Depending on what is considered an acceptable mortality rate, addressing one or more of the information gaps might be sufficient to determine whether mortality rates are within acceptable limits. For example, if the proportion of a population encountering the Annapolis TiGS is less than the acceptable mortality rate then further research would not be required to determine that the mortality occurring is within the acceptable limit.

There have been significant improvements in technology and methods since most of the research was carried out at the Annapolis TiGS, most notably acoustic tracking technology. Fish passage usage, immediate and delayed turbine mortality rates, residency times and the number of times that fish move past the causeway can now be evaluated using acoustic tracking equipment (e.g. Suzuki et al. 2017, Dodd et al. 2018). These methods are well developed for larger-bodied robust fish such as Atlantic Sturgeon, Atlantic Salmon, American Eel, and Striped Bass, and methods are continuously being improved for other species such as the adult *Alosa*. Tracking technologies are not yet developed for smaller-bodies animals such as the YOY *Alosa*. As such, for these species, capture with nets may be the most appropriate method, with emphasis placed on reducing the effects of capture and handling or of better accounting for it (Gibson and Myers 2002a).

During this review, COSEWIC designations and commercial, recreational or aboriginal (CRA) fisheries were used to identify the highest priority species for further research. This species prioritization could be further reviewed and refined prior to taking on additional studies. For example, only one Atlantic Wolffish was captured (in 1993) during these studies and hake were generally not identified to species. White Hake and Atlantic Wolffish may not be priorities for these reasons.

There are three species for which the level of impact was deemed to be high or extreme, but with high uncertainty. These are Annapolis River populations of Striped Bass, Atlantic Salmon, and Atlantic Sturgeon. For Atlantic Salmon, the key source of uncertainty is survival of smolts and adults migrating past the turbine. These are also unknown for Striped Bass and Atlantic Sturgeon. For Striped Bass, a key source of uncertainty is the status of the Annapolis River population and whether the river could sustain a production of Striped Bass under current conditions. For Atlantic Sturgeon, the presence and status of an Annapolis River population is a key source of uncertainty.

For populations for which the population-level impact is categorized as low, but with high or very high uncertainty (TOR 3), the major source of uncertainty is the population structure and whether a significant portion of the population would be exposed to the turbine. For these species, further research about population units and utilization of the Annapolis River estuary could be prioritized over research about survival passing through the turbine.

Research with respect to fish diversion and guidance systems at the Annapolis TiGS might best follow after sufficient information is obtained about the impacts of the station on specific species and whether they are within acceptable limits. Selection of fish diversion technologies (or other strategies to reduce impacts) for evaluation will depend on the focal species, as well as the magnitude of the effect required to bring mortality rates to within acceptable limits. New variants of deterrent systems such as acoustic barriers (Gibson and Myers 2002b, Jesus et al. 2018), and lighting devices (Ford et al. 2017, Elvidge et al. 2018) have been investigated and show promising results.

Table 9. Information available for assessing the population-level impacts of the Annapolis TiGS (Y = sufficient information, N = information gap). YOY = young-of-the-year.

Species	Population definition	Life history / resiliency	Proportion of the population that would encounter the turbine	Timing of turbine mortality relative to life history events	Rate of fish passage usage and survival during passage	Acute turbine mortality rates	Delayed turbine mortality rates	Expected number of passes through the turbine	Rates of other human-induced mortality	Reference level defining an acceptable mortality rate
Alewife	Y	Y	Y	Y	N	Y (YOY only)	N	N	Y	N
American Eel	N	?	?	Y	N	N	N	N	N	N
American Shad	Y	Y	Y	Y	N	Y	N	N	N	N
Atlantic Menhaden	N	N	N	N	N	N	N	N	Y	N
Atlantic Salmon	Y	Y	Y	Y	N	Y	N	N	Y	N
Atlantic Sturgeon	N	Y	N[1]	N	N	Y	N	N	Y	N
White Hake	Y	Y	Y	N	N	N	N	N	N	N
Lumpfish	Y	Y	Y	N	N	N	N	N	N	N
Rainbow Smelt	Y	Y	N	N	N	N	N	N	N	N
Spiny Dogfish	Y	Y	Y	N	N	N	N	N	N	N
Blueback Herring	Y	Y	Y	Y	N	Y (YOY only)	N	N	Y	N
Atlantic Wolffish	Y	Y	N	N	N	N	N	N	N	N
Striped Bass	Y (more than one population)	Y	Y for the Annapolis; N for others	Y	N	N	N	N	N	N

[1]If there a population natal to the Annapolis River, the proportion would be one for that population. For other populations the proportion is unknown.

SOURCES OF UNCERTAINTY

With respect to understanding the impacts of the Annapolis TiGS, there are many information gaps pertaining to fishway utilization, turbine mortality rates, and fish behavior in the vicinity of the TiGS that preclude definitive statements about its impacts (Table 9). These are described throughout the document. Additionally, the information about the fish community at the Annapolis TiGS is dated. Species composition and relative abundance may have changed since the last field tudy was conducted in 1999. As such, the information presented in this document reflects the current state of knowledge which may not be equivalent to the current state of the Annapolis River system. No studies occurred during winter and few in spring. Species expected to be most prevalent during these seasons are not well represented in the available studies.

With respect to the impact analysis provided under TOR 3, impacts were evaluated at the level of a closed population. This is appropriate in the context of an objective of maintaining the overall productivity of these fish stocks. However, the results would be expected to be different if a different objective was used. For example, if maintaining abundance locally to support a recreational fishery was the objective, say, then a high level of turbine mortality would result in a high level of impact, even if only a small portion of the population encountered the turbine. Similarly, if the management objective is to maintain a trophy fishery, a decline in the local population of older and larger individuals would be categorized as a high impact despite a potentially low impact to the population as a whole.

The focus of this document is turbine mortality at the Annapolis TiGS and its associated population-level impact. While this focus does address the effects of direct mortality, it does not fully address the impacts of this generation station on fish populations because there is the potential for indirect effects as well. Topics such as changes in habitat quality and quantity, prey availability, or sediment transport which may affect the productivity of clam fisheries in the Annapolis Basin, are not addressed herein. These topics warrant further consideration.

SUMMARY

Studies that pertain to turbine mortality, fishway utilization, fish diversion or population-level effects at the Annapolis TiGS were identified during this review. In total, 24 documents (including a previous review) were identified that are directly related to these themes. These were grouped as 12 studies for the purposes of this review. Together, they describe a diverse fish community consisting of at least 40 species. Research is dated; the most recent field study was completed in 1999. During the mid-1990's, when much of the research was conducted, the abundance of many species was high and the number of fish migrating past the Annapolis TiGS annual at that time is thought to be in the low millions.

There are three routes of passage for fish moving downstream at the Annapolis TiGS: the new fishway, the old fishway and through the turbine tube. Studies of fishway utilization are mostly based on relative catches of fish from ichthyoplankton nets at the three possible routes through the Annapolis causeway which do not fully quantify the proportion of the fish using each route. The studies do indicate that, with the exception of Atlantic Silverside, the majority of individuals pass downstream through the turbine.

There is evidence of fish mortality occurring at the Annapolis TiGS. This evidence includes observations of dead fish downstream of the causeway, and studies of the mortality rates of downstream migrating fish. Survival estimates are limited to experiments that quantify mortality rates shortly after passage through the turbine, and with the exception of adult American Shad, are limited to small fish amenable to capture in ichthyoplankton nets. From a study in 1999, mortality rate estimates range from 0.0% for Sea Lamprey to 23.4% (95% C.I. = 6.1% - 58.8%)

for young-of-the-year American Shad. Estimates of the mortality rate of 46.3% for young-of-the-year clupeids from the mid-1980's are questionable due to methodological issues associated with determining the cause of death.

The Annapolis River American Shad population is the only population for which a comparison of their biological characteristics before and after Annapolis TiGS became operational is available. Annapolis River American Shad assessments were undertaken before the Annapolis TiGS began operation, and then were continued at intervals of roughly one generation and two generations after operation began. Although differences in methods among the assessments preclude rigorous quantitative comparison of the results, the results do indicate a decrease in the oldest age captured, the size of the fish and the percentage of repeat spawners in the spawning run. Although the results are roughly consistent with a turbine mortality rate estimate for adult American Shad (21.3% from Hogans (1987)), the changes cannot be wholly attributed to the turbine without knowledge of fish passage usage as well as other changes that would affect their survival that may have occurred during the 15 years over which these assessments occurred.

Recreational fishing records of Striped Bass in the Annapolis River have indicated a decline in the proportion of large fish and average length of the fish. The reported decrease in the average size of Striped Bass caught in the recreational fishery in the Annapolis River is not evidence of a population-level change wholly attributable to the turbine. Studies have shown that Striped Bass undergo coastal migrations and forage in estuaries other than those at the mouths of their natal rivers. Striped bass tagged elsewhere, including USA rivers, have been captured in the Annapolis River estuary, and Striped Bass tagged in the Annapolis River have been recaptured in other areas. Additionally, directed surveys have demonstrated a lack of successful spawning (absence of young-of-the-year) in this river. The present composition of the Striped Bass assemblage in the Annapolis River is not known.

Fish diversion systems, intended to deter fish from passing through the turbine have been tested at the Annapolis TiGS. The use of sound has shown some promise for young-of-the-year *Alosa*, but it is presently unclear whether behavioral stimuli can be developed for many of the species and life stages present at the Annapolis TiGS.

As a result of technological advances in fish tracking, methods now exist for studying fish behavior in the vicinity of the Annapolis TiGS. For fish large enough to carry acoustic tags, these methods could be used to determine fish passage usage and survival at the Annapolis TiGS and could potentially help address questions surrounding population structure. For species and/or life stages that are too small to carry tags, further research pertaining to the effects of capture and handling on the survival of fish, as well as to the capture efficiency of the nets being used to monitor fish passage is necessary to better understand the effects of the Annapolis TiGS on these species and life stages.

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APPENDIX: SPECIES DESCRIPTIONS

Background information about the species found in the vicinity of the Annapolis Tidal Generating Station (Annapolis TiGS) is provided in this appendix. This information forms the basis for the qualitative risk assessment summarized under TOR 3 in the main document.

For each species a brief summary of the species distribution, commercial and recreational fishing status, COSEWIC and IUCN designations, and their life history is provided. An overview of information about the species in the vicinity of the Annapolis, such as capture frequency and turbine mortality rate estimates, is also provided if available. Mortality rate estimates provided in the species summaries are from one of three sources; Gibson and Myers (2002a), Stokesbury and Dadswell (1991) or Hogans (1987). When multiple estimates exist, the best available estimate was provided. Further information about the mortality rate estimates is provided in the main document under TOR 5. Resilience estimates were obtained from fishbase.com which uses intrinsic rates of population growth, von Bertalanffy growth coefficients, fecundity, generation time, and longevity, with set thresholds recommended by the American Fisheries Society to categorize values as high, medium, low, or very low to determine an overall estimate of resilience (Musick 1999). Information about the relative abundance of a species is a qualitative assessment of abundance in the vicinity of the Annapolis TiGS, and is based on the frequency that fish of that species are captured or observed at the Annapolis TiGS, as well as the likelihood they would be captured or observed with the monitoring equipment used in studies at this generating station.

The information for each species is used for a qualitative assessment of the risk the Annapolis TiGS poses to the fish populations that encounter this generating station. As described in the main document under TOR 3, risk categories are defined based on expected population-level abundance decline, and uncertainty categories are defined based on how well factors such as the proportion of the population that would encounter the generating station, behavior of fish in the vicinity of the station, proportion of the population that use the fishways, and turbine mortality rates are understood. Category definitions are provided in the main document under TOR 3.

Acronyms used in this Appendix are:

COSEWIC: Committee on the Status of Endangered Wildlife in Canada.

CI: Confidence interval. Note: Confidence intervals are about repeatability of the experiment. A 95% confidence interval is interpreted as meaning that if the same experiment was repeated 100 times, the resulting parameter estimate (e.g. the turbine mortality rate) would be expected to fall within the confidence interval 95 times, and outside the confidence interval 5 times. It does not say anything about the true value of the parameter. For example, if the experiment has inherent biases, these biases would occur throughout, if the same experiment was repeated over and over again.

IUCN: International Union for the Conservation of Nature.

SARA: Species at Risk Act.

YOY: Young of the year.

Alewife (*Alosa pseudoharengus*)

Level of impact (Uncertainty):	Medium (Moderate)
Life stages present:	YOY, juvenile, adult
Relative abundance:	High
Usage:	Upstream spawning migrations of adults in spring; Downstream migration of YOY in summer and fall; Nursery area for YOY in summer and fall
Turbine mortality rates:	Available for YOY only; Acute only Best available estimate for YOY is 7.7% (95% CI:1.5%-31.4%)
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Commercial, recreational and aboriginal (limited fishing in the Annapolis River and estuary)

The Alewife (*Alosa pseudoharengus*) is an anadromous river herring species that lives mostly at sea but enters freshwater habitats to spawn (Scott and Scott 1988). They live coastally in waters less than 100 m in depth in the Northwest Atlantic, spanning from Newfoundland to North Carolina (Loesch 1987, Scott and Scott 1988). Alewife home primarily to natal rivers to spawn, and as such, individual rivers tend to have unique populations (McBride et al. 2014). Populations often mix while at sea, with some individuals tagged in the Bay of Fundy having travelled as far south as North Carolina (Dadswell et al. 1986).

Alewife are fished commercially in Atlantic Canada under the name “gaspereau”, which includes both Alewife and Blueback Herring, using square, dip, gill, and trap nets (Gibson et al. 2017). They are also fished recreationally (NSDFA 2018). They are used for human consumption as well as a bait fish. Alewife have not been assessed by COSEWIC and are currently designated as least concern on the IUCN Red List (NatureServe 2013b).

The spawning migration of Alewife is triggered by water temperature (Loesch 1987). They begin moving inland toward freshwater when temperatures reach between five and 10 °C, which in the Bay of Fundy tends to be late April to early May. Spawning activity may last for up to two months once it has begun (Scott and Scott 1988). They spawn in slow-flowing, freshwater sections of streams or in ponds and lakes. Alewife mature between ages three to six with almost all fish having matured by age five (Loesch 1987; Scott and Scott 1988). They may spawn five to six times in their lives. During spawning runs, the older and larger fish generally migrate earlier in the season, although young fish (three to five years) are the most abundant. Alewife are a productive species; the maximum lifetime reproductive rate is estimated at 19.2 replacement spawners per spawner (Gibson et al. 2017), which is high relative to other fish species (Myers et al. 1999). Based on their life history characteristics, Alewife are considered to have a medium resilience to additional mortality (Froese and Pauly 2018).

In the Annapolis River system, adult Alewife spawn in the river during spring and then move back to sea quickly following spawning. Young-of-the-year Alewife may remain in the system until November (Gibson and Daborn 1995b). Information is available about fish passage utilization, turbine mortality rates and fish diversion for Alewife at the Annapolis TiGS. Throughout the studies, Alewife have been caught in high abundance (e.g. Gibson and Daborn 1995b, Stokesbury and Dadswell 1991) and have been recorded as traveling through the causeway using both the new fishway and the turbine. The old fishway does not appear to pass a large proportion of Alewife. Two studies provide different estimates of the acute turbine mortality rates for Alewife at the Annapolis TiGS (see TOR 5), but neither of these studies fully quantifies the mortality rates. Fish diversion methods have been tested on Alewife and Alewife do respond to high-frequency sound (Dunning et al. 1992, Gibson and Myers 2002b).

Population-level impacts are assessed as medium given the well-defined population structure with all of the population likely to encounter the Annapolis TiGS, and acute turbine mortality rate estimates that could lead to a decline in population size greater than 10%. However, information about fishway usage is limited and delayed mortality and movement patterns in the vicinity of the turbine are not known, leading to moderate uncertainty in the conclusion.

American Eel (*Anguilla rostrata*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Elvers, juveniles, adults
Relative abundance:	Moderate
Usage:	Resident (at least summer and fall); Upstream and downstream migration
Turbine mortality rates:	Unknown (most eel passing through the turbine at the Annapolis TiGS were captured alive)
Resiliency:	Low
COSEWIC/IUCN designation:	Threatened / Endangered
Fisheries:	Commercial, recreational and aboriginal in the Annapolis River and estuary

The American Eel (*Anguilla rostrata*) is a catadromous, semelparous species, which spends most of its life in fresh water (Scott and Scott 1988). Both juveniles and adults are present in most bodies of freshwater with a connection to the Atlantic Ocean, spanning from northern South America to Greenland. All adults migrate and spawn in the Sargasso Sea, resulting in one large panmictic population (Schmidt 1923, Avise et al. 1986, COSEWIC 2012a, Côté et al. 2013). Eels are fished commercially in Canada at a number of different life stages (elvers, yellow eels, silver eels) and are often caught recreationally as well. They are also of significant value to aboriginal communities, in particular the Mi'kmaq, who have fished them for subsistence for thousands of years and continue to carry on these traditions today (COSEWIC 2012a). They have been designated as threatened by COSEWIC (COSEWIC 2012a) and endangered by the Ontario Endangered Species Act (S.O. 2007, c. 6) and the IUCN Red List (Jacoby et al. 2017).

American Eel have many sensitive life stages that are easily affected by habitat degradation or barriers, overfishing, pollution, and climate change (COSEWIC 2012a). Adult eels from throughout their range migrate to the Sargasso Sea to spawn between February and August, with males leaving to spawn earlier than females (Schmidt 1923, Scott and Scott 1988, COSEWIC 2012a). Upon hatching, transparent larvae called leptocephali passively migrate northwestward, transforming into glass eels, their pre-juvenile state, once they reach the continental shelf. As the glass eels continue to move towards fresh water and into the mouths of estuaries, they begin to develop pigmentation and are called elvers (Scott and Scott 1988, COSEWIC 2012a). They remain as elvers for 3 to 12 months until they fully enter fresh water where they metamorphose into their juvenile state and eventually develop into reproductive adults ready for migration back the Sargasso Sea to spawn. American Eels have a generation time of 22 years although in some regions where they do not reach fresh water, the generation time is as short as nine years (COSEWIC 2012a). In Canada, the average size of mature females ranges from 76 to 102 cm and males average 61 cm (Scott and Scott 1988). Based on their life history characteristics, population resiliency is considered low (Froese and Pauly 2018).

American Eel have been caught in a number of studies in the Annapolis River and estuary in relatively high abundance (e.g. Daborn et al. 1979, Stokesbury 1985, Gibson and Daborn 1995b). There have been very few recorded mortalities at the Annapolis TiGS (Dadswell and Rulifson 1994, Gibson and Myers 2002a). Gibson and Myers (2002a) captured ten eels, nine of which were alive, but were not able to apply their turbine mortality model to these data.

Population-level impacts are assessed as low due to a panmictic population structure such that only a small portion of the population is likely to encounter the turbine, and low turbine mortality rate estimates. However, fishway usage, delayed mortality and movement patterns in the vicinity of the turbine are not known and information on population structure is limited, leading to moderate uncertainty in the conclusion. The cumulative effects of turbine mortality and fishing mortality for American Eel are not understood. Additionally, although the impacts of turbine mortality on overall American Eel status may be low, localized human-induced losses are a factor contributing to cumulative range-wide losses. For this reason losses attributable to the Annapolis River would be a factor in evaluating progress towards meeting the immediate and short term national eel management goals of reducing eel mortality from all sources by 50% relative to the 1997 to 2002 average (DFO 2010).

American Sand Lance (*Ammodytes americanus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Adult (others not known)
Relative abundance:	Unknown
Usage:	Unknown
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed
Fisheries:	Commercial (not in the Annapolis River and estuary)

The American Sand Lance (*Ammodytes americanus*) is a small benthic fish inhabiting mostly inshore regions in the Northwest Atlantic from Hudson’s Bay south to Virginia (Scott and Scott 1988). They prefer areas with sandy substrates to burrow into, but school when not burrowed in the sand. There are no commercial or recreational Sand Lance fisheries in Canada, although they are fished commercially in Europe and the USA (DFO 1996). They have not been assessed by COSEWIC or the IUCN.

American Sand Lance spawn inshore between December and January with spawning adults reaching a maximum of 22 cm in length and a lifespan of 12 years (Robards et al. 1999). They are a forage fish for many larger species of fish, birds, and mammals. Based on their life history characteristics, population resiliency is considered medium (Froese and Pauly 2018).

American Sand Lance were caught in low numbers at the Annapolis TiGS on multiple occasions (Stokesbury 1985, Gibson and Daborn 1993, Gibson and Daborn 1995b) but there are no estimates of turbine mortality rates. Population-level-impacts are likely low due to a broad geographic distribution suggestive that only a small proportion of the population is likely to encounter the turbine. However, fishway usage, acute and delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known and population structure is not well understood, leading to moderate uncertainty in the conclusion.

American Shad (*Alosa sapidissima*)

Level of impact (Uncertainty):	Medium (Moderate)
Life stages present:	YOY, juvenile, adult
Relative abundance:	High but less than historical
Usage:	Upstream spawning migrations of adults in spring; Downstream migration of YOY in summer and fall; Nursery area for YOY in summer and fall
Turbine mortality rates:	Estimates for both adults and YOY; Acute only; Best available adult estimate is 21.3% (90% CI: ± 15.2%) and best available YOY estimate is 23.4% (95% CI: 6.1% to 58.8%)
Resiliency:	Low
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Commercial, recreational and aboriginal (large recreational fishery in the Annapolis River and estuary; Commercial fishery in the Annapolis River and estuary is closed)

The American Shad (*Alosa sapidissima*) is an anadromous coastal migrant that naturally inhabits the Northwest Atlantic, ranging from Newfoundland and Labrador south to Florida (Scott and Scott 1988). Shad are an important species to commercial, recreational, and aboriginal fisheries. They are fished commercially in the Maritime Provinces, including the Bay of Fundy, but are no longer fished commercially in the Annapolis River (Melvin et al. 1985, Chaput and Bradford 2003). They are also kept as bycatch in gaspereau fisheries in the Maritimes. American Shad are fished recreationally in many rivers, including the Annapolis River. Currently, American Shad have not been assessed by COSEWIC and are designated as least concern by the IUCN Red List (NatureServe 2013c).

In Atlantic Canadian rivers, American Shad have genetically distinct spawning populations with significant genetic diversity among rivers, as well as groups spawning populations that make up genetically similar metapopulations (Hasselman et al. 2010). The Bay of Fundy metapopulation includes the large Annapolis River spawning population (Hasselman et al. 2010). In addition to genetics research, further evidence of rivers containing discrete populations comes from the behavior of adult shad, which show high fidelity to their river of previous spawning (Melvin et al. 1986).

Shad native of the Annapolis River are known to spawn in May-June, with males arriving to the river earlier than females (Williams and Daborn 1984). Following spawning, adult fish will leave the estuary, and if in the Bay of Fundy, make their way counter-clockwise around the Bay, and head back out to sea in the fall (Melvin et al. 1985, Dadswell et al. 1987). Larvae hatch at a size of 5 to 10 mm total length after 8 to 12 days. In the Annapolis River, YOY are present in the estuary in mid-July, and migrate seaward throughout the summer and fall (Gibson 1996a). Shad mature at ages three to six for males and at ages four to six for females (Chaput and Bradford 2003).

Based on life history characteristics, population resiliency is considered low (Froese and Pauly 2018). Many stocks over their North American range have declined in the past century and do not show signs of recovery (ASMFC 2007). Abundance declines or total collapses have occurred in rivers that historically had spawning populations, mainly following the installation of man-made barriers such as dams and causeways (Chaput and Bradford 2003). Local examples include the populations in the Petitcodiac, Saint John (above Mactaquac Dam) and St. Croix rivers of New Brunswick.

The American Shad is the most studied fish species in the Annapolis River watershed. Information is available about adult and juvenile turbine mortality rates, fishway utilization, fish diversion, and changes in life-history characteristics since the turbine came online. American Shad have been caught consistently throughout the studies at the Annapolis TiGS (e.g. Stokesbury 1985, Gibson and Daborn 1995b, Gibson and Myers 2002b). They have been documented moving through both the turbine and the fishways, although the majority of individuals have been captured in the tailrace of the turbine. Turbine mortality rates have been estimated for both adult and YOY American Shad (see TOR 5). Gibson and Myers (2002b) found that a high- frequency sound fish diversion system decreased American Shad passage through the turbine by 42%. Assessments of the American Shad population in the Annapolis River before and after the turbine came online indicate that there has been a decline in the mean size, mean age, maximum age, age at maturity, and percent of repeat spawners since the operation of the Annapolis TiGS began, along with an increase in the annual total mortality rates (see TOR 6).

Population-level impacts are assessed as medium based on a well-defined population structure with all of the population likely to encounter the Annapolis TiGS, high productivity and acute turbine mortality rate estimates that could lead to a decline population size greater than 10%. However, information about fishway usage is limited, delayed mortality and movement patterns in the vicinity of the turbine are not known and cumulative effects of fishing and passage at the Annapolis TiGS have not been studied, leading to moderate uncertainty in the conclusion.

Atlantic Herring (*Clupea harengus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	YOY, juvenile, adult
Relative abundance:	High
Usage:	Migrate inshore to feed from spring to fall; Nursery area for YOY in summer and fall
Turbine mortality rates:	YOY only; Acute only; 15.8% (95% CI:10.8%-22.1%)
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Commercial, recreational and aboriginal (limited fishing in the Annapolis River and estuary)

The Atlantic Herring (*Clupea harengus*) is a schooling, marine, pelagic fish (Scott and Scott 1988). They range seasonally from inshore to offshore waters in the North Atlantic (0 - 200m). Atlantic Herring are located on both the Northwest and Northeast Atlantic coasts, and in the Northwest Atlantic they range from Greenland to Cape Hatteras, North Carolina. There are active commercial, recreational, and aboriginal fisheries in the Northwest Atlantic for Atlantic Herring, including coastally in the Bay of Fundy (DFO 2017c). Commercially they are fished primarily with purse seines but gillnets, trapnets, and weirs are also used. Atlantic Herring have not been assessed by COSEWIC and are currently of least concern on the IUCN Red List (Herdson and Priede 2010).

Spawning in Atlantic Herring is spatially and temporally variable by stock with some spawning in the fall and some in the spring (Scott and Scott 1988, DFO 2017c). In the Bay of Fundy, there are many spawning areas and most spawning occurs in the fall, although there are some small spring spawning aggregations (DFO 2017c). Atlantic Herring undertake predictable seasonal migrations, moving inshore in the spring, remaining inshore to feed throughout the summer, and eventually moving offshore in the late fall to overwinter (MacDonald et al. 1984, Scott and Scott 1988, Dadswell and Rulifson 1994). Spring spawning occurs in shallow inshore waters while fall spawning occurs farther offshore at greater depths (Scott and Scott 1988). After spawning, larvae hatch at 4 to 10 mm in total length. Atlantic Herring reach sexual maturity between three and five years at which time they begin their annual seasonal migration cycle. They are an important prey species, and are eaten by larger fish, sea birds and marine mammals at all life stages.

Population resiliency based on life history characteristics is considered medium (Froese and Pauly 2018).

Atlantic Herring have been recorded in high numbers in the Annapolis River around the Annapolis causeway (e.g. Stokesbury and Dadswell 1989, Gibson and Myers 2002b).They are present within the estuary during at least spring and fall and have potential for multiple passes through the turbine. Acute turbine mortality rate estimates are available for YOY Atlantic Herring (see TOR 5) Additionally, fish passage studies indicate many Atlantic Herring utilize the new fishway when moving through the Annapolis TiGS (Gibson and Daborn 1993, Gibson and Myers 2002b), although the proportion of the fish using the fishways is not well understood. Although Atlantic Herring were commonly captured at the Annapolis TiGS, whether these are part of a population native to this estuary is unknown.

Population-level impacts are assessed as low due to a broad geographic distribution of the population suggestive that only a portion of the population is likely to encounter the Annapolis TiGS, recorded utilization of the new fishway, and medium resiliency. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine is not well known, there is limited information about population structure and cumulative effects of fishing and passage at the Annapolis TiGS have not been studied leading to moderate uncertainty in the conclusion.

Atlantic Mackerel (*Scomber scombrus*)

Level of impact (Uncertainty):	Low (Low)
Life stages present:	Adult (others not known)
Relative abundance:	Moderate
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Commercial, recreational and aboriginal (limited fishing in the Annapolis River and estuary)

The Atlantic Mackerel (*Scomber scombrus*) is a temperate, schooling, pelagic fish (Scott and Scott 1988). In the Northwest Atlantic, Atlantic Mackerel range from North Carolina to Labrador. Atlantic Mackerel are fished commercially, recreationally and by aboriginal communities throughout the Maritime provinces, including in the Bay of Fundy, Newfoundland and Labrador, and Quebec (DFO 2007a). Stocks appear to be declining in recent years although stock assessments have a high level of uncertainty due to estimated high levels of unreported catch coming from the growing recreational fisheries (DFO 2017a). Atlantic Mackerel have not been assessed by COSEWIC and are designated as least concern on the IUCN Red List (Collette et al. 2011).

Spawning of Atlantic Mackerel in the Northwest Atlantic occurs mainly between Cape Hatteras and Cape Cod, and in the Southern Gulf of Saint Lawrence (Sette 1943, MacKay 1979, Scott and Scott 1988), although some spawning activity also occurs in the inner Bay of Fundy (Minas Basin) and on the Scotian Shelf off Nova Scotia (MacKay 1979, DFO 2007a, Scott and Scott 1988). There is no concrete evidence of natal homing or fidelity and there is a high degree of mixing between fish greater than one year of age (Nesbø et al. 2000, Redding 2017). This supports the idea of a single mixed Northwest Atlantic population, although the north and south are managed separately by Canada and the U.S. (TRAC 2010).

In Canada, the spawning migration and activity is dictated by temperature (TRAC 2010). Adults move inshore as water temperatures reach between 9 and 12 °C and spawn from mid-June to mid-July in surface waters. Following spawning, adults remain inshore feeding until the fall when they return to the outer continental shelf to overwinter. Eggs hatch in roughly seven days and larvae are about 3 mm in length (MacKay 1979). Mackerel grow quickly and transform into juveniles at 50 mm at which time they begin to form schools. Sexual maturity is linked to size more than age with most fish maturing once larger than 30 cm (Scott and Scott 1988). Almost all fish are sexually mature by age four. The average size of Atlantic Mackerel is between 32 and 36 cm with a weight of 0.5 kg (Scott and Scott 1988). Atlantic Mackerel are a major prey source for many larger fish, including sharks and marine mammals, and are subject to high natural mortality at all life stages (Sette 1943, Scott and Scott 1988).

Population resiliency based on life history characteristics in considered medium (Froese and Pauly 2018).

Atlantic Mackerel have been captured at the Annapolis TiGS on many occasions, although the low numbers captured were insufficient to estimate turbine mortality rates (Gibson and Daborn 1995a, Gibson and Myers 2002a, Gibson and Myers 2002b). Dead Atlantic Mackerel have been recovered from below the turbine with injuries consistent with turbine mortality (Dadswell and Rulifson 1994).

Population-level impacts are assessed as low due to a broad geographic distribution of the population with a high degree of mixing suggestive that only a small portion of the population would encounter the Annapolis TiGS and medium resiliency. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact. Distribution and population structure is well understood and therefore uncertainty with this conclusion is low.

Atlantic Menhaden (*Brevoortia tyrannus*)

Level of impact (Uncertainty):	Medium (Very High)
Life stages present:	YOY (others not known)
Relative abundance:	Low
Usage:	Nursery area for YOY during summer and fall
Turbine mortality rates:	YOY only; Acute only; A single estimate for Clupeidae; see TOR 5
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Not fished in Canada

The Atlantic Menhaden (*Brevoortia tyrannus*) is a euryhaline schooling fish inhabiting the continental shelf of the Northwest Atlantic from the southern Gulf of Saint Lawrence to southern Florida (Scott and Scott 1988). Presence in Canada, aside from a regular appearance in the Saint John River, is unpredictable. They are not fished in Canada; however, they do support a large commercial fishery in the USA (Scott and Scott 1988). Data from this fishery indicate that the stock is not overfished, and overfishing is not occurring (ASMFC 2017). Atlantic Menhaden have not been assessed by COSEWIC and are designated as least concern on the IUCN Red List (Carpenter and Ralph 2015).

Atlantic Menhaden spawn mainly in large coastal bays from May through October in the northern parts of their range (Canada to Massachusetts). Larvae move into estuaries following hatching, using the estuaries as nursery grounds until fall when they migrate back out to sea (Stokesbury and Stokesbury 1993). Genetic diversity is low throughout their range and all individuals are considered part of a single large population (Lynch et al. 2010). They mature at around two years of age and remain at sea where they feed on plankton and are fed upon by large fish, sea birds, and marine mammals (Scott and Scott 1988). Population resiliency based on life history characteristics is considered medium (Froese and Pauly 2018).

Juvenile Atlantic Menhaden were captured at the same location in the Annapolis River estuary during fall seine surveys in 1985, 1986, 1989 and 1995 (Stokesbury 1985, Stokesbury and Dadswell 1991, Ruggles and Stokesbury 1990, Stokesbury and Stokesbury 1993, Gibson and Daborn 1995b). There are no species-specific estimates of turbine mortality rates for Atlantic Menhaden at the Annapolis TiGS; however mortality was estimated for juvenile clupeids which included Atlantic Menhaden (see TOR 5).

Population-level impacts are assessed as low due to a broad geographic distribution suggestive that only a small portion of the population is likely to encounter the Annapolis TiGS and a medium resiliency. However, the presence of juveniles in the Annapolis River estuary is atypical and their usage of the river is unknown. Combined with limited information about population structure, turbine mortality rates and movement patterns in the vicinity of the turbine, uncertainty is considered very high.

Atlantic Salmon(*Salmo salar*)

Level of impact (Uncertainty):	Extreme (High)
Life stages present:	Smolt, pre-spawning adults, kelts
Relative abundance:	Low
Usage:	Upstream spawning migrations (late spring to fall); Downstream migration of smolts and kelts (spring)
Turbine mortality rates:	Unknown
Resiliency:	Very low
COSEWIC/IUCN designation:	Endangered / Least concern (globally, but information is out of date)
Fisheries:	Commercial, recreational and aboriginal; All salmon fisheries are closed in Southern Upland, Nova Scotia

The Atlantic Salmon (*Salmo salar*) is an anadromous fish spanning both the Northwest and Northeast Atlantic (Scott and Scott 1988). In the Northwest Atlantic, they range from Hudson Bay down to the Connecticut River. Historically, Atlantic Salmon have been heavily fished commercially, recreationally, and by aboriginal communities. Commercial fishing in the Maritimes was closed in 1984 and subsequent restrictions and closures of the recreational fisheries have been set in place. Atlantic Salmon were fished recreationally in the Annapolis River, and DFO formerly ran a broodstock collection and stocking program in the river. This program was terminated in the 1980's. The recreational fishery for Atlantic Salmon in the Annapolis River is currently closed.

In Canada, Atlantic Salmon have been separated into 16 designatable units (DU) based on genetic and behavioral variability (COSEWIC 2010a). Designatable units in Canada range from not at risk to extinct. In the Bay of Fundy there are three DUs (the Inner Bay of Fundy, the Outer Bay of Fundy and Southern Upland DU's), all of which are designated "Endangered" by COSEWIC. The Annapolis River Atlantic Salmon population is a part of Southern Upland DU. Abundance of Atlantic Salmon in the Southern Upland has declined markedly and river specific populations extirpations have occurred (Gibson et al. 2010, Gibson et al. 2011). Atlantic Salmon on a global scale are designated as least concern on the IUCN Red List, although the last assessment was completed in 1996, where it was noted that the assessment was in need of updating (World Conservation Monitoring Center 1996).

Gibson and Bowlby (2013) describe the life cycle of Southern Upland Atlantic Salmon as follows:

> "Southern Upland 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 November. After spawning, adults (known as "kelts") may return to the sea or may remain in fresh water until the following spring. Although the proportion of kelts remaining in fresh water is not well studied, a recent (2010/11) acoustic tagging study on the St. Mary's River indicates that the proportion of salmon over-wintering in fresh water is likely very high (Gibson and Halfyard, unpublished data).

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". Parr in Southern Upland rivers typically remain in fresh water for two to four 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"). Within Southern Upland 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 is highly variable among populations. For example, in the West Branch of the St. Mary's River, the majority of salmon mature after one winter at sea, whereas in the East Branch of the river, there is a higher proportion of salmon that mature after two winters at sea. Three sea-winter salmon are now very rare or absent from most populations in the Southern Upland. Adult run timing is variable. In many years, the majority of salmon return to the rivers during late spring or early summer whereas, depending on both oceanic and freshwater environmental conditions, in other years the majority may return during the fall. 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 Southern Upland."

Population resiliency based on life history characteristics is considered medium (Froese and Pauly 2018). However, for endangered Southern Upland Atlantic Salmon the number of lifetime replacement spawners per spawner calculated for the LaHave and St. May's Bay Southern Upland Salmon populations are 0.84 and 1.02, respectively (Gibson and Bowlby 2013), which is very low for any fish species (e.g. Myers et al.1999).

Historically, the population of Atlantic Salmon in the Annapolis River has been small, owing to a lack of suitable habitat, mostly available in tributaries such as the Nictaux River, covering a much smaller area than other Southern Upland Rivers (Bowlby et al. 2014). Atlantic Salmon were caught in the most recent surveys of the river in very low numbers, which corresponds to the general trend seen throughout the Southern Upland DU (Gibson et al. 2011).

There are no records of Atlantic Salmon in any turbine related studies at the Annapolis TiGS. However, considering the timing of sampling for these studies (mid-summer to fall), migrating smolts and kelts would not be expected to be present.

Population-level impacts are assessed as extreme due to their very low maximum lifetime reproductive rate, well defined population structure and low population size. All of the population is expected to encounter the turbine, as smolts, as returning as adults and as post-spawning adults. However, due to cumulative effects with high mortality rates in other parts of their life cycle, and unknown fishway usage, turbine mortality rates and movement patterns in the vicinity of the turbine, the uncertainty in the conclusion is high.

Atlantic Silverside (*Menidia menidia*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Larvae, juvenile, adult
Relative abundance:	Very high
Usage:	Upstream and downstream migration; Resident from at least spring to fall
Turbine mortality rates:	Acute: 2.2% (95% CI: 1.1% - 4.1%)
	Acute + Delayed: 5.2% (95% CI: 2.3% - 11.2%)
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Commercial (PEI only in Canada)

The Atlantic Silverside (*Menidia menidia*) is a small, schooling, coastal species (Scott and Scott 1988). They mainly inhabit inshore areas from the Gulf of Saint Lawrence to Florida, moving into brackish estuaries, marshes, and streams in the spring to spawn. Silversides have been fished commercially in Prince Edward Island since 1973 but are not fished commercially or recreationally elsewhere (Scott and Scott 1988, DFO 2016a). Atlantic Silverside is designated as least concern on the IUCN Red List (Carpenter and Monroe 2015) and have not been assessed by COSEWIC.

Spawning occurs in the spring, mostly between April and June, and timing is latitude dependent with some of the most southern populations recorded to spawn as early as late February (Jessop 1983, Scott and Scott 1988). In the Bay of Fundy, and more specifically the Annapolis River, adult Atlantic Silverside begin to move into the estuary in April or May and spawning occurs in the middle reaches of the river mainly in June (Jessop 1983). Following spawning, the adults migrate downstream and spend the summer months in coastal marine waters to feed, returning to the estuary to overwinter in the northern reaches of their range (Jessop 1983). Eggs hatch after around 10 days and juvenile fish may remain in brackish waters until late summer, when they move to feed in coastal areas. Around 73% of each year-class dies annually prior to reaching maturity (Jessop 1983). Resident populations in the Bay of Fundy have a faster growth rate with a shorter growth period than southern populations, resulting in smaller individuals on average when compared with fish of southern populations of the same age (Jessop 1983). In the Bay of Fundy, Atlantic Silversides commonly live to two years of age, while other populations have a one year lifespan. These fish mature at one year and are an average size of 90 to 100 mm and 5.8 to 7.7 g with females being larger than males (Jessop 1983, Scott and Scott 1988). They are omnivorous, feeding almost exclusively on plankton, and are a major source of food for larger fish such as Striped Bass. Based on life history characteristics, population resiliency is considered high (Froese and Pauly 2018).

Based on seining surveys and sampling at the Annapolis TiGS, Atlantic Silversides are the most abundant fish in the Annapolis River estuary upstream of the Annapolis TiGS. In the Annapolis River, both adults and juveniles have been documented at high abundance around the causeway from August to November (Jessop 1983, Stokesbury 1985, Gibson and Myers 2002b). Turbine mortality rate estimates are available for Atlantic Silverside (see TOR 5). Additionally, many individuals preferentially travelled through the fishway rather than the turbine on their seaward migration (see TOR 4).

Population-level impacts are assessed as low due to low acute turbine mortality rate estimates, a high rate of fishway utilization, high abundance and high resiliency. However, population structure and behavior around the turbine is not well understood leading to moderate uncertainty in this conclusion.

Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*)

Level of impact (Uncertainty):	Scenario dependent: Low or High (High)
Life stages present:	Adult
Relative abundance:	Low
Usage:	Unknown. Two potential scenarios; transient individuals from Saint John River and/or local spawning population
Turbine mortality rates:	Unknown
Resiliency:	Very Low
COSEWIC/IUCN designation:	Endangered / Near threatened globally
Fisheries:	Commercial (Saint John River only), aboriginal (not in the Annapolis River and estuary)

The Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*) is a large, long-lived, anadromous fish that, as adults, inhabits the Northwest Atlantic from Labrador down to Florida to depths up to 200 m (Scott and Scott 1988, Dadswell 2006). There is a moratorium on fishing Atlantic Sturgeon along their entire U.S. range (Hilton et al. 2016). In Canada, there is a fishery in the Saint John River with an annual quota of 350 individuals (Dadswell 2006, DFO 2013, Bradford et al. 2016).There are also two First Nations licenses for Atlantic Sturgeon in the Maritimes. Although there are no targeted at-sea fisheries for Atlantic Sturgeon, they are caught as bycatch throughout the winter (November to April) throughout their range (Wirgin et al. 2015). They are currently designated as threatened by COSEWIC (COSEWIC 2011) and U.S. populations are designated as either endangered or threatened (Hilton et al. 2016). Globally, they are designated near-threatened on the IUCN Red List (St. Pierre 2006).

Atlantic Sturgeon spawn in fresh water , exhibiting fidelity for their natal rivers (Scott and Scott 1988). Genetic analyses have determined that each large spawning river along the east coast of North America has a unique population that is genetically distinct (Wirgin et al. 2012). A genetic analysis looking at bycatch from fisheries in the Gulf of Maine down to Cape Hatteras during the winter months found that at-sea assemblages of Atlantic Sturgeon were of mixed origin, with individuals from populations throughout their range (Wirgin et al. 2015). In the Bay of Fundy, spawning occurs in June, but may continue as late as August. Males arrive to the spawning rivers earlier than females, making their way up river to the spawning ground characterized by rocky-gravel substrate, high flow, deep pools and sometimes below waterfalls (COSEWIC 2011, Beardsall et al. 2016). After spawning, adults move downstream back into the marine environment. Males are known to return to spawn every one to five years, while females may return every two to five years (Hilton et al. 2016). Juveniles may remain in the estuarine waters of their natal rivers for two to six years before migrating to sea, where males mature at 16 to 24 years and females at 17 to 28 years in the northern parts of their range (Scott and Scott 1988, Bradford et al. 2016, Hilton et al. 2016). Mature adult females are larger than males, averaging 2 to 3 m in length, and 100 to 200 kg in weight, while males average 1.4 to 2.1 m and 50 to 100 kg (Scott and Scott 1988, COSEWIC 2011, Dadswell et al. 2016). Fishing records from the 1800’s to1900’s indicate a change in the population to one of a smaller size and faster growth in more recent years (Dadswell 2006, Balazik et al. 2010, Hilton et al. 2016). Population resiliency is considered very low (Froese and Pauly 2018). Based on historical fisheries records, many populations have seen substantial declines or have been extirpated (Dadswell et al. 2006, Balazik et al. 2010).

At present, it is not known whether there was, and still is, a spawning population of Atlantic Surgeon native to the Annapolis River. Huntsman (1922) states: “They [Atlantic Sturgeon] are known also in the Annapolis river, which they are said to ascend as far as Middleton.”, potentially indicating the presence of a population historically. From the time of the construction of the Annapolis TiGS to 2017, there have been 21 dead Atlantic Sturgeon (2 immature fish, 19 adults) found in the vicinity of the Annapolis TiGS whose death has been attributed to turbine passage (Dadswell et al. 2018). This number does not represent the total number that would have been killed at the generating station because some might not wash up on shore and there has not been a systematic survey to locate dead fish spanning this time period. Some of these larger individuals were ripe or ripening adults. However, no YOY, indicative of a spawning population, have been captured in the Annapolis River or estuary. Atlantic Sturgeon are known to undertake long migrations and to forage in estuaries other than near their natal river (Savoy and Pacileo 2003). Additionally, similar to other diadromous fish species, homing to natal rivers is not 100% and a small number of fish may spawn in non-natal rivers potentially leading to colonization of these other rivers (Savoy et al. 2017). Individuals observed in the Annapolis River may be fish from the Saint John River, or some from other population. The origin of these individuals is not known.

If a native spawning population exists in the Annapolis River, population-level impacts are assessed as high due to the evidence of turbine mortality and very low population resiliency. All of a native population would be expected to encounter the turbine, potentially several times throughout the year. In the absence of information on fishway usage, turbine mortality rates and movement patterns in the vicinity of the Annapolis TiGS, uncertainty is considered high. If the sturgeon found at the Annapolis TiGS are part of the Saint John River, or some other population, the population-level impact is assessed as low due to the low proportion of the population thought to encounter the turbine. Similar to with the Annapolis population, uncertainty remains high due to the lack of information about fishway usage, turbine mortality rates, movement patterns in the vicinity of the Annapolis TiGS, and the actual proportion of the population that encounters the turbine.

Atlantic Tomcod (*Microgadus tomcod*)

Level of Impact (Uncertainty):	Low (Very High)
Life stages present:	YOY, juvenile, adult
Relative abundance:	Unknown
Usage:	Resident
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Commercial, recreational and aboriginal (limited fishing in the Annapolis River and estuary)

The Atlantic Tomcod (*Microgadus tomcod*) is an inshore, marine fish inhabiting areas of the Northwest Atlantic from Labrador to North Carolina (Scott and Scott 1988). They rarely venture far from shore and are seasonally abundant in the Bay of Fundy. In Atlantic Canada, they are fished in small, local fisheries using bag nets, box nets, gill nets, and spears and are also caught as bycatch in the smelt fishery (DFO 2018a). They are also fished recreationally in the winter through the ice with hook and line (Scott and Scott 1988). Atlantic Tomcod have not been assessed by COSEWIC and are designated as least concern by the IUCN (NatureServe 2013f).

In Canada, Atlantic Tomcod spawn in early to mid-winter, moving inshore, often into rivers and estuaries, in December and moving back to sea in January swiftly following spawning (Scott and Scott 1988; Fortin et al. 1990). They reach sexual maturity at 3 to 3.5 years and 150 mm in length for males and 3 years and 180 mm for females (Fortin et al. 1990), and have a lifespan of four years (Scott and Scott 1988). Atlantic Tomcod population structure is poorly understood, and stock status is unknown. Population resiliency is considered medium based on life history characteristics (Froese and Pauly 2018). Atlantic Tomcod were captured in two studies at the Annapolis TiGS in low numbers (Gibson and Daborn 1993, Gibson and Daborn 1995b) and in greater, but still low, numbers upstream of the causeway during a third (Stokesbury 1985). However, considering the timing of sampling for these and other studies (summer and fall) high catches of winter spawning fish, such as the Atlantic Tomcod, would not be expected. There are no estimates of turbine mortality rates, but fishway utilization studies have captured Atlantic Tomcod in the tailrace, old fishway and new fishway (Gibson and Daborn 1993, Gibson and Daborn 1995b).

Population-level impacts are potentially low due to a broad geographic distribution that is suggestive that only a small portion of a population would encounter the Annapolis TiGS and medium resiliency. However, fishway usage, turbine mortality rates and movement patterns in the vicinity of the turbine are not known. Additionally, population structure is not known and a local population is possible. Timing of studies did not overlap with their expected presence, leading to very high uncertainty is the conclusion.

Atlantic Wolffish (*Anarhichas lupus*)

Level of impact (Uncertainty):	Low (Low)
Life stages present:	Adult
Relative abundance:	Very low
Usage:	Primarily a marine species but transient individuals may move into the Annapolis River temporarily
Turbine mortality rates:	Unknown
Resiliency:	Low
COSEWIC/IUCN designation:	Special concern / Unassessed
Fisheries:	Commercial in Newfoundland and Labrador; Not fished in the Annapolis River and estuary

The Atlantic Wolffish (*Anarhichas lupus*) is a large-bodied, solitary, benthic fish distributed in deep water along slopes on both sides of the North Atlantic. In the Northwest Atlantic, Atlantic Wolffish are found from West Greenland through to the Gulf of Maine, including the Scotian Shelf and Bay of Fundy (Scott and Scott 1988).There is a small commercial fishery for Atlantic Wolffish off the south coast of Newfoundland, but they are not fished commercially in Nova Scotia. However, Atlantic Wolffish are often caught as bycatch in both Nova Scotia and Newfoundland groundfish fisheries (COSEWIC 2012b). Atlantic Wolffish are designated special concern by COSEWIC (COSEWIC 2012b) and are listed on Schedule 1 of the Species at Risk Act (S.C., 2002, c.29). They have not been assessed by the IUCN.

Spawning is thought to occur in the fall with females producing a small number of eggs that are then guarded by the males until hatching occurs (COSEWIC 2012b). Maturity is thought to occur around 8 to 10 years of age and 50 to 60 cm length in the northern portions of their range, but may occur at a younger age and smaller size in the southern portions of their range (Scott and Scott 1988). McCusker and Bentzen (2010) suggest there may be multiple populations of

Atlantic Wolffish in Atlantic Canada. Based on life history characteristics, population resiliency is considered low (Froese and Pauly 2018).

A single Atlantic Wolffish was captured at the Annapolis TiGS in 1993 (Gibson and Daborn 1993) but there is no information on turbine mortality or fishway utilization.

Population-level impacts are assessed as low due to a broad geographic distribution and low catch at the Annapolis TiGS suggestive that only a small portion of the population would encounter the Annapolis TiGS. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact. Distribution and habitat preference are well understood and therefore uncertainty with this conclusion is low.

Blackspotted Stickleback (*Gasterosteus wheatlandi*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	YOY, adult
Relative abundance:	Moderate
Usage:	Resident
Turbine mortality rates:	Acute only; <0.1% (95% CI: <0.1- 5.6)
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	None

The Blackspotted Stickleback (*Gasterosteus wheatlandi*) is a small, benthopelagic species of the temperate Northwest Atlantic (Newfoundland – Massachusetts) found in mainly marine and sometimes brackish habitats (Scott and Scott 1988). They are not fished commercially or recreationally. They have not been assessed by COSEWIC and are designated as least concern by the IUCN Red List (Martins 2015).

The population structure of Blackspotted Stickleback is unknown, however they share similar characteristics with Threespine and Ninespine Sticklebacks and may exhibit similar population structuring. Spawning occurs in May and June during which time the fish migrate to brackish waters and the males build nests to care for the eggs and eventually their young. They grow to an average total length of 51 mm. Size at maturity and growth rate are unknown (Scott and Scott 1988). Males are generally smaller in size than females of the same age. They generally live one year and die shortly following spawning, although it has been found that some individuals may not spawn their first year and can live to two years of age (Martins 2015). Population resiliency is considered high (Froese and Pauly 2018).

Blackspotted Stickleback were caught in high numbers (>800) in at least one study at the Annapolis TiGS (Gibson and Myers 2002b) and potentially others where sticklebacks were not identified to species. (e.g. Gibson and Daborn 1995b). Acute turbine mortality was estimated to be less than 0.1%, based on a sample size of 68 fish (Gibson and Myers 2002a).

Population-level impacts are assessed as low due to a very low turbine mortality rate estimate and high resiliency. However, there is limited information about fishway usage and delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known, leading to moderate uncertainty is the conclusion. There is little information about population structure but a local population is likely.

Blueback Herring (*Alosa aestivalis*)

Level of impact (Uncertainty):	Medium (Moderate)
Life stages present:	YOY, juvenile, adult
Relative abundance:	High
Usage:	Upstream spawning migrations of adults in spring; Downstream migration of YOY in summer and fall; Nursery area for YOY in summer and fall
Turbine mortality rates:	YOY only; Acute only; Best available estimate for YOY is 8.1% (95% CI: 3.5%-17.2%)
Resiliency:	Medium
COSEWIC/IUCN designation:	Not recently assessed / Vulnerable
Fisheries:	Commercial, recreational and aboriginal (limited fishing in the Annapolis River and estuary)

The Blueback Herring (*Alosa aestivalis*) is an anadromous species often grouped with Alewife under the broader terms of gaspereau or river herring due to their physical similarities and overlapping geographic ranges (Loesch 1987). They range coastally throughout the Northwest Atlantic from the Gulf of Saint Lawrence down to Florida (Scott and Scott 1988). Blueback Herring are fished commercially with Alewife throughout their range, but the landings are not recorded separately for the two species (Loesch 1987, Scott and Scott 1988). They were assessed by COSEWIC in 1980 and designated as not-at-risk. Blueback Herring are listed as vulnerable on the IUCN Red List (NatureServe 2013a).

In Canada, spawning populations that are spatially closer together have been found to be more closely related and major genetic differences are found between the three distinct units in the Southern Gulf of Saint Lawrence, on the Atlantic coast of Nova Scotia, and in the Bay of Fundy (McBride et al. 2014). The timing of spawning in Blueback Herring is dictated strongly by temperature (Loesch 1987). They spawn in brackish or fresh water with high flow over rocky or hard bottoms, returning to sea directly following spawning (Loesch 1987, Scott and Scott 1988). Females eventually growing larger than males, with a maximum total length of 30 cm in Canadian waters (Scott and Scott 1988). Maturity occurs between ages three and six, with most maturing by age four. Individuals may spawn up to eight times throughout their lives. Population resiliency is considered medium (Froese and Pauly 2018). Blueback Herring are a significant prey species for larger fish, birds, mammals, and reptiles (Loesch 1987).

In the Annapolis River system, adult Blueback Herring spawn in the river during spring or early summer, and then move back to sea quickly following spawning. Young-of-the-year Blueback Herring may remain in the system until November (Gibson and Daborn 1995b). Information is available about fish passage utilization, turbine mortality rates and fish diversion for Blueback Herring at the Annapolis TiGS. High numbers of Blueback Herring have been caught throughout the studies (e.g. Gibson and Daborn 1995b, Stokesbury and Dadswell 1991) and have been reported as traveling through the causeway using both the new fishway and the turbine. Blueback Herring do not appear to use the old fishway as frequently. Two studies provide different estimates of the acute turbine mortality rates for Blueback Herring at the Annapolis TiGS (see TOR 5), but neither of these studies fully quantifies the mortality rates. Fish diversion methods have been tested on Blueback Herring and they do respond to high-frequency sound (Dunning et al. 1992, Gibson and Myers 2002b), although the effectiveness of the system that was tested may be density dependent (see TOR 7).

Population-level impacts are assessed as medium given a well-defined population structure with all of the population likely to encounter the Annapolis TiGS, high productivity and acute turbine mortality rate estimates that could lead to a decline in population size greater than 10%. However information about fishway usage is limited and delayed mortality and movement patterns in the vicinity of the turbine are not known leading to moderate uncertainty in the conclusion.

Bluefish (*Pomatomus saltatrix*)

Level of impact (Uncertainty):	Low (Very Low)
Life stages present:	Adult
Relative abundance:	Very low
Usage:	Infrequent visitor, Annapolis is the northern extent of the range
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Vulnerable
Fisheries:	Commercial and recreational in U.S.A.; Not fished in the Annapolis River and estuary

The Bluefish (*Pomatomus saltatrix*) is a migratory, temperate and subtropical, marine, pelagic species (Scott and Scott 1988). Geographically, they range almost globally, and in the Northwest Atlantic they winter off Florida, moving to the mid-Atlantic as waters warm, often moving as far north as the Bay of Fundy. They have no commercial value in Canada, however, they are important both commercially and recreationally in the U.S.A. (Scott and Scott 1988, ASMFC 2015). They have not been assessed by COSEWIC and are designated as vulnerable by the IUCN (Carpenter et al. 2015b). Bluefish have been observed in low numbers in the Bay of Fundy (e.g. Dadswell and Rulifson 1994) and at the Annapolis TiGS (Gibson and Myers 2002c).

Given their range, presence in the Annapolis Basin is likely infrequent and population-level impacts are expected to be low. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact. The range is well known leading to very low uncertainty in the conclusion.

Butterfish (*Peprilus triacanthus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, possibly adult
Relative abundance:	Low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Acute only; 8.71% (95% CI: 1.7- 34.5)
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed
Fisheries:	Commercial in U.S.A., Not fished in the Annapolis River and estuary

The Butterfish (*Peprilus triacanthus*) is a migratory, pelagic, schooling fish that inhabit coastal waters in the Northwest Atlantic. They are found from Newfoundland to the Gulf coast of Florida and their abundance is highest between the Gulf of Maine and Cape Hatteras (Scott and Scott 1988, Cross et al. 1999). They are fished commercially throughout the Mid-Atlantic and are considered one stock spanning their entire range (Adams 2018). Currently the stock is not considered to be in an overfished state, and overfishing is not thought to be occurring. They have little commercial value in Canada (aside from bycatch which is used in fish meal), but are a food source for larger commercially important species (i.e. groundfish) (Scott and Scott 1988). Butterfish have not been assessed by COSEWIC or the IUCN.

Butterfish spawning is temperature dependent and variable by location, occurring later in the summer at higher latitudes (Cross et al. 1999). In Canadian waters, Butterfish spawn between July and October. Upon hatching, larvae are 1.6 to 1.75 mm in total length and often take refuge within jellyfish for their first summer (Scott and Scott 1988; Cross et al. 1999). Butterfish are short lived and fast growing. They mature between one and two years of age at a length of roughly 12 cm. They live for approximately three years and can obtain a maximum size of 30 cm. Population resiliency is considered high (Froese and Pauly 2018).

Butterfish have been observed in the Bay of Fundy (MacDonald et al. 1984, Dadswell et al. 1986) and the Annapolis River estuary on a number of occasions (e.g. Stokesbury 1985, Dadswell and Rulifson 1994, Gibson and Myers 2002b). A single estimate of their acute turbine mortality rate at the Annapolis TiGS is available (see TOR 5), although there is some uncertainty in this estimate due to the small sample size.

Population-level impacts are assessed as low due to a low acute turbine mortality rate estimate and high resiliency, as well as a broad geographic distribution and management as a single stock suggestive that only small proportion of the population is likely to encounter the Annapolis TiGS. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known and population structure is not well understood leading to moderate uncertainty in the conclusion.

Cunner (*Tautogolabrus adspersus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	Low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Recreational in U.S.A.; Not fished in the Annapolis River and estuary

The Cunner (*Tautogolabrus adspersus*) is a marine, inshore fish that generally inhabits coastal waters close to shore and tends to avoid brackish and freshwater environments (Scott and Scott 1988). They are found on or near the sea floor in congregations around structures such as wharves, wrecks, and even seaweed. They are non-migratory, occupying small home ranges in the Northwest Atlantic from Newfoundland down to the Chesapeake Bay. Cunner are not presently fished commercially, but do support recreational fisheries in areas where they are highly abundant. They have not been assessed by COSEWIC and are listed as least concern on the IUCN Red List (Choat 2010).

In Canadian waters, Cunner spawn in the summer months from June to August with most spawning occurring off Southern Newfoundland, in the Southern Gulf of Saint Lawrence, and in St. Mary's Bay in Nova Scotia (Scott and Scott 1988). Spawning in the Bay of Fundy is generally less successful. Cunner mature at around 8 to 11 cm in size, and average less than 30 cm total length as adults. Females grow faster than males. Population resiliency is considered medium (Froese and Pauly 2018).

In the Annapolis River system, Cunner have been caught on a number of occasions (Daborn et al. 1979, Stokesbury 1985, Gibson and Myers 2000, Gibson and Myers 2002a, Gibson and Myers 2002b). All three individuals observed in Gibson and Myers (2002b) survived passage through the turbine. Delayed mortality was not assessed.

Population-level impacts are assessed as low given no evidence of a population specific to the Annapolis Basin, preference for marine habitat, medium resiliency and evidence they can survive passage at the TiGS. However, limited information about population structure suggests they are non-migratory with a small home range leading to moderate uncertainty in the conclusion.

Flying Gurnard (*Dactylopterus volitans*)

Level of impact (Uncertainty):	Low (Very Low)
Life stages present:	Adult
Relative abundance:	Rare
Usage:	Accidental; Annapolis is northern extent of the range
Turbine mortality rates:	Unknown
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	None

The Flying Gurnard (*Dactylopterus volitans*) is a benthic, tropical, warm water fish that generally ranges in the Western Atlantic from North Carolina to Argentina (Scott and Scott 1988).They are listed on the IUCN Red List as least concern and have no commercial, recreational, or aboriginal value (Carpenter et al. 2015a). They have been known to stray into Canadian waters in warmer months, and on a single occasion one was caught in the Annapolis River (Gibson and Myers 2000).

Given their distribution and habitat preference, presence in the Annapolis Basin is likely infrequent and population-level impacts are low. Distribution and habitat preference is well known and the single specimen captured at the Annapolis TiGS was a first Bay of Fundy record, leading to very low uncertainty in the conclusion.

Fourbeard Rockling (*Enchelyopus cimbrius*)

Level of Impact (Uncertainty)	Low (High)
Life stages present:	Adult (others are unknown)
Relative abundance:	Low
Usage:	Seasonal migrant which may move into Annapolis River during warmer months
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed/ Least concern
Fisheries:	None

The Fourbeard Rockling (*Enchelyopus cimbrius*) is a seasonal migrant of the North Atlantic (Scott and Scott 1988). In the Northwest Atlantic they range from Greenland to the Gulf of Mexico. They move into warmer shallow waters in the summer and burrow into sandy or muddy substrates. They have no economic value in North America, have not been assessed by COSEWIC and are designated as least concern by the IUCN (Iwamoto et al. 2015). In the Bay of Fundy, spawning occurs in late May and may continue as late as October, depending on water temperature. Adults have an average length of 15 to 30 cm, with a maximum length of 41 cm. Population resiliency is considered medium (Froese and Pauly 2018). Fourbeard Rockling were captured in the tailrace of the Annapolis TiGS on one occasion (Gibson and Myers 2002b).

Population-level-impacts are likely low due to a broad distribution in the Northwest Atlantic suggestive that only a small proportion of the population is likely to encounter the turbine and medium resiliency. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known and population structure is not well understood, leading to moderate uncertainty in the conclusion.

Fourspine Stickleback (*Apeltes quadracus*)

Level of Impact (Uncertainty)	Low (Moderate)
Life stages present:	Juvenile, Adult
Relative abundance:	Moderate
Usage:	Resident
Turbine mortality rates:	Unknown
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries	None

The Fourspine Stickleback (*Apeltes quadracus*) is a small, marine and brackish water fish found throughout the Northwest Atlantic from the Gulf of Saint Lawrence down to Virginia (Scott and Scott 1988). They have no commercial or recreational value, have not been assessed by COSEWIC and are designated as least concern by the IUCN (NatureServe 2013d). In Canadian waters, the Fourspine Stickleback spawns during the summer from May through July in the intertidal region, commonly among vegetation (Scott and Scott 1988). Males construct nests and then initiate courtship with females. During spawning, female deposit 15 to 20 eggs and then males follow them into the nest and fertilize the eggs. The male remains with the nest to guard the eggs and may build multiple nests on top of one another to breed with multiple females. Males often die at one year of age following hatching of young while females may survive to three years of age. Young hatch at roughly 4 mm in length and grow quickly, reaching a size-at-maturity of greater than 51 mm within their first year. They feed mainly on plankton and are preyed upon by larger fish, birds, and mammal. The population resiliency is considered high (Froese and Pauly 2018)

Fourspine Sticklebacks have been caught throughout the Annapolis River estuary including near the tidal generating station (Daborn et al. 1979, MacDonald et al. 1984, Stokesbury 1985).

Population-level impacts are assessed as low due to their small size, low turbine mortality rate estimates for other stickleback species and high resiliency. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known and population structure is not well understood, although a local population is likely, leading to moderate uncertainty is the conclusion.

Hake (White, Silver, Red)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile (others are unknown)
Relative abundance:	Unknown
Usage:	Unknown
Turbine mortality rates:	Juveniles only; Acute only; 8.7% (95% CI: 0.3- 20.9)
Resiliency:	Low (White), Medium (Silver, Red)
COSEWIC/IUCN designation:	Threatened (White) / Near Threatened (Silver)
Fisheries:	Commercial

Hake is the overarching term used in the Northwest Atlantic to refer to four species: Silver Hake (*Merluccius bilinearis*), White Hake (*Urophycis tenuis*), Red Hake (*Urophycis chuss*) and Longfin Hake (*Phycis chesteri)*. Generally, they range from Newfoundland down to North Carolina, with some absent in the Gulf of Saint Lawrence and Bay of Fundy (Scott and Scott 1988). They are fished commercially in Canadian and American waters in the groundfish fishery. COSEWIC designated the Atlantic DU of White Hake as threatened (COSEWIC 2013) The IUCN evaluated the Silver Hake and listed it as near threatened (Carpenter 2015). The other species have not been assessed by either COSEWIC or the IUCN.

Hake are benthic fish living over mud or sand bottoms at a range of depths. They typically carryout seasonal migrations influenced by temperature, moving to shallow inshore waters in the spring and summer, and returning back offshore for the winter. Silver hake spawn while inshore mostly around Sable Island on the Scotian Shelf between June and September, with spawning peaking in July and August (Scott and Scott 1988). Upon hatching, larvae are around 3 mm in length and mature at two to three years of age at a size of roughly 32 cm. They have a maximum lifespan of 12 years and maximum size of 37 cm for males and 65 cm for females. The fishing stock on the Scotian Shelf holds a healthy status with fishing mortality below the set reference point (DFO 2017b). Population resiliency of Silver Hake is considered medium (Froese and Pauly 2018).

Red Hake in the Bay of Fundy spawn in the late summer to early fall (September) prior to moving to deeper depths for the winter (Scott and Scott 1988). They are fast growing and relatively short lived, reaching sexual maturity at two years and roughly 30 cm in length. Population structure of Red Hake is unknown and catch of Red Hake is not usually differentiated from White Hake in Canadian fisheries. As such, there is no independent stock status for Red Hake in Canadian waters. Population resiliency of Red Hake is considered medium (Froese and Pauly 2018).

White Hake closely resemble Red Hake, but they reach a much larger size and typically inhabit deeper colder waters (Scott and Scott 1988). White Hake are highly fecund and young grow quickly, reaching sexual maturity by four years of age at a size of roughly 40 cm in length for males and 47 cm for females. Genetic variability exists for three groups of White Hake in Canada, the Southern Gulf of Saint Lawrence population, the Southern Newfoundland population, and the Scotian Shelf/Bay of Fundy/Northern Gulf of Saint Lawrence population (Roy et al 2012). This genetic variation has been used by COSEWIC to identify two DU: DU1 – Southern Gulf of Saint Lawrence, and DU2 – Atlantic and Northern Gulf of Saint Lawrence (COSEWIC 2013).

White Hake and Red Hake have been observed in estuaries and bays connecting to the outer Bay of Fundy, including Passamaquoddy Bay, St. Mary's Bay, the Saint John and Kennebecasis Rivers, and the Annapolis River and estuary (MacDonald et al. 1984; Scott and Scott 1988; Horne and Campana 1989). Hake were not identified to species in the studies at the Annapolis TiGS. Hake are present in at least the late summer and early fall (Horne and Campana 1989; Gibson and Myers 2002b). There is evidence that they pass through the turbine and an acute turbine mortality estimate is available for hake (see TOR 5).

Population-level impacts are assessed as low due to the presence of a healthy commercial fishery on the Scotian Shelf, low acute turbine mortality rate estimates and because only a small portion of their population is likely to encounter the Annapolis TiGS.. However, individuals were not identified to species, information on population structure is limited and resiliency is variable among species leading to moderate uncertainty in the conclusion.

Longhorn Sculpin (*Myoxocephalus octodecemspinosus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	Very low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed
Fisheries:	Commercial 1999-2006; Not fished in the Annapolis River and estuary

The Longhorn Sculpin (*Myoxocephalus octodecemspinosus*) is a coastal, bottom dwelling, resident of the Northwest Atlantic, living in shallow shoals and estuaries from spring to fall, and moving to deeper waters for the winter (Scott and Scott 1988, Comeau et al. 2009, Hyndman and Evans 2009). They range from Newfoundland to Virginia and are very common in the Bay of Fundy. They were fished commercially from 1999 to 2006 in St. Mary's Bay in the Bay of Fundy but are no longer the target of any fisheries; however they may still be retained as bycatch (Comeau et al. 2009). Longhorn Sculpin have not been assessed by COSEWIC or the IUCN.

The exact timing of spawning of Longhorn Sculpin is not known but appears to be variable by location, with fish at higher latitudes spawning later in the year (Scott and Scott 1988; Comeau et al. 2009). Longhorn sculpin in the Bay of Fundy spawn inshore over rocky substrate in the winter months somewhere between November and January (Comeau et al. 2009). They become sexually mature at around three years of age at a size of roughly 23 and 24 cm for females and males respectively (Beacham 1982; Scott and Scott 1988). They can grow to a maximum length of 46 cm but generally do not exceed 35 cm in length (Scott and Scott 1988). Longhorn Sculpin are opportunistic feeders, eating mainly crustaceans and small fish, and are themselves fed upon by larger fish and sea birds. Little is known about their population structure and population resiliency is considered high (Froese and Pauly 2018).

Longhorn Sculpin are highly abundant in the Bay of Fundy and have been recorded in the Annapolis River and estuary, although only in low numbers (MacDonald 1984; Horne and Campana 1989, Gibson and Myers 2002b, Comeau et al. 2009).

Population-level-impacts are likely low due high abundance throughout the Bay of Fundy suggestive that only a small proportion of the population is likely to encounter the turbine and a high resiliency. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known leading to moderate uncertainty in the conclusion. Population structure is not known, but a local population is possible.

Lumpfish (*Cyclopterus lumpus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	Very low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	Low
COSEWIC/IUCN designation:	Threatened / Unassessed
Fisheries:	Commercial (NL)

The Lumpfish (*Cyclopterus lumpus*) is a widely distributed fish species in the North Atlantic along both the east and west coasts. In the Northwest Atlantic, they range from Greenland to the Chesapeake Bay and are highly abundant around Newfoundland and in the Gulf of Maine, including the outer Bay of Fundy (Scott and Scott 1988). Genetic variation has led to the designation of three separate populations: the Western Atlantic, the Eastern Atlantic, and the Baltic Sea (Pampoulie et al. 2014). In Canadian waters, they are fished commercially off Newfoundland, mainly for their roe (Simpson et al. 2016). The status of the Lumpfish population in the Northwest Atlantic is currently unknown. Lumpfish are designated threatened by COSEWIC and have not been assessed by the IUCN.

As adults, Lumpfish are generally pelagic, feeding on small fish, although in the early spring they venture inshore to shallow waters to spawn and feed on benthic invertebrates (Scott and Scott 1988). Lumpfish build nests and spawn on rocky or gravel substrate. Young Lumpfish reside in the top one meter of water feeding on plankton for their first year. They mature after three years at length larger than 34 cm and typically grow to a maximum length of 46 cm. They are predated upon by marine mammals and large fish, including sharks, throughout their lives. Population resiliency is considered low (Froese and Pauly 2018).

Lumpfish are present in the outer Bay of Fundy, including the Annapolis Estuary, between summer and fall (Horne and Campana 1989; Dadswell and Rulifson 1994; Gibson and Myers 2002b).

Population-level-impacts are likely low due to a high abundance in the outer Bay of Fundy and a broad geographic distribution suggestive that only a small proportion of the population is likely to encounter the turbine. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known and resiliency is low, leading to moderate uncertainty in the conclusion. Population structure is not known, but a local population is possible.

Meek’s Halfbeak (*Hyporhamphus meeki*)

Level of impact (Uncertainty):	Low (Very Low)
Life stages present:	Adult
Relative abundance:	Rare
Usage:	Accidental
Turbine mortality rates:	Unknown
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	None

The Meek's Halfbeak (*Hyporhamphus meeki*) is a resident species in the Northwest Atlantic inhabiting waters from Massachusetts to the Gulf of Mexico, with rare straying into the Gulf of Maine (Gibson and Myers 2002; Collette et al. 2015). They are not actively fished throughout their range, although they are sometimes used as bait in offshore recreational fisheries (Collette et al. 2015). They have not been assessed by COSEWIC, and at the species level they are designated as least concern by the IUCN (Collette et al. 2015). The first record in Canadian waters was in the Annapolis Basin in 1999 (Gibson and Myers 2002c), although they have been misidentified in older surveys. This individual passed through the causeway using the new fishway and there are no estimates of acute turbine mortality rates for this species.

Given their range, presence in the Annapolis Basin is likely infrequent and population-level impacts are low. The range is well known and the single specimen captured at the Annapolis TiGS was a first Canadian record leading to very low uncertainty in the conclusion.

Mummichog (*Fundulus heteroclitus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	Moderate
Usage:	Resident
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	Commercial in USA for bait; Not fished in the Annapolis River and estuary

The Mummichog (*Fundulus heteroclitus*) is a small, euryhaline, schooling fish that is often found in high abundances in marshy or brackish habitats in the Northwest Atlantic, between the Southern Gulf of Saint Lawrence and the Gulf of Mexico (Scott and Scott 1988). There are no directed fisheries in Canada for Mummichog, although they are often used as a bait fish by fishermen and are a forage fish for many larger species (Scott and Scott 1988). Mummichog have not been assessed by COSEWIC and are designated as least concern on the IUCN Red List (NatureServe 2013e).

Mummichog reside near the surface and occupy small home ranges with no evidence of migratory behaviour (Scott and Scott 1988, Skinner et al. 2005). They spawn in shallow waters, either marine or brackish, between April and August, depending on temperature. They have high tolerances for warm temperatures, low oxygen concentrations, and a wide range of salinities, and have proven resilient to pollution, predation, and fishing (Scott and Scott 1988, Kneib 1986, Boudreau et al. 2005, Goldman et al. 2010). They are a small fish, growing to a maximum length of 13 cm and have a lifespan of four years (Scott and Scott 1988). They feed mainly on crustaceans, marine worms, and vegetation and are considered an important benthic predator (Kneib 1986). Population resilience is medium (Froese and Pauly 2018).

Mummichog have been observed throughout the Annapolis River, including in the vicinity of the causeway (Daborn et al. 1979, Stokesbury 1985, Gibson and Myers 2002a). Mummichogs travel through the fishways more frequently than the turbine. There are no estimates of turbine mortality rates, although for those captured in the tailrace (6 individuals) there was no acute or delayed mortality recorded (Gibson and Myers 2002a).

Population-level impacts are assessed as low due to their small size, high abundance, population-level tolerance to pollution, predation and fishing and medium resiliency. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known and population structure is not well understood, although a local population is likely, leading to moderate uncertainty in the conclusion.

Ninespine Stickleback (*Pungitius pungitius*)

Level of impact (Uncertainty):	Low (High)
Life stages present:	Juvenile, adult
Relative abundance:	Low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	None

The Ninespine Stickleback (*Pungitius pungitius*) is a small, marine or brackish water fish that inhabits nearshore weedy or well vegetated areas (Scott and Scott 1988). They are found along the Northeast coast of North America from the Canadian Arctic to New Jersey. They have no value to fishermen aside from acting as a forage fish for larger sport fish. They have not been assessed by COSEWIC and are designated as least concern on the IUCN Red List (NatureServe 2013k).

In Canada, Ninespine Stickleback spawns during the summer, moving into slightly brackish or fresh water (Scott and Scott 1988). They often spawn multiple times during the spawning season with females releasing clutches of 20 to 30 eggs at a time. Young grow rapidly in their first year and then growth slows, eventually reaching a maximum length of 51 mm in a lifespan of 3 to 3.5 years. Population structure of Ninespine Stickleback in Canada is unknown, however studies investigating Ninespine Stickleback in northern Europe found genetic variation between coastal and freshwater populations as well as between freshwater locations (Shikano et al. 2010). They feed on plankton, including small crustaceans and insects, and are themselves heavily preyed upon by fish, birds, and mammals. Population resiliency is considered to be medium (Froese and Pauly 2018).

Ninespine Stickleback have been observed throughout the Annapolis River, including in close proximity to the tidal generating station (Daborn et al. 1979; Stokesbury 1985). Ninespine Sticklebacks have not been captured during turbine mortality studies at the Annapolis TiGS.

Population-level impacts are assessed as low due to their small size, low turbine mortality estimates for other stickleback species and medium resiliency (which is lower than for other sticklebacks in the Annapolis River estuary). However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known and population structure is not well understood, although a local population is likely, leading to high uncertainty is the conclusion.

Northern Pipefish (*Syngnathus fuscus*)

Level of impact (Uncertainty):	Low (High)
Life stages present:	Juvenile, adult
Relative abundance:	High
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Acute only; 2.2% (95% CI: 0.7- 6.4)
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	None

The Northern Pipefish (Syngnathus fuscus) is a small, coastal, migrant, inhabiting inshore saltmarshes and estuaries in the warmer months and moving offshore to depths of 400 m to overwinter (Scott and Scott 1988; Lazzari and Able 1990). They span the coast of North America in the Northwest Atlantic from the Southern Gulf of Saint Lawrence to Florida and are often found in brackish water. They are not fished commercially or recreationally. Northern Pipefish have not been assessed by COSEWIC and are designated as least concern by the IUCN (Pollom et al. 2015).

Spawning time in Canadian waters is not well defined but is estimated to occur inshore between March and August (Scott and Scott 1988). Male pipefish have brood pouches and are responsible for the young until hatching which occurs at roughly eight to nine millimeters. While inshore, adults and juveniles live in seaweed and eelgrass beds and feed primarily on zooplankton. In Canada, they mature at around one year and can reach a maximum of 30.5 cm; however they rarely exceed 20 cm in length. Population resiliency is considered high (Froese and Pauly 2018).

Northern Pipefish are abundant in the Bay of Fundy and are present in the Annapolis River estuary in at least the summer and fall (Daborn et al. 1979, Stokesbury 1985, Gibson and Myers 2002a). An estimate of the acute turbine mortality rate is available (see TOR 5).

Population-level impacts are assessed as low due to low estimates of acute turbine mortality rates and high resiliency. However, information about fishway usage is limited and delayed mortality and movement patterns in the vicinity of the turbine are not known leading to high uncertainty in the conclusion. Population structure is not known, but reproduction in the Annapolis River estuary is probable and a population specific to the Annapolis River estuary is possible.

Pollock (*Pollachius virens*)

Level of impact (Uncertainty):	Low (Low)
Life stages present:	Adult
Relative abundance:	Low
Usage:	Transient
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed
Fisheries:	Commercial, recreational and aboriginal (limited fishing in the Annapolis River and estuary)

Pollock (*Pollachius virens*) is a demersal, marine species found in temperate regions of both the Northeast and Northwest Atlantic (Scott and Scott 1988, DFO 2011). They are fished commercially throughout much of their range and are caught also recreationally. Pollock within the Bay of Fundy are included in the Western Component Pollock population where they are fished using otter trawls and gillnets (DFO 2011). Pollock have not been assessed by COSEWIC or globally by the IUCN.

Pollock grow to an average of 60 cm total length with an average mass of seven kilograms, mature at roughly six years of age, and have a maximum recorded age of 23 years (DFO 2011). Pollock migrate long distances between spawning grounds and overwintering areas. In the Northwest Atlantic, these spawning grounds are in Massachusetts Bay in the southern Gulf of Maine (Southern Scotian Shelf and Bay of Fundy summer fish) and on the northern Scotian Shelf off Cape Breton for Newfoundland summer fish (Steele 1963, Scott and Scott 1988). Their spring/summer distribution includes the Bay of Fundy where smaller individuals tend to frequent western Nova Scotian areas while larger fish can be found in coastal New Brunswick waters (Steele 1963). Based on their life history characteristics of moderate growth rates and relatively long longevity, Pollock are thought to have medium resilience to increased mortality (Froese and Pauly 2018).

Pollock have been captured in a number of studies carried out within the Annapolis River and estuary, although in small numbers (Daborn et al. 1979, Stokesbury 1985, Gibson and Myers 2002b). There is no information about survival moving past the Annapolis TiGS.

Population-level impacts are assessed as low due to the large population size and low numbers observed at the Annapolis TiGS. These factors are suggestive that only a small portion of the population is likely to encounter the Annapolis TiGS. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact. Population size and structure is reasonably well known and therefore uncertainty with this conclusion is low.

Rainbow Smelt (*Osmerus mordax*)

Level of impact (Uncertainty):	Medium (High)
Life stages present:	Larvae, juvenile, adult
Relative abundance:	Moderate to High
Usage:	Upstream and downstream migration to and from spawning grounds; Nursery area for YOY; Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed/ Least concern
Fisheries:	Commercial, recreational and aboriginal (limited fishing in the Annapolis River and estuary)

The Rainbow Smelt (*Osmerus mordax*) is a schooling, anadromous species. They spend most of their adult lives near the coast or in estuaries and enter fresh water in the spring to spawn (Scott and Scott 1988). In the Northwest Atlantic they are found from Labrador to New Jersey. Additionally, there are many landlocked freshwater populations. Rainbow Smelt are an important commercial, recreational, and aboriginal resource in Atlantic Canada, in particular in the Southern Gulf of Saint Lawrence (DFO 2007b). Anadromous Rainbow Smelt have not been assessed by COSEWIC and are globally designated as least concern on the IUCN Red List (NatureServe 2013i).

Spawning of Rainbow Smelt occurs in the spring from April to June in Canada during which time adults migrate up rivers to freshwater (Scott and Scott 1988). Males arrive prior to females and larger fish spawn first. Many males die following spawning. Genetically, freshwater and anadromous Rainbow Smelt are highly differentiated and there is some evidence of heterogeneity between the anadromous groups along the Atlantic coast (Taylor and Bentzen 1993). There is evidence of very high site fidelity (95-99%) in Rainbow Smelt (Bradbury et al. 2008). Smelts are carnivorous, feeding on crustaceans, worms, and small fish, and are themselves a significant prey item for larger fish, birds, and mammals. Population resilience is considered to be medium (Froese and Pauly 2018)

Rainbow Smelt are abundant in the Bay of Fundy (MacDonald et al. 1984). They have been caught in high numbers between July and October during studies in the Annapolis River (e.g. Daborn et al. 1979, Stokesbury 1985, Gibson and Daborn 1995b) and certainly are present during other months as well. There is no information about their survival moving past the Annapolis TiGS.

Population-level impacts are assessed as medium because of their medium resiliency, and because the population is very likely native to the Annapolis River (all of the population is likely to encounter the Annapolis TiGS). However, information about fishway usage is limited and delayed mortality and movement patterns in the vicinity of the turbine are not known leading to high uncertainty in the conclusion.

Rock Gunnel (*Pholis gunnellus*)

Level of impact (Uncertainty):	Low (High)
Life stages present:	Adult
Relative abundance:	Low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed
Fisheries:	None

The Rock Gunnel (*Pholis gunnellus*) is a coastal, seasonal migrant that spends the warmer months, March through November, inshore in the intertidal zone often above the low water mark. They travel to deeper waters in the winter to spawn (Sawyer 1967, Scott and Scott 1988). Rock Gunnel are found from Labrador to the Delaware Bay in the Northwest Atlantic and are commonly seen in the Bay of Fundy. They are not targeted in any fisheries, although they are a source of food for some commercially important species. Rock Gunnel have not been assessed by COSEWIC or globally by the IUCN. There are a few accounts of Rock Gunnel in the Annapolis River and estuary (Horne and Campana 1989). There is no information about their survival when moving past the Annapolis TiGS.

Population-level-impacts are likely low due to a broad geographic distribution suggestive that only a small proportion of the population is likely to encounter the turbine. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known leading to high uncertainty in the conclusion. Population structure is not known, but a local population is possible.

Sea Lamprey (*Petromyzon marinus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	Moderate
Usage:	Resident (at least seasonally); Upstream migration for spawning; Downstream migration
Turbine mortality rates:	Acute and delayed mortality; 0% (all fish were alive)
Resiliency:	Low
COSEWIC/IUCN designation:	Unassessed/Least concern
Fisheries:	None

The Sea Lamprey (*Petromyzon marinus*) is a parasitic fish species that inhabits coastal and inland freshwater areas in the Northeast and Northwest Atlantic. In the Northwest Atlantic they range from Greenland to Florida with large inland freshwater populations, such as those in the Great Lakes (Scott and Scott 1988). Sea Lamprey are not fished commercially and are detrimental to many commercial and recreational fisheries, as adults may parasitize large, targeted species (e.g. salmon, trout). They have not been assessed by COSEWIC and are of least concern on the IUCN Red List (NatureServe 2013j).

After roughly two years at sea, Sea Lamprey return to rivers to spawn in fresh water (Scott and Scott 1988). Spawning is variable by location and has been estimated to occur in late spring to early summer. There is little genetic variation in anadromous fish along the Atlantic Coast and they may all be part of a single panmictic population (Bryan et al. 2005). Young, upon hatching, remain as ammocoetes in the river bed for six to eight years feeding on plankton. Once they reach around 25 cm in length, they metamorphose into adults and begin a seaward migration in the fall. At sea they feed on fish and grow rapidly, reaching 70 to 80 cm and 600 to 800 g. Adults die after spawning. Population resiliency of Sea Lamprey is considered low (Froese and Pauly 2018).

There is evidence of a spawning migration of Sea Lamprey in the Annapolis River (Beamish 1980) and individuals have been caught in turbine mortality studies (Stokesbury 1985; Gibson and Myers 2002a). Single pass acute and delayed turbine mortality rate was estimated at 0%, although there is a high level of uncertainty given the small sample size (Gibson and Myers 2000). Lamprey caught when monitoring during studies at the Annapolis TiGS in the 1990's were small, about 15 to 20 cm in length (J. Gibson, personal observation), and are likely seaward migrating juveniles.

Population-level impacts are assessed as low due to the very low acute turbine mortality rate estimates and potential for panmictic population structure such that only a small portion of the population is likely to encounter the turbine. However, fishway usage and movement patterns in the vicinity of the turbine are not known and population structure is not well understood, although a local population is possible, leading to moderate uncertainty is the conclusion.

Sea Raven (*Hemitripterus americanus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Unknown
Relative abundance:	Unknown
Usage:	Unknown
Turbine mortality rates:	Unknown
Resiliency:	Very Low
COSEWIC/IUCN designation:	Unassessed
Fisheries:	None

The Sea Raven (*Hemitripterus americanus*) is a benthic, marine fish living in the Northwest Atlantic from Labrador to the Chesapeake Bay, and are particularly common on the Scotian Shelf and the Bay of Fundy (Scott and Scott 1988). They are not targeted by commercial or recreational fisheries, although they are sometimes kept as bycatch and used at bait (Comeau et al. 2009). They have not been assessed by COSEWIC or the IUCN.

Sea Raven prefer warm shallow waters and infrequently venture into estuaries and tidal flats with records from estuaries in the outer Bay of Fundy (MacDonald et al. 1984).They are known to spawn in the late fall to winter and mature at a size of 36 cm in males and 28 cm in females (Beacham 1981, Scott and Scott 1988). Population resiliency is considered to be very low (Froese and Pauly 2018).

Sea raven have not been caught in fish passage studies at the Annapolis TiGS.

Population-level impacts are assessed as low due to the lack of preferred habitat within the Annapolis River system suggestive that only small proportion of a population is likely to encounter the Annapolis TiGS. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known and population structure is not well understood leading to moderate uncertainty in the conclusion.

Smooth Flounder (*Liopsetta putnami*)

Level of impact (Uncertainty):	Low (Low)
Life stages present:	Juvenile, adult
Relative abundance:	Low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed
Fisheries:	None

The Smooth Flounder (*Liopsetta putnami*) is a small flatfish that inhabits inshore habitats along the eastern coast of North America ranging from Labrador southward to Rhode Island (Scott and Scott 1988). They prefer estuaries, river mouths, and sheltered bays with mud or silt bottoms (Scott and Scott 1988, Armstrong and Starr 1994). They are not a target species for commercial or recreational fisheries, likely due to their small size. They have not been assessed by COSEWIC or the IUCN.

Little information is available for Smooth Flounder in Canadian waters; however spawning is believed to occur in late winter to early spring (Scott and Scott 1988). In the U.S. they spawn in the winter over inshore mudflats, which are important nursery grounds (Armstrong and Starr 1994, Armstrong 1997). Male Smooth Flounder mature in their first year of life and females in either their first or second year (Armstrong and Starr 1994). Females normally grow larger than males, to a maximum total length of 30 cm. As small carnivorous fish, they feed mainly on marine worms, and small crustaceans and molluscs (Scott and Scott 1988). They are also preyed upon by larger fish and sea birds, including Osprey and Cormorants. Population resiliency is considered to be medium (Froese and Pauly 2018).

Smooth Flounder have been documented in the Annapolis River on one occasion (Gibson and Myers 2002a).

Population-level impacts are assessed as low due to a broad geographic distribution and preference for habitat not widely available in the Annapolis River and estuary, indicative that only a small portion of the population is likely to encounter the Annapolis TiGS. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact. Distribution and habitat preference are well understood and therefore uncertainty with this conclusion is low.

Spiny Dogfish (*Squalus acanthias*)

Level of impact (Uncertainty):	Low (Low)
Life stages present:	Juvenile, adult
Relative abundance:	Very low
Usage:	Transient
Turbine mortality rates:	Unknown
Resiliency:	Very Low
COSEWIC/IUCN designation:	Special concern / Endangered
Fisheries:	Commercial (not fished in the Annapolis River and estuary)

The Spiny Dogfish (*Squalus acanthias*) is a small, temperate shark that schools in coastal areas, including estuaries, in the Pacific and Atlantic Oceans (Scott and Scott 1988). In the Northwest Atlantic, they range from Greenland to Florida. They are seasonally abundant in Canada and are fished commercially, mainly on the Scotian Shelf and in the Bay of Fundy where they are found in high abundance (COSEWIC 2010b). COSEWIC has designated Spiny Dogfish as special concern for their entire Canadian range, based on genetic evidence showing little variability throughout all of the North Atlantic (COSEWIC 2010b, Veriissimo 2010). The Northwest Atlantic unit of Spiny Dogfish has been designated as endangered on the IUCN Red List (Fordham et al. 2006).

In the Northwest Atlantic, Spiny Dogfish migrate long distances and can take up residence in specific regions (Campana et al. 2009). Generally, they overwinter offshore on the edge of the Scotian Shelf, move south out of Canadian waters or migrate across the Atlantic (Scott and Scott 1988). Adults, including pregnant females, usually move inshore in the late spring (June) and many take up seasonal residence in the Bay of Fundy until colder temperatures drive them back offshore (Campana et al. 2009). Mating and pupping occur offshore during the winter. Spiny dogfish are ovoviviparous and mate directly after pupping. Their gestation period lasts for 22 months and young are born at 25 to 30 cm in length. They have slow growth, reaching sexual maturity at about a length of 60 cm and six years of age in males, and about 80 cm length and 12 years of age in females. They have a lifespan of 30 to 40 years and may reach a maximum size of 95 cm in males and 120 cm in females. Their population resiliency is considered very low (Froese and Pauly 2018).

Population-level impacts are assessed as low due to a broad distribution and high abundance in the Bay of Fundy during the summer and fall, indicative that only a small portion of the population would encounter the Annapolis TiGS. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact. Distribution, abundance and habitat preference are well understood and therefore uncertainty with this conclusion is low.

Striped Bass (*Morone saxatilis*)

Level of impact (Uncertainty):	Annapolis: Extreme (High) Shubenacadie: Low (Moderate) USA origin: Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	Historically high, now low
Usage:	Native population was resident; Local knowledge indicated overwintering in headpond; Transient individuals from other populations
Turbine mortality rates:	Unknown
Resiliency:	Low
COSEWIC/IUCN designation:	Endangered / Least concern
Fisheries:	Commercial (not in the Annapolis River and estuary), recreational (historically large in the Annapolis River and estuary), aboriginal

The Striped Bass (*Morone saxatilis*) is a large, moderately long-lived, anadromous fish species (Scott and Scott 1988). They school in coastal inshore waters along the eastern coast of North America from the Gulf of St. Lawrence down to Florida (Scott and Scott 1988, COSEWIC 2012c). Striped Bass support important commercial, recreational, and aboriginal fisheries. In Canada, they are sometimes retained as bycatch from other finfish fisheries, but are particularly important as a recreational fish species (COSEWIC 2012c, Bradford et al. 2015). Throughout the Maritime Provinces there are also a number of First Nations licenses for Striped Bass (Bradford et al. 2015). The Annapolis River used to have an active recreational fishery for Striped Bass (Jessop and Doubleday 1976), but less fishing for bass occurs now.

Striped Bass spawn in fresh water in their natal rivers in the spring (Williams et al. 1984, Scott and Scott 1988) and may overwinter in the freshwater parts of rivers, in adjoining lakes (Rulifson and Dadswell 1995, Douglas et al. 2003) or in the marine environment. Striped Bass do not all spawn every year. Juveniles live in shallow shoreline areas that may be tidal or non-tidal (Bradford et al. 2015). Maturation occurs at three to four years and about 32 cm in fork length for males, and four to six years and about 50 cm in fork length for females (Douglas et al. 2003). Population resiliency of Striped Bass is thought to be low (Froese and Pauly 2018).

Wirgin et al. (1993) and Bentzen et al. (2009) determined through genetic analyses that there are several distinct groups of Striped Bass. Striped Bass in Canada have been assessed by COSEWIC as three designatable units. The COSEWIC designated the Southern Gulf of St. Lawrence DU as special concern, and both the Bay of Fundy and the St. Lawrence River DUs as endangered (COSEWIC 2012c). On a global scale, Striped Bass have been listed as least concern by IUCN (NatureServe 2013h).

Within the Bay of Fundy DU, there were three known spawning populations: Shubenacadie, Saint John, and Annapolis (COSEWIC 2012c, DFO 2014b, Bradford et al. 2015). The Annapolis population was considered extirpated by COSEWIC and the status of the Saint John River population was unclear.

Adult Striped Bass are found in many coastal rivers and estuaries in North America, including non-natal habitat, where they aggregate in spring and summer to feed (Dadswell et al. 1986, Waldman et al. 1990, Dadswell and Rulifson 1994, Rulifson and Dadswell 1995, Bradford et al. 2015). Both genetic and tagging studies have shown that Striped Bass undergo coastal migrations and forage in estuaries other than those at the mouths of their natal rivers (Waldman et al. 1990, Wirgin et al. 1994). Striped bass tagged elsewhere, including USA rivers, have been captured in the Annapolis River estuary (Waldman et al. 1990), and Striped Bass tagged in the Annapolis River have been recaptured in other areas (Harris 1988). In both cases, the actual river-of-origin of the fish is not known and the number of recaptured fish is small. The relative abundance of these natal and non-natal fish in the Annapolis River estuary would be expected to vary depending on many factors including the abundance decline of the Annapolis River population and the abundance of fish in other populations. In this document, Striped Bass are treated as three groups: the native Annapolis River population, the Shubenacadie River population (which is presently at high abundance), and the USA populations.

Changes to the Annapolis River Striped Bass population pre-date the construction of the tidal generating station. Prior to the construction of the Annapolis TiGS, there was concern with the egg survival of Striped Bass in the Annapolis River. While eggs were obtained from the Annapolis River in several years between 1976 (Williams et al. 1984) and 1994 (Jessop 1995), the last larvae captured was in 1976 (Williams et al. 1984). Seine surveys in 1976 (Williams et al. 1984), 1980 (Jessop 1983), 1985 (Stokesbury 1985), 1986 (Stokesbury 1986), 1992 (Jessop 1995), 1993 (Gibson and Daborn 1993), 1994 (Gibson and Daborn 1995b), 2009 ,2010, 2014 and 2015 (Clean Annapolis River Project - unpublished) have failed to capture any young-of-the-year Striped Bass. Together, these results provide evidence that Striped Bass have not been successfully reproducing in the Annapolis River since at least the mid-1970’s.

Creel surveys in 1976 indicated an increase in the size of fish which is consistent with reproductive failure as it is expected to lead to increase in the size of fish until the last fish are gone (Jessop and Doubleday 1976). However, a creel survey undertaken after the installation of the turbine identified smaller, younger fish than earlier creel surveys (e.g. Harris 1988). The presence of these younger fish in conjunction with the evidence of reproductive failure is suggestive that these fish may be from other populations. Based on the above information, the Annapolis TiGS is not the only factor that would contribute to the documented changes to the size distribution of Striped Bass in the Annapolis River estuary. The present composition of the Striped Bass assemblage in the Annapolis River estuary is not known.

There are records of dead Striped Bass in the vicinity of the Annapolis TiGS (e.g. Dadswell and Rulifson 1994) that can reasonably be attributed to passage at the Annapolis TiGS.

Here, population-level impacts of the Annapolis TiGS are assessed as extreme for the native Annapolis River population of Striped Bass due to their very low reproductive rate in this river, their well-defined population structure and their life history. All of a native population would be expected to encounter the turbine several times throughout their lives and potentially several times within a year. However, due to cumulative effects with high mortality rates in other parts of their life cycle; as well as unknown fishway passage rates, turbine mortality rates and movement patterns in the vicinity of the turbine, the uncertainty in the conclusion is high. For Striped Bass from non-native sources such as the Shubenacadie and U.S.A. population-level impacts are considered low due to the low proportion of each of these populations thought to encounter the turbine. However fishway usage, turbine mortality rates and movement patterns in the vicinity of the turbine are not known and the information on the proportion of the population encountering the Annapolis TiGS is limited, leading to moderate uncertainty in the conclusion.

Threespine Stickleback (*Gasterosteus aculeatus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	High
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	High
COSEWIC/IUCN designation:	Unassessed / Least concern
Fisheries:	None

The Threespine Stickleback (*Gasterosteus aculeatus*) is a small, marine, brackish, or freshwater fish with a circumpolar distribution that is found in coastal regions in the Northern Hemisphere (Scott and Scott 1988). In the Northwest Atlantic, they range from Baffin Island and Hudson's Bay to Cape Hatteras. They are not fished commercially or recreationally and have not been assessed by COSEWIC. They are designated as least concern on the IUCN Red List (NatureServe 2015).

Population structure in Canada is not well understood. However, Threespine Stickleback in Alaska share similar DNA throughout all marine areas sampled suggesting a panmictic oceanic population. Conversely, there is variability between marine and freshwater individuals and between freshwater individuals inhabiting different locations (Hohenlohe et al. 2010). They are planktivores, feeding on small crustaceans and insects, and are preyed upon by larger fish, birds and mammals.

In Canada, Threespine Stickleback spawn in fresh water in June to July, when males build nests and court females (Scott and Scott 1988). Females may deposit up to 600 eggs in a single clutch. Young grow quickly in their first year. They reach maturity at age one and may live up to 3.5 years of age. Threespine Stickleback have a high level of resilience based on life history characteristics (Froese and Pauly 2018).

Threespine Stickleback have been caught during the summer in the Annapolis River (e.g. Stokesbury 1985, Gibson and Daborn 1993). There is no information about turbine mortality rates or potential for multiple passes, but individuals utilize all available routes through the Annapolis causeway (Gibson and Daborn 1993).

Population-level impacts are assessed as low due to their small size, low turbine mortality estimates for other stickleback species and high resiliency. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known, and population structure is not well understood, although a local population is likely, leading to moderate uncertainty is the conclusion.

White Perch (*Morone americana*)

Level of impact (Uncertainty):	Low (Low)
Life stages present:	Juvenile, adult
Relative abundance:	Very low
Usage:	Primarily occupy habitat well upstream of the causeway
Turbine mortality rates:	Unknown
Resiliency:	Low
COSEWIC/IUCN designation:	Unassessed/ Least concern
Fisheries:	Recreational

The White Perch (*Morone americana*) is an anadromous species found on the eastern coast of North American from the Gulf of Saint Lawrence to South Carolina (Scott and Scott 1988). White Perch are not fished commercially in Canada, but have been in the U.S in the past. Although commercial value is low, they are an important fish for recreational angling in Atlantic Canada. Currently in the Annapolis River, they can be fished from April 1st through October 31st with a daily bag limit of 25 fish (Department of Fisheries and Aquaculture 2018). They have not been assessed by COSEWIC and are currently designated as least concern on the IUCN Red List (NatureServe 2013g).

Considered a non-migratory species, White Perch spawn in fresh and slightly brackish waters between May and June in Canada (Scott and Scott 1988). They are tolerant of a relatively small range of salinities and are often found in landlocked freshwater systems in Canada. Due to their limited tolerance to high salinities, populations are generally restricted to their home estuaries and there is little to no evidence of mixed populations on a large spatial scale. Within estuaries, White Perch exhibit partial migrations; parts of a single population remain freshwater residents while other individuals show brackish migratory behavior (Kerr et al. 2011). There has also been population mixing in river populations that share the same estuary (Kerr and Secor 2012). Individuals hatch at around 2 mm in size and growth is variable by location. White Perch have an average lifespan of six to seven years with an average total length of 13.5 cm. They first spawn at age three. Population resilience of White Perch is thought to be medium based on life history characteristics (Froese and Pauly 2018).

White Perch have been caught in the Annapolis River during studies both pre- and post-construction of the tidal generating station (Daborn et al. 1979, Stokesbury 1985, Jessop 1995). There is no information on turbine mortality rates for White Perch at the Annapolis TiGS.

Despite residency in the Annapolis River, Population-level impacts are assessed as low due to a preference for lower salinity that would limit encounters with the Annapolis TiGS. Given the low proportion of the population expected to encounter the turbine, a high mortality rate at the TiGS would still have a low population-level impact. Salinity preference is well understood and therefore uncertainty with this conclusion is low.

Windowpane (*Scophthalmus aquosus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	Low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Acute only; 8.8% (95% CI: <0.1- 59.4)
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed
Fisheries:	Commercial (bycatch)

The Windowpane (*Scophthalmus aquosus*) is a benthic flatfish that inhabits coastal waters over sandy bottoms (Scott and Scott 1988). They are tolerant of a wide range of temperatures allowing them to remain in coastal regions year around. Geographically, they range along the eastern coast of North America from the Southern Gulf of Saint Lawrence down to Florida. There are no directed fisheries for Windowpane in Canada, although bycatch from the groundfish fisheries is retained. Windowpane have not been assessed by COSEWIC or the IUCN.

In Canadian waters, spawning occurs in coastal marine environments in late spring to early summer (Scott and Scott 1988). The Windowpane is fast growing with maximum growth during the spring and summer. Individuals mature between ages 3 and 4, at a size of roughly 23 to 25 cm total length. The maximum total length of Windowpane is about 46 cm. Population resilience is considered medium, based on life history characteristics (Froese and Pauly 2018).

Population structure of Windowpane in Canadian waters has not been studied; however stocks have been divided in the USA based on location (Chang et al. 1999). In the Annapolis River estuary, Windowpane have been caught above and below the causeway. There is a single estimate of the acute turbine mortality rate for Windowpane at the Annapolis TiGS (see TOR 5).

Population-level-impacts are likely low due to a low acute turbine mortality rate estimate coupled with a broad distribution, suggestive that only a small proportion of the population may encounter the turbine. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known, and population structure is not well understood, leading to moderate uncertainty in the conclusion.

Winter Flounder (*Pseudopleuronectes americanus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile, adult
Relative abundance:	Low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Acute only; 5.8% (0.8- 31.2)
Resiliency:	Medium
COSEWIC/IUCN designation:	Unassessed
Fisheries:	Commercial, recreational

The Winter Flounder (*Pseudopleuronectes americanus*) is a bottom-dwelling species of flatfish that is tolerant to a wide range of salinities. Although they are considered a coastal marine species, they are often found in brackish rivers and estuaries (Scott and Scott 1988). They range from Labrador to Georgia in the Northwest Atlantic Ocean and are fished commercially, within the broader group of groundfish. In Canada, fishing is most intensive off the coast of Newfoundland, in the Gulf of Saint Lawrence, on the Eastern Scotian Shelf, and in the Bay of Fundy (Scott and Scott 1988, DeCelles and Cardin 2011). Winter Flounder are fished recreationally in the Bay of Fundy where fishing is only closed for the month of January and there is a bag limit of 10 fish (DFO 2018a). Winter Flounder have not been assessed by COSEWIC or the IUCN.

In the Bay of Fundy, spawning occurs in the spring, with individuals moving inshore in April or May and spawning at night (McCracken 1963, Scott and Scott 1988). Following spawning, they remain in the shallow inshore waters over muddy or sandy bottoms for the spring and early summer. Winter Flounder metamorphose from larval form to flatfish form around 2.5 to 3.5 months of age and mature at roughly 20 cm length for males and at about 25 cm length for females (Scott and Scott 1988). Age at maturity is not well understood for fish in the Bay of Fundy. Winter Flounder can grow to a size of 45 cm in total length with an average lifespan of 11 years for males and 12 years for females (DFO 2016b). They are preyed upon by larger fish, marine mammals and sea birds. Based on life history characteristics, Winter Flounder are thought to have a medium resiliency to increased mortality (Froese and Pauly 2018).

Winter Flounder have been recorded upstream of the Annapolis causeway on many occasions (e.g. Daborn et al. 1979, Stokesbury 1985, Gibson and Myers 2002a). The number of individuals that have been recorded moving past the causeway is low and likely represents only a small proportion of the overall population considering their high abundance throughout the Bay of Fundy (DFO 2018b). An estimate of the acute turbine mortality rate at the Annapolis TiGS is available for species (see TOR 5).

Population-level impacts are assessed as low due to the low acute turbine mortality rate estimate, and their broad distribution throughout the Bay of Fundy, suggestive that only a small proportion of a population may encounter the turbine. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known, and population structure is not well understood, leading to moderate uncertainty in the conclusion.

Wrymouth (*Cryptacanthodes maculatus*)

Level of impact (Uncertainty):	Low (Moderate)
Life stages present:	Juvenile (others are unknown)
Relative abundance:	Low
Usage:	Resident (at least seasonally)
Turbine mortality rates:	Unknown
Resiliency:	Unknown
COSEWIC/IUCN designation:	Unassessed
Fisheries:	None

The Wrymouth (*Cryptacanthodes maculatus*) is a benthic inshore burrower, living in areas with muddy and soft bottoms in the Northwest Atlantic from Newfoundland to New Jersey (Scott and Scott 1988). They are not the target of directed fisheries and have not been assessed by COSEWIC or the IUCN. Spawning may occur in winter and individuals can grow to a maximum of 90 cm, feeding mainly on benthic crustaceans. The population resilience of Wrymouth based on life history characteristics is unknown (Froese and Pauly 2018).

Wrymouth have been caught in the Bay of Fundy and adjoining bays and estuaries (Comeau et al. 2009), including the Annapolis on one occasion (Gibson and Myers 2002b).

Population-level impacts are assessed as low due to their broad distribution and low numbers encountered during fish passage studies at the Annapolis TiGS. This information is suggestive that only a small proportion of a population is likely to encounter the turbine. However, fishway usage, delayed turbine mortality rates and movement patterns in the vicinity of the turbine are not known leading to moderate uncertainty in the conclusion. Additionally, population structure is not known.

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Using macroalgal bioindicators to map nutrient plumes from fish farms and other sources at a bay-wide scale

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ABSTRACT: Human activities can elevate coastal levels of dissolved inorganic nitrogen (DIN). As macroalgae readily absorb and accumulate DIN, the elemental (total N and C:N ratio) composition of their tissues is less affected by temporal fluctuations compared to more direct measures of DIN concentration. Additionally, their isotopic ($\delta^{15}N$) composition can reflect that of the source, which could potentially be used to identify between multiple effluent sources. To investigate whether macroalgal 'bioindicators' could map and distinguish between multiple effluents, 2 species of macroalgae (*Chondrus crispus* and *Palmaria palmata*) were deployed in a bay containing a salmon farm and sewage treatment facility. Both species exhibited high total N and low C:N ratio near the salmon farm and sewage facility. However, the elemental composition of *C. crispus* was influenced over a greater distance than that of *P. palmata*. Differences were also observed between their isotopic composition, as *C. crispus* indicated that the salmon farm and sewage facility had distinct $\delta^{15}N$ signatures, whereas values of $\delta^{15}N$ in *P. palmata* had not changed after 10 d incubation in the field. Interestingly, the distinct isotopic signals observed in *C. crispus* were likely a result of higher DIN concentrations at the salmon farm, which likely caused macroalgae to fractionate and form biomass lighter in $\delta^{15}N$. Overall, this study suggests that macroalgal bioindicators can monitor and identify between multiple effluent sources, which could provide a useful tool for coastal management. However, some species of macroalgae may make more effective bioindicators than others, and the mechanisms underlying their fractionation require further investigation.

KEY WORDS: Aquaculture · $\delta^{15}N$ · Isotope · Macroalgae · Nitrogen · Salmon · Sewage · Wastewater

1. INTRODUCTION

Human activities can elevate nutrient levels in coastal environments (Nixon 1995, Smith 2003, Eddy 2005). Nitrogen (N) loading is of particular concern, as it can lead to algal blooms, oxygen depletion and biodiversity loss (Nixon 1995, Howarth et al. 2011). Given that net-pen aquaculture releases N and other wastes into the surrounding water, monitoring its effluents has developed into a very active area of research (e.g. Cloern 2001, Callier et al. 2013, Jansen et al. 2018). However, most water bodies also receive effluents from other sources, such as agriculture, urbanisation, industrialisation and wastewater treatment facilities (Leonard et al. 1997, Costanzo et al. 2005, Alquezar et al. 2013). Hence, there are calls for a change in the way aquaculture is monitored and managed, from the current approach where the localised effects of single farms are regarded, towards an ecosystem approach to aquaculture, whereby the

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effects of multiple farms and human activities are simultaneously considered at a wider scale (FAO 2007, 2010).

Fish farms release nutrients into the environment as either particulate or dissolved wastes. Particulate wastes derive from faeces and uneaten feed, and represent between 13 and 32% of all the N released from fish farms (Islam 2005, Holmer et al. 2007, Sanderson et al. 2008, Callier et al. 2013). These particulate wastes quickly settle onto the seafloor and rarely disperse more than a few hundred metres (Brager et al. 2015, Price et al. 2015, Bannister et al. 2016, Filgueira et al. 2017). Over the last 3 decades, the amount of particulate wastes produced by fish farms has been significantly reduced due to the development of more efficient feeds and feeding systems (Islam 2005, Sørensen 2012, Sprague et al. 2016). In contrast, dissolved wastes are excreted by fish directly into the water column and represent between 68 and 87% of N (Wang et al. 2012). Although this dissolved fraction makes up the majority of fish farm wastes, less is known about its dispersal and persistence within the marine environment (Price et al. 2015, Jansen et al. 2018).

Fish farm effluents are enriched in dissolved inorganic N (DIN), a term which includes nitrite (NO_2^-), nitrate (NO_3^-), ammonia (NH_3) and ammonium (NH_4^+). Up to 90% of all the N excreted by marine fish occurs as NH_3, which quickly converts to NH_4^+ at the pH of seawater (reviewed by Leung et al. 1999). Consequently, several studies have reported elevated NH_4^+ concentrations close to fish farms (Navarro et al. 2008, Sanderson et al. 2008, Jansen et al. 2018). However, a recent and comprehensive review showed that most studies to date have found no evidence of fish farms increasing DIN concentrations (Price et al. 2015). These varying and conflicting results can be attributed to several factors. First, background nutrient concentrations, and the release of nutrients from fish farms, both exhibit strong daily pulses and seasonal fluctuations (Karakassis et al. 2001). Second, fish farms are often purposefully located in areas of high water exchange, quickly dissolving and dispersing any dissolved wastes (Dalsgaard & Krause-Jensen 2006). Third, any inputs of DIN are rapidly assimilated by marine organisms and lost to the atmosphere through volatilization (Dailer et al. 2010). Hence, any increase in DIN is likely to be small, short-lived and difficult to detect. For these reasons, monitoring dissolved nutrient concentrations is prone to large margins of error, making it time and cost expensive due to the need for continual and repeated sampling (Dalsgaard & Krause-Jensen 2006).

Using macroalgae as biological indicators (or 'bioindicators') is an alternative method of assessing dissolved nutrient levels (García-Seoane et al. 2018). Macroalgae readily absorb and accumulate N, meaning the N content of their tissues is less affected by short-term fluctuations compared to more direct measures of DIN concentration (Chopin et al. 1995, García-Sanz et al. 2010, Carballeira et al. 2013). Also, macroalgal bioindicators only absorb the fraction of nutrients that are bioavailable (i.e. the fraction responsible for eutrophication) and can be logistically simpler and more time effective (Costanzo et al. 2001, Dailer et al. 2010). Lastly, and of growing interest, is the potential for their N isotope composition to identify and distinguish between multiple effluent sources (Heaton 1986, Costanzo et al. 2001, Lemesle et al. 2016).

The N isotope composition of a sample is generally measured as the ratio of ^{15}N to ^{14}N relative to air, and expressed on a delta scale ($\delta^{15}N$) in units of ‰ (Peterson & Fry 1987). Although isotopes undergo the same biochemical pathways, the cells of most organisms will preferentially take up the lighter ^{14}N isotope due to faster reaction times and metabolic processes (Mariotti et al. 1982, Dailer et al. 2010, Newton 2010). This process of 'fractionation' generally occurs throughout the trophic web, which is why $\delta^{15}N$ usually increases with trophic level (Mill et al. 2007). Interestingly, fractionation can cause different effluent sources to exhibit distinct isotopic signatures. For instance, sewage tends to be high in $\delta^{15}N$ (8~30‰), as it is derived from the wastes of humans and animals at high trophic levels (Dailer et al. 2010, García-Sanz et al. 2010). Also, bacteria involved in sewage treatment fractionate the available pool of N, further enriching the effluent in ^{15}N (Middlebrooks & Pano 1983, Bannon & Roman 2008). In contrast, fish farm effluents tend to have moderate $\delta^{15}N$ values (8~11‰) due to the use of feeds derived from both animal (^{15}N-enriched) and plant (^{15}N-depleted) ingredients (García-Sanz et al. 2010, Wang et al. 2014). Pulp mill effluents tend to have low $\delta^{15}N$ values due to their use of terrestrially derived wood chips (Wayland & Hobson 2001, Oakes et al. 2010). Lastly, agricultural effluents tend to have $\delta^{15}N$ values close to 0 because of the application of fertilizers enriched in NH_4^+ or NO_3^- fixed from atmospheric N (Costanzo et al. 2001, Dailer et al. 2010). As macroalgae are widely believed to take up ^{15}N in proportion to its availability, the $\delta^{15}N$ of their tissues can reflect the signature of the dominant source of N (Gartner et al. 2002, García-Sanz et al. 2010, Lemesle et al. 2015).

Macroalgal bioindicators have been used to monitor effluents in coastal waters since the late 1970s (reviewed by García-Seoane et al. 2018). The majority of these studies have relied on collecting native species occurring naturally within an area. However, a growing number have transplanted live individuals from one location to another. The advantages of this approach are that macroalgae can be: (1) deployed in areas where do they not naturally occur (e.g. in highly impacted areas or in deep water); (2) conditioned to be isotopically similar prior to their deployment; (3) starved of N so they absorb it more readily; and (4) exposed to effluents for controlled periods of time (Costanzo et al. 2001, Alquezar et al. 2013, García-Seoane et al. 2018). For example, Costanzo et al. (2001) used transplanted macroalgae to map wastewater effluents in Australia and were able to track improvements made to wastewater management practices within the region. Transplanted macroalgae have also been used to map effluents emanating from fish farms in Japan (Yokoyama & Ishihi 2010). However, to date, few attempts have been made to map aquaculture effluents in relation to other sources.

The aim of this study was to investigate whether macroalgal bioindicators could map the footprint of multiple effluents, and to gain a greater insight into the dispersal of dissolved aquaculture wastes. Two macroalgae species were deployed across a bay containing a salmon farm and a sewage treatment facility to test the following hypotheses: (1) macroalgae deployed near the salmon farm and sewage facility will display elevated $\delta^{15}N$ values, reflecting the composition of the dominant N source; and (2) the total N content of macroalgae will be higher near anthropogenic sources of N. If macroalgal bioindicators prove capable of mapping the footprint of multiple effluents, they could provide a useful tool in helping coastal management transition towards an ecosystem approach to aquaculture.

2. MATERIALS AND METHODS

2.1. Study region

Liverpool Bay is located along the southern shore of Nova Scotia in eastern Canada (Fig. 1). The bay measures approximately 6 km long and 2 km wide, and has a maximum depth of 40 m, a tidal range of 2 m and a flushing time of 65 h (Gregory et al. 1993,

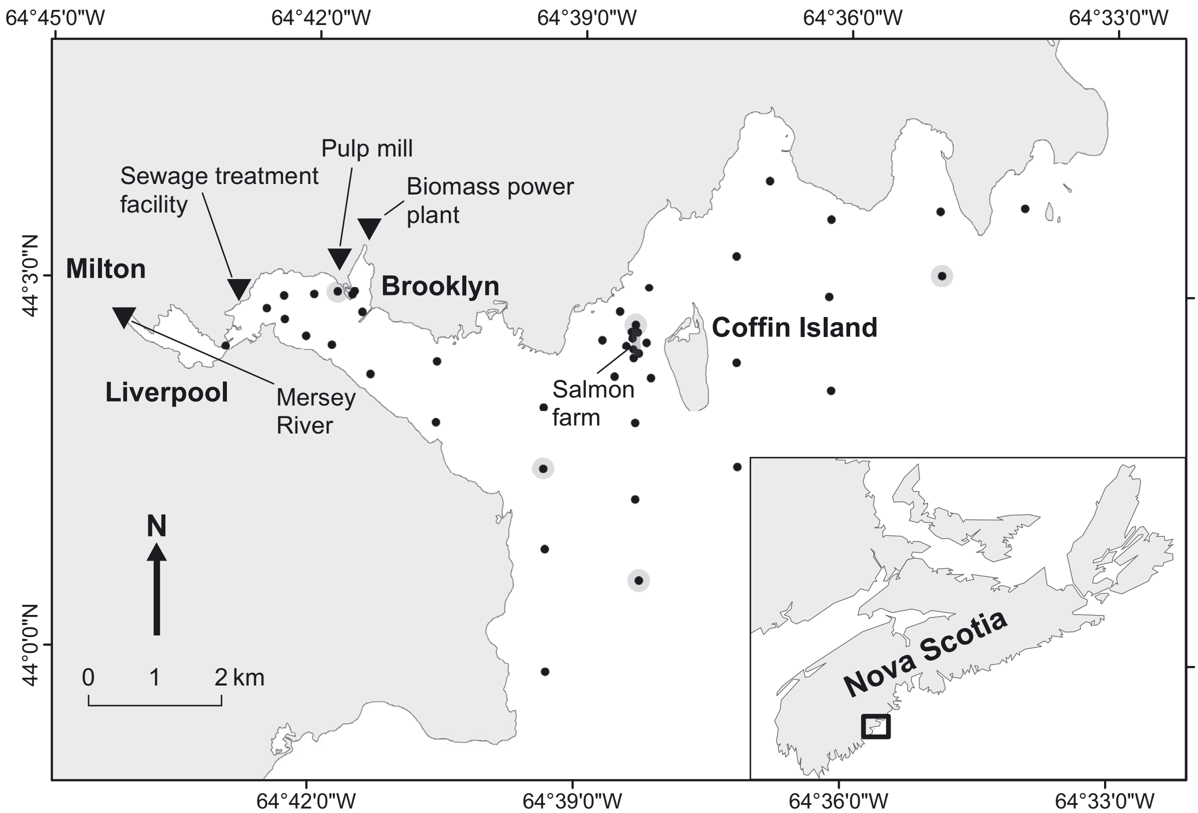

Fig. 1. Liverpool Bay, Nova Scotia, Canada, indicating towns, key geographical features and probable point sources. Points indicate the locations of the deployments of the macroalgae *Chondrus crispus* and *Palmaria palmata*. Grey circles indicate stations where surface water samples were taken

Stewart & White 2001). The Mersey River discharges into the northwest of the bay and has the largest outflow and watershed of all Nova Scotian rivers (Davis & Browne 1998). Between 2014 and 2018, the Mersey River exhibited the highest daily mean (±SE) discharge rate of 190 ± 21.4 m^3 s^{-1} in December, and the lowest daily discharge of 18 ± 8.6 m^3 s^{-1} in August (www.wateroffice.ec.gc.ca). The discharge of the Mersey River was <4 m^3 s^{-1} at the time of this study.

The drainage catchment of Liverpool Bay encompasses the towns of Brooklyn, Liverpool and Milton, which have a combined population of 4460 (Statistics Canada 2016). Industrial and urban developments in the area include a biomass power plant (in part-time operation), a scallop processing plant, commercial docking wharfs (>50 registered fishing vessels), a shipyard and a marina (Stewart & White 2001). A pulp mill, which has been decommissioned since June 2012, is believed to have created large deposits of sawdust on the surrounding seafloor (V. Fisher pers. comm.). There is also a municipal wastewater treatment facility consisting of 3 aerated lagoons capable of ultraviolet disinfection that release treated sewage from the surrounding towns into Liverpool Harbour (A. Grant pers. comm.). Lastly, a salmon (Atlantic salmon *Salmo salar*) farm was built to the west of Coffin Island in 2010 and consists of 14 pens, 32 m in diameter, situated at depths of 12–16 m. Salmon production is approximately a 2 yr cycle which starts with the stocking of smolts (body mass ~110 g) in the spring. These are later harvested as full-size adults (body mass ~6 kg) during the winter of the following year, and the farm is left fallow until the next spring.

2.2. Field sampling

Fieldwork was conducted in Liverpool Bay in August 2018. Forty stations (Fig. 1) were designated to cover as wide an area as logistically possible but were spatially biased to provide greater sampling in areas close to probable point sources (i.e. the salmon farm and sewage treatment facility). At each station, approximately 20 g of the macroalgae *Chondrus crispus* and *Palmaria palmata* were placed in separate transparent, perforated chambers and suspended in the water column at a depth of 3 m using a combination of buoys, leaded ropes, weights and anchors (see Costanzo et al. 2001, 2005). An additional 5 chambers were hung directly from the outer rim of the salmon pens using just ropes and weights. All chambers were then left to incubate for 10 d based on immersion times trialled by Lemesle et al. (2016). In addition, 1 l surface water samples (n = 10) were collected using sterilised containers, half of which were collected during sample deployment and the other half during sample retrieval. After retrieval, all macroalgae and water samples were immediately stored in the dark at 5°C for 12 h before being relocated to a −20°C freezer.

This experiment took place 3 mo before the salmon harvest, when salmon biomass was near its peak. This likely corresponded with high rates of feeding in order to finish the salmon at maximum harvestable size. In addition, field sampling took place during the summer when sea surface temperatures are highest and background NH_4^+ and NO_3^- concentrations are at their lowest (Gregory et al. 1993, Keizer et al. 1996). Hence, this experiment represents an extreme case scenario of low ambient nutrient concentrations and high fish production.

2.3. NO_3^- and NH_4^+ concentration analysis

Water samples were defrosted and filtered using a 0.45 µm sterile syringe filter. NO_3^- was measured by hot vanadium reduction of NO_3^- to nitric oxide using an Analytical Sciences NOx 5100 Thermalox detector (Braman & Hendrix 1989). For this, 2 ml of each water sample (in duplicate) were injected into the vanadium (III) solution and were run with bracketing NO_3^- standards of between 0 and 10 µM. Samples were also measured for NO_2^- concentration using the sulfanilamide and napthal-ethylenediamine colorimetric method (Pai et al. 1990). However, NO_2^- was absent, indicating the samples measured on the NOx detector were NO_3^- only. NH_4^+ concentrations were analysed using the phenol blue method and were measured on a Thermo Scientific Evolution 260 Bio UV-Visible Spectrophotometer (Solórzano 1969). For this, 2 ml per sample (in duplicate) were reacted with bracketing standards of ammonium chloride between 0 and 10 µM.

2.4. Source of macroalgae

Macroalgae can exhibit large spatial and temporal variations in isotope composition which could confound their ability as bioindicators (Raimonet et al. 2013, Lemesle et al. 2016). Consequently, the *C. crispus* and *P. palmata* used in this study were reared at a land-based hatchery (www.acadianseaplants.com) and were collected 24 h before deployment and stored in the dark at 5°C. Unlike wild-collected specimens, these macroalgae were grown from the same brood stock in identical environmental conditions, resulting in similar physical condition and isotopic composition prior to deployment. However, the 2 species were grown in separate tanks from one another and supplied with different sources of inorganic N. Hence, we expected isotope composition to differ between species.

2.5. Isotope analysis

Macroalgae samples were defrosted in filtered seawater and cleaned of any epibionts before being washed in deionised water and dried at 60°C for at least 48 h. This process was carried out on just the tips (outermost 10 mm) of fronds for *C. crispus*, whereas whole fronds were processed for *P. palmata*. The latter species exhibits uniform growth, meaning new tissues can form all across its fronds (Nunes et al. 2016). In contrast, *C. crispus* exhibits apical growth, meaning new tissues grow only at the tips (Chopin et al. 1990). Hence, the tips of *C. crispus* fronds should represent the newest tissues and be more representative of nutrient conditions in Liverpool Bay. Although somatic and reproductive tissues can have markedly different isotopic compositions (Fredriksen 2003), our samples did not contain any reproductive structures.

All dried samples were homogenised using an agate pestle and mortar and weighed to 3.25 ± 0.25 mg in tin capsules in triplicate. Analysis was performed with an elemental analyser coupled to a DeltaPlus XP – Conflo III continuous flow-isotope ratio mass spectrometer. This created estimates of total N, C:N ratio, $\delta^{13}C$ and $\delta^{15}N$ with an analytical precision of 0.02‰. $\delta^{15}N$ and $\delta^{13}C$ were calculated using the equations described by Peterson & Fry (1987).

2.6. Pigment extraction

Many of the *C. crispus* samples were noticeably paler in colour after the field incubation period. Hence, to investigate whether macroalgal pigment concentrations exhibited spatial trends, we measured the chlorophyll *a* (chl *a*) concentration in 1 *C. crispus* sample from each of the stations using the methanol extraction protocol outlined by Torres et al. (2014).

2.7. Spatial analysis

Mean values of $\delta^{15}N$, total N, chl *a* and C:N ratio were interpolated using the 'kernel interpolation with barriers' tool within ArcMap 10.5. This method uses the shortest distance between points without intersecting a barrier (i.e. the coastline), such that points on either side of a barrier have less influence on one another, allowing for contours to change abruptly along barrier edges (Gribov & Krivoruchko 2011). Bandwidth selection for the kernel interpolation was carried out using the visual inspection approach outlined by Wand & Jones (1995). This involved creating a series of exponential models from large (3000 m) to small (1750 m) bandwidths in increments of 250 m (Figs. S1–S7 in the Supplement at www.int-res.com/articles/suppl/q011p671_supp.pdf). The most efficient kernel function (e.g. polynomial, Gaussian and Epanechnikov) was then identified by selecting the function which generated the lowest root-mean-square error (RMSE) and mean prediction error (ME) closest to 0 (Table S1) as detailed by Lessio et al. (2014). Through this process, an exponential function with a 2500 m bandwidth was deemed the most appropriate. Lastly, to assess the accuracy of interpolated data, the coefficient of variation (CV) was calculated using the equation described by Costanzo et al. (2001).

A hotspot analysis was performed using the Getis-Ord Gi* statistic (Getis & Ord 1992). This tested whether high or low values clustered together more than if spatial patterns were generated by chance alone (Getis 2010). The resulting *z*-scores and p-values identified whether stations were 'hot spots' (high values surrounded high values) or 'cold spots' (low values surrounded low values) at the 95% significance level. A fixed distance band of 2500 m was used, and a false discovery rate correction was applied to account for multiple testing and spatial dependency.

3. RESULTS

3.1. Dissolved nutrient concentrations

NH_4^+ and NO_3^- concentrations were low across the study area (Fig. 2), with many samples containing concentrations below detection limit. NH_4^+ ranged from detection limit (these cases were assigned a value of 0) to 1.29 µM (±0.05) and NO_3^- ranged from 0 to 0.32 (±0.01). There were noticeable differences between the 2 sampling dates as the station closest to the pulp mill and sewage treatment facility had an NH_4^+ concentration of 0 µM on 18 August, which increased to 1.3 µM just 11 d later, the highest concentration of NH_4^+ recorded in this study. The other stations also displayed marginally higher NH_4^+ concentrations for this second sampling date. NO_3^- con-

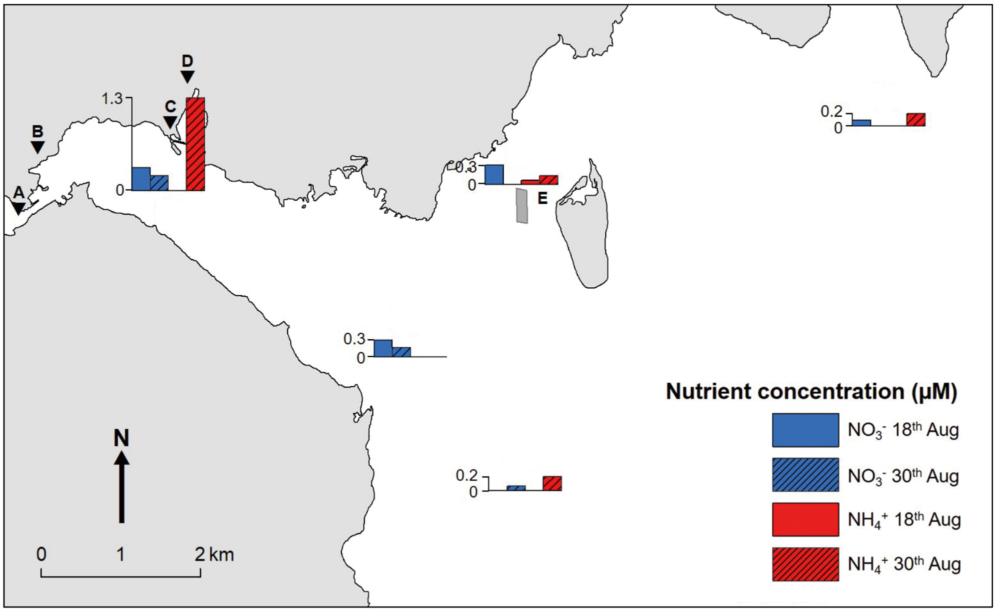

Fig. 2. NO_3^- and NH_4^+ concentrations of surface water samples collected at 2 different time periods. Chart position indicates sampling stations, and triangles indicate probable point sources (A: river mouth, B: sewage treatment facility, C: decommissioned pulp mill, D: biomass power plant). The grey box labelled E denotes the boundaries of the salmon farm

centrations were comparatively much lower than NH_4^+ and displayed no clear spatial or temporal trends.

3.2. End-member elemental and isotopic composition

Prior to their incubation in the field, the $\delta^{15}N$ and C:N ratio of *Chondrus crispus* were higher than those of *Palmaria palmata* (Table 1). In contrast, the total N content and $\delta^{13}C$ was lower in *C. crispus*. The $\delta^{15}N$ of the fish feed used at the salmon farm was slightly greater than that of *C. crispus.*

3.3. Elemental and isotopic composition of incubated macroalgae

Values of $\delta^{15}N$ in *C. crispus* (Fig. 3a) were lowest within the boundaries of the salmon farm (mean ± SE: 1.69 ± 0.06‰) and highest near the sewage treatment facility (4.43 ± 0.01‰). The $\delta^{15}N$ of all *C. crispus* samples deployed within the vicinity of the salmon farm had decreased by approximately 1‰ compared to initial values following the 10 d incubation period. The hot-spot analysis (Fig. 4a) indicated that $\delta^{15}N$ values were significantly low within the salmon farm ($p < 0.05$) and that these low values reached 2 km to the northeast and 2.4 km to the southwest of the farm. In contrast, the hot-spot analysis indicated that the inner bay area had significantly high $\delta^{15}N$ values ($p < 0.05$) which reached outwards over a distance of 3.5 km. Interpolations also suggested elevated $\delta^{15}N$ values 4 km northeast of the salmon farm, an area not known to have any obvious anthropogenic inputs of N. However, the hot-spot analysis indicated that these were not significant ($p > 0.05$). Unlike *C. crispus*, all *P. palmata* $\delta^{15}N$ values were negative and displayed a much narrower range of −2.32 to −1.14‰. These $\delta^{15}N$ values did not display any clear spatial patterns (Fig. 3b), and all samples were deemed statistically indistinguishable (Fig. 4b) by the hot-spot analysis ($p > 0.05$).

N content of the incubated macroalgae (Fig. 3c,d) was highest within the boundaries of the salmon farm for both species (*C. crispus* = 3.48 ± 0.01%; *P. palmata* = 4.05 ± 0.03%) and lowest at the northern outermost edge of the bay (*C. crispus* = 1.28 ± 0.02%; *P. palmata* = 2.17 ± 0.03%). The hot-spot analysis

Table 1. Isotope composition of hatchery-reared macroalgae *Chondrus crispus* and *Palmaria palmata* prior to incubation in the field, and of the fish feed used at the salmon farm. Error is ± 1 SE

Source	$\delta^{15}N$ (‰)	N (%)	$\delta^{13}C$	C:N ratio
C. crispus	3.36 ± 0.04	3.26 ± 0.01	−31.36 ± 0.01	10.21 ± 0.03
P. palmata	−1.72 ± 0.04	4.38 ± 0.02	−20.18 ± 0.1	7.8 ± 0.09
Fish feed	3.87 ± 0.09	7.1 ± 0.08	−20.38 ± 0.08	7.8 ± 0.06

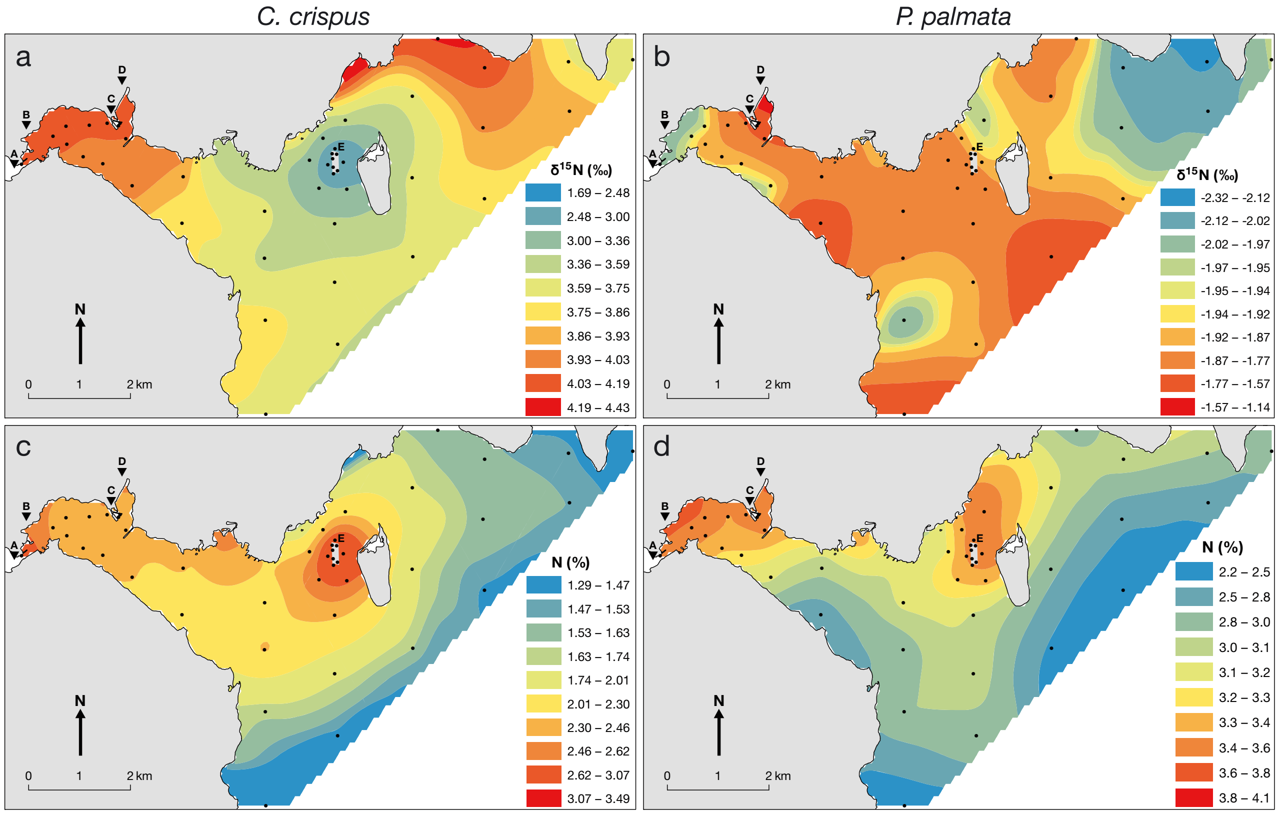

Fig. 3. (a,b) $\delta^{15}N$ and (c,d) total N content of the macroalgae *Chondrus crispus* (a,c) and *Palmaria palmata* (b,d). Points represent deployment stations. Triangles indicate probable point sources (labelled as in Fig. 2)

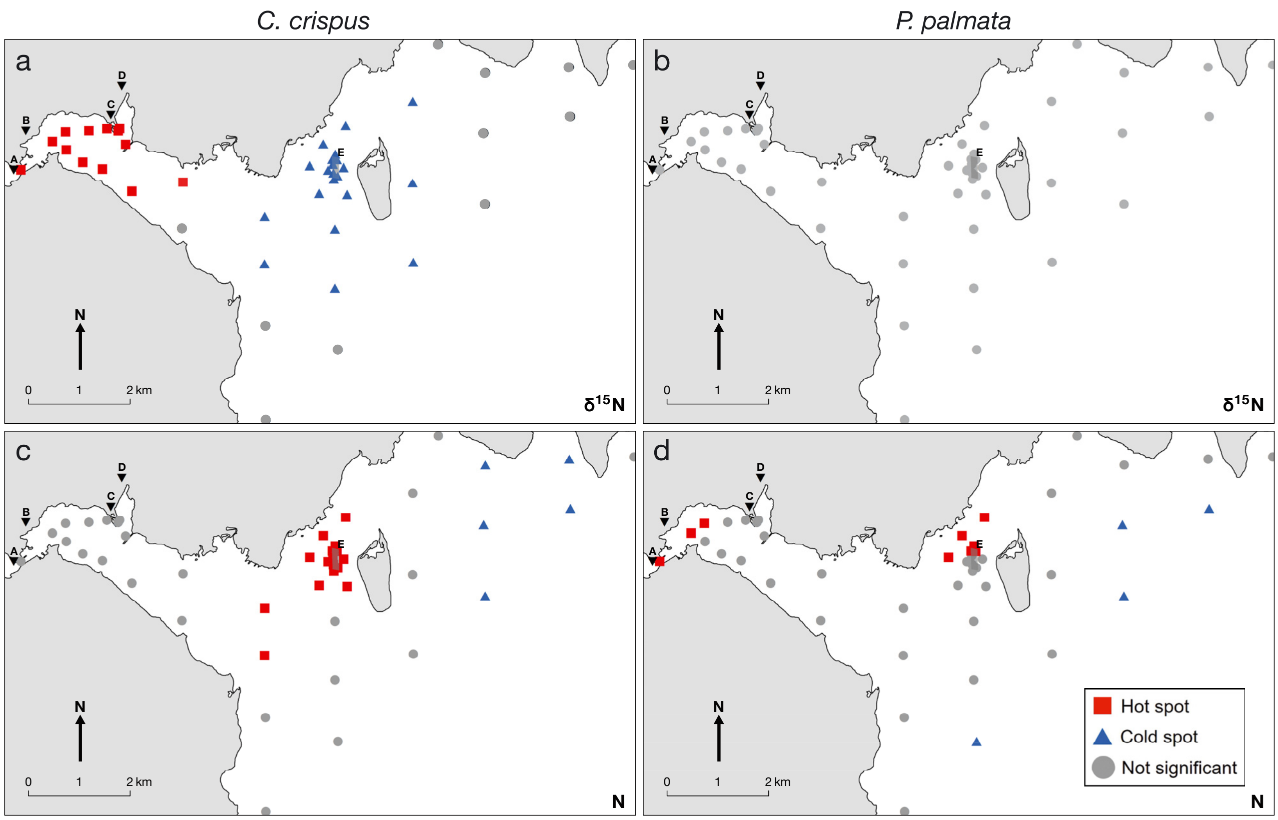

Fig. 4. Hot-spot analysis of (a,b) $\delta^{15}N$ and (c,d) N content of the macroalgae *Chondrus crispus* (a,c) and *Palmaria palmata* (b,d). Triangles indicate probable point sources (labelled as in Fig. 2)

(Fig. 4c,d) indicated that total N was significantly high ($p < 0.05$) within the boundaries of the salmon farm for both species. For *C. crispus*, these high N values reached 0.8 km to the north of the farm and 2.4 km to the southwest. In contrast, the high N values observed in *P. palmata* spread out from the farm in a predominantly northerly direction for 0.8 km. Only *P. palmata* exhibited significantly high values of total N ($p < 0.05$) within the inner bay area. Both species displayed significantly low total N values ($p < 0.05$) at the northern outermost edge of the bay.

Ratios of C:N (Fig. 5a) were lowest within the boundaries of the salmon farm for *C. crispus* (9.25 ± 0.01) and inner bay area nearest to the river (11.69 ± 0.02), and highest at the northern outermost edge of the bay (23.51 ± 0.06). *P. palmata* (Fig. 5b) was slightly different in that C:N ratios were lowest at the salmon farm (9.7 ± 0.04) but highest at the southern outermost edge of the bay (17.83 ± 0.22). The hot-spot analysis (Fig. 6a,b) confirmed that C:N ratios were significantly low ($p < 0.05$) within the boundaries of the salmon farm for both species. For *C. crispus*, these low C:N values reached 0.8 km to the north of the farm and 2.4 km to the southwest. In contrast, the low C:N values observed in *P. palmata* spread out from the farm in a northwest direction for just 0.3 km. The hot-spot analysis confirmed that the inner bay area also had significantly low C:N values ($p < 0.05$) which reached outwards over a distance of 2.8 km for *C. crispus* and 1.2 km for *P. palmata*. In contrast, C:N ratios were significantly high ($p < 0.05$) at the northern outermost edge of the bay for *C. crispus* and at the southern outermost edge of the bay for *P. palmata*.

Lastly, the chl *a* content of *C. crispus* (Fig. 5c) exhibited similar spatial trends to total N in that values were highest close to the salmon farm and inner bay area. However, all stations were deemed statistically indistinguishable in terms of chl *a* content (Fig. 6c). Overall, the interpolated data were between 80 and 90% accurate (Figs. S8 & S9).

4. DISCUSSION

The aim of this study was to investigate whether macroalgal bioindicators could map and identify between multiple effluent sources, and in doing so, provide a useful tool for helping management transition towards an ecosystem approach to aquaculture. Two species of macroalgae were incubated in a bay con-

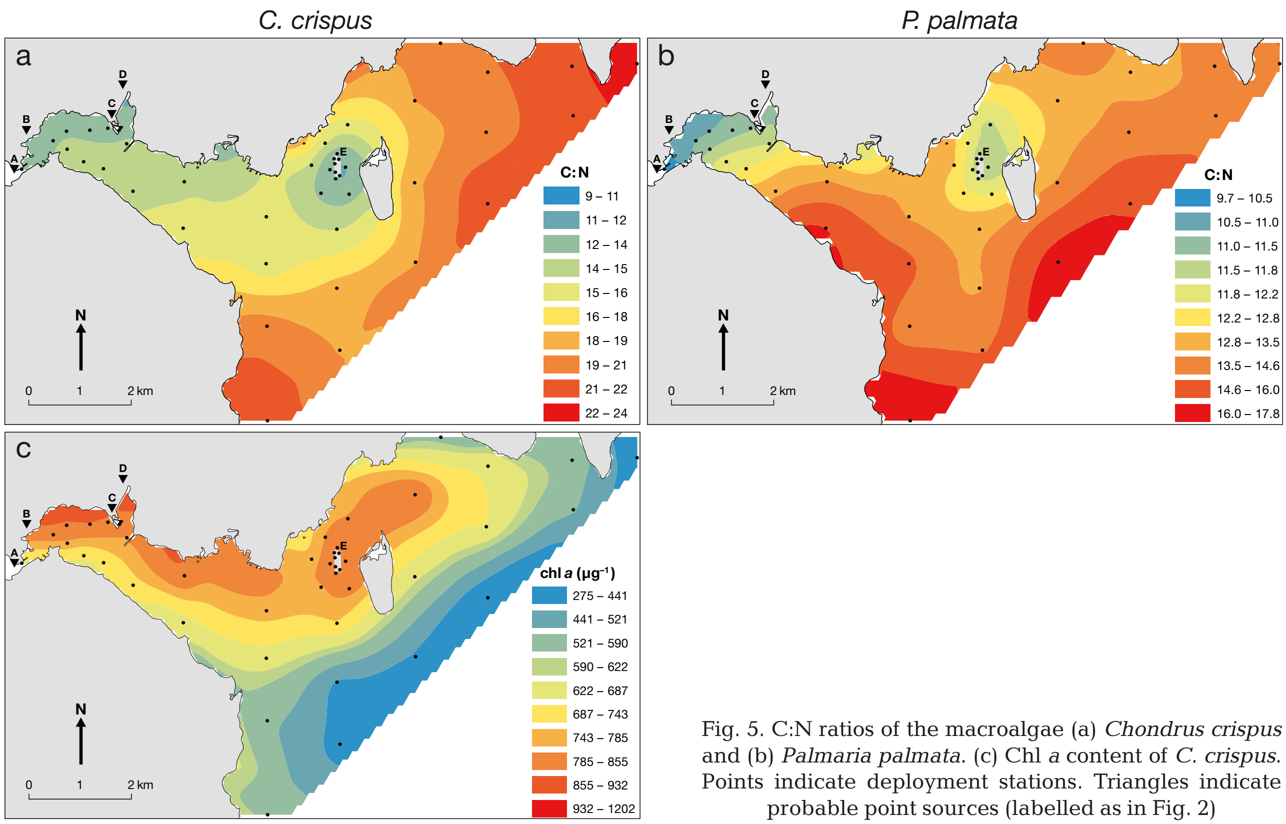

Fig. 5. C:N ratios of the macroalgae (a) *Chondrus crispus* and (b) *Palmaria palmata*. (c) Chl *a* content of *C. crispus*. Points indicate deployment stations. Triangles indicate probable point sources (labelled as in Fig. 2)

taining a salmon farm and sewage treatment facility, after which their elemental and isotopic composition was analysed.

4.1. Dissolved nutrient concentrations and the footprint of aquaculture wastes

The ambient nutrient concentrations of coastal waters, and the release of wastes from salmon farms and sewage treatment facilities, exhibit strong daily and seasonal fluctuations (Munksgaard & Young 1980, Karakassis et al. 2001). Such variation may explain why concentrations of NH_4^+ and NO_3^- differed greatly between the 2 sampling periods in our study. For example, NH_4^+ was undetectable in water samples taken near the sewage facility during the first sampling event, but this had increased to 1.3 ± 0.04 µM (mean ± SE) for the second sampling event. Similarly, the salmon farm had an initial NO_3^- concentration of 0.3 ± 0.01 µM but was undetectable during the second sampling event. Although DIN concentrations are lowest in Nova Scotia during the summer (Keizer et al. 1996), it is highly unlikely that NO_3^- and NH_4^+ concentrations were 0. Rather, these low and variable DIN concentrations were probably an artefact of experimental design, as the spatial and temporal coverage of water sampling was low.

4.2. Total N and C:N ratios of macroalgal bioindicators

Macroalgae take up N primarily in the form of NH_4^+ and NO_3^- (Hurd et al. 2014). When the supply of these nutrients exceeds what is needed for growth, macroalgae can accumulate N within their tissues for use during periods of low availability (Fong et al. 2004). Hence, the total N content of macroalgae tends to be higher after exposure to high DIN concentrations (Duarte 1992, Yokoyama & Ishihi 2010). In reverse, low C:N ratios tend to correspond with high DIN concentrations, as well as greater rates of photosynthesis and growth (Ahn et al. 1998, Umezawa 2002, Royer et al. 2013). As macroalgae deployed at the outer edges of the bay exhibited lower values of total N and chl *a*, and higher values of C:N, it suggests that they had grown under more N-limited conditions. In contrast, macroalgae near the salmon farm and sewage treatment facility exhibited higher values of total N and chl *a*, and lower C:N ratios, suggesting that the DIN emanating from these

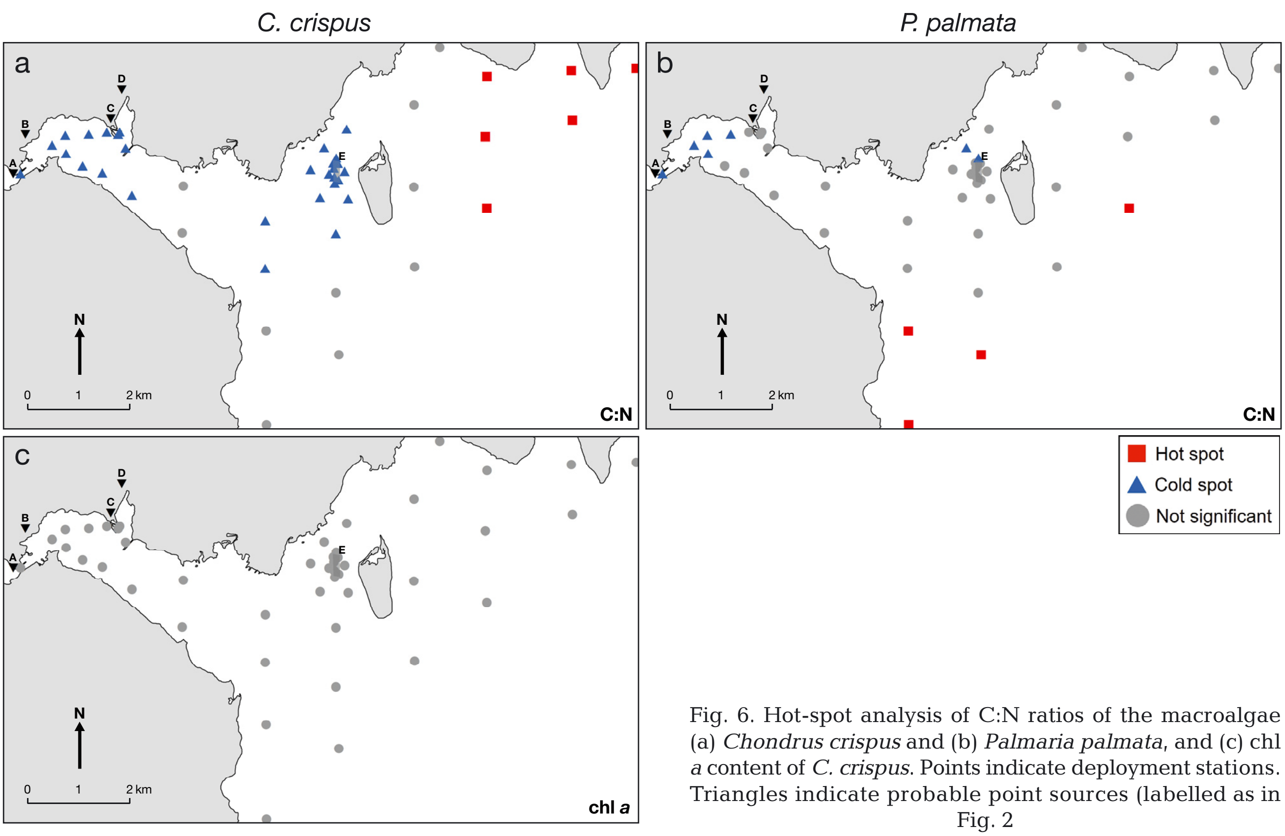

Fig. 6. Hot-spot analysis of C:N ratios of the macroalgae (a) *Chondrus crispus* and (b) *Palmaria palmata*, and (c) chl *a* content of *C. crispus*. Points indicate deployment stations. Triangles indicate probable point sources (labelled as in Fig. 2

effluent sources had promoted higher rates of growth, photosynthesis and N storage (Ahn et al. 1998, Umezawa 2002). Hence, our hypothesis that total N content would be higher near sources of anthropogenic N was supported.

One of the aims of this study was to use macroalgal bioindicators to provide greater insight into the dispersal of dissolved aquaculture wastes. The hot-spot analysis suggested that the tissues of *Chondrus crispus* exhibited significantly high total N values, and significantly low C:N ratios, up to 0.8 km northeast of the salmon farm and 2.4 km to the southwest. As both variables suggest that a transfer of N had occurred from the salmon farm to nearby macroalgae, these distances could indicate the extent of DIN emanating from the salmon farm. However, it must be noted that our results cannot be directly interpreted as a nutrient 'plume', but rather as a 'zone of influence' on organisms (García-Sanz et al. 2010, Carballeira et al. 2013).

Overall, our results suggest that the DIN released from the salmon farm and sewage treatment facility had increased the growth of macroalgal bioindicators. Determining to what degree these DIN inputs affect food webs and the wider ecosystem was beyond the scope of the present study. However, it is likely that other marine organisms (e.g. bacteria, phytoplankton, seagrass and other species of macroalgae) are also absorbing this N and experiencing faster rates of growth and reproduction (reviewed by Price et al. 2015). For example, Lapointe et al. (2005) attributed an increase in macroalgal blooms and invasions on coral reefs in Florida (USA) to increasing DIN inputs from sewage outfalls and septic tanks. Similarly, Robinson et al. (2005) reported that the development of salmon farming in the Bay of Fundy coincided with the growth of extensive algal mats along the surrounding shoreline. Hence, monitoring and reducing coastal N will undoubtedly continue to be an important aspect of coastal management as coastal human populations, wastewater treatment facilities and aquaculture, agricultural and industrial developments increase around the world (Yang et al. 2015, Clements & Chopin 2017).

4.3. Isotopic composition of macroalgal bioindicators

Several studies have reported elevated $\delta^{15}N$ levels in macroalgae exposed to effluents from aquaculture (Vizzini et al. 2005, Carballeira et al. 2013, Wang et al. 2014) and sewage treatment facilities (Gartner et al. 2002, Costanzo et al. 2005). Correspondingly, some of the highest $\delta^{15}N$ values observed in this study (4.32 ± 0.03‰) were from *C. crispus* samples deployed near the sewage treatment facility. However, the lowest $\delta^{15}N$ values (1.7 ± 0.06‰) were observed within the boundaries of the salmon farm. Hence, our hypothesis that macroalgae would display elevated $\delta^{15}N$ values close to the salmon farm and sewage facility was only partially supported.

There are several possible explanations why *C. crispus* near the salmon farm had such low $\delta^{15}N$ values. First, the effluents from the salmon farm may simply have been depleted in $\delta^{15}N$. However, salmon excretion is typically enriched by ~1.2‰ relative to their feed (Wang et al. 2014). As the salmon feed had a $\delta^{15}N$ of 3.87 ± 0.09‰, the effluent from the salmon farm was likely to be ~5.1‰. Alternatively, it may be that the salmon farm was producing very little in the way of dissolved wastes. However, this is unlikely, as the farm was at peak biomass at the time of sampling. Instead, effluents from the salmon farm could have negatively affected the physiology and growth of *C. crispus*, and consequently, its isotopic composition. However, total N, C:N ratio and chl *a* content suggested that *C. crispus* near the farm was growing faster in response to elevated DIN concentrations. Hence, the most probable explanation for these low $\delta^{15}N$ values is that *C. crispus* was fractionating near the salmon farm.

Many authors have claimed that fractionation within macroalgae is minimal to non-existent, meaning they should take up ^{14}N and ^{15}N in direct or close proportion to the supply (Gartner et al. 2002, Cohen & Fong 2005, Deutsch & Voss 2006, García-Sanz et al. 2010, Lemesle et al. 2015). However, several species of macroalgae have now been shown to fractionate when exposed to high DIN concentrations (Cohen & Fong 2005, Swart et al. 2014, Wang et al. 2014). For example, Swart et al. (2014) found that *Ulva lactuca* fractionated NH_4^+ at concentrations of ≥10 μM. Hence, the salmon farm could have increased DIN concentrations above the threshold required for fractionation, causing macroalgae to grow new biomass more isotopically negative than the source. This would explain why the $\delta^{15}N$ of *C. crispus* deployed within a 400 m radius of the salmon farm had decreased by an average of 1‰ compared to their initial nitrogen composition. This study therefore joins a growing number which suggest that the $\delta^{15}N$ of macroalgae is not simply a function of the source, but also of the rate of fractionation when levels of DIN are in excess (Chopin et al. 1995, Wang et al. 2014). However, these results should not be interpreted as a direct measure of farm-induced eutrophication since the DIN threshold required for *C. crispus* to fractionate remains unknown.

If elevated DIN concentrations caused *C. crispus* to fractionate within the salmon farm, why was the same response not observed near the sewage treatment facility? This difference may be due to differences in DIN composition between the 2 sources. While NO_3^- and NH_3 account for the majority of DIN discharged by sewage facilities, NH_4^+ is the principal form of N released by fish farms (Leung et al. 1999, Pehlivanoglu & Sedlak 2004). As macroalgae preferentially take up NH_4^+ because it is energetically 'cheaper' than NO_3^-, and because they fractionate NH_4^+ at a faster rate, this could have resulted in stronger fractionation rates near the salmon farm (Cohen & Fong 2005, Hurd et al. 2014). Additionally, DIN concentrations near the sewage treatment facility may have been below what was required to promote fractionation. In support of this, total N and C:N ratios in *C. crispus* suggested that DIN levels were higher at the salmon farm. However, the only way to truly understand whether *C. crispus* fractionated at the salmon farm would be to conduct a controlled lab-based study investigating the mechanisms (e.g. NO_3^- and NH_4^+ concentrations) underlying its fractionation dynamics.

4.4. Comparisons between species

This study used 2 different species of macroalgae to test and compare their suitability as potential bioindicators. To our knowledge, this is the first study to trial *Palmaria palmata* as a bioindicator, and only the second to use *C. crispus* (Lemesle et al. 2016). Prior to its deployment in the field, *P. palmata* was substantially lighter in $\delta^{15}N$ than *C. crispus.* As macroalgae depleted in ^{15}N typically respond faster to elevated $\delta^{15}N$ signals, this could have given *P. palmata* an advantage over *C. crispus* as a bioindicator (Cohen & Fong 2006, García-Seoane et al. 2018). However, the $\delta^{15}N$ values of *P. palmata* exhibited very little change after the 10 d incubation period and displayed no response to the effluents from the salmon farm and sewage facility. In addition, values of N and C:N ratios in *P. palmata* exhibited a much narrower range than in *C. crispus* and were less responsive to anthropogenic effluents. These results therefore suggest that $\delta^{15}N$ in *P. palmata* is not an effective bioindicator, and that N and C:N ratios in *P. palmata* are less effective bioindicators than they are in *C. crispus*.

The differences observed between these 2 macroalgae species might be because *P. palmata* has thicker and flatter fronds, and a different cell wall composition, and therefore different rates of diffusion and nutrient uptake (Chopin et al. 1999, Gartner et al. 2002, Dailer et al. 2010, Hurd et al. 2014). Alternatively, it may be due to differences in their growth characteristics and how tissue samples were prepared in the lab. In this study, only the tips of *C. crispus* fronds were processed and analysed, as this species is known to exhibit apical growth (Chopin et al. 1990). Hence, it was assumed that these tissues should be the newest, and therefore, provide a better reflection of recent nutrient concentrations and isotope composition in Liverpool Bay. In contrast, *P. palmata* is known to exhibit uniform growth, meaning fronds should have contained a combination of new and old tissues (Nunes et al. 2016). Consequently, whole fronds were processed and analysed, which may have diluted any signals.

4.5. Future research

Macroalgal bioindicators have become a popular tool for monitoring anthropogenic effluents in coastal areas. However, this approach is limited by the lack of a standardized methodology, with many studies employing different incubation times, depths, distances, pre-exposure conditioning procedures and tissue selection processes (reviewed by García-Seoane et al. 2018). Also, while bioindicator studies have been conducted at many different latitudes, they have been restricted to species that can be locally sourced or can survive being transplanted to the area of interest. Consequently, over 40 different macroalgae species have been used as bioindicators to date. This is problematic, as rates of nutrient uptake vary between species due to differences in morphology, tissue composition and growth rates (Gartner et al. 2002, Deutsch & Voss 2006). Hence, there is a need to test how results vary between species, and between tissues, in macroalgae transplanted to the same location. While this study and several others have tried to address this research gap, further research is required before they can be widely adopted as a biomonitoring tool.

4.6. Conclusions

This study investigated whether macroalgal bioindicators could be used to map and identify between multiple effluent sources. Maps of total N and C:N ratios suggested that both macroalgae species had absorbed N from the salmon farm and sewage treatment facility, but that the salmon farm had influenced the elemental composition of *C. crispus* over a greater distance than that of *P. palmata*. Differences were also observed between their isotopic composition, as *C. crispus* indicated that the fish farm and sewage treatment facility had distinctly different $\delta^{15}N$ signatures, whereas values of $\delta^{15}N$ in *P. palmata* remained largely unchanged after the incubation period. This evidence suggests that *P. palmata* is a poor bioindicator compared to *C. crispus*. There was also evidence that *C. crispus* was fractionating N in response to elevated DIN concentrations, which could complicate its use as a bioindicator. Overall, this study suggests that macroalgal bioindicators have the potential to monitor and identify between multiple effluent sources, which could provide a useful tool for helping the transition towards an ecosystem approach to the management of aquaculture.

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