COSEWIC
Assessment and Status Report

on the

American Eel
Anguilla rostrata

in Canada

THREATENED
2012

COSEWIC
Committee on the Status
of Endangered Wildlife
in Canada

COSEPAC
Comité sur la situation
des espèces en péril
au Canada

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COSEWIC
Assessment Summary

Assessment Summary – May 2012

Common name
American Eel

Scientific name
Anguilla rostrata

Status
Threatened

Reason for designation
This species is widespread in eastern Canada, but has experienced dramatic declines over a significant portion of its distribution (e.g., Lake Ontario and the upper St. Lawrence River). Although trends in abundance in other areas are highly variable, strong declines are apparent in several indices. Continuing habitat degradation, especially owing to dams and pollution, and existing fisheries in Canada and elsewhere may constrain recovery.

Occurrence
Ontario, Quebec, New Brunswick, Prince Edward Island, Nova Scotia, Newfoundland and Labrador, Atlantic Ocean

Status history
Designated Special Concern in April 2006. Status re-examined and designated Threatened in May 2012.

COSEWIC
Executive Summary

American Eel
Anguilla rostrata

Wildlife Species Description and Significance

The American Eel is an elongate, cylindrical fish, a member of the "freshwater eels", and the only North American representative of the genus. The American Eel has a protruding lower jaw and a small gill opening in front of the pectoral fins, which are located high up on the side of body. The dorsal fin is long, beginning at about mid-body and extending such that it is confluent with the caudal and anal fin. The American Eel lacks pelvic fins and grows to a maximum size of about 1 m. The American Eel plays an important role as a top aquatic predator, is the focus of many fisheries, is of great importance to Aboriginal peoples, and is considered as an excellent indicator of habitat integrity.

Distribution

The American Eel is a migratory species, widely distributed in freshwater habitats, estuaries, and coastal marine waters of the western North Atlantic Ocean coastline. The continental distribution of the American Eel ranges from northern South America to Greenland and Iceland. The historical Canadian distribution encompasses all accessible freshwater habitats, estuaries, and coastal marine waters connected to the Atlantic Ocean of Canada, up to the mid-Labrador coast. The distribution and abundance of American Eel has diminished over the past century in freshwater habitats impacted by human development.

Habitat

The American Eel uses a variety of marine and freshwater habitats over the course of its life history. Eels inhabit a variety of continental and oceanic habitats during their migrations to and from spawning areas in the Sargasso Sea. Habitat requirements during the overwintering period are poorly known, in both fresh and saltwater habitats. In fresh water, preferred habitat can be found in lakes and rivers including all waters extending from the high-water mark down to at least 10 m depth. In marine habitats, the continental phase includes predominantly shallow, protected waters. Growing eels are primarily benthic, using substrate (rock, sand, mud), and woody debris and submerged vegetation for protection and cover. American Eels commonly overwinter in mud bottoms in both bay and estuary habitats. Eelgrass and interstitial spaces are important to American Eel as cover, particularly during daylight hours.

Biology

American Eels spawn only once during their lives in the Sargasso Sea of the southern North Atlantic Ocean. Larvae develop a leaf-like shape and are termed "leptocephali". American Eel larvae disperse westward toward the continental shelf, where they metamorphose into small, transparent "glass eels", which have the serpentine shape of the adult form. As glass eels move into inshore waters, they develop pigmentation and grow into "elvers". Elvers become increasingly pigmented and are known as "yellow eels", the stage where sexual differentiation occurs. Density is thought to be an important factor influencing sex ratio, with high densities promoting the production of males. Females also dominate in many locations in Canada, especially in Lake Ontario and the upper St. Lawrence River, but sex ratios are more variable in rivers in the Maritime provinces and Newfoundland. During maturation, yellow eels go through morphological and physiological modifications to become "silver eels". In the upper St. Lawrence and Lake Ontario, the size at which silvering occurs is the largest in the species' range. Generation time for eels residing in fresh water is as high as 22 years. Generation time is much shorter in eels that reside permanently in salt water (roughly 9 years).

Population Sizes and Trends

Times series data used to estimate percent change in indices of abundance from the 1950s to the 2000s (three generations) were almost uniformly negative (from -7.1% to -96.2%) within the western portion of the species' range, while trends were mixed within the eastern portion of its range. Indices of abundance from fishery landings series indicated negative change. Abundance relative to the 1980s is very low for Lake Ontario and St. Lawrence River fish according to fisheries-independent data. Between 1996-1997 and 2010, estimates of the total number of maturing eels declined by 65% in the Great Lakes and upper St. Lawrence River area, despite the reduction in mortality from commercial fisheries (50% of fishing effort between 2002-2009). An index of year class strength indicated a substantial decline of juvenile eels migrating upstream in the Sud-Ouest River (lower St. Lawrence River) between 1999 and 2005. Trends in some areas (New Brunswick) were mixed while other areas (Newfoundland, southwest Nova Scotia) indicated some declines between the 1980 and the 2000s.

Threats and Limiting Factors

In fresh water, barriers erected in watercourses severely impede upstream migration of juvenile eels if no fish passage is provided. Impeded access to the Ottawa River, Lake Ontario, and Lake Champlain resulted in substantial cumulative loss in access by eels to formerly productive rearing habitat, e.g., at least 12,140 km² of eel freshwater habitat (10 m or less in depth) in the St. Lawrence River watershed. The turbines of hydroelectric dams also impose substantial mortalities (up to 40%) during passage through multiple dams as maturing fish migrate downstream toward the spawning grounds. Vulnerability to fisheries and bioaccumulation of contaminants are also important threats. An exotic swim bladder nematode parasite may be negatively affecting eels. The parasite has been found in Nova Scotia (Cape Breton Island), New Brunswick, and Lake Ontario. Climate change and shifting oceanographic conditions and supplementation of eels by stocking of wild-recruits (now suspended) may also pose risks.

Protection, Status, and Ranks

The American Eel was assessed as Special Concern by COSEWIC in April 2006. The status was re-examined by COSEWIC in May 2012 and designated Threatened. It currently has no status under the federal *Species at Risk Act*. In Ontario, the American Eel was listed as Endangered and became protected under the Ontario *Endangered Species Act, 2007*. In Québec, the American Eel is ranked as *vulnerable*. In Newfoundland and Labrador, it has been listed under the provincial listing (ESA) as *vulnerable*. The American Eel has been ranked as *secure* in New Brunswick and Nova Scotia, *apparently secure-secure* in Prince Edward Island, *apparently secure* in Canada and globally by NatureServe (as of 2006).

TECHNICAL SUMMARY

Anguilla rostrata

American Eel   Anguille d'Amérique

Range of occurrence: Lake Ontario and St. Lawrence River system in Ontario and Québec, and adjacent brackish waters; Gulf of St. Lawrence in Québec, Maritime regions of New Brunswick, Nova Scotia, Prince Edward Island, as well as Newfoundland and southern Labrador and adjacent marine waters.

Demographic Information

Generation time (average age of parents in the population) *A few areas appear to have American Eels that live their whole lives in saltwater (in addition to the "typical" catadromous life style of eels from other areas); many Canadian areas produce predominantly females.*	Freshwater: 22 yr Saltwater: 9 yr
Is there an observed continuing decline in number of mature individuals? *A 65% decline has been reported over the last 14 years in maturing eels from Lake Ontario and St. Lawrence River area and substantial declines have been observed in some areas of the Maritimes.*	Yes, but variable among areas
Estimated percent of continuing decline in total number of mature individuals within 5 years or 2 generations. *A 65% decline has been reported over the last 14 years in maturing eels from Lake Ontario and St. Lawrence River area and substantial declines have been observed in some areas of the Maritimes.*	Unknown overall
Inferred percent reduction in total number of mature individuals over the last 3 generations.	Unknown overall, but > 70% in some areas
Projected or suspected percent reduction in total number of mature individuals over the next 3 generations.	Unknown
Inferred percent reduction in total number of mature individuals over any 10 year period, over a time period including both the past and the future. *Unknown overall, but a 65% decline has been reported over the last 14 years in maturing eels from Lake Ontario and St. Lawrence River areas*.	Unknown overall, but > 70% in some areas
Are the causes of the decline clearly reversible and understood and ceased? *Causes of potential declines are understood in some areas, but not easily reversible.*	No, causes are only partially understood
Are there extreme fluctuations in number of mature individuals?	Probably not

Extent and Occupancy Information

Extent of occurrence	2,138,676 km²
Index of area of occupancy (IAO)	997,042 km²
Is the total population severely fragmented (*sensu* IUCN)?	No
Number of locations (total) *Unknown, but in freshwater probably at least several hundred*.	Unknown
Is there an inferred continuing decline in extent of occurrence?	No
Is there an inferred continuing decline in index of area of occupancy?	No overall; yes in Lake Ontario and St. Lawrence River
Is there an observed continuing decline in number of populations? *Eels in some tributaries of the Ottawa River and Lake Ontario drainage have been lost.*	No overall
Is there an observed continuing decline in number of locations?	Unknown
Is there an [observed, inferred, or projected] continuing decline in [area, extent and/or quality] of habitat?	Yes, mainly in freshwater
Are there extreme fluctuations in number of populations?	No

Are there extreme fluctuations in number of locations?	Unknown, but probably not
Are there extreme fluctuations in extent of occurrence?	No
Are there extreme fluctuations in index of area of occupancy?	No

Number of Mature Individuals (in each population)

Population:	**N Mature Individuals**
Estimate for	Data not available
Lake Ontario and upper St. Lawrence River area estimated as 153,044 (116,480 – 189,608; 95% confidence limit) fish in 2010 using pooled Peterson estimates. There will be some mortality of these fish during their oceanic migration to the spawning grounds.	Unknown, but likely over one million overall.
Total	Unknown

Quantitative Analysis

Probability of extinction in the wild is at least [20% within 20 years or 5 generations, or 10% within 100 years].	No quantitative analysis (necessary data not available)

Threats (actual or imminent, to populations or habitats)

Actual

- Dams (habitat fragmentation, loss of access to upstream habitat by impassible dams, and turbine mortalities for mature eel)
- Habitat degradation
- Fisheries (mainly commercial, on elvers, yellow and silver eels). Not in Ontario (closed).
- Chemical and biological contamination
- Introduced parasite: *Anguillicoloides crassus*

Potential:

- Climate change and effects on ocean circulation, productivity, and pH
- Hydro development (at least in Ontario). Proposed and actual new facilities have the potential to offset any new mitigation measures that are currently being discussed between hydro producers and OMNR.
- Effects of stocking programs (e.g., changes to sex ratios)

Rescue Effect (immigration from outside Canada)

Status of outside population(s) *U.S. populations also in decline*	
Is immigration known or possible? *Likely for freshwater areas given apparent panmixis in eel, mature American Eel all spawn in one area – the Sargasso Sea*	Yes
Would immigrants be adapted to survive in Canada? *Stocking of eels from eastern Canada into Ontario has resulted in some males being found in inland waters, where historically, almost all eels were female. The implications of this are not known, but could be serious*.	Probably
Is there sufficient habitat for immigrants in Canada?	Yes
Is rescue from outside populations likely? *Possible, but American Eels in United States' portion of the range are also in decline.*	Possible

Current Status

COSEWIC: Designated Special Concern in 2006. Status re-examined and designated Threatened in 2012. Ontario ESA: Endangered (2007) Newfoundland and Labrador (NL) ESA: Vulnerable (2011) SARA: No status

Status and Reasons for Designation

Status: **Threatened**	**Alpha-numeric code:** **A2b**
Reason for Designation: This species is widespread in eastern Canada, but has experienced dramatic declines over a significant portion of its distribution (e.g., Lake Ontario and the upper St. Lawrence River). Although trends in abundance in other areas are highly variable, strong declines are apparent in several indices. Continuing habitat degradation, especially owing to dams and pollution, and existing fisheries in Canada and elsewhere may constrain recovery.	
Criterion A: Meets Threatened A2b, because declines in abundance of multiple indices over much of the range are inferred to exceed 30% over the past three generations (in some areas declines exceed 90% since 1972, about two generations). Various threats, including dam construction and operations and habitat degradation, have impacted the species for more than three generations.	
Criterion B: Not applicable; exceeds all criteria.	
Criterion C: Not applicable; exceeds all criteria.	
Criterion D: Not applicable; exceeds all criteria.	
Criterion E: Not applicable; data to assess abundance over the entire range are unavailable.	

PREFACE

The American Eel was designated a species of Special Concern by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2006). The status was re-examined and designated Threatened in May 2012 but has not yet received protection under Schedule 1 of the federal S*pecies at Risk Act* (SARA). At the time of the last assessment, sufficient concern about the status of the species across its range was expressed such that it was decided to re-assess the species after five years rather than the usual ten. Since the last assessment, the American Eel has been listed as an *Endangered Species* under Ontario's *Endangered Species Act, 2007* (ESA) (OMNR, 2007). In December 2010, the Draft Recovery Strategy for the American Eel in Ontario was released for public review and posted on the Ontario Species at Risk website (MacGregor, *et al.,* 2010). In Newfoundland and Labrador (NL), the American Eel is listed as vulnerable under NL *Endangered Species Act, 2011* (ESA). The Management Plan for the American Eel in Newfoundland and Labrador was released in 2011 (Wildlife Division, 2011).

Since the COSEWIC Status Report in 2006, significant research and management plans have been initiated and conducted across the Canadian range of the American Eel. Bernatchez *et al.* (2011) concluded that the American Eel reproduces as a single population within the Sargasso Sea. In 2004, the ministers responsible for fisheries in Ontario, Québec, and nationally (DFO) announced a plan to stop the decline of the American Eel. Since then, the National Management Plan for American Eel in Canada has been drafted by the Canadian Eel Working Group (CEWG, 2009). One of the short-term goals of the plan is to reduce eel mortality from all anthropogenic sources by 50% relative to the 1997-2002 average. Long-term management goals are to rebuild overall abundance of the American Eel in Canada to its level in the mid-1980s. There have been extensive discussions and consultations between the governments of Ontario and Québec, DFO, Aboriginal groups, commercial fishers, hydropower companies, and other interested parties to develop coordinated actions to reduce mortalities and enhance spawning escapement. Negotiations with hydropower companies in Ontario (Ontario Power Generation, OPG) and Québec (Hydro-Québec, HQ) have led to development of action plans to try and reduce dam-related mortalities. Ontario Power Generation and HQ have been providing funds toward short-term actions of reducing fishing mortality, replenishing eels in the upper St. Lawrence and Lake Ontario system, as well as research on trapping and transferring of eel (Trap and Transport Program). In Ontario, commercial and recreational harvests were closed in 2004 and 2005, respectively. Ontario also increased the passage efficiency of eels during upstream migration through the Moses-Saunders Dam by making improvements to the existing ladder. In Québec, fishing mortalities have been reduced by approximately 50% by reducing the number of licences (Richelieu River fishery closure in 1998; buyout of Québec commercial licences from 2002-2009). In the Atlantic Provinces, minimum sizes for adults have increased and elver quotas have been reduced.

In 2006, the Great Lakes Fishery Commission (GLFC) formed a binational American Eel Task Group to set science priorities, coordinate funding strategies, and develop a recovery strategy for eels in the upper St. Lawrence River and Lake Ontario system. The Task Group involves representatives of the U.S. Fish and Wildlife Service (USFWS), DFO, the provinces of Ontario and Québec, and New York State (MacGregor *et al.*, 2008; 2009). Formal discussions between Canada and the United States are underway to develop an effective memorandum of understanding that will establish coordinated binational management and science across the existing North American Eel range (MacGregor *et al*., 2009). Since 2008, the Lake Ontario Management Unit (LOMU) has worked closely with Canadian federal agencies, provincial governments, various U.S. federal and state agencies, and non-government partners to develop and implement plans to protect and restore American Eel in Lake Ontario (OMNR, 2009). Given the Canadian distribution of American Eels over a huge area that encompasses many jurisdictions, aspects of the biology and status of eels are discussed, where possible, by National Freshwater Biogeographic Zone in the following report.

COSEWIC HISTORY

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) was created in 1977 as a result of a recommendation at the Federal-Provincial Wildlife Conference held in 1976. It arose from the need for a single, official, scientifically sound, national listing of wildlife species at risk. In 1978, COSEWIC designated its first species and produced its first list of Canadian species at risk. Species designated at meetings of the full committee are added to the list. On June 5, 2003, the *Species at Risk Act* (SARA) was proclaimed. SARA establishes COSEWIC as an advisory body ensuring that species will continue to be assessed under a rigorous and independent scientific process.

COSEWIC MANDATE

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assesses the national status of wild species, subspecies, varieties, or other designatable units that are considered to be at risk in Canada. Designations are made on native species for the following taxonomic groups: mammals, birds, reptiles, amphibians, fishes, arthropods, molluscs, vascular plants, mosses, and lichens.

COSEWIC MEMBERSHIP

COSEWIC comprises members from each provincial and territorial government wildlife agency, four federal entities (Canadian Wildlife Service, Parks Canada Agency, Department of Fisheries and Oceans, and the Federal Biodiversity Information Partnership, chaired by the Canadian Museum of Nature), three non-government science members and the co-chairs of the species specialist subcommittees and the Aboriginal Traditional Knowledge subcommittee. The Committee meets to consider status reports on candidate species.

DEFINITIONS
(2012)

Wildlife Species	A species, subspecies, variety, or geographically or genetically distinct population of animal, plant or other organism, other than a bacterium or virus, that is wild by nature and is either native to Canada or has extended its range into Canada without human intervention and has been present in Canada for at least 50 years.
Extinct (X)	A wildlife species that no longer exists.
Extirpated (XT)	A wildlife species no longer existing in the wild in Canada, but occurring elsewhere.
Endangered (E)	A wildlife species facing imminent extirpation or extinction.
Threatened (T)	A wildlife species likely to become endangered if limiting factors are not reversed.
Special Concern (SC)*	A wildlife species that may become a threatened or an endangered species because of a combination of biological characteristics and identified threats.
Not at Risk (NAR)**	A wildlife species that has been evaluated and found to be not at risk of extinction given the current circumstances.
Data Deficient (DD)***	A category that applies when the available information is insufficient (a) to resolve a species' eligibility for assessment or (b) to permit an assessment of the species' risk of extinction.

* Formerly described as "Vulnerable" from 1990 to 1999, or "Rare" prior to 1990.

** Formerly described as "Not In Any Category", or "No Designation Required."

*** Formerly described as "Indeterminate" from 1994 to 1999 or "ISIBD" (insufficient scientific information on which to base a designation) prior to 1994. Definition of the (DD) category revised in 2006.

Environment Canada / Environnement Canada
Canadian Wildlife Service / Service canadien de la faune
Canada

The Canadian Wildlife Service, Environment Canada, provides full administrative and financial support to the COSEWIC Secretariat.

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WILDLIFE SPECIES INFORMATION

Name and Classification

The American Eel, *Anguilla rostrata* (LeSueur, 1817), belongs to the Class Actinopterygii, Order Anguilliformes, Family Anguillidae, and Genus *Anguilla* (Figure 1). Alternative (but unoffical) common names include Atlantic eel, common eel, freshwater eel, silver eel, yellow-bellied eel, green eel, black eel, bronze eel, elver, whip, and easgann (Scott and Crossman, 1973; Scott and Scott, 1988). In French, the species is called *Anguille d'Amérique, anguille argentée, anguille jaune, anguillette*, and *civelle.* The Mi'kmaq use *ka't or g'at* (Prosper, 2001; Prosper and Paulette, 2002; GMRC 2008a); the Algonquins *pimzi* or *pimizi* (Allen, 2008); the Ojibwe *bimizi* (Barara, 1878 in MacGregor *et al.,* 2010), the Seneca *goda:noh* (Bardeau, 2002 in MacGregor *et al.,* 2010), and the Cree *Kinebikoinkosew* (MacGregor *et al.*, 2008).

Figure 1. The American Eel, *Anguilla rostrata* (from United States Fish and Wildlife Service).

Members of the genus *Anguilla* are termed freshwater eels, although some species (including the American Eel) are able to complete their life cycle in saltwater (Tsukamoto *et al.*, 1998; Arai *et al.*, 2004; Lamson *et al.*, 2006). The American Eel is the only North American species of the genus. Its closest relative is the European Eel (*Angullia anguilla*), which occupies a similar latitudinal range in western Europe, and the two species come into contact in Iceland (see Avise *et al.* 1990).

Morphological Description

The American Eel has an elongated and serpentine body (Figure 1). Its single continuous dorsal fin extends posteriorly from a point about one third of the body length behind the head, around the tail to the vent. The pectoral fin is supported by 7 to 9 fin rays (up to 11 in young specimens). The mouth is terminal and the lower jaw slightly longer than the upper. The teeth are small and arranged in several rows on the jaws and palate. A tongue is present, and the lips are thick. The lateral line and the palatopterygoid arch are well developed. The gill openings are not confluent, and the frontal bones are paired (Tesch, 1977).

Tesch (1977) described three morphological features that persist through all stages from larvae to maturing eels: the total number of vertebrae (mean 107.2), the number of myomeres (mean 108.2; evaluated at 106.84 by Kleckner and McCleave, 1985), and the distance between the origin of the dorsal fin to the anus (mean 9.1% of total length). Other morphological characteristics can only be used comparatively if the individuals are at the same stage of development (e.g. leptocephalus, glass eel, elver, yellow eel, silver eel).

Head shape and size dimorphism has been observed for some time by the Mi'kmaq. In the Bras d'Or community, Aboriginal people use the head shape to sex the eels; the ones with a broad head are considered female, and the ones with a narrow head are considered male (Denny *et al.*, 2011). For the European Eel, *Anguilla anguilla,* shape can be related to food source. Proman and Reynolds (2000) reported that the ratio of head width: total length (HW:TL) increased significantly ($P<0.05$) from glass to yellow eel. They also reported that in three Dutch lakes, 78% of "broad-headed" eels consumed large and/or hard-bodied organisms, while 83% of "narrow-headed" eels consumed exclusively small and/or soft-bodied prey (see also Lammens and Visser 1989).

Population Spatial Structure and Variability

Anguillid eels of the North Atlantic Ocean have been divided into two species based on morphological (Ege, 1939; Tesch, 1977) and molecular genetic (Avise *et al.*, 1986; Aoyama *et al.*, 2001; Wirth and Bernatchez, 2003) characters. The American Eel inhabits continental waters on the western side of the Atlantic Ocean, while the European Eel is found in continental waters on the eastern side of the Atlantic. Both species spawn in the Sargasso Sea in the southern North Atlantic Ocean (Schmidt, 1922, Figure 2).

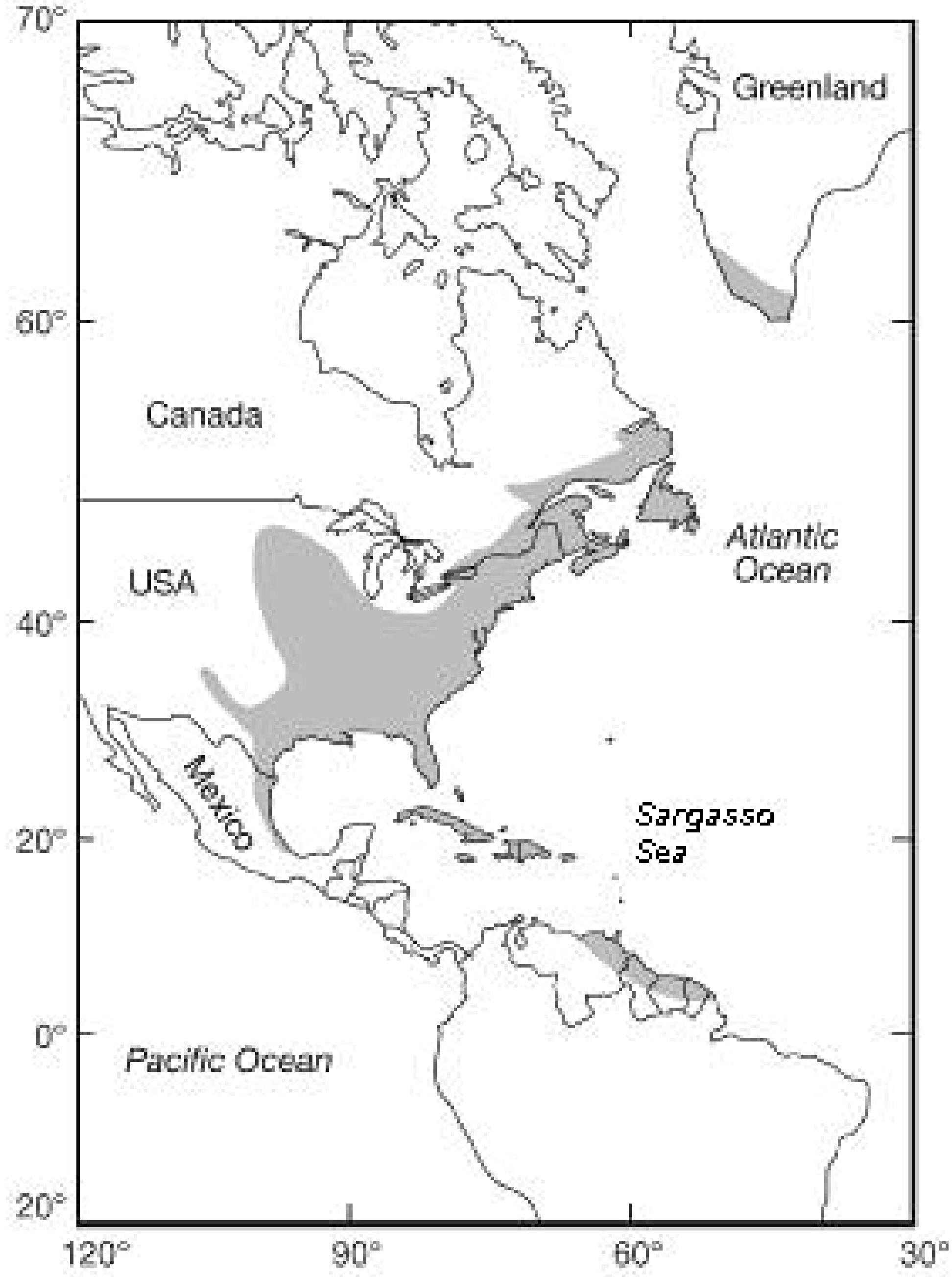

Figure 2. Global range of the American Eel (*Anguilla rostrata*). From http://www.mnr.gov.on.ca/ (Fisheries and Oceans Canada, American Eel, Underwater World (Reproduced by MRN with the permission of Her Majesty the Queen in Right of Canada, 2006).

Although American and European Eel are two distinct species, they are capable of hybridizing with each other (Albert *et al.*, 2006, van Ginneken and Maes, 2005). Hybrids have been observed almost exclusively in Iceland (Avise *et al.*, 1990), but hybridization occurs in the Sargasso Sea, which suggests that hybrids do not have the same dispersal behaviour as American or European Eels (Als *et al.*, 2011). Because these species are close relatives, some information from European Eel studies has been applied to the American Eel for the purpose of this status report.

Molecular genetic data have provided evidence both supporting and rejecting the hypothesis that each eel species comprises a single, randomly mating (panmictic[1]) population (reviewed in Maes and Volckaert, 2007), but compared to the European Eel, less effort had been devoted to the genetics of American Eel until recently. Since 2007, ongoing effort has been raised to test the hypothesis of panmixia in American Eel by molecular and experimental approaches across the species' range. Recent population genetic analysis based on the genotyping of over 2,500 individuals from 34 locations and nine year classes at 18 microsatellite loci revealed very low levels of F_{ST}, the parameter that estimates the proportion of the total variation in allele frequencies that represents differences among groupings of fish (Bernatchez *et al.*, 2011; Table 1). While the levels of F_{ST} are low, at least one value, that reported for glass eels (F_{ST} = 0.00014), was well below the minimum level (F_{ST} = 0.00034) that could be tested for statistical significance with reasonable power (e.g., β = 0.9, Bernatchez *et al.* 2011; cf. Als *et al*. 2011). The value of F_{ST} for yellow eel samples (F_{ST} = 0.0036) was more than twice as high as reported for glass eels, but was not statistically significant (P = 0.59, Table 1). In addition, tests for differentiation were conducted by collection site (N = 17) and not at any other scale of geographic organization (e.g., NFBZ or larger groupings). Therefore, tests against the hypothesis of panmixia may have lacked sufficient statistical power (glass eels), did not test regional groupings, or have generally not assessed differences amongst groups of maturing eels (cf. Palm *et al.* 2009). The data that do exist, however, support the idea of panmixia (glass and yellow eels analyzed by site). On balance, and in combination with results from Avise *et al.* (1986) with mitochondrial markers and Wirth and Bernatchez (2002), the most recent results for maturing European Eels, which have a very similar life history (see Palm *et al.* 2009), current data are consistent with the idea that American Eel comprise a single panmictic population with respect to neutral markers. It is possible, however, that one or more spawning populations of American Eels exist within the Sargasso Sea, but that large effective population sizes coupled with even low levels of gene flow between such populations would make detection of divergence at neutral loci difficult, especially when the samples consist of juvenile eel and not spawning adults. Larger samples of glass eels, more regional-based analyses, and, ideally, samples of adults on the spawning grounds would provide the most powerful tests of the panmixia hypothesis. The evidence for a lack of genetic structure of American Eels based on neutral genetic markers does not necessarily mean, however, that there are no genetic differences between eels from different geographic areas (see below).

[1] A panmictic breeding system is one in which all members of a species mate randomly as a single breeding population. In panmictic species, analyses of genetic structure typically indicate no geographical heterogeneity at loci immune to the influence of natural selection ("neutral loci").

Table 1. Genetic differentiation based on F_{st} estimated at both the spatial and temporal level with 18 microsatellite loci genotyped in glass eels and yellow eels from multiple locations in northeastern North America (Bernatchez *et al.* 2011).

Dataset	Sum of squares	Variance components	F_{ST}	P-value
Among life stages (N=2) Glass eels and Yellow eels (N=2575)	4.598	-0.00131	-0.00019	0.99609
Among sites (n=17) Glass eels (N=872)	109.477	0.00104	0.00014	0.91887
Among sites (N=15) Yellow eels (N=1270)	98.193	0.00239	0.00036	0.59042
Among cohorts (N=10) Yellow eels (N=1121) (hatching years 1995-2004)	71.169	0.00389	0.00055	0.17009
Among cohorts (N=12) Yellow eels (N=1121) (hatching years 1995-2004) Glass eels (N=100) (hatching years 2005-2006)	86.962	0.00421	0.00059	0.15249

Bernatchez *et al.* (2011) demonstrated quantitative (functional) significant genetic differences among glass eels collected from two different areas. Genetically based phenotypic differences in growth and size distribution of females were observed in controlled and similar conditions on glass eels and/or elvers collected in the Mira River (Atlantic coast of Nova Scotia) and the Blanche River (tributary of the St. Lawrence South Shore estuary) (Côté *et al.*, 2009).

Indeed, although eels show no evidence of population structure at neutral loci, groups of individuals are not necessarily the same genetically and, therefore, may not have the same propensity for dispersal according to their genotype and may have differential survival to any given encountered environmental conditions. Young eels of a panmictic species that settle in a given system may genetically differ from those of other areas in functional aspects of the genome (Bernatchez *et al.*, 2011).

Designatable Units (DU)

As noted above, the most recent and comprehensive molecular genetic study conducted on American Eel found no neutral genetic differences throughout the species' range (Bernatchez *et al.*, 2011; Table 1), and the American Eel has a migratory life cycle and no obvious disjunctions are seen in its distribution in Canada (Figure 2). Notwithstanding the occupancy of American Eels in several NFBZ, differences in life history (e.g., size at maturity, sex ratio), and the strong potential for genetic differences relevant to the persistence of eels in distinct environments within and among NFBZ (Côté *et al*., 2009; Bernatchez *et al*. 2011; Gagnaire *et al*. 2012), the life cycle of American Eels, where fish from several NFBZs probably interbreed in one or more areas in the Sargasso Sea, coupled with the absence of evidence of neutral genetic population structure suggest that the species should currently be assessed as a single DU.

Special Significance

The American Eel is a major predatory fish in freshwaters and in marine systems and is probably an important prey species. Consequently, the American Eel likely plays an important ecological role in a variety of aquatic communities (e.g., Smith and Saunders, 1955; O'Connor and Power 1973). The American Eel is of great historical and contemporary significance to Aboriginal peoples throughout its range (Prosper; 2001; Social Research for Sustainable Fisheries (SRSF), 2002; Casselman, 2003; Prosper and Paulette, 2003a; Paulette and Prosper, 2004b; Bourget, 1984, cited in Robitaille *et al.*, 2003; CEPI, 2004; Davis *et al.*, 2004; Prosper, 2004; CEPI, 2006; COSEWIC 2006; Lickers 2008; Regional Aboriginal Species of Concern Working Group 2008; GMRC, 2008a,b). Archaeological evidence indicates the existence of eel fishing for over 4,000 years (MacGregor *et al.*, 2008; 2009). In 1999, the American Eel was the basis of a Supreme Court of Canada decision (Regina *v.* Marshall) that upheld treaties signed by Mi'kmaq and Maliseet in 1760 and 1761. This decision confirmed a communal right to hunt, fish and gather in pursuit of a "moderate livelihood" (Prosper, 2003; Prosper and Paulette, 2004a,b; Davis *et al.*, 2004; Indian and Northern Affairs Canada, 2008).

DISTRIBUTION

Global Range

The American Eel is a migratory species, widely distributed in fresh water (streams and lakes), estuaries, and coastal marine waters of the western North Atlantic Ocean coastline (Figure 2) from Venezuela to Greenland and Iceland (Scott and Crossman, 1973; Tesch, 1977; Helfman *et al.,* 1987). The American Eel occupies the most extensive range, e.g. over 10,000 km of continental coastline between latitudes 7°N and 55°N) of any fish in the Americas (Helfman *et al.*, 1987; Edeline, 2007). The American Eel spawns in unknown habitats within an area encompassing several hundred thousand square kilometres in the Sargasso Sea (Figure 2; Schmidt, 1922), east of the Bahamas and southwest of Bermuda (22-27°N; 60-78°W; McCleave *et al.,* 1987).

Canadian Range

The historical Canadian range encompasses all accessible fresh water, estuaries, and coastal marine waters connected to the Atlantic Ocean of Canada, up to the mid-Labrador coast (Figure 3). Continental shelves are used by juvenile eels arriving from the spawning grounds, and by silver eels returning to the spawning grounds, the Sargasso Sea (Schmidt, 1922).

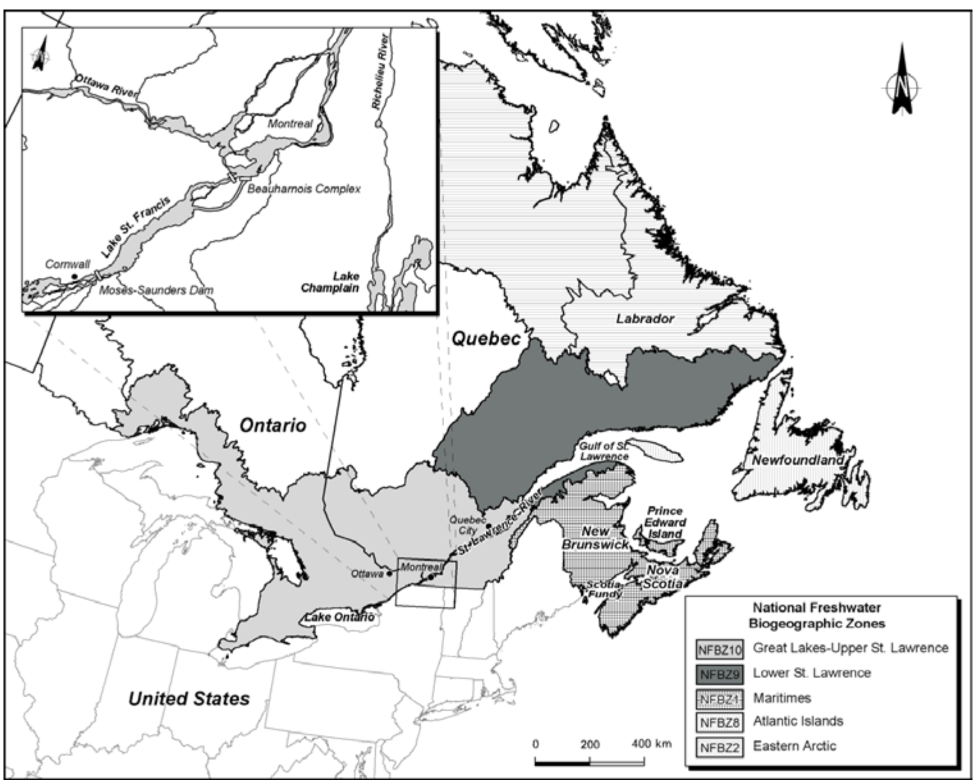

Figure 3. Canadian geographic range of the American Eel, *Anguilla rostrata*.

Niagara Falls was the natural upstream limit of the American Eel's distribution in the Great Lakes and occurrences above Niagara Falls (Lakes Erie, Huron and Superior) are probably the result of stocking and/or of recent dispersal through the Erie and Welland canals (Scott and Crossman, 1973; COSEWIC, 2006). As further archaeological information and traditional knowledge from Aboriginal people are collated, however, the possibility that eels found access somewhere over the Niagara Escarpment to Lake Erie and were historically native to the upper Great Lakes remains (MacGregor *et al.*, 2008; 2010). Access by possible routes identified from the Ottawa River to Lake Nipissing and then via the French River to Lake Huron and/or from the Trent Severn Waterway to Lake Huron rather than the Welland Canal should also be investigated given the propensity of eels to use damp substrates to surmount obstacles.

The range of eels has changed within the upper St. Lawrence River and Lake Ontario system. Indeed, the distribution and abundance of American Eel in freshwater habitats have diminished over the past century in areas affected by human development (e.g., NFBZ 10, MacGregor *et al.*, 2009).

On the mid-Labrador coast, eels regularly occur up to Hamilton Inlet-Lake Melville (53°15' N; 60°10' W; Scott and Crossman, 1973). Based on electrofishing surveys, however, the English River (about 120 km north) is now considered the northern limit of the American Eel in Canada (Kaipokok Bay: 54°58'N; 59°44'57''W), near Postville, Labrador. The extent of occurrence was estimated to be 2,138,676 km². The biological area of occupancy (AO) for a species should refer to the occupied habitat for the species (Table 2). Because the American Eel uses continental shelves during migration, a buffer of 370 km (200 nautical miles) from shore was included when calculating the index of area of occupancy (IAO). In Lake Ontario, only the area between the shoreline and 10 m of depth was also included in calculating the IAO, which was calculated to be 997,042 km² (2 km x 2 km grid).

Table 2. Extent of occurrence and area of occupancy (km²) of American Eels from each National Freshwater Biogeographic Zone (NFBZ) (from COSEWIC, 2006).

National Freshwater Biogeographic Zones (NFBZ)	Extent of occurrence [A]	Biological Area of occupancy (km²)
NFBZ10 (Great Lakes – Western (Upper) St. Lawrence)	391,515	97,400 (5.9%)
NFBZ9 (Eastern (Lower) St. Lawrence)	546,122	161,400 (9.8%)
NFBZ1 (Maritimes (New Brunswick, Nova Scotia, Prince Edward Island, and the central and southern parts of Québec's Gaspé Peninsula))	292,923	635,200 (38.4%)
NFBZ8 (Atlantic Islands (Newfoundland))	177,586	627,500 (38.0%)
NFBZ2 (Eastern Arctic (Labrador))	75,472	130,700 (7.9%)
Total	2,065,932	1,652,200 (100%)

A: The Sum of EO by NFBZ does not add up to the Canadian total because of the method used to calculate EO. A minimum convex polygon is constructed around the points for each NFBZ or the entire Canadian range. Because of the geography of Eastern Canada, more area is encompassed for the entire Canadian range than for the sum of NFBZ-specific polygons (N.E. Mandrak, DFO, pers. comm. 2011).

HABITAT

Habitat Use and Requirements

The American Eel uses a very broad diversity of habitats (Helfman *et al.,* 1987). During their oceanic migrations, eels occupy salt water and in their continental phase (growth in continental waters), they use all salinity zones. Catadromy is no longer seen as obligatory for eels, but rather is a facultative life history option (Tsukamoto *et al.,* 1998; Jessop *et al.,* 2002; Morrison *et al.,* 2003; Arai *et al.,* 2004; Lamson *et al.*, 2006; Thibault *et al.*, 2007a).

In freshwater habitats, preferred habitat can be found in both lentic and lotic waters including all waters extending from the high-water mark down to at least 10 m depth for all reaches currently or formerly used by the American Eel. In the Ottawa River, eels are found in habitats up to 15 m in depth (Smith, 2010). This includes all rivers, streams, and rivulets[2], both permanent and ephemeral (MacGregor *et al.*, 2010).

Given the high abundance of eels often observed in tributaries, these waters seem to comprise a very important component of eel habitat (Machut *et al.,* 2007). Habitat in tributaries is often of high quality and less disturbed than other areas (Machut *et al.,* 2007).

Growing eels are primarily benthic, and use substrate (rock, sand, mud), and bottom debris such as woody debris and submerged vegetation for protection and cover (Scott and Crossman, 1973; Tesch, 1977). Eelgrass (*Zostera* spp.) and interstitial spaces comprised of rock piles, logs, and other complex structures are important to American Eel as cover, particularly during daylight hours.

On the Québec north shore of the St. Lawrence estuary and Gulf (NFBZ9), Agence Mamu Innue Kaikusseht (AMIK) has an ongoing project in eelgrass habitat that demonstrates the importance of eelgrass to eels. Ichthyological surveys conducted in 2009 and 2010 resulted in capture of many eels (Essipit, Uashat Mak Mani-Utenam, Ekuanitshit, Unamen Shipu) (AMIK, 2010).

Beginning in 2007, ichthyological surveys in eelgrass habitat conducted in June and September in Forillon National Park (NFBZ1) have frequently captured eels (Daniel Sigouin, Parks Canada, pers. comm. May 2011). In Ontario, different sources (ATK, local community knowledge, archaeological information, historical records, scientific papers) all document the behaviour of large and small eels exiting water and moving considerable distances along damp substrates such as moss, grass, rocks and cement, which suggests that riparian areas are important habitats for eels (Machut *et al*. 2007; MacGregor *et al.*, 2010; Richardson *et al.* 2010).

[2] Rivulet = very small stream

In freshwater streams, eels generally do not demonstrate consistent preferences for habitat type, cover, substrate, or water temperature (Hawkins, 1995; Smogor *et al.,* 1995). In Prince Edward Island, eels are abundant in freshwater ponds formed by dams but are rare in most freshwater streams (Cairns *et al.*, 2007). Overwintering site (burrows) requirements and usage remain incompletely understood, in both fresh and saltwater habitats (Tesch, 1977; Feunteun *et al.,* 2003; Jessop *et al.*, 2009).

Recently, however, Tomie (2011) found that American Eels commonly winter in mud bottoms in both bay and estuary habitats in the southern Gulf of St. Lawrence. Based on records of winter spear fisheries, wintering eels appear to aggregate in certain areas within such habitats, especially zones of groundwater seepage. Mud and cobble bottoms were used by eels held overwinter in ambient temperature conditions (Tomie 2011). Eels also conceal themselves in the bottom during daytime in non-winter periods. Tomie (2011) estimated that American eels in eastern Canada spend about 74% of their entire yellow phase concealed in the substrate. The continental phase (growth phase) in marine habitats would be mainly restricted to shallow protected waters. Cairns *et al.* (2012) produced a distribution map and classification of aquatic habitat on the east coast of Canada. On the basis of commercial, recreational, and research fishing records, the “sheltered zone”[3] can be considered an approximation of the brackish and saltwater habitat of yellow eels. The sheltered zone comprises 9,626 km^2, which is 1.1% of all marine habitat from the coastline to the 500 m isobath. The proportion of sheltered habitat per region is highest in the St. Lawrence Estuary (419 km^2, 3.5%), followed by the Gulf of St. Lawrence (3,088 km^2, 1.4%), and the Atlantic Ocean and Bay of Fundy (6,120 km^2, 0.9%, Cairns *et al.* 2012).

Eel densities typically diminish with distance from the sea in medium and large rivers (Smith and Saunders, 1955; Gray and Andrews, 1971; Smogor *et al.,* 1995; Ibbotson *et al.*, 2002; Imbert *et al.*, 2008). In Europe, it has been reported that dispersal of eels into freshwater can be heavily influenced by density-dependent effects (Feunteun *et al.*, 2003), i.e., the higher the density, the stronger the motivation to continue to move upstream. The European Eel also appear to disperse randomly, equivalent to random dispersion of particles (Ibbotson *et al.*, 2002; Edeline *et al.,* 2007; Lambert and Rochard, 2007). Dispersal is influenced by many factors (density-dependent effects of eels, prey and predator density and distribution, physical factors such as obstacles), including anthropogenic factors.

[3] Sheltered zones were defined with the aid of a 1.5 km diameter circle. The circle was moved toward an inlet until it touched the coast at two points. A line drawn between these points was taken as the outer boundary of the sheltered zone. In the case of estuaries, the sheltered zone included waters to the approximate limit of salt penetration, as drawn on the base map (Cairns *et al.,* 2012).

For instance, White and Knights (1997) reported that barriers to upstream migration had a greater effect on European Eel density in some areas than distance from the ocean. The ability of eels to overcome obstacles is size-dependent. Small eels (less than 10 cm long) are able to creep up damp vertical barriers (Legault, 1988), but larger eels are generally unable to bypass large waterfalls and dams (McCleave, 1980; Barbin and Krueger, 1994). Hence, larger eels attempting to move upstream require unobstructed passage or eel ladders (Moriarty, 1987). Connectivity among important inland habitats is crucial to ensure eels are able to disperse effectively and take advantage of the best growth conditions in various rearing habitats. Additionally, safe and adequate passage to and from the oceanic spawning grounds is required to complete their life cycle (MacGregor *et al.*, 2010).

Survival is affected by environmental conditions in any habitat (oceanic, estuarine, fresh water) occupied during any life cycle phase, and by anthropogenic factors. Mortalities of maturing eels in their seaward migration has been associated with passage through hydroelectric turbines (Desrochers, 1995; Normandeau Associates and Skalski, 2000), fisheries (Castonguay *et al.,* 1994a; Caron *et al.,* 2003; Verreault and Dumont, 2003), and obstructions that produce free falls of more than 13 m (Larinier and Travade, 1999).

Habitat requirements for spawning in the Sargasso Sea (Schmidt, 1922) and incubation are poorly understood. Kleckner and McCleave (1988) related the northern limit of spawning by Atlantic eels (*Anguilla* spp.) in the Sargasso Sea to thermal fronts and surface water masses, with spawning taking place south of east-west thermal fronts that separate southern Sargasso Sea surface water from the mixed Subtropical Convergence Zone water to the north. Estuarine and oceanic migrations of the juvenile (yellow) and reproductive (silver) stages of the American Eel are currently being studied under the Ocean Tracking Network (OTN) Canada (2010-2012). In 2010, acoustic lines were deployed in the system at four locations within the St. Lawrence River. Overall, 76% of silver eels (N = 62) and 33% of yellow eels (N = 30) implanted with sonic tags have been detected downstream. Detections farther downstream will be informative on the migration route and timing. A listening line composed of 30 VR2W (Vemco) acoustic receivers is located south of Cabot Strait, another one is effective in Canso Strait, and a final listening line is located off Halifax, Nova Scotia. Along with U.S. efforts, detections may also be extended beyond to the Gulf of Maine, mid-Atlantic Bight, and Bermuda OTN acoustic curtains (J.J. Dodson, Université Laval, and M. Castonguay, DFO, pers. comm., 2010).

Habitat Trends

Freshwater habitat deterioration, migratory barriers generating habitat loss and fragmentation for upstream migrants, and turbine mortality for downstream migrants, are major habitat perturbations that have occurred since European colonization of North America. Such habitat degradation, along with fishing, are frequently proposed to explain the declines of the American Eel (Castonguay *et al.* 1994a; Haro *et al.* 2000; Verreault *et al.* 2004; see **Limiting Factors and Threats**). By analyzing four decades of abundance data in the St. Lawrence River, de Lafontaine *et al.* (2009b) suggested that the cumulative impact of mortality due to hydroelectric dams and recruitment overfishing are the most probable causes of the decline to the mid-1980s.

The St. Lawrence River watershed has a freshwater runoff averaging 10,100 m^3/s annually, and represents approximately 19% of the total freshwater runoff in the species' range (Castonguay *et al*., 1994a). In the St. Lawrence River watershed, 8,411 dams are present (Verreault *et al.,* 2004). These dams are estimated to block access to 12,140 km^2 of eel habitat in the St. Lawrence River watershed (Verreault *et al.,* 2004).

American Eel abundance has declined in the Ottawa River throughout tributaries of the middle and upper reaches of the watershed and coinciding with the construction of large hydroelectric dams on the mainstem of the river (MacGregor *et al.*, 2010). The same scenario is reported for the Mississippi River (Ontario) watershed, and for the tributaries of the upper St. Lawrence River and Lake Ontario. Eel distribution has contracted to the lower reaches primarily because of reduced recruitment and the construction of numerous dams and hydroelectric facilities (MacGregor *et al*., 2009; Casselman and Marcogliese, 2010a,b). Closer to Lake Ontario, eels have persisted over a greater range of habitats, which is likely attributable to the lower number of hydroelectric facilities on some of these systems (MacGregor *et al.*, 2010).

The National Management Plan (CEWG, 2009) contains objectives and actions focusing on habitat restoration, e.g., mapping of obstructions. Fisheries and Ocean Canada Science and Habitat Management are conducting an American Eel Barrier Study to identify priority watersheds for mitigating barriers to migration and opening up habitat access. A geographic information system (GIS) tool is being developed to aid restoration of American Eel by, for instance, evaluation of eel passability at dams in Québec (Tremblay *et al.,* 2011). Passability ranks were assigned to each category of dam based on three assessment criteria: the height of the dam, the materials used in its construction, and its use. This analysis suggests that the problem of passability (but not mortality) is more significant for upstream passage than it is for downstream passage. Once added to the GIS tool, passability ranks will assist managers in setting priorities for mitigation. Lambert *et al.* (2011) simulated eel dispersal in the Rimouski River watershed (828 km^2) in order to prioritize mitigative measures to overcome obstacles to migration and to increase spawning escapement[4] from continental waters to the Sargasso Sea (OMMER model). For now, the model extension to other watersheds is limited by the availability of relevant data for a diversified set of habitats and watersheds.

[4] Spawning escapement is considered to be the number of silver eels that survive fisheries, turbines, and other continental mortality factors to reach the open sea on their way to the Sargasso Sea spawning grounds.

Habitat Protection and Ownership

In Canada, the American Eel occurs primarily in publicly owned waters. The species' habitat, including ocean habitat used during migrations, may receive protection against harmful alteration and destruction by the Canadian *Fisheries Act*. In addition, habitat protection occurs within the *Canadian Environmental Protection Act*, and numerous provincial acts including the Ontario *Environmental Protection Act*, the Ontario *Water Resources Act*, the Québec *Environmental Quality Act* and *Act Respecting the Conservation and Development of Wildlife*, and the New Brunswick *Clean Water Act*, and *Clean Environment Act*. Habitat that lies within National Parks, Provincial Parks, National Wildlife Areas, and Marine Protected Areas may be subject to additional protection through the *Canada National Parks Act,* the *Loi sur les Parcs* in Québec, the Ontario *Provincial Parks and Conservation Reserves Act, 2006* and the *Canada Wildlife Act*. In Ontario, the American Eel was designated as an endangered species and became protected under Ontario's *Endangered Species Act*, *2007* (S.O. 2007, c. 6) on June 30, 2008. Its habitat will be protected under the general habitat provisions of the act as of June 30, 2013. It will receive habitat protection prior to this if a habitat regulation is developed for the species before June 30, 2013 (B. Walpole, OMNR, pers. comm., 2010).

Rules governing parks and conservation areas, however, do not necessarily prohibit exploitation, where it is otherwise allowed, and do not automatically protect eels from other threats (see **Threats and Limiting Factors**).

BIOLOGY

Life Cycle, Migratory Behaviour, Growth, and Reproduction

The life history of the American Eel is exceptionally complex and encompasses oceanic, coastal, estuarine, and freshwater environments. This complexity is reflected in the large number of life-history stage-specific names given to various forms of eels (e.g., "glass eels", "yellow eels"). Eels are semelparous; spawning occurs once per life-time in the Sargasso Sea (Figure 2; Schmidt, 1922; Helfman *et al.,* 1987). The terminology of eel life history differentiates stages according to migration patterns and morphological characteristics as described below.

Spawning and hatching take place in the Sargasso Sea across several hundred thousand square kiliometres (Schmidt, 1922; Als *et al.* 2011). None of these two activities have been observed, but are mainly inferred from sampling of larvae in the Sargasso Sea (Schmidt, 1922). Temperature fronts in the Sargasso Sea may act as cues that help adult eels locate the spawning area (McCleave, 1987; Kleckner and McCleave, 1988). The latitude of the surface expression of the 22.5°C isotherm is typically near the frontal zone at the northern limit of eel spawning (Kleckner and McCleave, 1988; Tesch and Wegner, 1990; Friendland *et al.*, 2007).

Given that exact spawning areas within the Sargasso Sea are unknown, there is a considerable gap in our understanding of both the timing and the stages of embryogenesis, as well as early larval development. Oliveira and Hable (2010) have studied embryogenesis in the American Eel by *in vitro* fertilization. Their findings demonstrated that the stage of the oocyte at the time of induced ovulation is a crucial factor for successful fertilization.

Recent investigations using otolith microchemistry report three main movement patterns: saltwater residency, freshwater residency, and inter-habitat shifting (Jessop *et al.,* 2002; Cairns *et al.,* 2004; Thibault *et al.,* 2007ab; Lamson *et al.*, 2006; 2009; Jessop, 2010). The proportion of non-catadromous eels (saltwater resident) within and among areas, however, remains unquantified. In the St. Jean River on the Gaspé Peninsula (NFBZ1), some freshwater resident eels make very short forays into brackish or saltwater (Daverat *et al.*, 2006; Caron *et al.*, 2009). Inter-habitat shifting is more frequent in systems where dams do not hinder movements (Jessop *et al.,* 2002; Cairns *et al.,* 2004). Seasonal local movements (Hammond and Welsh, 2009; Hedger *et al.*, 2010) associated with overwintering may also involve habitat needs and environmental requirements in terms of water temperature, oxygen concentration, and water quality, but winter habitat requirements are poorly known (Tesch, 1977; Feunteun *et al.,* 2003). Based on the known positive correlation between salinity and strontium (Sr) in water, habitat use of eels from the St. Lawrence River and Lake Ontario has been inferred from otolith Sr/Ca ratio variations. For all life stages, all of the eels remained exclusively in fresh water (J. Fitzsimons, DFO, pers. comm., 2011). Other factors (predation, food, and habitat availability) undoubtedly affect whether estuarine or freshwater residence or periodic inter-habitat migrations are optimal behaviours (Jessop, 2010).

After a number of years (typically 9 in saltwater to 22 in freshwater; Table 3[5]) as yellow eels in rearing habitats, eels mature into silver eels and migrate back to the spawning grounds in the Sargasso Sea (Figure 2). Life history traits or reproductive characteristics (growth rate, length and age at maturation) are variable across areas in Canada and different life history strategies are detected between male and female American Eel. Males appear to have a time-minimizing strategy, whereas females have a size-maximizing strategy (Helfman *et al.*, 1987; Oliveira, 1999; Tremblay, 2004; 2009a; Jessop, 2010) (see **Biology**-Silver Eel).

[5] Table 3 refers to female freshwater eels.

Table 3. Migration periods, mean length and age of maturing female silver American Eels exiting Canadian freshwater systems.

Site	NFBZ[A]	Migration period	n	Length (mm)	Growth (mm/yr)	Age	Reference
Upper St. Lawrence River	10	June–October	200	915	43.2	19.7	Casselman 2003
			53	976			McGrath *et al.* 2003
			30	1,001	47.9	20.9	Tremblay 2004; 2009a
Richelieu River	10	June-October	494	1,019			Dumont *et al.* 1998
Upper St. Lawrence estuary	10	September-October	474	840			Couillard *et al.* 1997
			4,529	853			Verreault *et al.* 2003
			30	837	41.6	20.1	Tremblay 2004; 2009a
Sud-Ouest River (South shore of the St. Lawrence)	9	August-November	693	1,015	44.4	21.4	G. Verreault, MRNF
			30	1,043	46.2	22.6	Tremblay 2004; 2009a
Rivière aux Pins	9		100	600			Couillard *et al.* 1997
Petite Trinité River (North shore of the St. Lawrence)	2	August-October	74	674	32.1	19.0	Fournier and Caron 2005
			30	679	35.5	20.0	Tremblay 2004; 2009a
Long Pond (Prince Edward Island)	1	August-October	30	693	32.2	19.5	Tremblay 2004; 2009a
Long Pond and Campbells Pond (Prince Edward Island)	1	August-October	82	714	36.5	17.8	Cairns *et al.* 2007
Margaree River (Lake Ainslie (Nova Scotia)	1	September	71	645	26.5	21.9	Cairns *et al.* 2007
LaHave River (Nova Scotia)	1	August-November	352	610	28.3	19.4	Jessop 1987
East River, Chester (Nova Scotia)	1		26	468	23.8	17.1	Jessop *et al.* 2004
Medway River (Nova Scotia)	1		276	555	25.7	19.2	Jessop 1987
Topsail and Indian ponds, Salmon River, Burnt Berry Brook, Topsail Barachois (Newfoundland)	8	August-October	92	694	51.2	12.3	Gray and Andrews 1971
Castors River (Newfoundland)	8	August-October	50	664	30.5	19.7	Jessop *et al.* 2009
Dog Bay (Newfoundland)	8	August-October	94	778	54.9	13.0	Bouillon and Haedrich 1985
Hollyrood Bay	8	August-October	90	722	51.0	12.9	Bouillon and Haedrich 1985

A: National Freshwater Biogeographic Zones: 1) Great Lakes-Upper St. Lawrence; 2) Lower St. Lawrence; 3) Maritimes; 4) Atlantic Islands
B: From Jessop (2010)

Egg

The egg probably hatches within a week of deposition in the Sargasso Sea. McCleave *et al.* (1987) suggested that hatching peaks in February and may continue until April. Wang and Tzeng (2000) proposed, on the basis of otolith back-calculations, that hatching occurs from March to October and peaks in August. Cieri and McCleave (2000), however, argued that these back-calculated spawning dates do not match collection evidence and may be explained by resorption at the edges of the otolith. Bonhommeau *et al*. (2010) hypothesized that the deposition of growth rings in the otoliths of eel larvae would not be daily owing to suboptimal growth conditions in certain areas. A validation of growth increment formation in otoliths of Atlantic *Anguilla* leptocephali is required to assess whether daily increments in otoliths could be confirmed on reared leptocephali.

Leptocephalus

The leptocephalus is the larval form of eel. Leptocephali are transparent willow leaf-like, laterally compressed larvae that are passively transported west and north to coastal waters on the eastern coast of North America, by the surface currents of the Gulf Stream system (Schmidt, 1922; Tesch, 1977; Kleckner and McCleave, 1982) (Figure 4). The time required for a leptocephalus to complete its larval phase is believed to be 7 to 12 months (Kleckner and McCleave, 1985; Avise *et al.*, 1986; Wang and Tzeng, 2000). Bonhommeau *et al.* (2010) used ocean circulation models to simulate the trans-Atlantic movements of American Eel leptocephali. The mean migration duration was estimated to be less than one year; which is considered a short migration compared to the 21-month migration for the European Eel (Bonhommeau *et al.*, 2009, 2010). Vertical distribution is usually restricted to the upper 350 m of the ocean (Kleckner and McCleave, 1982; Castonguay and McCleave, 1987). Growth has been evaluated at about 0.21 to 0.38 mm per day (Kleckner and McCleave, 1985; Castonguay, 1987; Tesch, 1998; Wang and Tzeng, 2000).

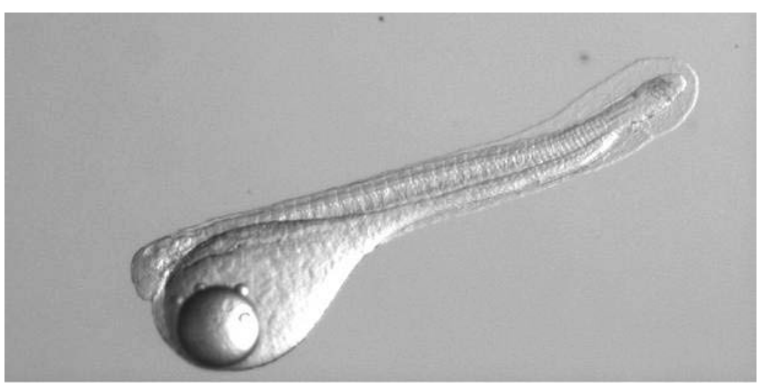

Figure 4. Leptocephalus, larval stage of the American Eel, *Anguilla rostrata* (from K. Oliveira and Whitney Hable, University of Massachusetts Dartmouth)

Glass Eel

As they enter the continental shelf, leptocephali metamorphose into glass eels (Figure 5), which have the typical elongate and serpentine eel shape (McCleave *et al.,* 1987). Developmentally, glass eels represent the juvenile period of development and the term refers to all developmental stages from the end of metamorphosis of the leptocephalus to pigmentation (Tesch, 1977). Metamorphosis occurs when leptocephali are about 55 to 65 mm long (Kleckner and McCleave, 1985). Mean age at this metamorphosis has been evaluated at 200 days and estuarine arrival at 255 days; giving 55 days between glass eel metamorphosis and estuarine arrival (Wang and Tzeng, 2000). Glass eel migration in the Gulf and the estuary of the St. Lawrence River has been recently described by Dutil *et al.* (2009). Ichthyoplankton survey data suggest that glass eels enter the Gulf of St. Lawrence primarily in May, migrate at the surface at night, and disperse broadly once they have passed Cabot Strait. They arrive in estuaries beginning in mid-June and through the month of July. Migrations extend west up to Québec City, in the freshwater zone of the St. Lawrence Estuary, 1,000 km west of Cabot Strait. Straight-line ground speed between Cabot Strait and estuaries was estimated to be 10-15 km per day (Dutil *et al.*, 2009).

Figure 5. Glass eel, unpigmented elver, post-larval stage of the American Eel, *Anguilla rostrata* (from G. Verreault, MRNF).

Elver

Glass eels become progressively pigmented as they enter tributaries; these eels are termed elvers. The melanic pigmentation process (Bertin, 1951; Élie *et al.,* 1982; Grellier *et al.,* 1991) occurs when the young eels are in coastal waters. At this phase of the life cycle, the eel is still sexually undifferentiated. The elver stage lasts about three to twelve months. Elvers that enter fresh water may spend much of this period migrating upstream (Haro and Krueger, 1991; Jessop, 1998a). Elver influx is linked to increased temperature and reduced flow early in the upstream migration season, and to the influence of the tidal cycle later on (Tesch, 1977; Kleckner and McCleave, 1982; Martin, 1995; Jessop, 2003b).

Elver length and arrival date increase from south to north along the Atlantic coast of North America (Vladykov, 1966; Haro and Krueger, 1988). Jessop (2010) demonstrated that elvers distributed along the Atlantic coast of North America increase in length with increasing latitude and distance from the spawning ground. In Atlantic coastal Nova Scotia, elver migration peaks between late April and late June, although small numbers of elvers may continue to enter rivers until mid-August (Jessop, 1998a). Total length averaged 60.14 ± 0.17 mm (50.4 - 70.5 mm) in 2000 on the East River, Chester, Nova Scotia (Jessop, 2003c). On the Murray River (Prince Edward Island), elvers were caught between the end of June and the end of August (Cairns *et al.*, 2007). In the Petite Trinité River (north shore of the Gulf of St. Lawrence), most individuals were already pigmented (elvers) in early July, but arrived until the end of July (Dutil *et al.,* 1989) and averaged 62.4 mm (59 - 69 mm). The average size of elvers in Canada varies roughly between 60 and 67 mm (Jessop *et al*., 2004; CEWG, 2009; Jessop, 2010).

Environmental predictors for the elver run are variable and can change between years (Overton and Rulifson, 2009). In Rhode Island, Haro and Krueger (1988) found that elver abundances peaked during the greatest increase in stream temperature and the lowest water levels. In New York and New Jersey, colonization of upper reaches appears to be contingent on water temperature when reaching a threshold of 10° to 15°C (Overton and Rulifson, 2009; Sullivan *et al.*, 2009). Elver runs in Canada are initiated at colder temperatures; the first elver in Fundy National Park was observed at a water temperature of 5°C (Goodbrand and Austin, 2009). The number of elvers that reach the mouth of the river leading into the watershed proper may, however, have much more to do with factors that impact eels outside the watershed in question.

During the elver run in the East River (Chester, Nova Scotia), the degree of elver pigmentation increased progressively over the run, and glass eels were rarely found after the end of May (Jessop, 2003a). On the north shore of the Gulf of St. Lawrence, in the Petite Trinité River, glass eels occurred in the second half of June and were rare compared to elvers (Dutil *et al.,* 1989).

Young eels can use selective tidal stream transport to move up estuaries (Kleckner and McCleave, 1982). As they enter coastal waters, they transform from a pelagic oceanic organism to a benthic continental organism.

Yellow Eel

The yellow stage is the major growth phase of the species. The belly colour varies from yellowish to greenish or olive-brown, with the back remaining dark (Scott and Crossman 1973; Tesch 1977). The skin is thick and tough and may secrete copious amounts of mucus, which acts as a protective cover. Unlike the well-developed scales of most fishes, eel scales are rudimentary and embedded deeply within the skin. Lake Ontario and Lake Champlain are the single largest freshwater rearing habitats for the American Eel within its geographic range.

In the Sud-Ouest River (south shore of the St. Lawrence River) and in the Petite Trinité River (northwestern Gulf of St. Lawrence), upstream migrations occurred between June and August (Dutil *et al.* 1989; Verreault 2002; Fournier and Caron 2005). At Chambly, Beauharnois and Moses-Saunders dams, migration generally peaks in July and August (Casselman *et al.*, 1997; Casselman, 2003, Verdon *et al.*, 2003; Desrochers, 2009). During upstream migration between Beauharnois and Moses-Saunders dams, Verdon and Desrochers (2003) have estimated that yellow eels can travel at a speed of 2.3 km per day.

Sexual differentiation occurs during the yellow stage. Oliveira and McCleave (2000) estimated that sexual differentiation was completed by 270 mm total length. Sexual differentiation is strongly influenced by environmental conditions (Krueger and Oliveira, 1997; 1999; Oliveira 1997). Although Krueger and Oliveira (1999) and Oliveira *et al.* (2001) suggested that density was the primary factor influencing the sex ratio of eels in a river with high densities promoting the production of males, the relative roles of various factors responsible and the mechanisms underlying sex determination in anguillid eels are incompletely understood (Lambert and Rochard, 2007; Jessop, 2010).

In the upper St. Lawrence River and Lake Ontario system, the observation of more males (38%, Pratt and Threader, 2011) during the monitoring of the species' after stocking compared to before stocking suggests that such enhancement initiatives may influence life history of American Eels and that there may be a genetic component to sex determination. First, historically, eels in the upper St. Lawrence River and Lake Ontario system were at much higher densities than present stocking densities and were nearly all female. More than 95% of sexually differentiated eel are female in most Canadian waters (Gray and Andrews 1970; Dolan and Power 1977; Dutil *et al.* 1985; Jessop 1987; Fournier and Caron 2005). Second, the sex ratios observed during stocking experiments match the sex ratios (and growth/maturation characteristics) of the donor stocks. Indeed, in Canada, males appear to be more common in the Scotia-Fundy Region. In the Saint John River, males were 7.4% of a sample of 970 eels (Ingraham 1999). In stream portions of the East River (Chester, Nova Scotia), more than 55% of yellow eels sampled were males (Jessop *et al.*, 2006). A previous stocking experiment of elvers from the Bay of Fundy in a lake on the south shore of the St. Lawrence River (NFBZ9) also produced a higher proportion of males, i.e., 27.2% males after four years of growth, than observed historically (Verreault et al., 2009).

Yellow eels tend to occupy home ranges which vary with habitat type (stream, lake, tidal, creek, marsh, estuary) and movement patterns (freshwater residency, saltwater residency, inter-habitat shifting (Morrison and Secor, 2003)). Home range was estimated to be relatively small (up to 2 ha) in estuarine habitats such as salt marshes (Ford and Mercer 1986) and tidal streams (Bozeman *et al.* 1985; Dutil *et al.* 1988), but in Lake Champlain, LaBar and Facey (1983) reported home ranges up to 65 ha. Some American Eels, however, have been shown to make seasonal migrations in spring and fall, establishing home ranges in summer, and some may inhabit thermal refuge areas over winter (Hammond and Welsh, 2009). Eels from areas that have ready access to brackish or salt waters often undertake downstream spring migration from freshwater to forage in the saltwater environment during the summer and migrate back into freshwater during the fall for overwintering (Medcof 1969; Jessop 1987; Caron *et al.*, 2009). Such migrations between freshwater versus brackish/marine waters habitats influence growth rates (Jessop *et al*., 2004; Thibault *et al*., 2007a; Cairns *et al.,* 2009; Lamson *et al*., 2009; Velez-Espino and Koops, 2010).

Anguillid growth is regulated by a number of factors including salinity, water temperature, productivity, food availability, and geography (Tesch, 2003; Jessop *et al.*, 2004; Jessop, 2010) and by differential effects of physiological factors (Edeline and Elie, 2004; Côté *et al.,* 2009). Eels utilizing brackish and salt waters grow much more rapidly than those in fresh waters (Jessop *et al.* 2002, 2004; 2009; Cairns *et al.* 2004, 2009; Lamson *et al.*, 2009). In the southern Gulf of St. Lawrence, eels resident in salt water grew on average in length 2.2 times faster than freshwater residents, which took an estimated 2.4 times longer than saltwater residents to reach the silvering stage (Lamson *et al.*, 2009). Côté *et al.'*s (2009) results on growth patterns of glass eels monitored in controlled conditions support the hypothesis that geographic origin of glass eels and salinity both influence growth patterns. Côté *et al.'*s (2009) results support the hypothesis of a genetic basis to differences in growth and sex ratio between eels from different areas. This could imply that stocking of areas where only females are found (such as the upper St. Lawrence River and Lake Ontario system) with glass eels and/or elvers from areas with variable proportions of males (Maritimes) may reduce the proportions of females produced in areas subject to stocking. This hypothesis may explain why stocked glass eels and/or elvers of Canadian stocking programs resulted in yellow eels showing pronounced regional differences in sex ratio between donor (Maritimes) and recipient areas (Lake Morin, upper St. Lawrence River and Lake Ontario) (see **Threats and Limiting Factors - Stocking**).

During their yellow phase, eels at Canadian latitudes spend roughly 74% of their time burrowed in the sediment and/or hidden in the substrate (Tomie 2011) (Figure 6). In the summer, eels forage at night and spend the daytime burrowed in the bottom substrate. Excavation experiments showed that eels typically burrow in the mud with their head close to the surface, often with the snout just at the surface. Eel burrows often have two or more openings. Dye experiments showed that eels buried in mud breathe by drawing water from the water column. If the head is below the surface, there is a tunnel between the surface and the mouth to permit water intake (Tomie, 2011).

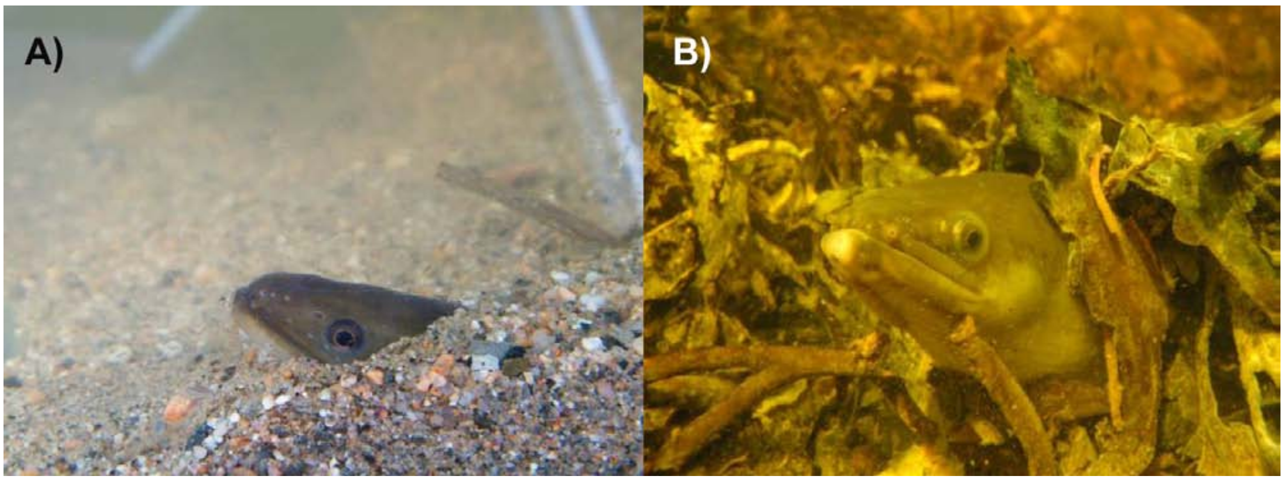

Figure 6. Yellow eel of the American Eel, *Anguilla rostrata*, hidden in the substrate (a) in the sand (from J. Tomie, UNB); (b) in the aquatic vegetation (from P. Rousseau, UQTR).

According to the Mi’kmaq, in the winter eels go in the grass or bury themselves in the mud (CEPI, 2006). Overwintering habitats include freshwater, brackish estuaries, and bays with full strength sea water (Smith and Saunders 1955; D.K. Cairns, DFO, pers. obs.). Eels in estuaries often concentrate in mud that has freshwater upwelling, but they may also be found in sediments that have no freshwater influx (D.K. Cairns, DFO, pers. obs.). Winter burrows, unlike summer burrows can be visually identified and resemble a fist-sized depression on the surface of the sediment. In winter, a visible cavity forms around burrow entrances, but there is no such cavity in summer. Colonial burrowing was observed in winter when large numbers of yellow eels are found in spring holes (Tomie 2011).

Eels are reported to enter torpor (complete inactivity) in mud during the winter at temperatures below 5^{o}C (Walsh *et al.* 1983). Smith and Saunders (1955), however, caught two eels in a stream trap in Prince Edward Island in January and February. Eel “balling” (aggregations of eels) in winter has been well documented by Aboriginal people and commercial fish harvesters who speared large numbers through the ice (Prosper and Paulette, 2002; Denny *et al.*, 2011). Eels speared through the ice in a Prince Edward Island estuary in January had stomachs bulging with fresh and undigested Atlantic Silversides (*Menidia menidia*) suggesting recent feeding activity

(D.K. Cairns, DFO, pers. comm. 2010). These observations suggest that despite what was thought, eels are occasionally active during what are normally thought of as periods of torpor.

Silver Eel

As the maturation process proceeds, the yellow eel metamorphoses into a silver eel. The silvering metamorphosis results in morphological and physiological modifications that prepare the organism to migrate to the Sargasso Sea. The eel acquires a greyish colour with a whitish or cream colouration ventrally (Gray and Andrews 1971; Scott and Crossman 1973; Tesch 1977). The digestive tract degenerates (Pankhurst and Sorensen 1984; Durif 2003), the pectoral fins enlarge (Pankhurst 1982a; McGrath *et al.* 2003; Durif 2003), eye diameter expands, visual pigments in the retina adapt to the oceanic environment (Vladykov 1966; Pankhurst 1982b; McGrath *et al.* 2003), the integument thickens (Tesch 1977; Pankhurst and Lythgoe 1982), percentage of somatic lipids increases to supply energy for migrating and spawning (Larsson *et al.* 1990; Tremblay 2004; 2009a), gonadosomatic index (GSI) increases (Verreault 2002; McGrath *et al.* 2003; Tremblay 2004; 2009a), oocyte diameter increases (Couillard *et al.* 1997), gonadotropin hormone (GTH-II) production increases (Durif *et al.* 2005), and osmoregulatory physiology changes (Dutil *et al.* 1987). While migrating, silvering eels continue to develop their reproductive traits.

There are large variations in life history characteristics (length, weight, age at maturation) of female silver eels across the species’ range (Nilo and Fortin, 2001; Cairns *et al.,* 2008; Tremblay, 2004; 2009a; Jessop, 2010), and such traits and life history strategies are of fundamental importance to ecological understanding and management of the American Eel (Jessop, 2010). Jessop (2010) examined the latitudinal variability in length, age and annual growth rate of silver eels along the Atlantic coast of North America. Phenotypic plasticity reflects the influence of environmental variation on life history traits. These traits may be adaptive if phenotypic differences result from the response of a given genotype to environmental variation (Jessop, 2010). Côté *et al*. (2009) and Bernatchez *et al.* (2011) demonstrated that the differences among traits across the species’ range can have a genetic basis and may be adaptive.

There is considerable geographic variation in the size of silver eels. For example, silver eels from northern areas show greater length, weight, and age at migration (Hurley, 1972; Facey and LaBar, 1981; Helfman *et al.,* 1987; Cairns *et al.,* 2008; Jessop, 2010), but this can vary between the sexes (Helman *et al.*, 1987; Oliveira, 1999; Jessop, 2010). Female silver eels were typically significantly longer and older than male eels from a given site over their geographic range (Jessop, 2010). Within Canada (upper St. Lawrence River, Lake Ontario and Gulf of St. Lawrence), female length at migration increases more with increasing longitude west (distance from the spawning ground) than with latitude north. Thus, female American eels from upper St. Lawrence River and Lake Ontario system (NFBZ10) are, on average, the largest in the species’ range, those of the lower St. Lawrence River (NFBZ10-9) are only slightly smaller, and those from the lower Gulf of St. Lawrence (NFBZ1) and western Newfoundland (NFBZ8) are smaller yet (Vladykov, 1970; Cairns *et al.*, 2006; Jessop *et al.*, 2009; Jessop, 2010).

Although the growing season and the number of degree-days ≥ 10°C declined with increasing latitude, growth rates of females and males were constant or increased in the St. Lawrence River and Gulf. Female growth rate adjusted for the number of degree – days increased in Canada suggesting countergradient variation in growth.
The temperature-size life history rule (increase in body size at lower temperatures) evidently applies to American Eel females, but not to males (Jessop, 2010).

Being less variable within a given area, size, rather than age, appears to be the main cue triggering maturation and migration (Helfman *et al.,* 1987; Oliveira, 1999; Tesch, 1977; Verreault, 2002; Tremblay, 2004; 2009a). Pre-spawning female eels from the St. Lawrence River and its tributaries are generally much larger (means 837-1043 mm) than those from other freshwater rearing sites in Canada (means 650-694 mm, Table 3). While studying the silvering process of the European Eel, Durif *et al.* (2005) concluded that the factors responsible for initiation of sexual maturation remain unknown. No current life history model provides an explanatory mechanism for anguillid life history traits and strategies (Jessop, 2010).

In Canada, there can be a very broad age distribution from samples of female silver eels from a given region (Cairns *et al.,* 2008; Jessop, 2010). Size at migration of silver eels within the same region can also vary temporally (de Lafontaine *et al.*, 2009b). Because the American Eel is a long-lived species, abundance indicators of any continental stage other than glass eels/elvers typically include numerous year classes. The mean age at spawning migration of female silver eels leaving Canadian freshwater sites is 18.6 years (SD 3.1, range 12-23, Table 3). Age of silver eels from brackish and salt habitat is poorly known because such eels are difficult to sample as they leave for the spawning grounds. Silver eels leaving Long Pond, Prince Edward Island, had a mean length of 693 mm (Tremblay, 2004; 2009a; Table 3). If eels from the same geographic area have similar sizes at silvering, the age of silvering of eels reared in saltwater can be estimated from growth trajectories of resident saltwater eels. Eels from saltwater bays a few km from Long Pond, whose exclusive use of saltwater habitat was confirmed by strontium-calcium analysis, showed an annual growth of 55.8 mm/yr (Lamson, 2005). Given this growth rate, these saltwater residents would reach 693 mm at about age 7. Eel growth rate increases sharply with salinity (Lamson, 2005), so the phenomenon of eels silvering much younger in marine than freshwater is probably widespread and general. Annual growth rate of 55.8 mm/yr (Lamson, 2005) is indeed well above the overall mean of mean growth of Canadian female silver eels leaving freshwater sites: 33.8 ± 9.9 mm/yr (Jessop, 2010; Table 3).

American Eel fecundity increases with body size (Wenner and Musick 1974; Barbin and McCleave 1997; Tremblay 2004; 2009a). Because female silver eels are of large body in the St. Lawrence River system, they are more fecund compared to elsewhere in the species' range (Table 3). The freshwater eels are, thus, producing predominantly larger and more fecund females than in brackish estuarine habitats. In five areas within the St. Lawrence River and Gulf, Tremblay (2004; 2009a) found that the female body size is positively related to absolute fecundity. Absolute fecundity ranges from 3.4 to 22 million eggs for body lengths ranging from 532 to 1110 mm and body weight ranging from 260 to 3,340 g. Relative fecundity (oocyte number/mean weight), however, is significantly lower ($p < 0.001$) in large-bodied eels such as the ones from the upper St. Lawrence River. Large-bodied eels average about 6.5 million oocytes per kg while small-bodied eels have more than 10 million oocytes per kg. Given their large size, high fecundity, and exclusively female composition, silver eels exiting the upper St. Lawrence and Lake Ontario have the highest absolute potential egg production per exiting migrant American Eel.

Male silver eels are more common in areas south of the St. Lawrence River and Gulf, including the Scotia-Fundy Region and the east coast of the United States. According to Oliveira *et al.* (2001), the proportion of male silver eels seems to be inversely related to the amount of lacustrine habitat. In a small Nova Scotian river (East River, Chester), Jessop *et al.* (2002) reported that 56.5% of silver eels (N=62) were males. In the Annaquatucket River (Rhode Island), sex ratios of silver eels ranged from 77% males in 1977 to 94% males in 1991 (Krueger and Oliveira 1997). Because males develop into silver eels at much smaller sizes than females, sex of silver eels can usually be evaluated by size alone. Winn *et al.* (1975) used a length threshold of 400 mm to identify females. Male silver eels from the Annaquatucket River had a mean length of 337.3 ± 0.4 mm (N=2,998) and a mean age of 10.9 ± 0.1 years (N= 853; Oliveira 1999). Compared to female silver eels, males have a more constrained size and age at migration. Unlike females, which have large body sizes to increase fecundity, males appear to migrate back to the Sargasso Sea at the minimum size necessary to survive the spawning migration (Helfman *et al.* 1987).

Distance travelled to reach the Sargasso Sea varies substantially over the geographic range of the American Eel. A silver eel from the most distant rearing habitats in western Lake Ontario or the English River (Labrador) migrate more than 5,500 km to reach the spawning grounds, whereas a maturing eel from the closest Canadian rearing habitat in southern Nova Scotia migrates about 2,000 km. Disparities in migration distance generate disparities in the timing of the onset of the migration, probably related to synchronous arrival in the Sargasso Sea, permitting spawning between February (peak) and April (Kleckner *et al.* 1983; Kleckner and McCleave 1985; Helfman *et al.* 1987). Maturing eels begin to descend the Richelieu River in May (Dumont *et al*. 1998). Eels from Lake Ontario and the upper St. Lawrence begin outmigrating in mid-June, and continue until very early October (McGrath *et al.* 2003; NYPA, 2010). Silvering eels in downstream pre-reproductive migration are caught from September to late October in the St. Lawrence estuary fishery (Verreault *et al.*, 2003; De Lafontaine *et al.*, 2009ab). In Nova Scotia, which is much closer to the spawning grounds, silver eels do not outmigrate until November (Jessop, 1987). Downstream migration occurs primarily at night and is typically associated with heavy precipitation and high-flow events (Tesch, 1977; Boubée *et al.*, 2001; Haro, 2003).

During mark-recapture experiments completed in 2010 between Cap-Santé (upstream Québec city) and Kamouraska (190 km stretch), transit time of silver eels has been evaluated to be longer at the beginning of the experiment (5 km/day), and shorter at the end of it (30-35 km/day). Although seasonal patterns of abundance appear similar among years at the Moses-Saunders Power Dam (2000-2010; NYPA, 2010), differences in transit time may be explained by the migration synchrony driving eels by the end of their pre-reproductive migration in the St. Lawrence watershed (G. Verreault, MRNF, pers. comm., 2011). Speed would therefore increase as the pre-reproductive migration progresses toward the downstream end of the St. Lawrence River watershed. In three European watersheds, Daverat *et al.* (2005) have evaluated migration speed of the freshwater residents travelling across estuaries at 3 to 4 km per day.

In 2006, 22 silvering European Eels were equipped with miniaturized pop-up satellite archival transmitters (PSAT) that were released in Ireland to study their oceanic spawning migration. Over a distance of 1,300 km southwest from release, the horizontal migration speed varied from 5 to 25 km per day, much lower than required to reach the Sargasso Sea for spawning in April, which could indicate partial impairment of swimming ability owing to such tags.(Burgerhout *et al.* 2011). Eels underwent diel vertical migrations from depths of 282 m (11.7°C) at night to 564 m (10.1°C) during the day, on average (Aarestrup *et al.*, 2009).

Age Estimation

The American Eel shows considerable variability in size at age, advising caution in applying an age-length key for this species. Age estimation is also recognized to be extremely difficult, in part because of their migratory life cycle. In order to standardize aging and to improve age estimation procedures, a Workshop on Age Reading of European and American Eel (WKAREA) took place in 2009 (ICES, 2009ab). Two methods are internationally accepted to age eels from otolith structures: grinding and polishing (and mostly staining), and cutting and burning.

In eel aging, the zero band (or elver check) is considered the start of first growth check outside the nucleus from where continental growth and age determination commence on the otolith structure. Age interpretation is representative of the continental phase (ICES, 2009ab). According to Bonhommeau *et al.* (2010), the mean American Eel migration duration from the Sargasso Sea would be less than one year. Therefore, the first migrating year of the larvae is not conventionally interpreted on the otolith structure.

Age estimation can be validated with otoliths from eels of known age, perhaps stocked elvers marked with OTC (Verreault *et al.,* 2009; Pratt and Threader, 2011). These otolith references will help to identify and describe growth increments over time in order to discriminate between false and real annual marks. In general, age estimation does not correspond automatically to true age, and raises concerns about stress factors inducing abnormal marks, such as waterfalls, dams, marking/handling procedures. Migration barriers could modify the growth trajectory by investing more energy in migration instead of somatic growth (ICES, 2009a). Also, for eels sampled early in the season (before growth has resumed) the last true annual check is not yet visible or situated very close to the outer margin (ICES, 2009a). It is possible that reported age estimations for eels were not conducted with one of the two standardized methods, and may not have been validated with more than one reader/reading. Depending on the reader awareness in age estimation, some may have overestimated systematically aging due to these false growth checks.

Mean generation time of female American Eels can be calculated as age at silvering + 2, to account for the time of migration from and to the spawning grounds; generation time for eels reared in Canadian fresh waters is approximately 22 years. Mean generation time for female American Eels reared in salt water is poorly known, but is probably roughly nine years, based on an estimate for Prince Edward Island marine habitats (COSEWIC, 2006).

Nutrition

Leptocephalus

A recent review by Miller (2009) suggests that leptocephali do feed. Oval objects that are likely zooplankton fecal pellets are also occasionally seen inside the guts of leptocephali collected in offshore areas far from the higher productivity coastal waters. These types of observations disprove the hypothesis that leptocephali do not feed and only get their nutrition by absorbing dissolved organic carbon (DOC). Observations of the fine structure of leptocephalus guts have suggested that they are adapted for intestinal water absorption, so DOC in the water surrounding the particulate matter is also likely absorbed, providing additional nutrition (Otake *et al.* 1995; Otake 1996; Ozaki *et al.* 2006a cited in Miller, 2009). Leptocephali also consume detrital particles such as fecal pellets or particles such as discarded shelters of larvacean tunicates (Otake *et al.,* 1993; Mochioka and Iwamizu. 1996; Riemann *et al.*, 2010).

Glass Eel and Elver

According to Bardonnet and Riera (2005), glass eels may use the estuary not only as a simple migration route, but also as a feeding habitat. By contrast, laboratory experiments on European glass eels by Lecomte-Finiger (1983) suggested that they were morphologically and physiologically unable to feed. Tesch (1977), however, found that elvers at a later stage of pigmentation, stage VIA4 (Élie *et al.* 1982), were feeding. Stomach examination of elvers caught during their upstream migration in the Petite Trinité River on the north shore of the Gulf of St. Lawrence revealed that elvers fed primarily on insect larvae (Dutil *et al.* 1989).

Yellow Eel

The yellow eel is essentially a nocturnal benthic omnivore. Prey includes fishes, molluscs, crustaceans, insect larvae, surface-dwelling insects, worms, and plants. The eel prefers small prey animals that can easily be attacked (Tesch 1977). Food type varies with body size (Ogden 1970, cited in Tesch 1977). Stomachs of eels less than 40 cm and captured in streams contained mainly aquatic insect larvae, whereas larger eels fed predominantly on fishes and crayfishes. In Lake Champlain, food sources were mainly fish (38%), decapods (30%), and insects (10%), and insect abundance decreased in larger eels (Facey and LaBar, 1981). Feeding activity decreases or stops during the winter, and food intake ceases as eels physiologically prepare for the spawning migration.

POPULATION SIZE AND TRENDS

Historically (prior to 1980), the American Eel exhibited one of the largest ranges of any diadromous fish in the Western Hemisphere and held a dominant position in freshwater fish communities by numbers and biomass (Smith and Saunders, 1955). In many freshwater systems, eels accounted for more than 50 percent of the total fish biomass (Smith and Saunders, 1955; Lary *et al.,* 1998). In Ontario, although eels are at the inland extremity of the species' range, they were widely distributed and abundant as far as Niagara Falls and the headwaters of the Ottawa River, including the Mississippi River subwatershed (Casselman, 2003). Baselines and perceptions of former abundance and distribution of the American Eel, however, have shifted over time and strong declines of eels have occurred in some areas (MacGregor *et al.,* 2008; 2009; 2010). Archaeological records and ATK indicate the former high abundance of eels throughout the Lake Ontario, St. Lawrence River and Ottawa River watersheds was sufficient to support local commercial fisheries (MacGregor *et al.*, 2009). Currently, where eels continue to persist in inland rivers and lakes, their abundance is now very low, and eels are approaching extirpation from all inland watersheds in Ontario (MacGregor *et al.*, 2010; Casselman and Marcogliese, 2010a,b). In more eastern portions of the range, American Eels are still widespread, although indices of abundance appear to be variable. Empirical studies on eel densities and abundances in Canada, however, are limited. Types of available data vary among NFBZs, and reported landings are available for all NFBZs.

The decline of American Eel is well documented with fisheries-independent data in the upper St. Lawrence River and Lake Ontario system (NFBZ10) and, therefore, most available information is from this area. Region-specific status indices show that abundance relative to the 1980s is very low for the upper St. Lawrence River and Lake Ontario system (DFO, 2010). Edeline (2007) proposed that the decline of the European Eel was less dramatic in saltwater than in freshwater habitats, suggesting that eel diadromy may be a conditional strategy and that anthropogenic changes are acting selectively on freshwater habitats. This idea may also apply to American Eels because the upper St. Lawrence River and Lake Ontario systems (NFBZ10) are freshwater habitats and these eels show the most evidence of declines compared to eels in the Gulf of St. Lawrence and Atlantic Region. Moreover, because Côté *et al.* (2009) suggested that differences in growth and sex ratio in American Eel have a genetic basis, it is possible that the silver eel mortalities observed in the upper St. Lawrence River and Lake Ontario system (NFBZ10) over the last 50 years (approximately 75% every year Verreault and Dumont, 2003) reduced the prevalence of highly migratory ecotypes; i.e., those that migrate more than 5,500 km from spawning areas in the Sargasso Sea.

Lake Ontario and St. Lawrence River Region

<u>NFBZ10 Great Lakes - Upper St. Lawrence (Ontario and western and central Québec)</u>

The number of eels in NFBZ10 (Figure 7) declined greatly from the mid-1980s through the 1990s (Castonguay *et al.,* 1994a; Casselman *et al.,* 1997; Casselman, 2003). Commercial catches in Ontario declined sharply despite increased price per kg and fishing effort (Casselman, 2003). Eel abundance has been severely reduced in Lake Champlain, the Ottawa and Richelieu river basins, the upper St. Lawrence River, and Lake Ontario. Declines have also been documented in three major tributaries on the U.S. side of Lake Ontario: the Oswego (including Oneida Lake), Genesee, and Black rivers.

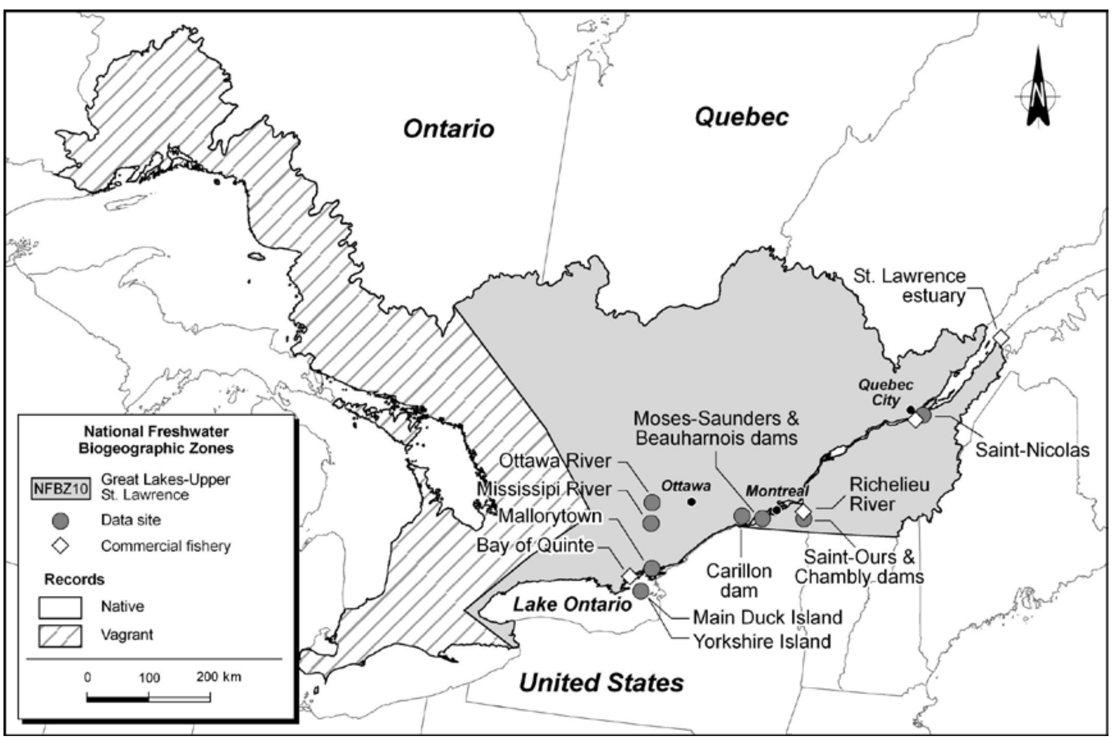

Figure 7. Data sites and commercial fisheries for American Eel within the National Freshwater Biogeographic Zone NFBZ10, Great Lakes Upper St. Lawrence (records from Mandrak and Crossman 1992).

The declines in American Eel in these tributaries appear to be largely due to recruitment failure driven by mortalities associated with a series of migration barriers, including hydroelectric facilities (Casselman, 2003; MacGregor *et al.,* 2009; 2010; Casselman and Marcogliese, 2010a).

In the Ottawa and lower Mississippi (Canada) rivers, recent survey data and incidental observations show that eels are still present in the lower portions of the watershed (below the most downstream dam in the system, the Carillon Dam), but in very low numbers, suggesting they are close to extirpation there (Casselman and Marcogliese, 2010a,b). According to trap net catches in lakes of the Mississippi River watershed (1961-2009), eel catches have been declining through time and are now approaching zero (J.M. Casselman, unpublished data). Based on the present survey, past reports, local observations, and historical abundance data, eel abundance in the lower Ottawa and Mississippi rivers has followed the dramatic declining trend reported for the upper St. Lawrence River and Lake Ontario system (Casselman and Marcogliese, 2010a,b), and in the Richelieu River and Lake Champlain system (Verdon *et al.*, 2003).

There are four ongoing abundance indices for the upper St. Lawrence River and Lake Ontario system: a long-term count of juvenile eels ascending the eel ladder at the Moses-Saunders Power Dam (1974–2010; Figure 8), a trawl survey index in the Bay of Quinte (Lake Ontario) (1972–2010; Figure 9a), an index of abundance based on an electrofishing survey in eastern Lake Ontario (1984-2010; Figure 9a), and dam tailwater surveys of emigrating adults at Moses-Saunders Power Dam (2000-2010; Figure 9b). The first three, longer-term Ontario abundance indices showed significant negative trends over the past 10 to 36 years (Figures 9a,b) indicating that eels are at about 3% of their mid-1980s abundance, far from the objective of rebuilding stocks to mid-1980s levels (DFO, 2010).

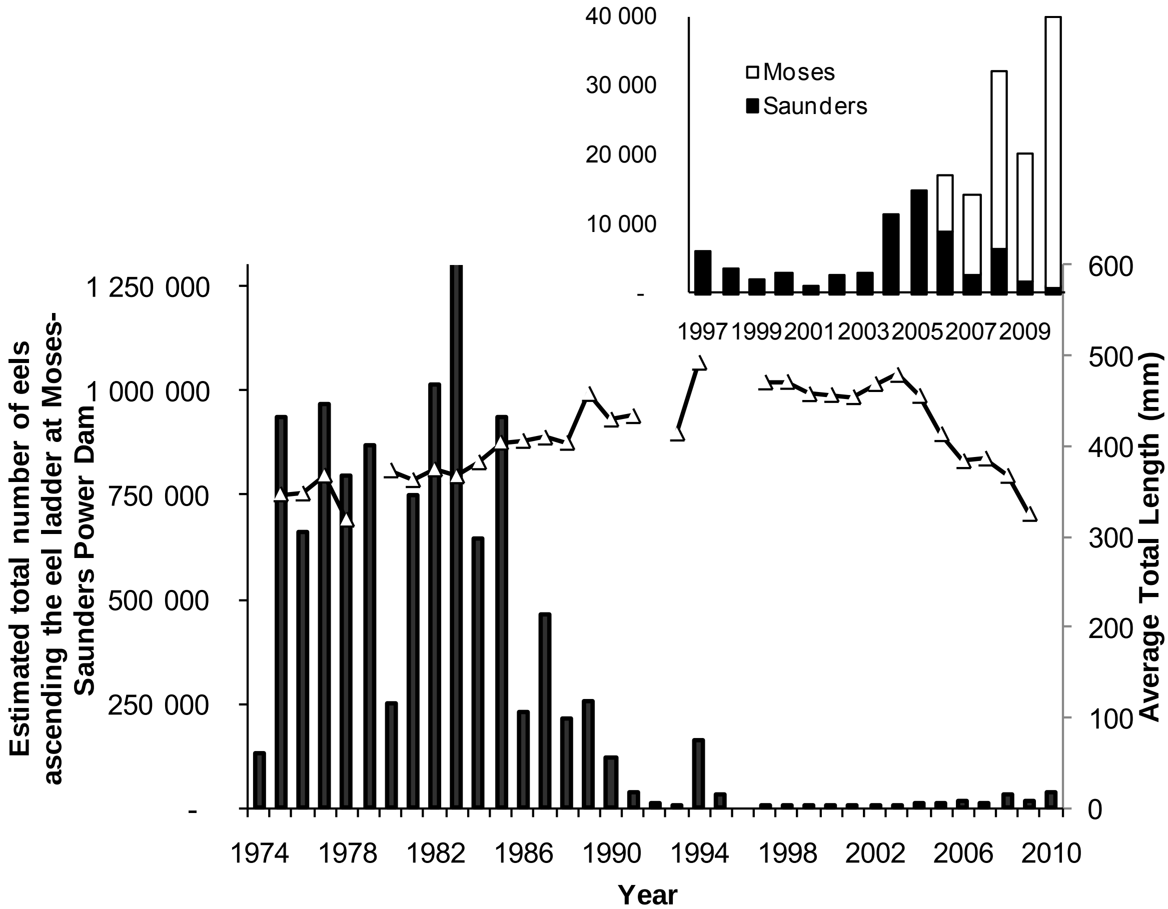

Figure 8. Estimated number of immature American Eels annually ascending the Moses-Saunders Dam eel ladder per day in the upper St. Lawrence River at Cornwall, Ontario, and New York Power Authority (Massena, New York) eel ladders combined. The insert shows recruitment in recent years, including the addition of a second ladder in 2006 on the Moses (United States) side of the generating station (from A. Mathers, OMNR). No counts are available for 1996. Number of eels represents total passage, not a standardized index of abundance because of variability in operating conditions. Days of operation for the Moses-Saunders Dam eel ladder varied greatly among years (33 to 154 d). Operation of a second ladder (the Moses ladder) began in 2006.

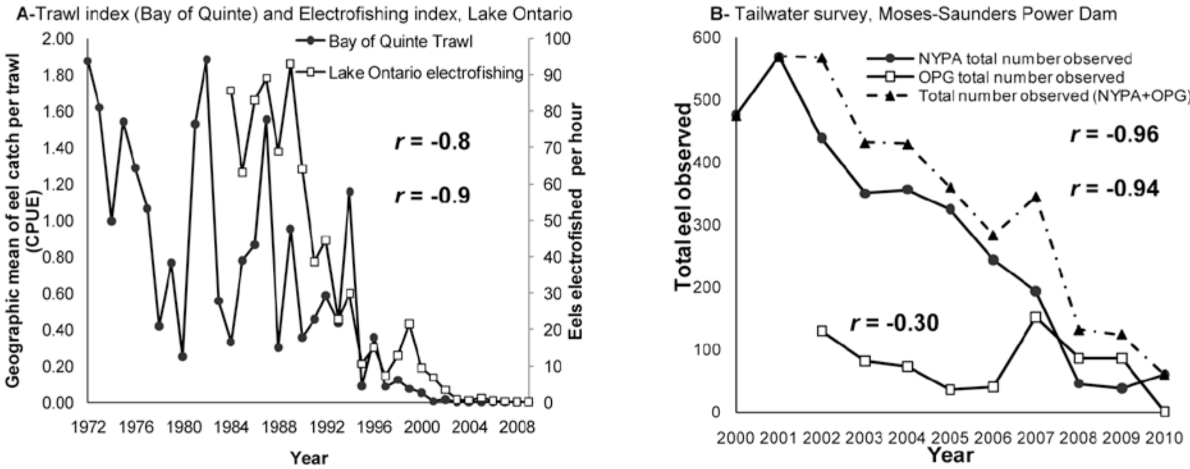

Figure 9. Fisheries-independent indices of abundance of American Eels in NFBZ10 (from A. Mathers, OMNR): (A) bottom trawl CPUE from Bay of Quinte, Lake Ontario (1972-2010) and electrofishing CPUE from eastern Lake Ontario (1984-2008) (yellow eel stage). Correlation coefficients for both time series are highly significant (both P < 0.001); (B) Counts of outmigrating, maturing American Eel detected downstream of the Moses-Saunders Power Dam during tailwater surveys (2000-2010) by: (i) Ontario Power Generation (OPG-Canada), (ii) New York Power Authority (NYPA-US) (from K. McGrath, NYPA), and (iii) the total of both counts. Only the OPG-Canada count trend, which incorporated a design change in 2007, was not significant (all other P < 0.001).

Indices of Abundance from Eel Ladders

The longest-term data set on yellow eel recruitment in NFBZ10 is for upstream migrants ascending the eel ladder at the Moses-Saunders Power Dam first operated in 1974 (Castonguay *et al.,* 1994a; Casselman, 2003; Mathers and Pratt, 2011). Counts of ascending juvenile eels at Moses-Saunders Power Dam averaged over 600,000 eels annually in the 1980s. Despite increasing numbers of smaller eels in recent years (Casselman, 2008), the average abundance for the 2000s (11,949 eels per year) is only 2% of the average observed in the 1980s, and 1994 was the last year of sizable movement (163,518 eels) of eels up the ladder (Figure 8).

A second fishway, the Moses eel ladder, however, has been operated since 2006 by the New York Power Authority (NYPA) on the U.S. side of the dam. Since this new ladder came into service, the use of the Moses-Saunders index (31-days[6]) alone is no longer valid as a comparison to earlier years (because some eels now use the Moses ladder). The counts of eels migrating upstream at both ladders at the dam are a valuable indicator of the number of eels in the system in that a strong year class moving in would be detected (A. Mathers, OMNR, pers. comm., 2010).

[6] In the past, an index of the abundance of eels migrating upstream was calculated based on the daily passage during the 31-day period that represented the peak of eel migration during each year. This index was used to address variation in the operating conditions and counting strategies which varied widely over the early years of ladder operation. The total numbers of eels moving upstream at the ladder located on the Canadian side of the dam is strongly correlated (r^2=0.94) with the ladder index (see Pratt and Mathers, 2011 for details).

During 2006, the first year of operation for the Moses ladder (U.S.), the numbers at the two eel ladders were very similar. During 2007-2008 around four times more eels were estimated to move through the Moses ladder (U.S.) at the south end of the dam compared to the Saunders ladder (Canada). In 2009, this ratio went up to ten times (Figure 8). Combined, the ladders recorded 17,144 ascending eels in 2006, 14,204 in 2007, 32,330 in 2008, and 20,214 eels in 2009 (Mathers and Pratt, 2011).

Even if recent (post-2006) numbers of eels passing the ladders are higher than what was observed since 2001 the abundance of upstream migrants entering the upper St. Lawrence River and Lake Ontario system are still less than 3% of the numbers migrating upstream during the early 1980s—over 1 million eels per year moved upstream in 1982 and 1983 (OMNR, 2009; Mathers and Pratt, 2011). This represents a decline of ascending juvenile eels of up to 98% over a little more than a single generation (i.e., 26 years). Even with the closure of the commercial (2004) and recreational fisheries (2005) in Ontario, the abundance of yellow eels in the upper St. Lawrence River and Lake Ontario system remains low (OMNR, 2009; Mathers and Pratt, 2011).

The age of eels observed at the Saunders ladder has changed over time (Marcogliese and Casselman, 2009). Eels ascending the ladder had a mean age of about 6 yr in 1975 (Liew, 1976), of 5 yr in the 1980s (Casselman *et al.,* 1997), and mean age increased to about 12 yr in the 1990s (Casselman, 2003). Then, the mean age of eels ascending the Saunders ladder decreased from about 11 yr in 2003 to 6 yr in 2006-2007 (Casselman, 2008). Whereas very few young fish were present in the samples from 2003 and 2004, a strong new pulse of young recruits has been seen in 2005 at Moses-Saunders, where 56.7% of eels were in the 5 yr to 8 yr age range (Casselman, 2008).

Downstream of Moses-Saunders Power Dam, young eels ascend eel ladders at the Beauharnois Dam on the Québec side of the upper St. Lawrence River (insert of Figure 3; Figure 10a). Eel counts were monitored in 1994-1995 and from 1998 to 2010 (Desrochers, 2009; Figure 10a). Before 1993, there was an accumulation of eels below the Beauharnois Dam before the ladders became operational. Eels ascending the ladders included some of the older and larger eels from previous years. Counts dropped from 24,721 in 1994 to 5,441 in 1998, and then rose to 43,111 in 2002 (Figure 10a). Since 2002, more than 50,000 eels have been observed to ascend the ladder annually. Although total counts fluctuate (Desrochers, 2009), the increases seen at Beauharnois Dam in recent years mirror the increasing numbers of small eels seen at Moses and Saunders ladders (Figure 10a). As reported for Moses-Saunders Power Dam, however, the rising numbers at Beauharnois Dam also correspond to a decrease in mean eel size, suggesting a declining mean age of eels that arrive at the dam. In 2004, Verreault and Tardif (2006) estimated the mean age of eels caught at Beauharnois Dam in 2004 at 6 yr. The pronounced reductions in mean size of ascending eels are unclear.

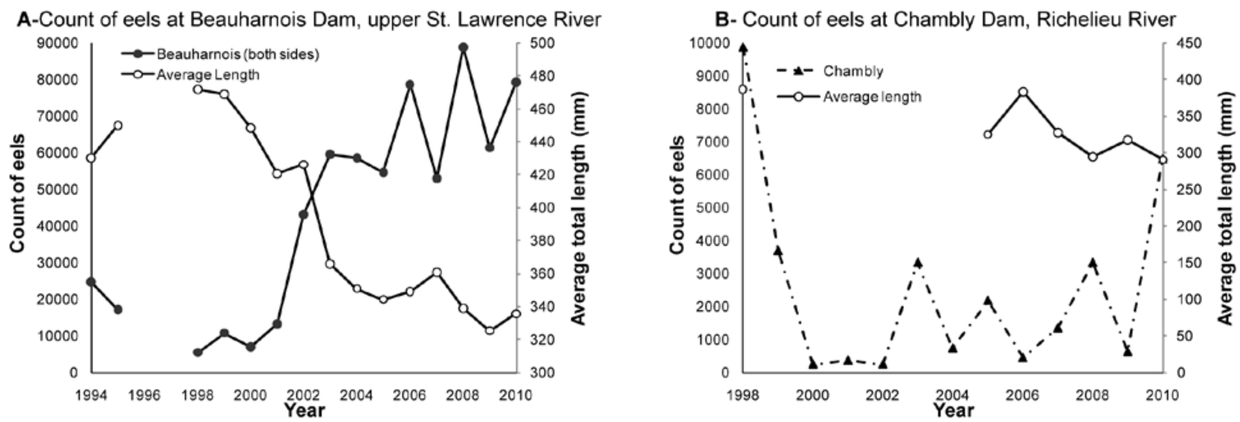

Figure 10. Fisheries-independent index of abundance of immature American Eels ascending the eel ladders in NFBZ10 (A) at the Beauharnois Dam in the St. Lawrence River, and (B) at the Chambly Dam in the Richelieu River (from J. Caumartin, Hydro-Québec).

Counts of eels ascending the eel ladder of the Chambly Dam in the Richelieu River, a tributary of the St. Lawrence River, provides some information on the abundance of eels in the St. Lawrence River (Figure 10b). Upstream movements of yellow eels have been monitored at the Chambly Dam since 1998 (Desrochers, 2009). Numbers were high in the first two years that the ladder operated (1997: 9,875 eels; 1998: 3,695 eels; Figure 10b). Since 2003 there have been irregular increases and decreases. The highest numbers were recorded in 2010, with 6,476 eels counted (Desrochers, 2009; Figure 10b). As at Beauharnois Dam, eels sampled from the Chambly Dam ladder represent a large number of age classes, ranging from 1 to 11 yr in the samples from 2009 (DFO, 2010). In 2009, however, 38.3% of the young eels ascending the Chambly Dam eel ladder were initially stocked above the dam in 2005, 2006, or 2007, moved downstream and then re-ascended the ladder toward Lake Champlain, which confuses the interpretation of the trend in recruitment of eels at this facility and could create a strong bias in the recruitment data series (P. Dumont, MRNF, pers. comm., 2010; DFO, 2010). As for Beauharnois Dam counts, numbers of eels ascending the Chambly Dam ladder are insufficient to restore historical abundance levels in upstream waters. The former commercial fishery in the Richelieu River (efficiency of 66%) harvested a mean of 34,000 kg (or approx. 23,000 individuals) of silver eels annually between 1920 and 1980 (Verdon *et al.*, 2003; COSEWIC, 2006).

Indices of Yellow Eel Abundance

The abundance of larger yellow eel in the upper St. Lawrence River and Lake Ontario system was measured by two assessment programs and each indicates that American Eel remain at very low levels in Lake Ontario. Bottom trawling in the Bay of Quinte has been conducted since 1974 as part of the fish community index program. The average catch of American Eel from 1974 to 1994 was 0.94 eel per trawl; however, no eels were captured in the 364 trawls conducted between 2003 and 2009 (Figure 9a). This suggests that eels are at a very low abundance in this area (Mathers and Pratt, 2011).

Quantitative electrofishing along transects has been conducted by J.M. Casselman and L.A. Marcogliese (Queen's University) in the upper St. Lawrence River in the Mallorytown area and in the east end of Lake Ontario (Main Duck Island and Yorkshire Bar) for 16 years and 26 years, respectively. Fishing is conducted during both the daytime and the nighttime. Detections (or observations, or catch) in 2008 and 2009 were the lowest reported since fishing was conducted. At both locations and times of day, catches were not statistically different than the previous four years ($P > 0.05$) and have not been statistically greater than 0 since nighttime catches in 2005, even with the inclusion of eels from stocking programs (Casselman and Marcogliese, 2009).

Catches in the Bay of Quinte (Lake Ontario) trawl survey index (CPUE) and the electrofishing index in the eastern part of Lake Ontario are currently not significantly greater than zero and show strong and significant negative trends over the past 10 to 36 years (Figure 9a; Casselman and Marcogliese, 2009). Both of these indices are strongly correlated with the decline of the eel ladder counts at the Moses-Saunders Power Dam. The best correlation ($r = 0.78$) between immigration at the eel ladder and trawl catches in the Bay of Quinte was with a 4-year lag. Electrofishing catch was most strongly correlated ($r = 0.89$) to the number of eels that ascended the ladder five years earlier. These indices clearly reveal a severe decline as a consequence of reduced recruitment to the upper St. Lawrence River and Lake Ontario system (Mathers and Pratt, 2011).

Indices of Spawning Escapement

Fully mature silver eels do not occur naturally in Ontario. Fish displaying the early stages of sexual maturity, however, have been identified because many downstream migrants show some characteristics of their final metamorphosis into silver eels (McGrath *et al.,* 2003; Tremblay, 2004; 2009a). Surveys of the tailwaters downstream of the Moses-Saunders Power Dam have identified dead and injured eels and provide an index of the number of silvering eels leaving this system on their downstream spawning migration since 2000 (NYPA, 2010). The tailrace survey is conducted in a systematic manner each year, sampling twice per week from mid-June through very early October; this broad period encompasses the outmigrating period for maturing eels in the upper St. Lawrence River and Lake Ontario system. The number of these eels has declined by approximately 20% annually since the start of the survey in 2000 (Figure 9b; NYPA, 2010). Allowing for a 10- to 15-year yellow eel growth phase in the upper St. Lawrence and Lake Ontario (Casselman, 2003), this decrease in number of downstream migrants is consistent with the decline observed in upstream juvenile migrants beginning in the mid-1980s. The best correlation ($r = 0.87$) between the abundance of upstream migrants and the numbers of eels observed in the tailwaters is with an 18-year lag (Mathers and Pratt, 2011; Pratt and Mathers, 2011). In 2010, tailwater surveys suggest that there are only 8% of the silver eels leaving the system compared to 2000, or fewer than 20,000 silver eels leaving the upper St. Lawrence River and Lake Ontario system annually (NYPA, 2010). A similar protocol has recently been established on other generation stations to document the presence of silver eels during tailwater surveys at hydro generation facilities in the Mississippi and Ottawa rivers (Thompson *et al.,* 2010 cited in Mathers and Pratt, 2011).

In 1996 and 1997, spawning escapement and fishing mortality on silver eels originating from NFBZ10 were estimated by means of mark-recapture experiments in the St. Lawrence estuary (Caron *et al.,* 2003). At that period, migrants upstream of Québec City were estimated to number 492,845 (383,693 - 633,091; 95% confidence limit) in 1996, and 410,895 (353,591 - 477,492; 95% confidence limit) in 1997. Based on these data, Verreault and Dumont (2003) estimated silver eel departures from the upper St. Lawrence River and Lake Ontario from a model based on eel passage, commercial landings, percentage of migrating eels in the commercial catch, and turbine survival rate. Because exploitation rate by the commercial fishery in the estuary was estimated to be 19% in 1996 and 24% in 1997, total spawning escapement was estimated to be 396,000 in 1996 and 302,000 in 1997. This mark-recapture experiment was repeated in 2010, using the same methods. The total number of migrants upstream of Québec City was estimated to be 153,044 (116,480 - 189,608; 95% confidence limit) fish in 2010 using pooled Peterson estimates. The exploitation rate by the commercial fishery is now estimated to be 10.5% (16,113 eels), and the total spawning escapement is currently estimated to be 136,932 eels (G. Verreault, MRNF, pers. comm., 2010). Total spawning escapement has, therefore, decreased by 65% in the 14 years despite the reduction in mortality from commercial fisheries (50% of fishing effort between 2002 and 2009) (G. Verreault, MRNF, pers. comm., 2011).

A fisheries-independent abundance index is provided by catches of eels at the Saint-Nicolas experimental trap fishery, near Québec City, between 1971 and 2009 (de Lafontaine *et al.*, 2009ab). Most eels caught in the fall (1 September - 1 November) are silver eels, although colour has not been systematically recorded. In the 1970s, fall catches showed intermittent high peaks (Figure 11a). Between the late 1970s and the late 1990s, fall catches were relatively stable, and fall catches in the late 1990s and early 2000s again showed intermittent peaks. From 2001 to 2009, fall catches showed a constant decline and over the entire 40 year time series there is a strongly significant trend of decline in abundance (Figure 11a; de Lafontaine *et al.*, 2009a,b). Another index consists of the catch records at commercial weirs (fixed gears) compiled from the personal logbooks of one fisher that fished with relatively constant effort 1 km upstream from the experimental trap (1965-1998; Figure 11b). These two CPUE indices indicate a significant declining trend in eel catch rate by between 65% and 80% since the early 1970s which has been suggested to indicate declining juvenile recruitment to the system (de Lafontaine *et al.* 2009a).

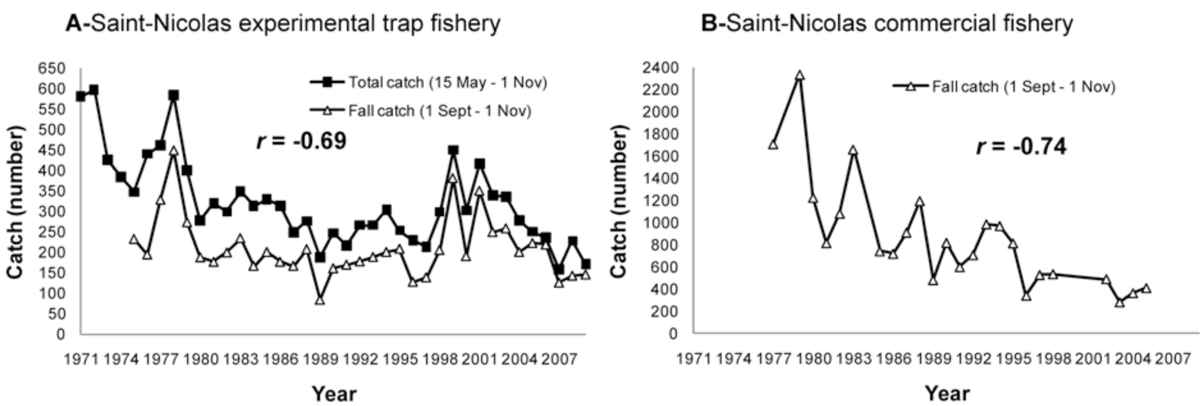

Figure 11. Total catches of maturing American Eels recorded at Saint-Nicolas, near Québec City (from de Y de Lafontaine, Environment Canada) (A) experimental trap fishery (1971-2009); (B) commercial fishery 1 km upstream (1977-2005). No catches are available for 1978, 1984, 1999-2001. The beginning date of the commercial fishery varied among years but the fall catch was always entirely covered. Both trends are highly significant ($P < 0.001$).

Landings

In Ontario, the commercial fisheries were closed in 2004, and the recreational fishery for eels was closed in 2005. In Québec, fisheries are now restricted with the implementation of a major buyout of licences between 2002 and 2009 (Table 4; from DFO, 2010). These reductions in effort, along with a decline in abundance of eels from the upper St. Lawrence River and Lake Ontario, have contributed to reduced catches in recent years.

Table 4. Chronology of licence buyout programs in the commercial fishery for American Eel in the Lake Saint-Pierre, Pont Laviolette and Île d'Orléans area, and lower estuary of the St. Lawrence River in NFBZ10 (modified from DFO, 2010).

NFBZ10 Region	Fishing Gear	Harvest composition	Year of buyout	Licences bought out	Remaining licences
Lake Saint-Pierre	Hoopnet	Mostly yellow eels, few silver	2002 2005 2006 2007	6 17 1 12	36 19 18 6
Pont Laviolette – Île d'Orléans	Hoopnet	Mostly yellow eels, few silver		0	30
	Trapnet	Mostly silver eels, few yellow	2009	8	2
Lower St. Lawrence estuary	Trapnet	Silver eels	2009	46	21
Total			2002-2009	90	59

Total landings for yellow eels in Ontario increased from 50 t in 1960 to peak at 230 t in 1978. Except for two low catches in 1982 and 1983, due to market collapse associated with Mirex[7] contamination, annual catches exceeded 100 t between 1972 and 1993 and then declined rapidly to an average of 23.8 t (range 11 to 41 t) from 1997 to 2002, until the closure of the commercial eel fishery in Lake Ontario in 2004 (Figure 12). This decline occurred despite sustained high prices; well above the long-term mean (Casselman and Marcogliese, 2007). Total reported landings for Ontario have declined in parallel with the Moses and Saunders ladders "recruitment index" (Figure 8).

[7] Mirex = synthetic odourless insecticide. Mirex is a snow-white crystalline solid.

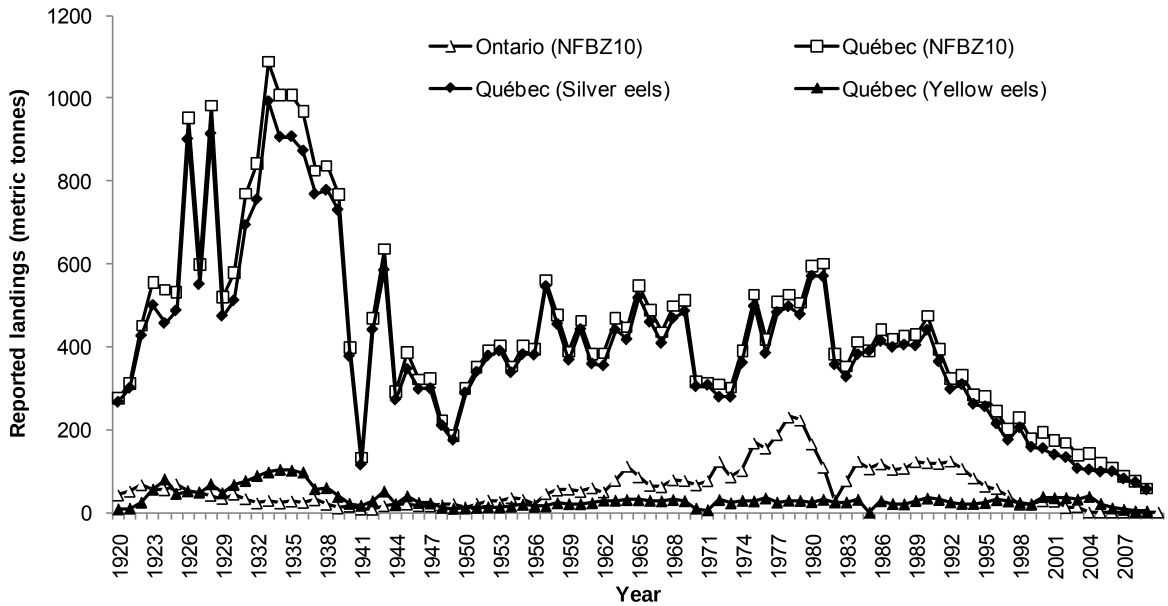

Figure 12. Commercial indicators of American Eel abundance within NFBZ10 (1920-2009): Reported landings in Ontario (upper St. Lawrence River and eastern Lake Ontario; from A. Mathers, OMNR), and for Québec (yellow and silver eels) (from Y. Mailhot,MRNF).

Because silver eels from the upper St. Lawrence River and Lake Ontario system are primarily captured in the Québec lower estuary fisheries of the St. Lawrence River, Québec silver eel landings from the Richelieu River and the St. Lawrence estuary are considered under NFBZ10 (Verreault *et al.* 2003; Verdon *et al.* 2003). Annual landings show dramatic declines since the 1980s (Figure 12). From 72.9 t caught in 1981, the Richelieu River fishery collapsed in 1997 with a total catch of less than 5 t and was closed in 1998 (Dumont *et al.* 1997; Verdon *et al.* 2003). In the Richelieu River, the decline has been partly related to the reconstruction of two old cribwork dams in the 1960s. No passage facilities for eels were provided, so upstream migration to Lake Champlain was impeded (Verdon *et al.* 2003). Between 1987 and 1997, average eel weight increased by 50%, which reflected an aging process that was not supplemented by recruits (Verdon *et al.* 2003). In the St. Lawrence estuary, total catch declined from 452 t in 1980 to less than 82 t in 2004 (Figure 12). Fishing effort in this area was relatively constant until 1996, and thereafter declined. To try and compensate for hydroelectric turbine-based mortality, a fishing licence buyback program was initiated in 2009 for the St. Lawrence River (Table 4). A fishing licence buyback for Yellow Perch (*Perca flavescens*) conservation in Lac St. Pierre between 2002-2009 resulted in an estimated reduction in total mortality of eels from fishing (measured as landings) of greater than 50% relative to mortality from 1997 to 2002. During 1997 to 2002, the mean catch of yellow eels, primarily from the Lac St.-Pierre fishery, was 29.5 t (range: 20 to 37 t). From 2005 to 2009, the mean catch of yellow eels was 8.7 t, a 71% reduction from the 1997 to 2002 mean. From 1997 to 2002, the mean catch of silver eels was 160.7 t (range: 133 to 205 t) compared to a mean catch of 80.8 t (range: 53 to 99 t) between 2005 and 2009, a reduction of 50% relative to between 1997 and 2002 (Figure 12). For both the yellow and silver eel phases, the total reduction of the catch between 2005 and 2009, compared to the 1997 to 2002 reference period, was 53% in the Québec waters of the St. Lawrence River (Table 5, DFO, 2010).

Table 5. Summary of the reductions in absolute mortality and in the rate of mortality of yellow and silver American Eels between 2004 and 2009 relative to between 1997 and 2002. For Ontario, the declines in absolute mortality and mortality rate refer to silver eel mortality associated with fishing in Lake Ontario and from hydro-dams. For all the other regions, the mortalities and rates are from fisheries only. An average weight of 1 kg was assumed for eels in Ontario (from DFO, 2010).

NFBZ Region	Change in absolute mortality (weight)	Change in mortality rate (number)
NFBZ10-Ontario	-90 %	-50%
NFBZ10-Québec	-53%	
NFBZ1-Gulf	+46%	-20%
NFBZ1-Maritimes	-27%	
NFBZ8-Newfoundland	-5%	
Total	-22%	

The reduction in the number of silver eels harvested in 2009 (exploitation rate estimated at 10.5% in 2010) in the lower St. Lawrence estuary fishery as a direct result of the buyout of silver eel fishery licences was calculated using the significant negative linear regression of catches of silver eel versus year for the period 1998 to 2008 (r^2 = 0.94). In 2009, the predicted catch in the absence of buyout would have been 57.5 t compared to a realized catch of 23.8 t, or a saving of 33.7 t of silver eel spawners. For this portion of the fishery, the reduction in absolute mortality between 2008 and 2009 was 58.6% (DFO, 2010).

<u>NFBZ9 Lower St. Lawrence (eastern Québec)</u>

Indices of Juvenile Abundance

An index of eel abundance, consisting of a partial count of eels ascending a natural barrier, has been developed for the Sud-Ouest River (Figure 13), tributary to the estuary of the St. Lawrence River on its south shore (Verreault and Tardif, 2009; Caron *et al.*, 2006). Eels ascending the falls have a mean length of about 250 mm and ages ranging from 1 to 11 yr with most eels being 2 to 6 yr old. Because partial counts over the short time series are highly variable and composed of several cohorts, an index of year class strength (YCSI) has been calculated based on the age structure of sampled eels to show the abundance. The YCSI indicates a significant negative trend in abundance in relative year class strength for the 1998 to 2005 recruiting year classes (spawning year) (Figure 14); the YCSI decreased from 1.37 in 1999 to 0.88 in 2005 (~36%, G. Verreault, MRNF, pers. comm., 2011).

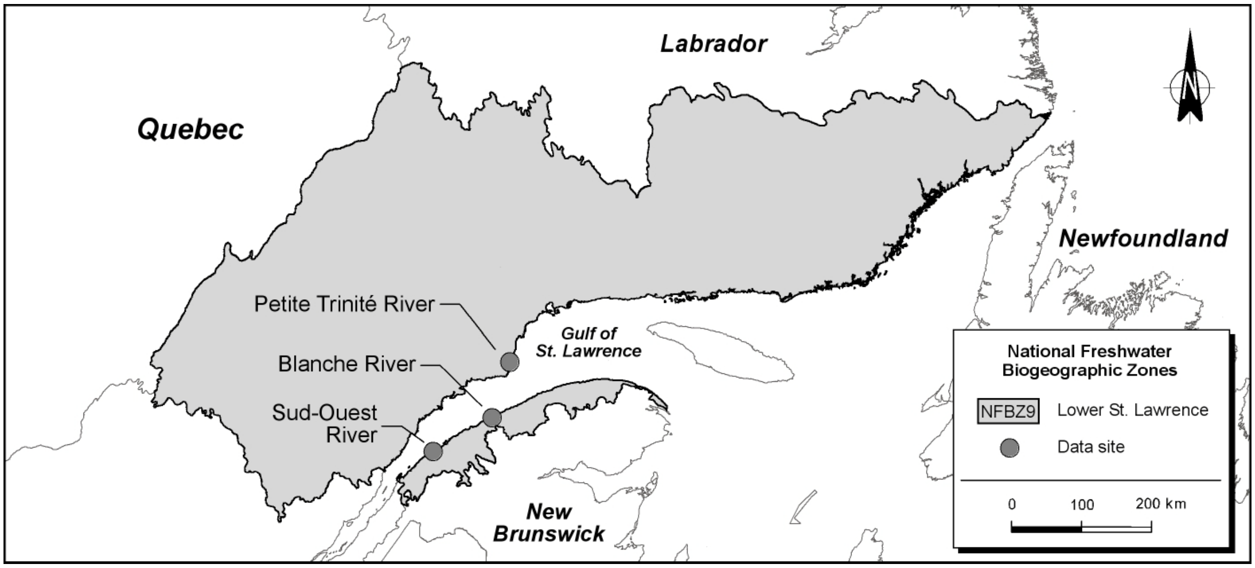

Figure 13. Data sites within the National Freshwater Biogeographic Zone NFBZ9, Lower St. Lawrence River.

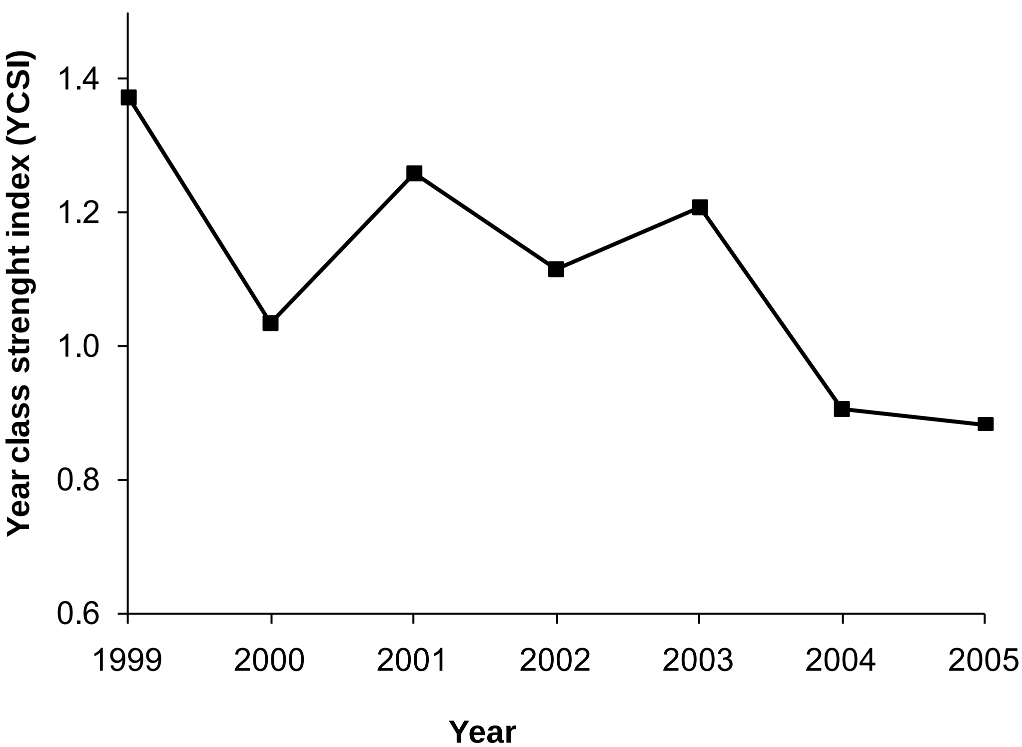

Figure 14. Relative year class (spawning year) strength (YCSI) of immature American Eels for the Sud-Ouest River, lower estuary of the St. Lawrence River (1995-2010, from G .Verreault, MNRF. The correlation coefficient is significant at P = 0.026).

Gulf of St. Lawrence and Atlantic Region

<u>NFBZ1 Maritimes (New Brunswick, Nova Scotia, Prince Edward Island, and the central and southern parts of Québec's Gaspé Peninsula)</u>

Index of Recruitment

A recruitment index is available for NFBZ1 (Figure 15), based on elver catches and counts in the East River, Chester (Nova Scotia). The river is also the site of an authorized elver fishery, downstream of the elver trap sites (Bradford *et al.*, 2010). The East River abundance series began in 1996, but gaps are present between 2003 and 2006. The index shows wide annual fluctuations in elver recruitment but no trend is apparent (Figure 16a; Bradford *et al.*, 2010). The total run of elvers surpassed the time-series mean in 1996-1997, 2002, 2008 and 2009. The highest run was in 2009, with elver counts of 1.9 million. Elver run sizes since 2008 were within the range reported for 1996-2002 or higher (1.9 million in 2008, low of 0.6 million in 2009) (Bradford *et al.*, 2010; DFO, 2010). The strength of the elver run to East River, Chester (NS) is positively correlated with the CPUE from the commercial elver fishery (Figure 16b; Bradford *et al.*, 2010), but these data are not currently used to monitor the fishery.

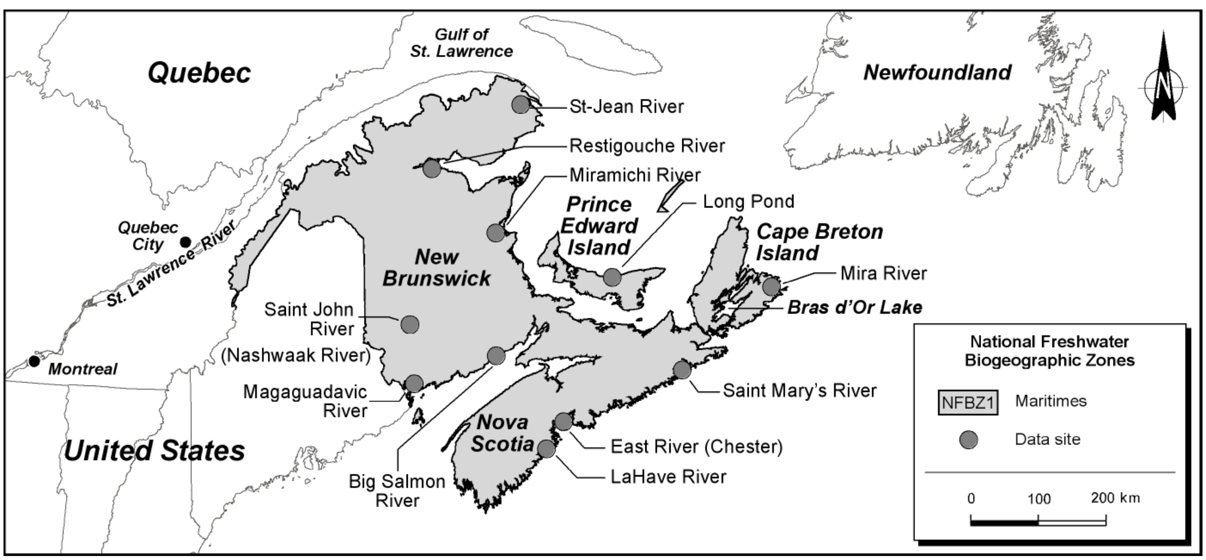

Figure 15. Data sites for American Eel within National Freshwater Biogeographic Zone NFBZ1, Maritimes.

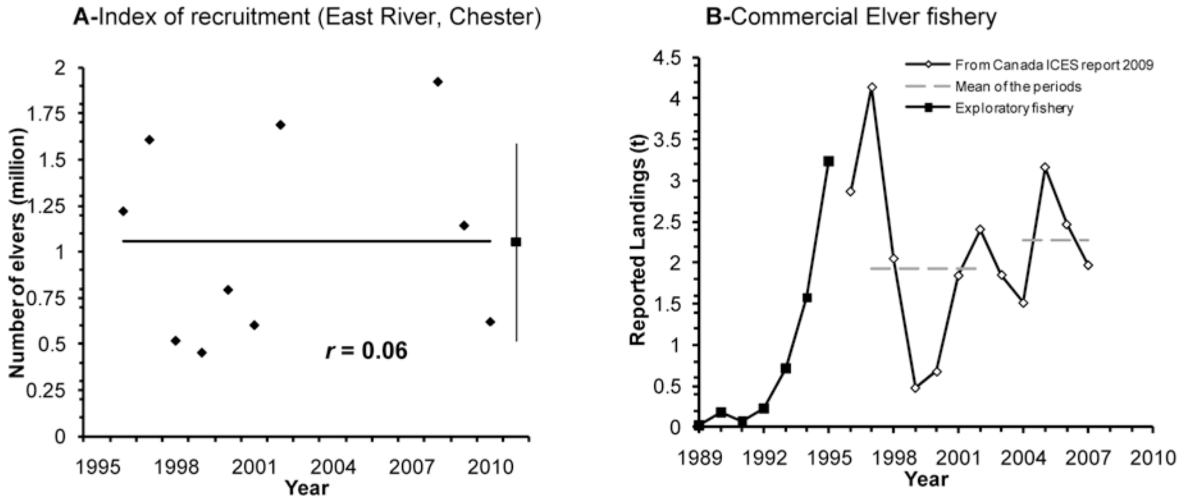

Figure 16. (A) Index of recruitment based on number of American Eel elvers in the East River, Chester (NS), 1996 to 2010. The solid square symbol is the average (± one std. dev.) of the time series. (B) Reported landings (t) of elvers in the Scotia-Fundy Region for 1989-2008. During 1989 to 1995, the elver fishery was an exploratory fishery. Since 1996, the elver fishery has been managed under an integrated fisheries management plan.

Whereas no recruitment data were available in Québec waters of NFBZ1 besides sporadic evaluation of elver abundance (Dutil *et al.*, 1989; Vladykov, 1955, Vladykov and Liew, 1982; Vladykov, 1966; Dolan, 1975), a new program monitoring elvers has started in 2009 in the Saint-Jean River estuary (M. Dionne, MRNF, pers. comm., 2011), whose watershed is included in NFBZ1 (Maritimes; Figure 15). No trend can be detected so far because only two years of monitoring have been completed. From mid-May until end of July, the monitoring uses fyke net traps to capture elvers. To date, abundance peaks at the end of May, and the length and weight of elvers progressively decreases through the season (M. Dionne, MRNF, pers. comm., 2011). Elver abundance is significantly affected by environmental factors such as the water temperature and tide height (M. Dionne, MRNF, pers. comm., 2011).

Indices of Juvenile and Yellow Eel Abundance

For NFBZ1, the longest fisheries-independent abundance indicators for the American Eel come from electrofishing surveys for salmonids. Although salmonid surveys are not optimal for determining abundance in eels (owing to habitat use differences), indices of juvenile and yellow eel abundance of eels in freshwater in the Miramichi River (1952-2009) and the Restigouche River (1972-2009), New Brunswick indicate highly variable trends in both rivers (Figure 17). Both indices indicate that the average abundance in the 2000s has generally been higher than the abundance in the 1980s and higher than the long-term mean, but instances of below average abundances have also been observed (Figure 17). In the Miramichi River, eel densities varied in the 1950s and 1960s, peaked in the early 1970s, and then declined to a minimum by the late 1980s until 1995, when they began to increase again (the overall trend in abundance shows no significant increase or decrease, Figure 17a). Mean estimated densities in the Restigouche River were 0.50 eels per 100 m² during the 1972 to 2009 period, peaked in 2001 and 2002, and declined overall to 0.80 eels per 100 m² during the 2005 to 2009 period (Figure 17b; DFO, 2010). Some of the highest, but most variable, abundances in the early 2000s have driven a significantly positive trend in abundance over the entire 37 years (Figure 17b).

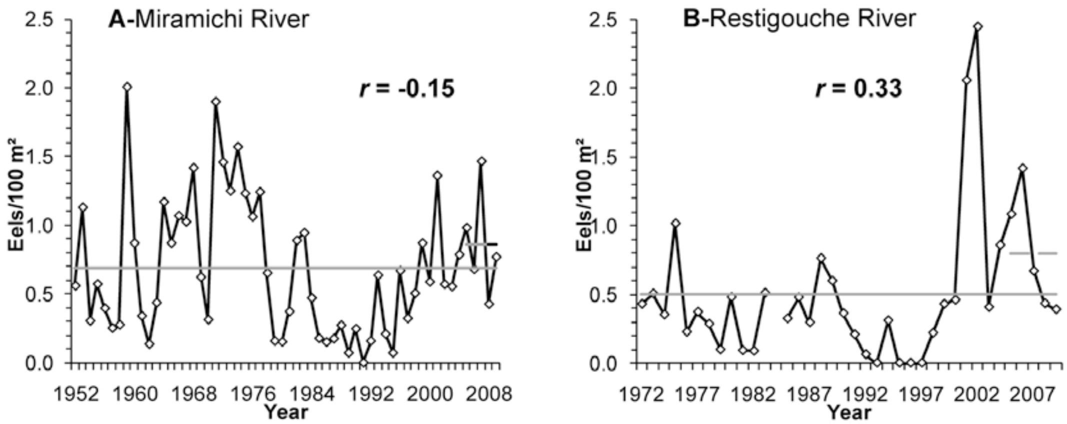

Figure 17. Abundance indices (fish per 100 m²) of immature American Eel (yellow eel stage) from electrofishing surveys in two rivers of the southern Gulf of St. Lawrence. The solid horizontal line represents the time-series mean and the dashed line the recent five-year mean (from D.K. Cairns, DFO): (A) Miramichi River (1952-2009); (B) Restigouche River (1972-2009). For the Miramichi River, the database 1952-1969 was reconstructed using original paper records (Cairns *et al.* 2008). Some densities calculated from the new database differ from those reported by Cairns *et al.* (2007). The negative trend in (A) is not significant ($P = 0.22$) while the positive trend in (B) is statistically significant ($P = 0.049$).

Electrofishing indices from salmonid surveys are also available for four rivers in the Scotia-Fundy Region flowing into the Bay of Fundy and the Atlantic Ocean (Figure 18). The Nashwaak River (1991-2009) and the Big Salmon River (1996-2009) are located in New Brunswick, whereas the St. Marys River (1985-86; 1995-2009) and LaHave River (1995; 1997; 2000-2009) are located on the Atlantic coast of Nova Scotia (Bradford, 2010). Time series from New Brunswick electrofishing surveys show no significant overall trends in abundance, but suggest declines since the early 2000s (Figure 18a; Bradford, 2010; DFO, 2010). By contrast, in the two Nova Scotia sites, the electrofishing series suggest significant negative trends in abundance, declines of about 75% in the St. Marys River over the last 25 years (approximately one generation) and 86% in the LaHave River over the last decade (Figure 18b).

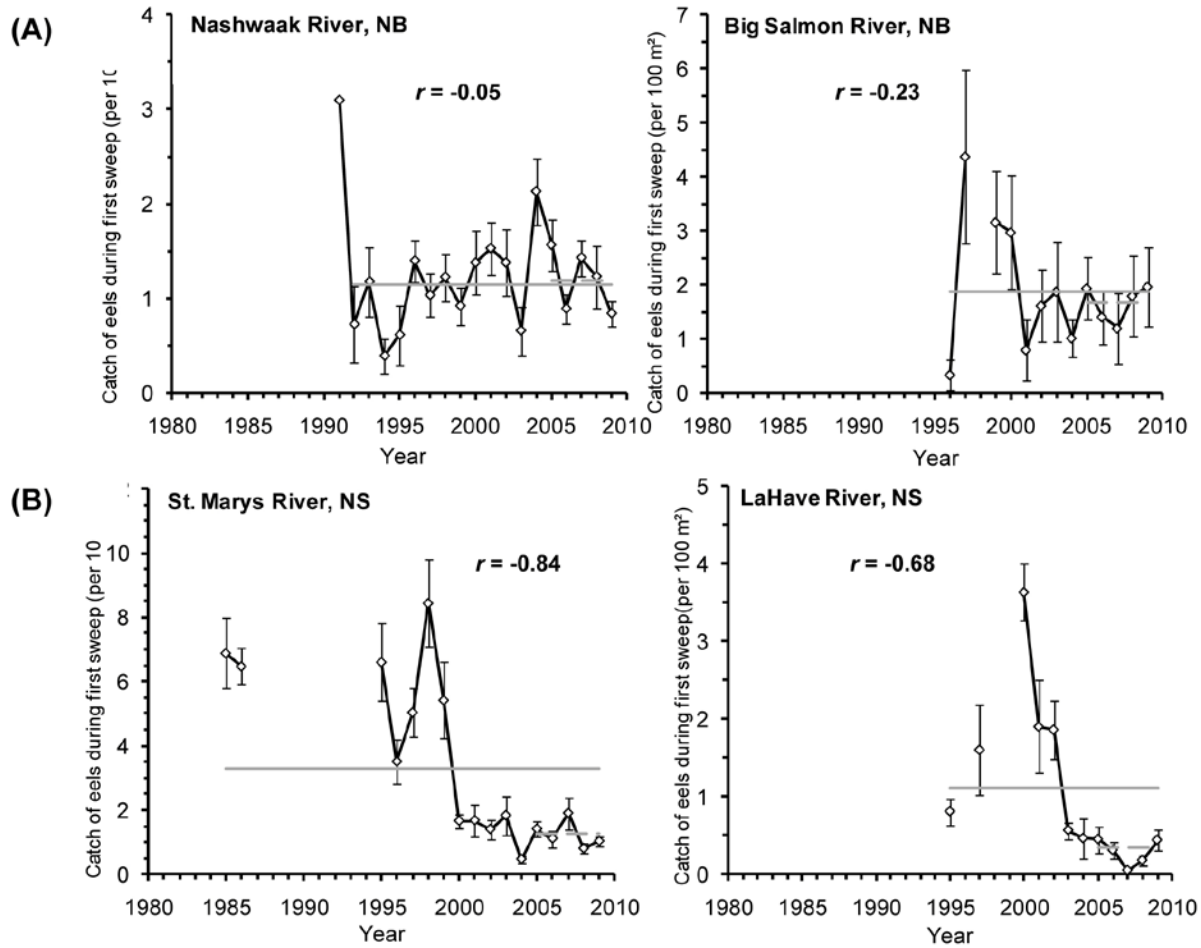

Figure 18. Annual indices of abundance (catch of eels per 100 m² during the first sweep; mean ± one std. dev.) of immature American Eel (yellow eel stage) from electrofishing surveys in (A) New Brunswick [Nashwaak River (1991-2009), Big Salmon River (1996-2009)], and (B) in Nova Scotia [St. Marys River (1985-86; 1995-2009), LaHave River (1995; 1997; 2000-2009)], rivers that flow into the Bay of Fundy and the Atlantic Ocean. The solid horizontal line represents the time-series mean and the dashed line the recent five-year mean (from R. Bradford, DFO). The overall trends are not significant in the New Brunswick rivers, but highly significant in the Nova Scotia rivers (maximum P = 0.014).

Indices of Yellow Eel Abundance

Between 2005 and 2008, American Eel densities (at the yellow stage) were estimated at 27 sites in brackish and saltwater habitats of the southern Gulf of St. Lawrence by surveys conducted on a glass bottom boat (GBB) equipped with underwater lights (Hallett *et al.,* 2010). Eels were counted at night during zigzag transects that ran across bays and estuaries. Visibility conditions typically limited survey depth to 2 to 2.5 m depth. Mean densities (and 95% confidence limits), estimated from a bootstrap routine, were 34.1 (23.0-45.5) eels/ha for Gulf New Brunswick, 40.9 (24.8-57.3) eels/ha for Gulf-Nova Scotia, and 155.7 (86.3-235.9) eels/ha for Prince Edward Island. Uncertainties in standing stock estimates, however, arise from sampling error in density estimates as analyzed by the bootstrap procedure, and limited knowledge regarding eel densities in waters deeper than 2.5 m and in the proportion of eels that remain buried in the substrate during the night (Hallett *et al.*, 2010). Therefore, these estimates cannot be used for abundance trends in NFBZ1.

Monitoring of the American Eel in the St. Jean River, estuary, and associated lakes in the eastern Gaspé Peninsula, Québec (NFBZ1) (Thibault *et al.* 2007ab; Caron *et al.*, 2009) in 2005, and 2008-2009 has shown that eels are present in substantial numbers, and with a range of ages (2 - 24 years). This suggests that recruitment to the area is continuing, although no time series of abundance is available. Catch per unit effort of adult eels (yellow and silver) in the estuary (from 17.4 to 19.3 eels/ha) are more elevated than in surrounding lakes (from 2.3 to 9.5 eels/ha) (Caron *et al.*, 2009; M. Dionne, MRNF, pers. comm. 2011). Growth is faster in the St. Jean River estuarine habitat and lower in the freshwater lakes, as reported for other watersheds in the NFBZ9 (Petite Trinité River; Fournier and Caron, 2005; Cascapédia River; Caron *et al*. 2007; York River, unpublished data; cited in Caron *et al.,* 2009) and for the species across its range (Jessop *et al.*, 1998). Based on mark and recapture experiments conducted in the spring (mid-May to mid-June) between 2003 and 2009, estimated abundance was 13,481-40,021 eels in the Saint-Jean River system (1 124 km²; 115 km of main river; 5.8 km² of estuary; 12 lakes of more than 5 ha) (Caron *et al.*, 2009; M. Dionne, MRNF, pers. comm. 2011).

Landings

Beside the elver fisheries, most eels harvested in NFBZ1 are yellow eels, but some silver eels are harvested as well. In DFO Gulf region, reported landings averaged 127.9 t between 1997 and 2002 and 187.0 t between 2004 and 2008, an increase of 46.2%. Short fishery-dependent abundance indices of catch per unit effort in three fisheries in Gulf of Nova Scotia and PEI showed a generally increasing trend in eel abundance in the southern Gulf of St. Lawrence during the 1997 to 2008 period (Figure 19). Mean CPUE in Gulf of Nova Scotia was 1.96 kg per fyke net-day and 3.43 eels per spear-hour in 1997 to 2009, versus 2.56 kg per fyke net-day and 4.63 eels per spear-hour between 2005 and 2009. Mean CPUE in PEI was 0.82 kg per fyke net-day between 1996 and 2009 and 1.12 kg per fyke net-day between 2005 and 2009 (D. Cairns pers. comm., 2010; DFO, 2010).

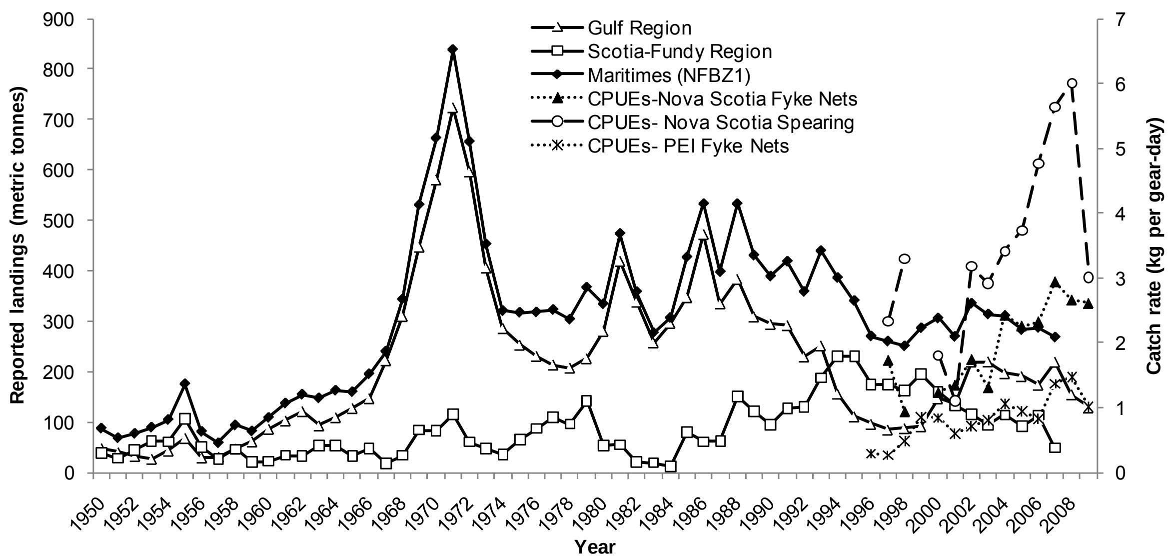

Figure 19. Commercial indicators of American Eel abundance within NFBZ1: Eel landings in the Maritimes (1950-2007), in the Southern Gulf of St. Lawrence Region (1950-2009), in the Scotia-Fundy Region (1950-2007), and catch per unit effort indices in three commercial eel fisheries of the southern Gulf of St. Lawrence (1996 to 2009) (from D.K. Cairns, DFO).

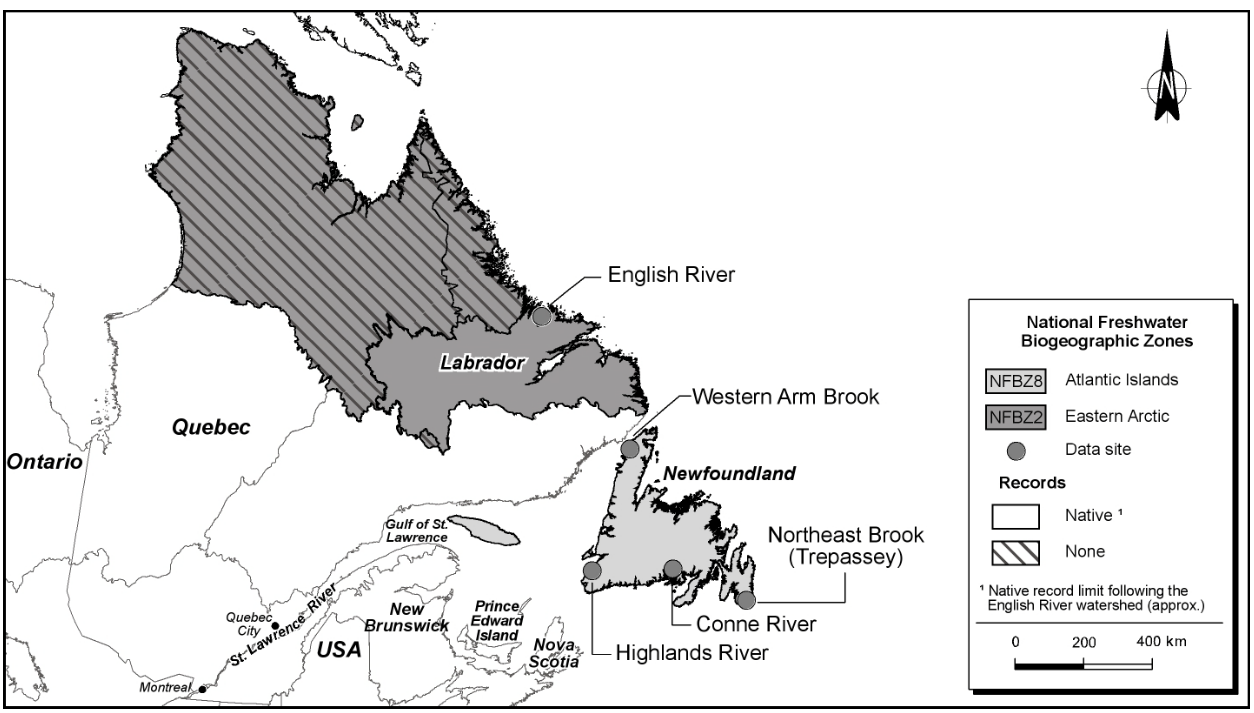

Figure 20. Data sites for American Eel within National Freshwater Biogeographic Zones NFBZ8 (Atlantic Islands) and NFBZ2 (Eastern Arctic).

In the Scotia-Fundy Region, reported landings of yellow and silver eels averaged 142 t between 1997 and 2002 and 103 t between 2004 and 2007, a decrease of 27% (Figure 19). Decreased landings may be confounded by issues of underreporting in 2004 to 2007 (DFO, 2010). Records after 2007 are not yet available; however, there are no indices of abundance based on fisheries in the DFO Maritimes Region that would allow assessment of present status relative to that of the 1980s (DFO, 2010).

Mi'kmaq elders from the Bras d'Or Lakes area believe that the lakes cannot sustain a commercial harvest as poor catches have been reported after the opening of a commercial fishery 10 years ago (CEPI, 2004; 2006). To protect the food fishery, elders recommended a complete closure of the commercial harvest (CEPI, 2004 and 2006).

The only sustained fishery for American Eel elvers and/or glass eels in Canada occurs in the Scotia-Fundy area of the Maritime Provinces (NFBZ1). Landings of elvers increased by 18% during 2004 to 2007 relative to the average landings during 1997 to 2002 (Figure 16b). In 1989, the first two licences were issued to fish experimentally for American Eel elvers in the lower reaches of rivers draining into the Bay of Fundy in Nova Scotia and New Brunswick (Jessop 1998b). Within a geographic region, the elver catch depends on elver abundance, fishing effort, and elver catchability (availability, fishing efficiency depending on gear type), and the distribution of elvers through the run. In 1996, nine commercial licences were issued. No new licences have been issued since 1998 (Bradford, 2010). The option for licence holders to apply for a 30% quota increase was removed (to be reviewed annually) and there was a 10% across the board quota reduction for each licence holder; the individual licence holder quota is now 900 kg. An additional 100 kg (10% of reduced quota), however, can be requested if that additional catch is destined for stocking in Canadian waters (Bradford, 2010; DFO, 2010).

NFBZ8 Atlantic Islands (Newfoundland) and NFBZ2 Eastern Arctic (Labrador)

Indices of Juvenile and Yellow Eel Abundance

Available fisheries-independent indices of eel abundance in Newfoundland are composed of electrofishing surveys conducted for salmonids. In Newfoundland, two electrofishing time series covering the period of the 1980s ended in the mid- to late 1990s: Northeast Brook (1984-1996); Highlands River (1980-1981; 1993-1999) (Figure 21). American Eel abundance was relatively stable within Northeast Brook from 1984 to 1990 (Figure 21a). With the exception of 1986, which had low eel abundance, average catches of eels ranged from a low of 4.5 per station to a high of 8 per station. The data for the post-1990 period, with the exception of 1992, indicate lower overall abundance of eels in the electrofishing stations. Average eel catches in this period, excluding 1992, ranged from 1 to 2.8 eels per station. The overall pattern from 1984-1996 is for a significant negative trend in abundance and an estimated decline of 88%. For the Highlands River, data are discontinuous, with only two sampling years in the early 1980s, and indicate a significant negative trend in abundance as observed in Northeast Brook; between 1980 and 1999 this index of eel abundance has declined by about 95%.

(Figure 21b). More recent time series are available for two rivers at salmonid counting fences: Conne River (1986-2008), and Western Arm Brook (1994-2008). These two series show variable counts of eels; whereas the average annual count for 2003-2008 years was greater than the long-term average at Conne River, the opposite was true for Western Arm Brook and neither trend was statistically significant (Figure 22).

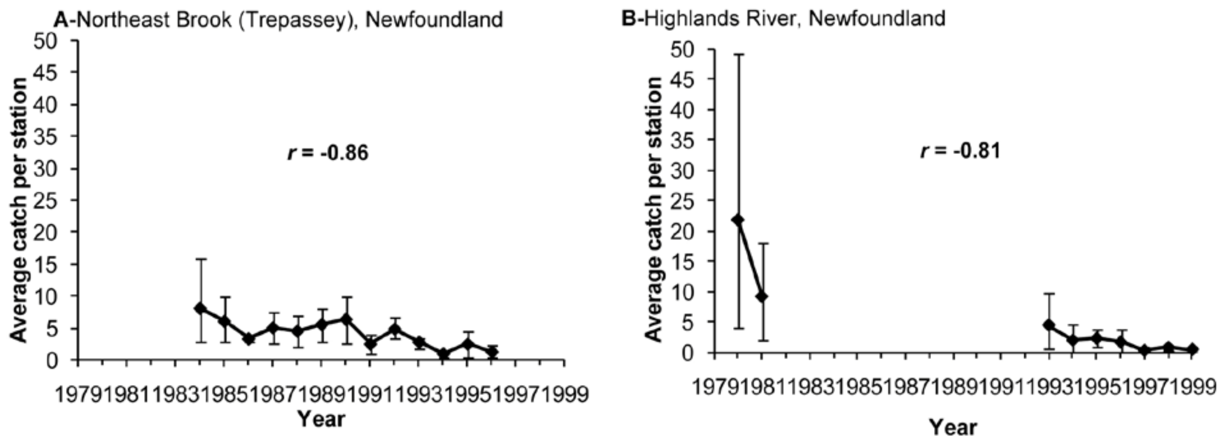

Figure 21. Mean number of immature American Eels (yellow eel stage) caught (± 95% confidence intervals) per 100 m² in two rivers of Newfoundland, estimated from electrofishing surveys (from K.D. Clarke, DFO): (A) Northeast Brook (1984-1996); and (B) Highlands River (1980-1981; 1993-1999). The negative trends associated with both time series are significant (maximum P = 0.008).

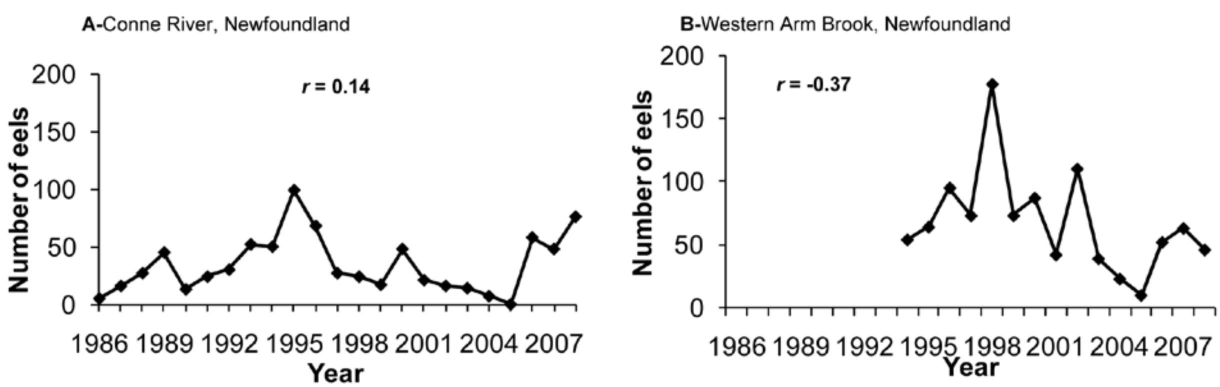

Figure 22. Counts of immature American Eel (yellow eel stage) at the salmonid counting facilities in Newfoundland (from K.D. Clarke, DFO) (A) Conne River (1986-2008), and at (B) Western Arm Brook (1994-2008). Neither abundance trend is statistically significant (minimum P = 0.18).

In Labrador (NFBZ2), data regarding eels are scarce. Electrofishing and fyke net studies have been conducted throughout the English River watershed over six years (1999-2005). This watershed flows in an easterly direction into Kaipokok Bay. The lower portion of the catchment is dominated by English River Pond (Clarke *et al.* 2004). The only eels captured since 1999 (N = 3) were in the lower mainstem of the system in 2004. This section of the river is characterized by braided channels, created by islands, with a predominance of cobble/gravel substrate (Clarke *et al.* 2004). Given this, while eels certainly can and do inhabit this area, they appear to be rare (K.D. Clarke, DFO, pers. comm., 2010). These captures, however, extend the known Canadian distribution of the species farther north than previously thought (Figures 3 and 20). Indeed, the English River is about 120 km north of Hamilton Inlet-Lake Melville (Scott and Crossman, 1973).

Landings

Landings during the 1997 to 2002 period averaged 62 t. During the recent five-year period (2005 to 2009), landings averaged 59 t, a decrease of 5% from the average of the 1997 to 2002 period (Figure 23). The landings of 33 t in 2009 represent a 47% reduction from the 1997 to 2002 average. This reduction may be mostly attributable to market conditions rather than abundance (DFO, 2010, Figure 23). Eel price peaked in 2007 at around $2.70 per pound, sharply declined in 2008 to around $1.70 per pound, and slightly increased in 2010 compared to 2008 and 2009 (A. Firminger, South Shore Trading Co., pers. comm., 2011).

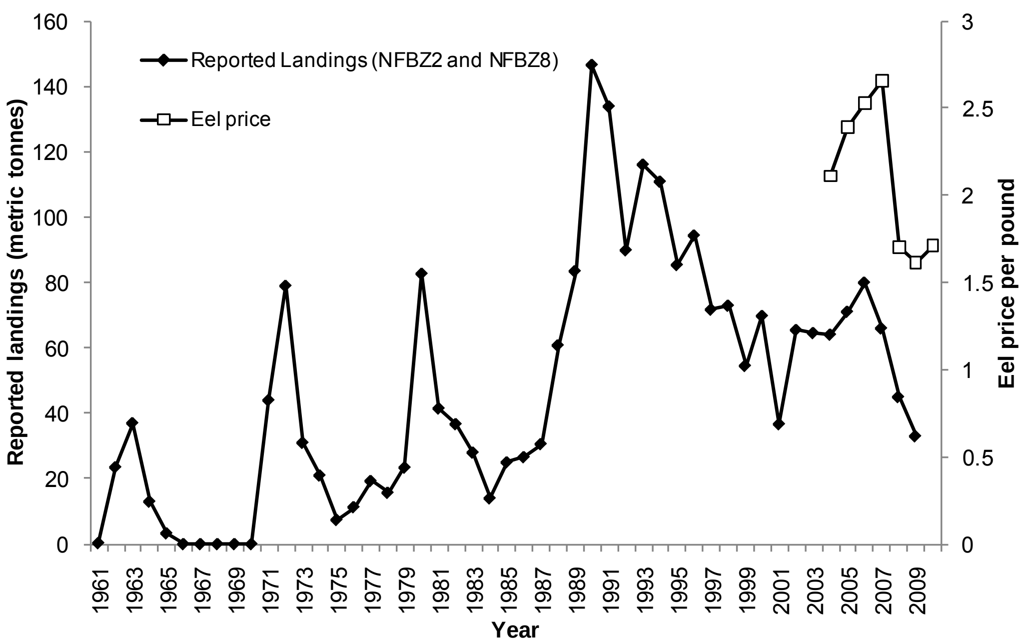

Figure 23. Reported landings (t) of American Eels in Newfoundland (NFBZ8) and Labrador (NFBZ2) from 1961 to 2009, and eel price in Newfoundland from 2004 to 2010 (from A. Firminger, South Shore Trading).

All NFBZ

Throughout its range, all continental life stages of the American Eel are harvested (Figure 24). The total North American harvest (Canada-U.S.; Figure 25) increased from an average of 1,430 t annually between 1950 and 1955 to an unprecedented peak of 3,145 t in 1979. By the early 1990s, North American harvests began to decline. By 2005, eel harvests fell rapidly to less than 975 t. In 2009, landings of less than 600 t had been reported in North America (Figure 25). This decline occurred despite sustained high prices; well above the long-term mean and strong demand in European and Asian markets (Casselman and Marcogliese, 2007). Overall trends in Ontario and Québec commercial harvests parallel those of Canada and the United States (Figures 24 and 25). There is evidence that overfishing has occurred for some time in other parts of the species' range such as Delaware and Chesapeake Bay (Clark, 2009; Weeder and Uphoff, 2009).

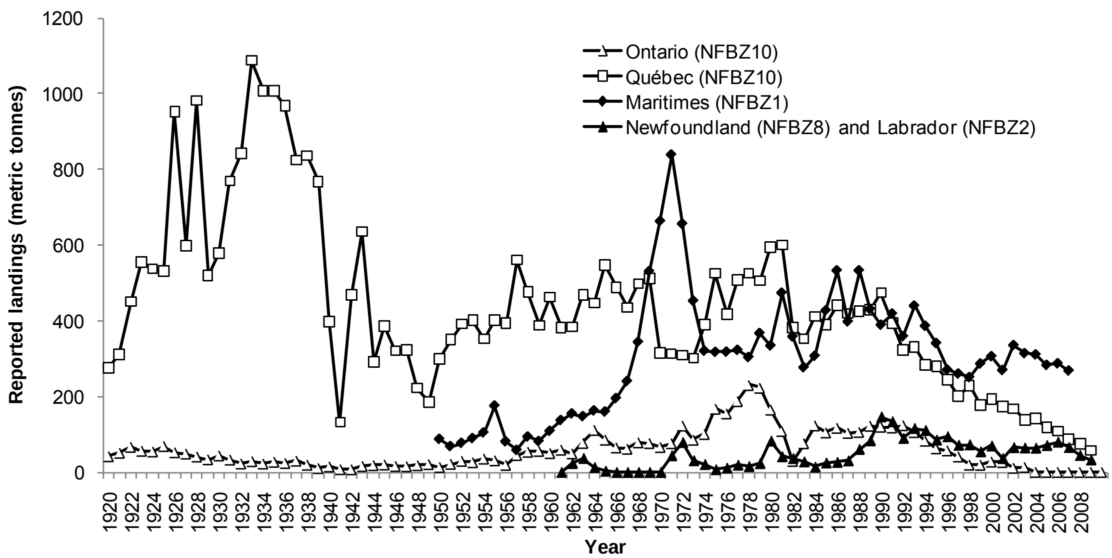

Figure 24. Reported landings (t) of American Eel for each NFBZ: Ontario (NFBZ10; 1920-2009); Québec (NFBZ10; 1920-2009); Maritimes (NFBZ1; 1950-2007), Newfoundland (NFBZ8; 1961-2009) and Labrador (NFBZ2; 1961-2009).

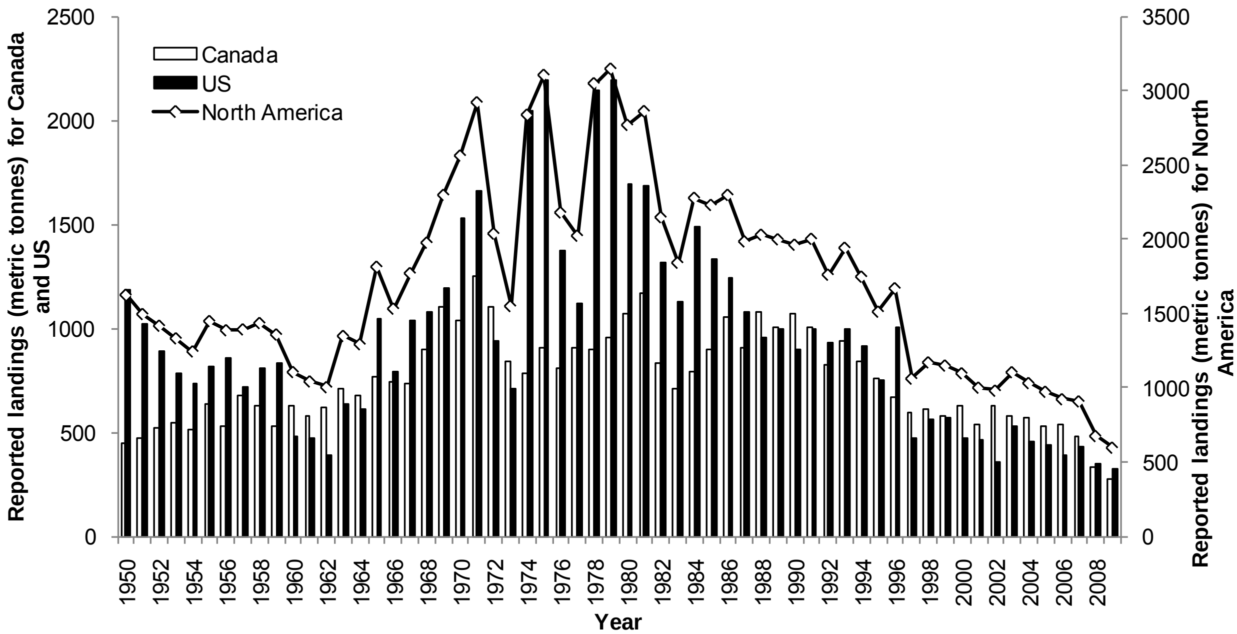

Figure 25. Reported landings (t) of American Eel in Canada and in the United States (from G. Nesslage, ASMFC), and pooled for North America (1950-2009).

Figure 26. Difference in total length at maturity between female American Eels during downstream migration: (A) naturally produced silver eel, and (B) stocked silver eel (from G. Verreault, MNRF).

With the objective to increase escapement in the long term, a reduction in mortality rate, expressed as a proportion killed, has been suggested to be an appropriate measure of performance for the American Eel (DFO, 2010). For all of eastern Canada, the reduction in mortality is estimated to be 22% during the recent five-year period relative to the 1997 to 2002 base period (Table 5). The degree of mortality reduction is, however, non-uniform and concentrated in NFBZ10 because the change in absolute mortality (weight) in Ontario and Québec reached -90% and -53%, respectively, whereas it has reached only -27% in the Maritimes, and -5% in Newfoundland. In the Gulf Region, an increase in absolute mortality has been reported (+46%).

The greater mortality rate for the upper St. Lawrence and Lake Ontario (-53% in mortality rate by number and -90% in absolute mortality by weight) system is not due to changes in turbine mortality rate, but due to closure of fisheries, lower numbers of fish in this system and a higher proportion of eels now being produced below Moses-Saunders Power Dam (DFO, 2010). For other NFBZs, the estimated lower mortality rate is due to regional fishing reductions. Because the short-term goal was to reduce eel mortality from all sources by 50% relative to the 1997 to 2002 average, the current 22% reduction in mortality observed has not met the current management goal.

Glass eels (and/or elvers) are harvested by fishers in NFBZ1 and the United States. Although glass eels from Canadian fisheries are the only available source of glass eels for stocking efforts in NFBZ10, the vast majority of Canadian glass eel harvest is exported, primarily to Asia (Jessop, 1997; ASFMC, 2000; Weeder and Uphoff, 2009).

Summary of Changes in Eel Abundance

Using mean generation times of 22 years for freshwater eels and nine years for eels that reside in salt water, and considering 2011 as “present,” three generations ago encompasses the period 1945-2011 for freshwater eels and 1984-2011 for eels reared in salt water. Eight Canadian and one U.S. data series are available for comparisons over three generations (Table 6). Three of the Canadian series are landings. From the five independent-fisheries series, three are from NFBZ10 (indices from Moses-Saunders total counts and Bay of Quinte trawl; electrofishing from eastern Lake Ontario), and two are from NFBZ1 (electrofishing in Restigouche and Miramichi Rivers). Finally, the single U.S. series is landings.

Table 6. Mean values of American Eel data series for indices of abundance for the period of three generations (approximately 65 years) by National Freshwater Biogeographic Zone (NFBZ). Series include scientific indices and landings data. Three of the Canadian series are landings, three are indices from NFBZ10 (indices in Moses-Saunders Dam total counts; indices in Bay of Quinte trawl; electrofishing in eastern Lake Ontario), and two are from research electrofishing in NFBZ1 (Restigouche and Miramichi Rivers). The single U.S. series is landings. All indices except for landings and the St. Nicholas experimental trap fishery are for juvenile life stages.

Parameter and NFBZ	1950s to 1970s		1980s		1990s		2000s		Percent change to 2000s		
	Years	Value	Years	Value	Years	Value	Years	Value	From 1950s-1970s	From 1980s	From 1990s
Landings (t), Ontario (NFBZ10)	1970-1979	142	1980-1989	106	1990-1999	75	2000-2009	8	-94.5	-92.6	-89.6
Silver eel landings (t), Québec (NFBZ10)	1970-1979	387	1980-1989	421	1990-1999	268	2000-2009	104	-73.0	-75.3	-61.1
Moses-Saunders Dam total count (NFBZ10)	1974-1979	726,024	1980-1989	608,044	1990-1999	41,257	2000-2009	11,948	-98.4	-98.0	-71.0
Bay of Quinte trawl index (NFBZ10)	1972-1979	1.20	1980-1989	0.90	1990-1999	0.37	2000-2009	0.01	-98.8	-98.4	-96.2
Lake Ontario electrofishing (NFBZ10)			1984-1989	80	1990-1999	27	2000-2009	2		-94.8	-91.4
Beauharnois Dam total passage (NFBZ10)					1994-1999	14,482	2000-2009	51,744			257.3
Chambly Dam total passage (NFBZ10)					1998-1999	6,780	2000-2009	1,280			-81.1
St. Nicolas exp. fall captures (NFBZ10)	1975-1979	295	1980-1989	179	1990-1999	195	2000-2009	210	-28.9	17.0	7.5
Landings (t), NFBZ1 NFBZ8 and NFBZ2 [A]	1970-1979	481	1980-1989	450	1990-1999	438	2000-2007	361	-24.9	-19.7	-17.5
Prince Edward Island commercial CPUE (NFBZ8)					1996-1999	0.47	2000-2009	0.83			75.2
Restigouche electrofishing densities (NFBZ1)	1972-1979	0.41	1980-1989	0.40	1990-1999	0.16	2000-2009	1.02	149.2	153.8	537.5
Miramichi electrofishing densities (NFBZ1)	1952-1979	1.08	1980-1989	0.37	1990-1999	0.37	2000-2009	0.82	-24.1	121.6	121.6
Naswaak open electrofishing counts (NFBZ1)					1991-1999	1.18	2000-2009	1.30			10.5
Conne counting fence (NFBZ8)			1986-1989	23.25	1990-1999	40.40	2000-2008	32.00			-20.8
Western Arm Brook counting fence (NFBZ8)					1994-1999	89.33	2000-2008	52.44			-41.3
Landings (t), U.S.	1970-1979	1,589	1980-1989	1,291	1990-1999	809	2000-2009	421	-73.5	-67.4	-48.0

A: There is no eel fishery in NFBZ9.

Out of the eight Canadian series allowing percent change evaluation from the 1950s through to the 2000s, seven show negative values (from -17.5% to -96.2%), including data from all areas (Table 6). All of the four landings series showed negative change. Four of the five survey indices were negative. Comparisons from the 1980s to the 2000s, and from the 1990s to the 2000s, show mixed results. Seven of 10 series showed negative change from the 1980s to the 2000s, and 10 of 16 series showed negative change from the 1990s to the 2000s. The most reliable long-term series are the fisheries-independent surveys in the upper St. Lawrence River and Lake Ontario system (NFBZ10), and the southern Gulf of St. Lawrence (NFBZ1; Table 6). Despite varying changes between early and recent periods for these series, very steep declines (>90%) are reported in Ontario (NFBZ10); declines are also reported in the St. Nicolas experimental trap (-28.9%; NFBZ10). The two electrofishing series data from the southern Gulf (NFBZ1), however, reported relatively stable indices of abundance (Table 6). Shorter time series data for other sites in New Brunswick, Nova Scotia, and Newfoundland, show either stable values post-1995 (New Brunswick), stable values in some areas and moderate declines in others (Newfoundland) to steep declines (Nova Scotia, Figures 18, 21). While the clear majority of indices strongly suggest consistent and steep declines both in juvenile and adult indices of abundance within the Lake Ontario - upper St. Lawrence River area, trends across the rest of the range are highly variable both temporally and spatially.

Several factors limit the reliability of these series to indicate changes in American Eel abundances. First, some of the electrofishing surveys are designed for salmonid assessments and are not likely ideal for eels given differences in body size and habitat use between these taxa. Second, fisheries landings are an indicator of minimum biomass, because landings can never exceed biomass. Beyond that, landings have limited value as indicators of abundance. Landings are influenced by regulations, price per kg (market), alternative opportunities in fishing and other employment, and by changing gear efficiencies (de Lafontaine *et al.*, 2009b). North American Eel fisheries have been strongly affected since 1970 by market factors, and by tightened regulations. Nevertheless, using the Canadian Consumer Price Index, the price per kg of eels in inflation-adjusted dollars has increased by roughly a factor of 1.8 between 1980 and 2004 (Cairns *et al.*, 2008), and steep declines in landings occurred despite sustained high prices (Casselman and Marcogliese, 2007).

Rescue Effect

If the American Eel became extirpated or severely depleted in one or more NFBZs in Canada, the possibility of rescue in one area will depend on its geographic proximity to eels from others areas and the status of eels across areas. Rescue of eels in the Lake Ontario/upper St. Lawrence river area is, under current conditions, more likely from the rest of the range than vice versa or by rescue of eels from areas in the United States. Conversely, rescue of eels within the eastern portions of the Canadian range is perhaps more likely to occur from eels from areas within the United States given the relatively poor status of eels in the Lake Ontario/upper St. Lawrence river area and the broader distribution of eels across areas of the United States. Across the entire range in Canada, rescue of America Eels can be considered under the following two scenarios:

1) Eels become extirpated or severely depleted in Canada, but there is no substantial change in American Eels from the United States. In this scenario, young eels from the Sargasso Sea, primarily from the United States parentage, could continue to colonize the whole continental rearing area, including eastern Canada. External eels would thus probably “rescue” the species in Canada. Whether or not such rescue would return eels to sustainable levels of abundance is, however, unknown. Alternatively, egg production in the Sargasso Sea might be substantially reduced due to the lack of female Canadian-reared spawners, so total recruitment of young eels to Canada and elsewhere could be much lower than normal, and recruitment toward range limits could be reduced if migration is density-dependent.

2) Eels become extirpated or severely depleted in both Canada and the United States. In this scenario, egg deposition in the Sargasso Sea would be drastically lowered, and recruitment of young eels would likely fall to low or negligible levels in all rearing areas, including Canada. The rescue effect for eels in Canada would therefore be limited, particularly if migration is density-dependent.

If American Eels are truly panmictic, progeny of eels that grew and began maturation in Canadian freshwaters could colonize the full range of the species in an apparently random fashion. Therefore, in the case of declines in abundance (severe reduction or extirpation) that affects both Canadian and the United States eels and all else being equal, reproductive output from Canadian-reared eels would be redistributed in the United States as much as reproductive output from United States-reared eels would be redistributed in Canadian areas. The results of Bernatchez *et al.* (2011), however, suggest that it may not be possible to rescue genetically controlled phenotypes across the entire species distribution, especially the ones from the extremes of the geographic range (e.g., upper St. Lawrence River and Lake Ontario) by progeny from eels originating from different freshwater regions (L. Bernatchez, Université Laval, pers. comm., 2010).

In the United States, Weeder and Uphoff (2009) reported that American Eels in the Chesapeake Bay region may already depend on recruitment from other areas and there is a trend of declines in United States eels (landings and abundance). Therefore, scenario 2 is more likely; rescue of Canadian eels from the United States, while possible, would be limited.

Based on assessments of stocking of eels from eastern Canada into Ontario, risks are present. For instance, translocation has resulted in some males being found in inland waters, where historically, almost all eels were female (Pratt and Threader 2011). The implications of this are not known, but could be serious because the upper St. Lawrence River and Lake Ontario system (NFBZ10) has been originally composed of more than 95% of large female silver eels, which would contribute to a relatively large proportion of the total number of American Eel eggs released on the spawning grounds (Gray and Andrews 1970; Dolan and Power 1977; Dutil *et al.* 1985; Jessop 1987; Fournier and Caron 2005; COSEWIC, 2006; MacGregor *et al.*, 2010).

THREATS AND LIMITING FACTORS

Because of its extended lifespan, semelparous reproductive system, and long migrations, the American Eel faces an array of natural and anthropogenic mortality factors across its range. According to Bonhommeau *et al*. (2008a,b), cumulative anthropogenic effects may have increased the American Eel’s sensitivity to environmental conditions. This is particularly true in freshwater habitats where vulnerability to migration barriers, fisheries, and bioaccumulation of contaminants is a more serious threat than in marine habitats (Edeline, 2007; Lamson *et al.*, 2009).

Natural Limiting Factors

Identifying factors and processes influencing natural mortality is fundamental to the understanding of population dynamics (Bevacqua *et al.*, 2010). The productivity and climate conditions within both the marine and estuarine areas are thought to be key factors influencing eel survival during early life history. At present, there is little understanding of specific factors that influence the production of eels in fresh water or nearshore estuarine areas. Some studies have estimated the instantaneous daily mortality rate (*Z,* the proportion of individuals dying at any particular time) for eels at different juvenile life stages and suggested links with specific environmental features. For instance, Jessop (2000a) used trap count data to estimate *Z* of 0.0612 for elvers entering the East River (Chester, NS), considerably higher than rates reported for European Eel elvers (0.0107 and 0.0233; Berg and Jørgensen, 1994, cited in Jessop, 2000a). Jessop (2000a) suggested that the apparently higher rate of mortality for Chester River elvers could stem from the toxic effects of low pH of the river, but this contrasts with the experiments conducted by Reynolds (2011) who demonstrated that elvers showed little to no lethal or sub-lethal responses to pH as low as 4.0. In other cases, estimates of mortality rates are provided, but there is no understanding of what drives these rates, what drives variation from locality to locality, or what the impact of marking-based mortality on these estimates might be. For instance, annual estimates of disappearance rate include both natural mortality and adult emigration and are based on the assumption that recruitment is stable through time. On the Sud-Ouest River (NFBZ9), Verreault (2002) estimated an annual disappearance rate of 26.4% (instantaneous rate = 0.307) for emigrating eels aged 9 to 17. A population model suggested that 27% of eels that enter Lake Ontario survive to reach the open Gulf of St.

Lawrence as pre-spawning silver eels. A stochastic life table model estimated a disappearance rate of 22.9% (instantaneous rate = 0.26) per year in an unexploited eels in Prince Edward Island (ICES, 2001).

Recently, using life history characteristics of eels from freshwater habitats based on sampling from the southern Gulf of St. Lawrence (NFBZ1), Chaput and Cairns (2011) estimated that the instantaneous mortality rate for females is in the range of 0.08 to 0.10 with a mean age at maturity of female silver eels set at 20 years (Jessop, 2010). This evaluation is based on Monte Carlo simulations using low density to high density predictions of equations of Bevacqua *et al.* (2010) for the European Eel. This evaluation is in the range of instantaneous rate (0.16 ± 0.06) established by Morrisson and Secor (2003), in the Hudson River, with a corresponding annual disappearance rate of 15%.

Oceanic and regional coastal currents may affect regional and local recruitment (McCleave, 1993; Castonguay *et al.*, 1994b; Jessop, 1998b). Indeed, global climate change and other environmental shifts may alter the Gulf Stream system by generating a northward deviation (Castonguay *et al.*, 1994b; Knights, 2003; Miller *et al.,* 2009), which reduces oceanic productivity (Dekker, 1998). The potential effects of ocean currents on recruitment have been described by Friedland *et al.* (2007), Bonhommeau *et al.* (2008a), and Miller *et al.* (2009).

Given the long larval duration (up to a year), weakening currents could interfere with their dispersal from the Sargasso Sea to continental waters and increase mortality either by starvation or by unfavourable transport patterns that extend the duration of oceanic migration (Knights, 2003; Friedland *et al.*, 2007; Bonhommeau *et al.*, 2008a; Miller *et al.,* 2009). Both of these outcomes could lead to reduced recruitment to continental areas. Bonhommeau *et al.* (2008b) demonstrated that long-term fluctuations in sea temperature have a greater effect on recruitment than variation in latitude and strength of the Gulf Stream. They found a strong correlation between the survival of eel larvae and the food availability during early life stages. The decrease in primary production through climate-driven processes would have therefore affected the recruitment of eel populations (Bonhommeau *et al.*, 2008ab). Even if global warming is a concern (Miller *et al.*, 2009), anthropogenic factors during the continental life stages should be considered as potential threats because the eel declines have generally preceded the oceanographic changes (Friedland *et al.*, 2007).

Anthropogenic Threats

Eels are sensitive to environmental conditions of different environments [freshwater (river, lake, stream), estuaries, saltwater] and can be impacted by migration barriers, poor water quality (land use practices), contamination and fisheries and the cumulative effects of multiple stressors that are likely operating. For instance, explanations by Mi'kmaq elders of the decline of eel fishing at Paq'tnkek (east of Antigonish, Nova Scotia) included migration barriers, chemical contamination, seaweed harvesting, introduction of foreign species, loss of habitat due to deforestation, agricultural practices, decline of eelgrass, and overfishing (Prosper, 2002; Prosper and

Paulette, 2003a; Davis *et al.*, 2004). Depending on their position in a watershed and if no passage is provided, barriers (discussed in more detail below) erected in watercourses restrict access to upper reaches of watersheds and severely impede upstream dispersal of juvenile eels in fresh water, negatively affecting abundance and subsequent recruitment (Haro *et al*., 2000; Larinier *et al.*, 2006; MacGregor *et al.*, 2008; 2009; 2010). A succession of multiple obstacles in a watershed has a cumulative impact on upstream passage of juvenile eels (Legault et Porcher, 1990 cited in Dagrève 2005), and leads to severe habitat fragmentation and strongly affects population dynamics by restricting eel dispersal and recruitment (Larnier *et al.*, 2006).

During downstream migration, obstacles affect eels by preventing or delaying movements, and increasing mortalities and sublethal injuries (Couillard *et al.*, 1997). The turbines of hydroelectric dams also pose a hazard during downstream passage by killing eels during their pre-reproductive downstream migration, and reducing spawning escapement (McCleave, 2001; ICES, 2003; 2006; Allen, 2008). Provisions for safe downstream passage of eels at hydroelectric facilities are rare, and turbine mortalities of downstream migrants can be high (Larinier and Dartiguelongue 1989; Travade and Larinier 1992; Normandeau Associates and Skalski 2000; Verreault and Dumont 2003). Turbine mortality is positively correlated to eel length and inversely proportional to blade spacing. It also varies with turbine type (Francis, Kaplan, and propeller), turbine size, and operating conditions such as flow and generating efficiency (Montén 1985; Larinier and Dartiguelongue 1989; Travade and Larinier 1992). During downstream migration, numerous successive facilities on a particular watershed will also increase cumulative mortality and injuries.

Dams (habitat fragmentation and turbine mortalities)

<u>Lake Ontario and St. Lawrence Region</u>

NFBZ10 Great Lakes - Upper St. Lawrence (Ontario and western and central Québec) and NFBZ9 Lower St. Lawrence (eastern Québec)

In Canada, the St. Lawrence River watershed contains some 8,411 dams of at least 2.5 m in height (Verreault *et al.,* 2004). Overall, these obstacles are estimated to prevent, restrict or delay access to at least 12,140 km² of eel freshwater habitat in the St. Lawrence River and Lake Ontario system (10 m or less deep[8]; LaBar and Facey, 1983; Verreault *et al.,* 2004; Table 7).

[8] Based on a recent study on the Ottawa River in Lac des Chats (Smith, 2010), local anglers have been catching incidental eel while fishing for walleye in the summer at depths as great as 15 m.

Table 7. Surface areas of American Eel freshwater rearing habitat that is upstream of restrictive dams in the St. Lawrence River watershed, and estimated potential annual escapement (modified from Verreault *et al.* 2004).

Site (subwatershed)	Estimated rearing habitat (0-10 m depth) above dams (km²)	Potential annual unutilized escapement (individuals)
Upper St. Lawrence - River Lake Ontario	5,800	399,700 [A]
Ottawa River	3,700	255,000
Richelieu River - Lake Champlain	1,200	82,700 [B]
Others	1,440	99,200
All	12,140	836,600

A: Access re-opened in 1974 at Moses-Saunders Dam and in 2002 at Beauharnois Dam; B: Access re-opened between 1997 (Chambly Dam) and 2001 (Saint-Ours Dam)

Table 8. Summary of estimated total numbers of American Eel stocked as glass eels and/or elvers in Canadian waters. The program has now been suspended.

Year	Lake Morin	Richelieu River	Lake Ontario	Canada
1999	40,000			40,000
2005		600,000		600,000
2006		1,000,000	167,000	1,167,000
2007		421,500	437,000	858,500
2008		746,000	2,001,000	2,747,000
2009		0	1,303,000	1,303,000
2010		0	143,000	143,000
Total	**40,000**	**2,767,500**	**4,051,000**	**6,858,500**

There are 5,260 dams in Québec watersheds alone that drain into the St. Lawrence River (Tremblay *et al.*, 2011). In Ontario, at least 953 dams exist within the eel's historical range, 40 of which are erected on the Ottawa River (MacGregor *et al.*, 2010). In this watershed, as in much of North America, dam construction increased in the early 1900s and peaked between 1950 and 1970. On the mainstem of the St. Lawrence River below Lake Ontario, there are two hydro complexes. The Moses-Saunders Dam was completed in 1959. Construction of the Beauharnois Dam began in the late 1920s and was completed in 1961 (Verdon and Desrochers 2003). Shipping locks at these two dams offered upstream passage opportunities, but upstream passage has been provided by permanent eel ladders at the Moses-Saunders Dam since 1974 and at the Beauharnois Dam by a passage facility that operated from 1994 to 1995 and from 1998 to the present. Eel ladders have been available to give access to Lake Champlain via the Richelieu River at Chambly since 1997 and at Saint-Ours since 2001.

Extensive habitat loss due to barriers has occurred throughout the American Eel's range (de Lafontaine *et al.*, 2009a; Casselman and Marcogliese, 2010a,b; MacGregor *et al.*, 2010). Impeded access to Lake Ontario and Lake Champlain resulted in range contraction and substantial cumulative loss in access by eels to formerly productive rearing habitat, and has limited the capacity of the upper St. Lawrence River and Lake Ontario system to produce large, highly fecund females.

Because turbine mortality is positively correlated to eel length and inversely proportional to blade spacing, large female eels from the upper St. Lawrence River - Lake Ontario system are at greatest risk of turbine mortality (Larinier and Dartiguelongue 1989; Travade and Larinier 1992; Normandeau Associates and Skalski 2000; Verreault and Dumont 2003). Mortality rate of emigrating eels with mean length of 88 cm has been estimated at 16% for a Francis turbine and at 24% for a propeller turbine at the Beauharnois Dam (Desrochers 1995). Eels with a mean length of 102 cm passing a propeller turbine in the Moses-Saunders Dam suffered an estimated mortality of 26.4% (Normandeau Associates and Skalski 2000). Migrants leaving Lake Ontario must pass through both of these turbine complexes during their spawning migration. Verreault and Dumont (2003) have estimated that silver eels from NFBZ10 exiting Lake Ontario are subjected to an accumulated turbine mortality of 40% after their passage through Moses-Saunders and Beauharnois generating stations. This additive turbine mortality at the Moses-Saunders and Beauharnois dams in the upper St. Lawrence River contributes to almost 75% of the anthropogenic mortality during downstream migration and reduces the annual spawning escapement by 40%. Turbine mortality figures should be regarded as a minimum because undetected sub-lethal injuries could further reduce the number of females that reach and successfully spawn in the Sargasso Sea (Couillard *et al.*, 1997).

Gulf of St. Lawrence and Atlantic Region

NFBZ1 Maritimes (New Brunswick, Nova Scotia, Prince Edward Island, and the central and southern parts of Québec's Gaspé Peninsula), NFBZ8 Atlantic Islands (Newfoundland) and NFBZ2 Eastern Arctic (Labrador)

In the Scotia-Fundy portion of NFBZ1, numerous rivers are blocked by hydro-electric dams, including the one at Mactaquac on the Saint John River. There is only one hydro dam in the Gulf of St. Lawrence drainages of the Maritime Provinces. There are numerous low-head, non-power dams in this sector, especially in Prince Edward Island where there are some 800 dams (MacFarlane, 1999). These dams, however, are not necessarily harmful to eels because they can frequently colonize their headponds, where prey are abundant.

Traditional knowledge from the Atlantic Provinces (NFBZ1 and NFBZ8) suggests that the dams have caused loss or declines of many species because of their impacts on water quality and fish passage (Regional Aboriginal Species of Concern Working Group, 2008). Members of Eel River Bar First Nation reported a significant decline in American Eel after construction of a dam on the Eel River in 1963 (GMRC, 2008a). The only monitored eel ladder in the Maritime Provinces is at the Morgan Falls hydro facility on the LaHave River on the south shore of Nova Scotia. The ladder has operated since 2002 (R. Bradford, DFO, pers. comm., 2005). A migration study has been conducted on silver eels (N = 25) at a recently reconstructed hydroelectric facility on the Magaguadavic River, New Brunswick. Downstream movements of many eels were delayed at the dam and tagged fish moved extensively in the reservoir, presumably searching for an exit. All 19 eels that entered the turbines died (Carr and Whoriskey, 2008). Six eels survived by passing the dam either via a fish bypass chute (four), by spilling over the dam (one), or through a fish ladder for upstream migrating fish (one). The efficiency of the downstream fish bypass at this site might be improved by altering water management strategies (Carr and Whoriskey, 2008).

Fisheries and Oceans Canada (Gulf Region) has examined culverts in watersheds to determine whether the height of the culvert drop allows for fish migration and the degree of impact on fish. Pilot projects are being undertaken in Richibucto and Shediac rivers' watersheds in New Brunswick; Mabou River in Cape Breton and Trout River in Nova Scotia. For the Miramichi River watershed alone, approximately 100 km² of fish habitat has been restored through culvert removal (Regional Aboriginal Species of Concern Working Group, 2008).

Preliminary data on number and locations of dams in Newfoundland was collected through a joint effort of the Government of Newfoundland and Labrador and DFO. According to Nicholls (2011), in southern Labrador (northern latitudinal limit of the range of American Eel), the potential sources of impact on the eel population are several municipal water supplies. In total, one of the municipal water supplies has a dam (Charlottetown) and the remainder are water intakes in streams or ponds. In Newfoundland, there are 393 potential impact locations for eels associated with surface water supplies. This includes 310 different sites, with some including dams along with other surface water removal. In insular Newfoundland, 234 dams have been identified that are associated with hydroelectric development and 81 dams associated with water supplies. In total 39 major dams (at least 10 metres in height) have been identified in Newfoundland by the Canadian Dams Association Registry (2003, cited in Nicholls, 2011). Further work is needed to evaluate habitat loss for eels because most of the dams have not been fully evaluated as to their potential impact on eels and it is not clear whether all dams and/or water supplies are having an impact on eels or how extensive these impacts may be (E. Hardman, Government of Newfoundland and Labrador, pers. comm., 2011).

Habitat Degradation

Poor land use practices (timber harvest, farming practices, and urbanization of watersheds) and unprotected riparian zones resulting in poor water quality, erosion, and sedimentation are all factors impairing overall stream quality (Machut *et al.*, 2007). Sediment arising from such practices also contains contaminants, making eel flesh less safe to eat and posing risks to reproductive success (MacGregor *et al.*, 2010).

Water level fluctuations can negatively affect eels during movements or migrations, or impact habitat use at overwintering sites and shallow areas such as wetlands and riparian zones. Alteration of important overwintering habitat has not been assessed, including desiccation of these important habitats during winter drawdowns. Winter drawdown of reservoirs can also cause ice scouring and removal of aquatic vegetation in the littoral zone, which eels use for cover and protection in other seasons. In the Ottawa River, water management regimes resulting in fluctuating water levels have been shown to be affecting fish community structure by removing all available food for benthivore juvenile fishes in the littoral zone, and thereby potentially be affecting growth and survival of eels (Haxton and Findlay, 2009).

Fisheries

Commercial, Aboriginal subsistence, and some recreational fisheries exist in eastern Canada. Exploitation activities of eels are managed by the three administrative regions of DFO in eastern Canada (Newfoundland and Labrador, Maritimes, Gulf) and by the provinces of Québec and Ontario in their respective jurisdictions. The draft American Eel management plan has called for a 50% reduction in American Eel anthropogenic mortality in Canada (CEWG, 2009).

Because the American Eel is semelparous, all eel fisheries target pre-spawners (Richkus and Whalen, 1999). All continental life stages are subject to commercial exploitation in Canada. Fisheries occur in most habitats including freshwater (lakes, rivers), estuaries, brackish and saltwater bays. Fisheries for elvers and silver eels occur during narrow time windows. The yellow eel stage may last many years, so fisheries that target this stage may produce high cumulative mortality even if fishing mortality rate per year is low. Eel fishing effort is unevenly distributed within the Canadian range of the American Eel. In some regions, there are intensive fisheries while in other regions eels are unexploited.

Since 1992, the eel fishery has been governed by the Aboriginal Fisheries Strategy (AFS) agreement, with summer and winter harvesting methods being listed as spear, rod and reel, fyke net, and pots.

Canadian commercial American Eel fisheries occur in the four Atlantic Provinces (NFBZ1 and NFBZ8), and in Québec (NFBZ9 and 10); the Province of Ontario (NFBZ10) closed their commercial fishery in 2004. In Canadian waters, yellow and silver stages of the American Eel are fished commercially in the St. Lawrence River system and tidal waters of the southern Gulf of St. Lawrence. Elsewhere in Eastern Canada eels are fished in only a minority of their habitat. Commercial fishing zones in Québec are located on the St. Lawrence River upstream and downstream of Montréal, and also in the St. Lawrence River estuary where silver eels are caught during their pre-reproductive migration. Eels originating in NFBZ9 are not exploited because Québec fisheries target eels from NFBZ10. Most commercial fishing in the Atlantic Provinces occurs in the southern Gulf of St. Lawrence, New Brunswick’s Saint John River, various locations on the Atlantic drainages of the Maritime Provinces, and various locations in

Newfoundland. Commercial fisheries target yellow eels in tidal waters, coastal waters and estuaries. Winter recreational spear fisheries also contribute to anthropogenic mortality of yellow eels in the Southern Gulf of St. Lawrence. In the Scotia-Fundy region, eel fishing occurs in both fresh and marine waters, but many rivers and coastal habitats are unfished. Elver fisheries in Canada occur only in Scotia-Fundy and the south coast of Newfoundland. In Newfoundland (NFBZ8) and Labrador (NFBZ2), yellow and silver eels are fished principally in rivers, but there are many rivers which are not exploited. There is no eel fishing in nearly all of the Gaspé Péninsula, most of the fresh waters draining into the southern Gulf of St. Lawrence, or in the north shore of the Gulf of St. Lawrence.

Fishing mortality rate is poorly known for yellow and silver American Eels. Instantaneous fishing mortality on mostly yellow eels in exploited waters of Prince Edward Island was estimated at 0.5 per year (ICES, 2001). The great majority of eels taken in this fishery are yellow eels, and are exposed to fishing mortality over several years. Assuming no density dependence in survival rates, the model estimated that fishing in exploited Prince Edward Island waters reduced spawner escapement by 90% below what it would have been in the absence of fishing. In the St. Lawrence estuary silver eel fishery, mark-recapture experiments yielded estimates of exploitation rates of 19% in 1996 , 24% in 1997 (Caron *et al.*, 2003), and 10.5% in 2010 (G. Verreault, MRNF, pers. comm., 2011).

The unreported eel catch was historically believed to be less than 5% in Lake Ontario and 8% in the St. Lawrence estuarine fishery (ICES, 2001). These rates are likely lower (or zero in Lake Ontario) given the fisheries closures (Ontario) and reductions (Québec). In the Scotia-Fundy region, reported landings are closely correlated with effort and are not considered to reflect abundance (R. Bradford, DFO, pers. comm., 2010).

Casselman (2003) and Casselman and Marcogliese (2007) described trends in United States and Canada commercial catch data for eels (Figure 25). While there are temporal differences across regions in the onset of major declines in catches, overall trends in Ontario commercial harvests parallel those of Canada and the United States (Casselman and Marcogliese, 2007). Between 1950 and 2003, Ontario commercial eel harvests averaged 80.1 t, but rose substantially in the 1970s to an unprecedented 228.2 t. This harvest constituted 20% of total Canadian landings in that year. At the same time, North American harvests also increased dramatically in response to high prices and strong markets, and concern grew over their sustainability. Ontario harvests declined substantially thereafter (in the 1990s) in synchrony with strong harvest declines across North America as did the important silver eel harvests in Québec (Figures 24 and 25), despite an increase in price per kg (Casselman, 2003); well above the long-term mean (Casselman and Marcogliese, 2007).

The Canadian elver fishery targets arriving glass eels and elvers as they ascend estuaries in the Scotia-Fundy Region (NFBZ1). Eels are harvested as glass eels and elvers primarily for aquaculture in Asia and to some extent Europe, as yellow eels for bait and food, and as silver eels for food (ASFMC, 2000; Weeder and Uphoff, 2009). The establishment of commercial fisheries for American Eel elvers is a recent development in North America compared to European and Japanese eels (Jessop, 1997). Elvers were first fished in the U.S. for export to Asia during the early 1970s, while the elver fishery in Canada began in 1989 (Jessop 1997). The total elver harvest has increased from 26 kg in 1989 to almost 1.6 t in 1994 (Jessop, 1995), a total catch that represents an annual estimated gross landed value of up to CAN $1,600,000 for the commercial elver fisheries in Canada (Jessop, 1997). Between 1996 and 1998, Jessop (2000b) estimated that elver fishers took 31 to 52% of arriving elvers in the East River, Chester, Nova Scotia. During 2004 to 2007, landings of elvers increased by 18% relative to the average landings during 1997 to 2002 (Bradford *et al.*, 2010; Figure 16b). In Scotia-Fundy, the number of potential licences is tightly limited (N = 9), each licence covers a specific geographic area, elver fishing is generally not permitted in streams where fisheries exist for larger eels, an overall catch quota is set for each license, catch limits are set for each fishable stream, and a record of daily fishing activity is required. Although the quota was reduced by 10% since 2005 for stocking, it resulted in no fishing reduction because the quota (1,000 kg by licence) set up at implementation is high and has never been reached. The continuing high value of elvers (e.g., $2000/kg in 2010) may encourage the development of illegal fisheries, such as reported in the United States (MacGregor *et al.*, 2008).

Chemical and Biological Contamination

Contaminants in eels in the St. Lawrence River and Lake Ontario system are still at elevated levels but their impact on eel survival, migration success, reproduction, and recruitment is not well understood or quantified (Castonguay *et al.,* 1994a; Hodson *et al.*, 1994; Couillard *et al.*, 1997). J. Tomie (UNB, pers. comm., 2010) pointed out the potential high exposure risk of eels to contaminants when burrowing in the sediment overwinter.

There is reason for serious concern as the level of measured concentrations of some contaminants has been demonstrated to have adverse effects on the reproductive success of silver eel (ICES, 2009d). According to Palstra *et al.* (2006), the environmental levels of dioxin-like PCBs and the decline of eel coincide worldwide.

The decline in some contaminants in the St Lawrence River in recent decades may mean that newer "legacy" chemicals such as fluorine-containing compounds [e.g. Teflon, polytetrafluoroethylene (PTFE)], perfluoroalkyl contaminants or brominated flame retardants could contribute to some of the observed toxic effects (Byer *et al.*, 2010a,b). In Canada, an understanding of contaminant effects on aquatic ecosystems has been limited with the cessation in 2005 of two programs on toxicology helping to understand the evolution of contaminants in the St. Lawrence River system (C. Couillard, DFO, pers. comm., 2010). When such programs were active, high summer mortalities of silver eels in the freshwater part of the St. Lawrence River in the early 1970s were attributed to acute toxicity from environmental contaminant levels (Dutil 1984). Eels from the St. Lawrence River estuary tributaries have been reported to be less contaminated with Mirex than those from Lake Ontario (Hodson *et al.,* 1994). Renaud *et al.* (1995), however, found a greater concentration of Mirex during the 1990s than between 1947 and 1950 in polluted tributaries of the St. Lawrence River (St-François and Sainte-Anne rivers). Therefore, deterioration of habitat quality could affect eel survival throughout its range depending on pollution level. Aquatic pollution has been pointed out as one of the most common reasons to explain rare eel fishing activity by Gespe'gewa'gi in the Gaspé Peninsula of Québec and in northern New Brunswick (GMRC, 2008ab).

In Lake Ontario, contaminants levels have decreased significantly from 1970s levels (Luckey *et al.*, 2007), and there is little evidence to suggest that human-related contaminants [PCBs, DDT, Mirex, Dieldrin (insecticide), dioxins, furans, mercury] are currently impacting natural reproduction and health of Lake Ontario benthos, plankton or fish on a lakewide basis. According to a monitoring program targeting introduced Coho Salmon (*Oncorhynchus kisutch*), total PCBs concentrations have decreased threefold and Mirex has reduced twofold since 1970 (Luckey *et al.*, 2007). On the other hand, exposure to emerging contaminants (new pesticides, pharmaceuticals, alkylphenols, flame retardants, etc.) has increased in the last decades and could impact eel health and reproduction. Endocrine disruption and altered development has been reported in White Perch (*Morone americana*) from the Lower Great Lakes regions (Kavanagh *et al*., 2004). The observation of intersex individuals suggest that these fish were exposed to estrogenic endocrine-disrupting substances of industrial or domestic origin. The potential effects of these substances on Lake Ontario American American Eels have not been studied.

One of the most important factors influencing contaminant dynamics in Lake Ontario is the increasing proliferation of invasive species (e.g., Zebra and Quagga mussels, *Dreissena* spp.) because they alter both fish community composition and food web energy flows (Luckey *et al.*, 2007). Thus, subsequent changes to pathways and fate of contaminants have resulted in altered bioaccumulation rates in portions of fish communities as evidenced by recent spikes in contaminant burdens. Alterations to the forage base of fish communities have resulted in diet shifts and in some cases, the consumption of a more contaminated prey, which produces elevated body burdens of contaminants (Luckey *et al.*, 2007).

Eels are particularly sensitive to bioaccumulation of lipophilic contaminants due to specific ecological and physiological traits. In polluted waters, eels are heavy bioaccumulators because they are long-lived benthic species with a high fat content that accumulates lipophilic contaminants such as PCBs (polychlorinated biphenyls), pesticides (DDT), dioxins, and furans.

The toxic effects can occur throughout the eel's life cycle, e.g., during growing, silvering, migration (freshwater and marine habitats), the development of reproductive cells, and larval stage. Concerns that contaminant burdens may influence reproductive success and possibly lead to population declines have received more research attention for the European Eel. Hence, contaminants are believed to be an important issue in understanding the reasons for the decline of the European Eel (Geeraerts and Belpaire, 2010). The ICES Eel Working Group (WGEEL) has described the risks of deteriorated biological quality of eels. An increasing level of evidence on the detrimental impact of contamination on the eel has been made available (ICES 2006; 2007; 2008; 2009d), and two recent reviews have described the effects of contaminants on the European Eel (Geeraerts *et al*., 2010; Elie and Gerard, 2009 cited in ICES, 2009d).

The European Eel has been proposed as a model for evaluating the chemical status within the Water Framework Directive[9] because eel contaminant profiles seem to be a fingerprint of the contamination pressure of a specific site (Belpaire and Goemans, 2007; Belpaire *et al.*, 2008). ICES (2009d) has cited multiple studies that have revealed that the European Eel represents one of the fish species accumulating the highest quantity of contaminants such as PCBs, polycyclic hydrocarbons, pesticides, and heavy metals. Different levels of contaminants have been found according to eel length and age, supporting the bioaccumulation hypothesis for PCBs (Tapie *et al*., 2006 cited in ICES, 2009d), organochlorines (Bruslé, 1994 cited in ICES, 2009d), and cadmium (Pierron *et al*., 2007; 2008a,b). In terms of long-term accumulation, high contaminant loads might not be related directly to the sampling site where eels were captured (Ramade, 1989; Tapie *et al*., 2006; 2009 all cited in ICES, 2009d).

Few ecotoxicological studies have evaluated the influence of contaminants at different life stages in eels and their subsequent influences on spawner quality. Among the few, Palstra *et al*. (2006) reported the transfer of PCBs from female silver eels to gonads as well as a correlation between increased PCB levels in females and reduced survival time in embryos, with levels over 4 ng TEQ/kg[10] in gonads being associated with no embryo survival. Such deleterious effects of contaminants (PCBs, cadmium, DDTs, Mirex) on eel fertility by impairing egg quality, embryonic development, the lipid accumulation process (fitness), and reduced oxygen consumption during spawning migration have been pointed out for the European Eel (Robinet and Feunteun, 2002; Pierron *et al*., 2008b; Van Ginneken *et al.*, 2005; 2009), and the American Eel by Hodson *et al.* (1994) and Couillard *et al.* (1997). Because migrating females are fasting (Pankhurst and Sorensen, 1984), contaminants recirculate into the blood system, and chemical levels in the eggs could be even higher at hatching, increasing the likelihood of toxicity to the larvae (Hodson *et al.* 1994). Couillard *et al.* (1997) used the relationship between tissue mirex concentration and body mass to identify the origin of migrating silver eels taken in the St. Lawrence estuary. Those with high loadings were presumed to have come from the upper St. Lawrence River and Lake Ontario area. The authors

[9] The EU Water Framework Directive integrated river basin management for Europe: http://ec.europa.eu/environment/water/water-framework/index_en.html

[10] PCBs units: nanogram toxicity equivalent quantities/kilogram

observed a relationship between the site of origin of the eels and the prevalence of pathological lesions, specifically pre-neoplastic lesions in the liver indicating exposure of eels to genotoxic contaminants such as polycyclic aromatic hydrocarbons in Lake Ontario/Upper St. Lawrence River (Couillard *et al*., 1997). Neoplastic liver lesions have been reported in white sucker in Lake Ontario (Hayes *et al*., 1990).

Pollution might also impact reproductive success through genotype effects. A significant negative correlation between heavy metal pollution and eel genetic variability was reported by Maes *et al.* (2005). Reduced genetic variability was observed in strongly contaminated eels, as well as a negative correlation between levels of bioaccumulation and multilocus heterozygosity (Maes *et al*., 2005). Reduced genetic variability could be associated with population size declines or a reduced ability to adapt to environmental change.

Productivity and Food-web Changes

Profound ecological changes have occurred in Lake Ontario since the 1970s. Exotic Zebra and Quagga mussels have substantially altered water quality and trophic relations in Lake Ontario (Mills *et al.*, 2005). By affecting prey species, one of the far-reaching impacts of Zebra and Quagga mussel colonization on aquatic ecosystems throughout the Great Lakes basin is their influence on the distribution of American Eel in Lake Ontario and the St. Lawrence River. Declines in nearshore catches of American Eel with electrofishing in Lake Ontario during the day are positively correlated with increased water transparency caused by *Dreissena* colonization in the early 1990s (Casselman, 2003). By increasing water clarity, the invasion of dreissenid mussels is forcing eels into deeper and thermally less preferred waters (Casselman, 2003; Mills *et al.*, 2005).

The Round Goby (*Neogobius melanostomus*), introduced in 1990, is now widespread in the Great Lakes and the mainstem of the St. Lawrence River (Reyjol *et al.,* 2010; Taraborelli *et al.,* 2010). It has been shown that Mottled Sculpin (*Cottus bairdi*), Johnny Darter (*Etheostoma nigrum*) and Logperch (*Percina caprodes*) populations, associated with the bottom of lakes and rivers, declined drastically after the colonization of Round Goby in the Great Lakes. The negative effects of the Round Goby on these and other fishes is likely related to its aggressive behaviour for food or space, and to a series of biological traits that enhance colonization success, such as a broad diet, multiple spawning periods and parental care (Corkum *et al*., 2004 cited in Reyjo *et al.*, 2010). Important changes in the pathways of nutrient and contaminant transfer in the Great Lakes - St. Lawrence River system are likely (Vanderploeg *et al*., 2002 cited in Reyjo *et al.*, 2010).

<u>Introduced Parasite: *Anguillicoloides crassus*</u>

The exotic swim bladder nematode parasite, *Anguillicoloides crassus*, is a native parasite of Japanese Eels (*Anguilla japonica*) that has spread from aquacultural imports from Asia (e.g., Køie 1991, cited in Barse and Secor, 1999). Dispersal of *A. crassus* within aquatic systems is generally through the natural movements of infected definitive, intermediate, and paratenic hosts. Spread between distant localities is generally through human transport of infected eels (Machut and Limberg, 2008).

In North America, the parasite was originally discovered in the United States. In 1995, the first confirmed report of *A. crassus* was from a single American Eel captured in Winyah Bay, South Carolina (Fries *et al*., 1996). Since then, the parasite has been detected in American Eels from different locations in the United States, e.g., in the Hudson River and Chesapeake Bay (Barse and Secor, 1999; Morrison and Secor, 2003; Machut and Limburg, 2008); Florida (Barse *et al.*, 2001); and Connecticut and New Jersey (Sokolowski and Dove, 2006). During recent surveys, all eels that were examined from rivers in New England ranging from the Pawcatuck River (Rhode Island) to Massachusetts, and to the East Machias River (Maine) had the parasite with some degree of variation in prevalence (7 to 76%; Aieta and Oliveira, 2009). Once introduced into freshwater and estuarine environments, *A. crassus* has been documented to spread rapidly among eels from different areas (Machut and Limberg, 2008). In Europe, prevalences increase from 10% to 50% within a year (Belpaire *et al*., 1989; Koops and Hartmann, 1989) and may even reach 100% prevalence within a year (Kennedy and Fitch, 1990).

Between 2002 and 2005, all eels examined through different programs were free of *A. crassus* in Canada. From the juvenile eels annually examined for swim bladder parasites in the upper St. Lawrence River and Lake Ontario system, as well as from the silver eels examined in downstream migration, and prior and during stocking efforts, parasites have been detected in this area only within the last year (Barker, 1997; Dumont *et al.,* 2005; J.M. Casselman, Queen's University; G. Verreault, MRNF, pers. comm. 2010, DFO, unpubl. data, 2012). The parasite was first detected in Nova Scotia as far north as Cape Breton Island in 2007 (Rockwell *et al.*, 2009; Campbell *et al.*, 2010). Campbell *et al.* (2010) conducted the first systematic survey of the presence of *A. crassus* in the Maritime Provinces. Necropsies of 1,966 eels collected from 175 sites distributed within 63 drainages showed the parasite exhibited a disjunct and limited (to six drainages) distribution. Overall prevalence and mean (± standard deviation) intensity (nematodes per eel) of infection within the six drainages were low: 10.1% and 2.6±4.1.

The effects of this parasite on viability of American Eel, however, remain not well understood, but Palstra *et al.* (2007) have linked the collapse of the European Eel to *A. crassus* and suggested that migrating silver eels with severely infected or damaged swim bladders are unable to reach the spawning grounds. According to Nagasawa *et al.* (1994 cited in Rockwell *et al.*, 2009), the parasite causes little or no pathology to its native host *A. japonica*. In anguillid eels other than *A. japonica*, however, infection of the swim bladder by *A. crassus* may affect eel survival by directly causing bladder dysfunctions and by decreasing host energy level (Rockwell *et al.*, 2009). Infestation rates are lower for inland American Eel (Machut and Limburg, 2008) probably because transmission from secondary hosts is reduced (Schmidt *et al.,* 2009).

Heavy infections can lead to hemorrhagic lesions, swim bladder fibrosis or collapse, skin ulceration, decreased appetite, and reduced swimming performance (Barse and Secor, 1999). Sokolowski and Dove (2006) first reported the pathogenesis of *A. crassus* infections in wild American Eels. Gross observations included opacity of the normally translucent swim bladder and dilation of blood vessels. The swim bladders of infected eels showed histological changes, and consistently observed pathologies included abnormal appearance, damage, and destruction of the mucosa, submucosa, hyperplasia of the lamina propria, dilation of the blood vessels and exposition of the mucosal blood vessels; bacterial infections in the submucosa and muscularis mucosa; and larval penetration of the tissues of the swim bladder and migration through the rete mirabile (Sokolowski and Dove, 2006).

<u>Stocking of Eels</u>

The first large-scale eel stocking experiment occurred in the Richelieu River, a tributary to Lake Champlain, in 2005 (Dumont *et al.* 2005). In 2006, stocking efforts were extended to Lake Ontario. From 2005 to 2010, a total of seven million elvers (three million in the Richelieu River; four million in the upper St. Lawrence River and Lake Ontario) have been stocked in Canadian waters (Verreault *et al.*, 2009; Pratt and Mathers, 2011; Pratt and Threader, 2011).

Stocking initiatives can be considered as a potential threat because their effects are uncertain, manifestation of some effects may only be apparent years after, and because of the documented negative effects of stocking of artificially propagated fishes in other taxa, particularly salmonids (see review by Araki and Schmidt 2010). Key issues that needed to be addressed during stocking programs is whether these efforts will ultimately contribute to species escapement are recruitment or whether the stocked eels will develop into a phenotype similar to naturally recruited eels migrate back to the Sargasso Sea. Such uncertainties are being assessed under a monitoring program to evaluate the species' response after stocking (survival, growth rates, density, and sex ratio) and to determine if stocked eels' characteristics differ from the natural recruits (Verreault *et al.*, 2010; Pratt and Mathers, 2011; Pratt and Threader, 2011). The results of Côté *et al.* (2009) support the hypothesis of a possible genetic basis for differences in growth and sex ratio between eels from different areas. This could imply that stocking of areas where only females are found (such as the upper St. Lawrence River and Lake Ontario system) with glass eels and/or elvers from areas with variable proportions of males (Maritimes) may reduce the proportions of females produced in water being stocked. Also, if slower-growing eels tend to migrate farther upstream in order to avoid competition with faster-growing individuals as proposed by Edeline *et al.* (2007), such stocking practice could potentially be detrimental for the remnant local eels (Côté *et al.*, 2009). These and other concerns have led to a suspension of the stocking program in Ontario.

Climate Change and Oceanographic Processes

All three Northern Hemisphere eels, the American Eel, European Eel, and Japanese Eel, show consistent patterns of declines in abundance through major portions of their ranges (e.g., Knights 2003; B. Knights, pers. comm. 2012). The general synchronicity of these declines, the variable nature of changes to freshwater habitats of all three species, and that fact that all species are heavily reliant on open ocean habitats for spawning and early life history suggest that marine conditions may play key roles in contributing to these declines. In fact, negative correlations between glass eel recruitment and North Atlantic Oscillation, Gulf Stream Position, and El Niño-Southern Oscillation indices have been recorded over recent decades (reviewed in Knights 2003). The veracity and biological basis for these associations are not well understood, but the latter may involve either or both of oceanographic influences on spawner migration and reproduction, larval drift or survival from variation in water temperature and/or food supply (Knights 2003). Only longer time series and a better understanding of the spawning and early life history of American Eels in the ocean can help evaluate the potential effects of climate change on eels. By contrast, Reist *et al.* (2006) pointed out that North Atlantic eels are basically subtropical fishes limited in their northern distribution by cold Arctic waters and that it is conceivable that warming trends could result in eels colonizing new habitats at the northern portions of their current range.

PROTECTION, STATUS, AND RANKS

Legal Protection and Status

The American Eel was assessed as Special Concern by COSEWIC in April 2006. The status was re-examined and designated Threatened in May 2012, but a decision by the Government of Canada on whether or not to list the species under Schedule 1 of the federal *Species at Risk Act* (SARA) has not been made. As such, the species is not currently protected under the Act. More general protection throughout Canada may occur, however, through the federal *Fisheries Act* and the *Environmental Protection Act*. In Québec, aquatic habitats are generally protected by the *Environmental Quality Act*. Fish habitat is also protected by the *Act respecting the conservation and development of wildlife* that, under articles 128.1 to 128.18, controls activities that could modify biological, physical or chemical components peculiar to fish habitat.

In Ontario, the American Eel was listed as Endangered and became protected under Ontario's *Endangered Species Act*, *2007* (S.O. 2007, c. 6) on June 30, 2008. Its habitat will be protected under the general habitat provisions of the act as of June 30, 2013. It will receive habitat protection prior to this if a habitat regulation is developed for the species before that date (B. Walpole, OMNR, pers. comm., 2010). In Ontario, no commercial or recreational fishing has been allowed since 2004 and 2005, respectively.

In Québec, the American Eel is on a list of species likely to be designated as threatened or vulnerable under the *Act Respecting Threatened or Vulnerable Species* (R.S.Q. c. E-12.01; October 2006; D. Bussières, MRNF, pers. comm., 2010; Caron *et al.*, 2007). In Newfoundland and Labrador (NL), the American Eel is listed as vulnerable according to the provincial listing under the NL *Endangered Species Act* (Management Plan for the American Eel, March 2011; S. Pardy Moores, Government of Newfoundland and Labrador, pers. comm., 2011).

Non-Legal Status and Ranks

Under the General Status of Wild Species (GS) process in Canada, the American Eelbeen ranked (2005) as *secure* overall in Canada, but *sensitive* in Québec and Newfoundland and Labrador, *may be at risk* in Ontario and Nova Scotia, and *secure* in New Brunswick and Prince Edward Island (CESCC 2006). In Prince Edward Island (PEI), the province has not ranked this species under the *Wildlife Conservation Act* (R. Curley, PEI Dept Environment, Energy and Forestry, pers. comm., 2010).

The NatureServe status for American Eel is *Apparently Secure* both globally (G4; last reviewed in 2006), and nationally (N4; last reviewed in 2005) in the United States and Canada. NatureServe designations for eels are: *Critically imperiled (S1?)* in Ontario and South Dakota*, Imperiled (S2)* in seven states (Illinois, Indiana, Kansas, Ohio, Vermont, West Virginia, Wisconsin), *Vulnerable (S3)* in Québec and seven states (Geogia, Iowa, Massachusetts, New Hampshire, New York, Oklahoma, Tennessee), *Apparently secure (S4)* in Labrador, Prince Edward Island, and five states (Arkansas, District of Columbia, Kentucky, Maryland, North Carolina), *Secure (S5)* in New Brunswick, Nova Scotia, Newfoundland**,** and 11 states (Alabama, Connecticut, Delaware, Louisiana, Maine, Mississippi, New Jersey, Pennsylvania, Rhode Island, Texas, Virginia), and *Presumed Extirpated* (SX) in New Mexico (NM).

The last revision of the conservation status of imperilled North American freshwater and diadromous fishes by Jelks *et al.* (2008) did not include the American Eel. It was decided to exclude the American Eel from the list primarily because it remains one of the most widespread, albeit diminishing, freshwater fishes in North America.

The *Endangered Species Act* administered by USFWS initiated a status review of American Eel in U.S. waters in 2004 at the request of the Atlantic States Marine Fisheries Commission (ASMFC) by means of a petition launched in 2005 to list American Eel under the federal *Endangered Species Act* (Watts and Watts, 2004). In their review, the USFWS placed more emphasis on short-term data series relating to glass eel abundance, rather than on long-term data series (MacGregor *et al.*, 2008), and concluded that protecting the American Eel as an endangered or threatened species under the *Endangered Species Act* was not warranted according to USFWS 12-month finding period (U.S. Office of the Federal Register, 2007; USFWS, 2007). Therefore, the American Eel is also not listed under the National Marine Fisheries Service's Species of Concern Program, a proactive conservation program for species that may become threatened or endangered.

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BIOGRAPHICAL SUMMARY OF REPORT WRITER

The report was prepared by V. Tremblay**,** aquatic biologist with AECOM (2, Rue Fusey, Trois-Rivières, Québec G8T 2T1, valerie.tremblay@aecom.com). V. Tremblay received an M.Sc. in Biology from the Université du Québec à Rimouski in 2005. Her thesis topic was the reproductive strategy of female American Eels in the St. Lawrence River watershed. She wrote the first status report for the American Eel in 2006 under the aegis of the Canadian Eel Science Working Group (CESWoG), which consists of biologists from the Ontario Ministry of Natural Resources (OMNR), the Québec Ministère des Ressources naturelles et de la Faune (MNRF, Secteur Faune Québec) and the Department of Fisheries and Oceans (DFO) who coordinate eel research and assessment in Canada. V. Tremblay, a CESWoG member, is associated with various mandates on eels in the upper St. Lawrence River and Lake Ontario system and frequently works on diadromous species and aquatic species at risk.

COLLECTIONS EXAMINED

No collections were examined.