Received: 3 December 2021 | Revised: 16 June 2022 | Accepted: 17 July 2022

DOI: 10.1002/rra.4034

RESEARCH ARTICLE

WILEY

A contemporary estimate for the abundance of juvenile American Eel *Anguilla rostrata* attempting to migrate past a barrier in the Ottawa River

Steven M. Woods[1] | Shannon D. Bower[2] | Nicolas W. R. Lapointe[1]

[1]Canadian Wildlife Federation, Kanata, Ontario, Canada

[2]Infinity SES, Ottawa, Ontario, Canada

Correspondence
Shannon D. Bower, Infinity SES, 507 McLeod St., Ottawa, ON K1R 5P9, Canada.
Email: shannonbower@infinityses.com

Abstract

American Eel (*Anguilla rostrata*) travels from the Sargasso Sea to fresh waters of eastern North America and back in a lifetime, and once provided one of the most abundant eel fisheries in the world. Many American Eel populations are now at risk worldwide. Dams act as barriers to the upstream migration of juvenile American Eels, which can be partially mitigated by installing eel ladders. To inform mitigation decisions and provide baseline data, the number of eels approaching barriers should be estimated; however, estimation is difficult for this now rare and cryptic species, especially in large rivers. In St. Lawrence and Ottawa River system, American Eels are among the largest and most fecund of the species, and local populations in the Ottawa River are almost entirely composed of large, female eels. American Eel in this area has declined to less than 1% of historic abundance, yet no local population estimates are available to inform recovery strategies and management actions. We, therefore, evaluated data from an unpublished study to estimate the abundance of American Eel attempting to migrate upstream past a barrier. American Eels ($n = 339$) were captured at the Carillon Generating Station over 36 days (July 12, 2010–August 17, 2010). Results were fit to the POPAN Jolly-Seber model in program MARK. Future studies could be improved by sampling throughout the migration season and deploying multiple traps spanning downstream features. While confidence intervals in the best-fitting model were wide, the estimate nonetheless provides a baseline to inform future work and management.

KEYWORDS
Jolly-Seber, migration, POPAN, population estimates, species at risk

1 | INTRODUCTION

American Eel (*Anguilla rostrata*, Lesueur 1821) are facultative catadromous fish that exhibit arguably one of the most extraordinary migrations in the world. Their life history involves migration from the Sargasso Sea first as larvae, then as glass eels, followed by a rearing phase in one of as many as six salinity profiles from highly brackish to entirely freshwater (Thibault et al., 2007) in systems that drain to the Atlantic and Caribbean (COSEWIC, 2006, 2012). This facultative status is believed to be driven mainly by energy status, where eels that are slow-growing avoid competition by migrating further into fresher water (Edeline, 2007).

For many, this journey includes eventually reaching the St. Lawrence River in Canada, a distance of approximately 2,500 km, where they arrive as elvers (COSEWIC, 2012). Once they arrive in freshwater, American Eel remain for years, maturing into yellow and eventually silver eels before out-migrating back to the Sargasso Sea to spawn. In the St. Lawrence River system, this final physiological shift into outmigration form (silver eel) does not occur until they reach the St. Lawrence estuary (McGrath, Bernier, Ault, Dutil, & Reid, 2003). American Eel are semelparous; thus, successful migration at both ends of the lifecycle is required for continued recruitment.

The American Eels that reside in the upper St. Lawrence River system, including Lake Ontario and the Ottawa River, remain in the system for years. Adults typically delay spawning migrations until reaching the age of 10–25 years (Casselman, 2003; COSEWIC, 2006), though in Ontario some adults are both larger and older (up to 42 years, J. Casselman, unpublished data, as cited in MacGregor et al., 2013). These populations are also almost exclusively composed of large, highly fecund females (Casselman, 2003; COSEWIC, 2006). Declines in the out-migration success of this population may therefore have far-reaching implications for the species (Venturelli et al., 2010).

While there is research indicating that oceanic conditions (Bonhommeau et al., 2008), overfishing (see examples in MacGregor et al., 2013), and environmental conditions such as low dissolved oxygen (Hill, 1969) are factors in the decline of this species, barriers to passage are widely considered the most pressing threat (COSEWIC, 2006; COSEWIC, 2012; Jacoby, Casselman, DeLucia, & Gollock, 2017; Verreault & Dumont, 2003). Juvenile eels are prevented from reaching rearing grounds, and out-migrating adult eels commonly experience issues such as impingement (trapped against screens), entrainment or diversion, and turbine mortality, where eels die attempting to pass through turbines (Elvidge et al., 2018). The American Eel is listed as "Endangered" on the International Union for the conservation of Nature's Red List (IUCN, 2021), declared depleted by the Atlantic States Marine Fisheries Commission (ASFMC) in 2012, and listed under Ontario's Endangered Species Act (2007) as endangered, and assessed as "Threatened" by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 2012.

Anguilla *spp*. are at risk worldwide, too. Jacoby et al. (2017), in their global assessment of eel populations, found that only two of 13 species examined were rated at "Least Concern" or by International Union for Conservation of Nature (IUCN) standards. The authors similarly noted that tropical anguillids are in urgent need of increased monitoring and study (Jacoby et al., 2017).

Recruitment of American Eel has decreased by >99% from historical abundance in the late 1970s throughout the Ottawa River and St. Lawrence River watersheds (Casselman, 2003; Dekker et al., 2003; MacGregor et al., 2009). In the St. Lawrence River, American Eels need to pass through two hydroelectric dam facilities, and annual turbine mortality at these locations is estimated at a combined 39.5% (Verreault & Dumont, 2003). The Ottawa River and its tributaries have multiple hydroelectric dams that act as barriers to juvenile migration and significant sources of mortality for out-migrating female eels. Likelihood of survival declines significantly for eels that mature upstream of multiple hydro facilities. For example, the out-migration survival of eels that mature in the lowest impounded reach of the Ottawa river is approximately 80%, whereas out-migration survival of eels departing from Mississippi Lake in the Mississippi River (a tributary of the Ottawa) is estimated at 2.8% due to turbine mortality at six facilities (MacGregor, Haxton, Greig, Dettmers, & McDermott, 2015) and reduces further to 1.4% when lower St. Lawrence fisheries are included as a factor. Eels are now rarely captured upstream of main stem barriers in the Ottawa River (Casselman & Marcogliese, 2010; MacGregor et al., 2009, 2013, 2015).

Eel ladders are not yet installed at the Carillon Generating Station, the first barrier in the Ottawa River and no estimates of potential recruitment for these populations are available. Some upstream migration does occur (Casselman, 2012, 2013, 2014), possibly through a recreational boat canal and locks around the dam, but without eel ladders and associated passage counts, estimating potential recruitment is challenging. The most accurate estimates of eel recruitment come from visual or automated counts at eel ladders. Baseline data are also lacking on how many eels approach the barrier, and on what proportion of these fish pass successfully.

Capture-mark-recapture (CMR) studies are one tool that can be used to generate baseline data on how many eels approach a barrier as they can be used to generate an estimate for population size. Population estimates were often the focus of the initial Jolly-Seber (JS) model (Jolly, 1965; Seber, 1965), and new developments in the JS models focused on population growth (Schwarz, 2001) included modified methods better adapted to estimate populations for species with complex life histories (e.g., POPAN; Schwarz & Arnason, 1996). These modified methods have been used to estimate population sizes of rare and migratory species (Carroll et al., 2011; Constantine et al., 2012; Williams, Frederick, & Nichols, 2011), including the American Eel (Morrison & Secor, 2004). This study, therefore, used a previously unpublished 2010 CMR dataset made available by Hydro Quebec (Guindon & Desrochers, 2010) to estimate the population size of juvenile American Eel exhibiting migratory behaviour at a barrier on the Ottawa River.

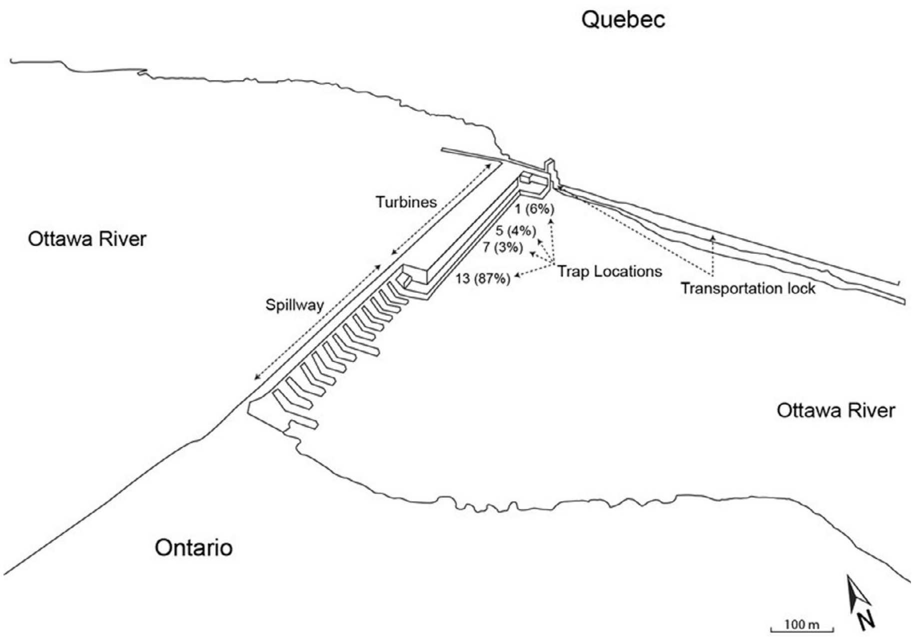
2 | METHODS

2.1 | Data collection

Our model used data from a study performed by Guindon and Desrochers (2010). The study was carried out at the Carillon Generating Station (CGS; 18 T, 548111 m E, 5046245 m N), the first major barrier in the Ottawa River from the St. Lawrence River. This station is 828 m wide and is equipped with 14 Kaplan turbines (Figure 1). A recreational navigation lock is located on the northern shoreline (Figure 1). The lock is typically operational from mid-May to Mid-October, a period that coincides with eel upstream migration (June–October in the Upper St. Lawrence River, Casselman, 2003). Though no eel ladder is installed at this complex, some upstream migration occurs, possibly via the lock. Relative abundances of American Eel are disproportionally higher downstream of the generating station than upstream (Casselman, 2012, 2013, 2014).

To estimate the relative abundance of migrating juvenile eels at the CGS, eels were captured, tagged with passive integrated

FIGURE 1 Design of the Carillon Generating Station (18 T, 548111 m E, 5046245 m N), showing position of the spillway, turbines, transportation lock, and trap locations (at turbine # 1, 5, 7 and 13) used to capture American Eel attempting to pass the barrier in 2010. Numbers in brackets represent the proportion of American Eel captured in each trap.



transponders (PIT tags), and released downstream of the dam over a 36-day period from July 12, 2010, to August 17, 2010 (Guindon & Desrochers, 2010). According to previous studies (e.g., Desrochers, 2001), this sampling period was estimated to be the peak of the upstream migration. Studded plastic ladders with an attractant flow of 9.6 L/s and an operation flow of ±0.6 L/s funnelled into a net were used to trap upstream migrating eels downstream of four non-operating turbines (turbines 1, 5, 7, and 13) at the CGS. Captured eels were anesthetized (0.1 mL clove oil/L), measured, and tagged with a Destron Fearing PIT tag (model TX1411SST, 12.50 mm X 2.07 mm, 134.2 kHz ISO, 0.1020 g), after being scanned for an existing PIT tag. Tags were inserted under the skin, posteriorly to the skull along the anteroposterior axis (i.e., behind the skull). Eels were then released 30 m downstream from their trap site. Over the 36-day sampling period, traps were visited every two days, giving 18 sampling events. The raw data summarized in Guindon and Desrochers (2010) was used in the analysis.

The traps in this study were designed to capture eels showing migratory behaviour at the barrier by using an attraction flow in otherwise calm waters in proximity to very turbulent waters (mean turbine outflow 2,289 m^3/s). Using this method, eels migrating upstream to find suitable habitat to settle in ("pioneers" and "nomads"; Feunteun et al., 2003) were the targeted population. Estimating the number of eels potentially migrating upstream is valuable as it provides an index of river use.

2.2 | Data analysis

To estimate population size based on the CMR data, we generated an open population model using the POPAN option (Schwarz & Arnason, 1996) in program MARK (White and Burnham 1999; Schwarz & Arnason, 2017), based on the Jolly-Seber model

$$\mathrm{N} = B_0 + B_1 + B_2 + B_3 + B_4 + \ldots + B_{k-1}$$

$$\begin{array}{ccccccccccc}
b_0 & & b_1 & & b_2 & & b_3 & & b_4 & & \mathrm{N} \\
\phi_1 & & \phi_2 & & \phi_3 & & \phi_4 & & & & \\
t_1 & \rightarrow & t_2 & \rightarrow & t_3 & \rightarrow & t_4 & \rightarrow & t_5 \ldots & & \\
\uparrow & & \uparrow & & \uparrow & & \uparrow & & \uparrow & & \\
p_1 & & p_2 & & p_3 & & p_4 & & p_5 & &
\end{array}$$

FIGURE 2 The POPAN parameters survival (Φ), probability of capture (*p*), and probability of entry into the population (*b*) at each sampling period (*t*) generate an abundance estimate *B* at each sampling period. N is, therefore, the population estimate derived from the sum of *B* values generated for sampling periods

(Jolly, 1965; Seber, 1965) and using a log link function. Together, these options (the POPAN and log link function) account for inconsistencies in the entry process of the original Jolly-Seber model (Schwarz & Arnason, 1996). The model estimates survival (Φ), probability of capture (*p*), and probability of entry into the population (*b*) at each sampling period (*t*), which are used to obtain a population size (N). Note that in the model, emigration (leaving the study area) and death are considered indistinguishable. Thus, we are unable to differentiate between death and emigration. Hereafter, we use the term survival to include survival and retention in a population.

Here, the models were parameterized using combinations of time-dependent survival (Φ), capture (*p*), and entry parameters (*b*) either changing through time according to sampling period (*t*) or remaining fixed. (Figure 2). To begin with, capture histories in sampling data are coded per individual capture in a matrix using the binary

(1,0). The number of individuals with the same binary profile is then counted together as a group. These profiles are then transformed to a count form of {Φ(*t*) *p*(*t*)}. The resulting matrices (Parameter Index Matrices), place constraints on the parameter estimate because there is a limit on the number of possible outcomes (based on the number of shared binary profiles, White & Burham, 1999). A Design Matrix is a matrix that is multiplied by the parameter vector (the likelihood parameters) to approximate the data used in the model (White & Burnham 1999). This specifies a linear model to link the parameter vector to the data (via a log link function), effectively linking the Design Matrix and the Parameter Index Matrices. This provides both a starting point for the model and values for the changing parameters (through the linear model). More theory and detailed model processes can be found in Schwarz & Arnason, 1996, 2017.

To make model parameters identifiable, it was necessary to assume the initial and final capture (p_1 and p_k) values. In this study, we assumed an initial estimate of survival (Φ_1) of 0.36 (CI = 0.26–0.47), with SE = 0.05 and probability of capture for eels was estimated at 5%.

Akaike's information criterion, corrected for small samples [AICc], was used to select the most appropriate model. This model was selected following the rule that the lowest AICc scores fit the data best (Burnham & Anderson, 2003) and have substantial support (i.e., the lowest delta AICc is zero; Burnham & Anderson, 2004) and models having higher AICc values (e.g., five and higher; Burnham & Anderson, 2004) have little or no support.

$$\mathrm{N} = B_0 + B_1 + B_2 + B_3 + B_4 + \quad + B_{k-1}.$$

$$b_0 \quad b_1 \quad b_2 \quad b_3 \quad b_4 \quad \mathrm{N}$$

$$\phi_1 \quad \phi_2 \quad \phi_3 \quad \phi_4$$

$$t_1 \rightarrow t_2 \rightarrow t_3 \rightarrow t_4 \rightarrow t_5 \ldots$$

$$\uparrow \quad \uparrow \quad \uparrow \quad \uparrow \quad \uparrow$$

$$p_1 \quad p_2 \quad p_3 \quad p_4 \quad p_5$$

The assumptions of a Jolly-Seber model are: (1) animals retain their tags, (2) survival probabilities are equal between sampling times for marked and unmarked fish, (3) tags are read correctly, (4) sampling is instantaneous, (5) the study area is constant, and (6) capture probability is equal for unmarked and marked fish. We believe that all assumptions were sufficiently met by the design of this study. For example, PIT tags inserted in dorsal muscle rarely lead to loss or mortality in fish the size used in this study (Morrison & Secor, 2003; Musselman, Worthington, Mouser, Williams, & Brewer, 2017; Rude, Whitledge, & Phelps, 2011), addressing the assumptions 1 and 2. We assume that tags were read properly (i.e., no malfunction in the PIT tag reader) and correctly (assumption 3), given that the study was conducted by professional consultants. The short time period and high survival rate allowed for instantaneous sampling (assumption 4), and the study area remained constant throughout the study (assumption 5). A crucial assumption for the Jolly-Seber models is that catchability must remain the same for marked and unmarked animals at each sampling occasion (assumption 6). This can be influenced by behaviour, biological, or environmental factors. Traps were designed to capture the population of eels showing migratory behaviour at the generating station by using an attraction flow (Piper, Wright, & Kemp, 2012). Trap shyness was assumed to be minimal, given that trap avoidance by eels has been shown to occur for only one day following capture (Morrison & Secor, 2004). When faced with a barrier, American Eels exhibit searching behaviour to find a suitable passage (Brown, Haro, & Castro-Santos, 2009; Haro, Castro-Santos, & Boubée, 2000) and will likely spend a considerable amount of time attempting upstream migration, especially if there is a driver for migration (Feunteun et al., 2003). All eels captured were in the early "yellow" phase of their life cycle, and sizes of eels captured were normally distributed. Further, a sex differentiation in catchability would not occur since all fish caught were likely female (MacGregor et al., 2013; Oliveira & McCleave, 2000).

3 | RESULTS

Guindon and Desrochers (2010) captured and tagged a total of 339 eels in the 18 sampling events. Over the course of the study, 9 (2.7%) eels were recaptured. The highest number of eels was captured between July 18 and July 22, 2010 (n = 140) (Figure 3), particularly July 22 (n = 55), followed by July 20 (n = 51). There was another spike in catches between July 30 and August 5, 2010 when 111 eels were caught. Eel catches declined toward the end of the sampling period. Most of the eels (87%) were captured downstream of Turbine 13 (n = 295), located in the centre of the river at the junction of the spillway and the generating station.

With POPAN, after parameter counting, the best-fitting model was $p(.)\Phi(.)b(t)$, a model that assumed a constant rate of capture probability over time (p), a constant rate of survival (Φ), and a variable rate

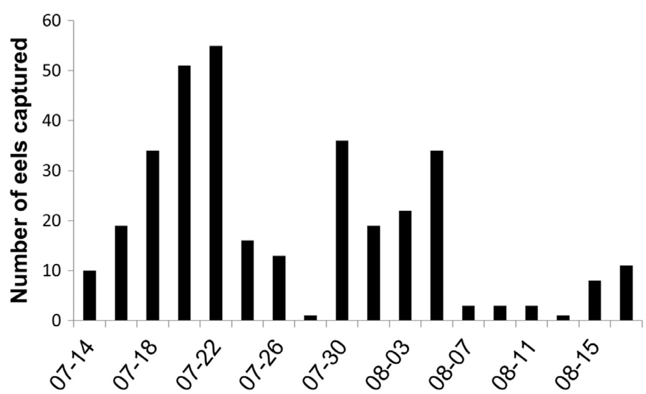


FIGURE 3 Number of American Eels captured in ladder traps set at the Carillon Generating Station between July 14, 2010 and August 17, 2010

TABLE 1 Model results from POPAN option in program MARK for eel population estimate (N) at the base of Carillon Generating Station in 2010

Model	AICc	Delta AICc	AICc Weights	Model Likelihood	Num. Par	Deviance	N	Lower CI	Upper CI
p(.), *Φ* (.), *b*(t)	**253.07**	**0.00**	**0.88**	**1.00**	**20**	**−1,502.98**	**4,367**	**2,293**	**9,491**
p(t), *Φ* (.), *b*(t)	258.07	5.00	0.07	0.08	37	−1,538.68	16,362	4,141	67,757
p(t), *Φ* (t), *b*(.)	259.58	6.51	0.03	0.04	35	−1,532.15	9,020	9,020	9,020
p(.), *Φ* (t), *b*(t)	261.66	8.59	0.01	0.01	36	−1,532.57	5,376	2,807	10,607
p(t), *Φ* (t), *b*(t)	282.25	29.18	0.00	0.00	51	−1,551.65	5,804	1,472	26,558
p(.), *Φ* (.), *b*(.)	12,800.42	12,547.35	0.00	0.00	4	11,078.89	771	550	1,213
p(.), *Φ* (t), *b*(.)	12,884.61	12,631.53	0.00	0.00	20	11,128.55	330	330	330

Notes: The models estimate survival (Φ), probability of capture (*p*), and probability of entry into the population (*b*) at each sampling period (*t*). Models were also compared with time varying (*t*) and constant/fixed (.) parameters. Values for parameters changing with *t* were calculating using a log of the likelihood function, L = probability (first capture) × probability (subsequent recaptures) × probability (loss on capture). The best-fitting model according to AIC criteria is highlighted in bold.

TABLE 2 Parameter estimates for the best-fitting model, p(.),*Φ* (.), *b*(t), over the 18 sampling periods (*t*) including standard error, and confidence intervals for the initial probability of capture assumption (p), the initial probability of survival estimate (*Φ*), the probability of entrance at each sampling period (*b*), and the resulting population size estimate (N)

	Parameter	Estimate	SE	Lower CI	Upper CI
1	*Φ*	0.36	0.05	0.26	0.47
2	p	0.05	0.02	0.02	0.10
3	*b*	0.07	0.02	0.04	0.12
4	*b*	0.12	0.03	0.08	0.19
5	*b*	0.18	0.03	0.12	0.25
6	*b*	0.15	0.03	0.10	0.22
7	*b*	0.00	0.00	0.00	0.00
8	*b*	0.02	0.02	4E-03	0.09
9	*b*	0.00	0.00	0.00	0.00
10	*b*	0.15	0.03	0.11	0.22
11	*b*	0.03	0.02	0.01	0.13
12	*b*	0.07	0.02	0.04	0.13
13	*b*	0.08	0.02	0.05	0.14
14	*b*	0.00	0.00	0.00	0.00
15	*b*	3E−06	8E−04	0.00	1.00
16	*b*	0.01	0.01	1E−03	0.04
17	*b*	2E−09	0.00	2E−09	2E−09
18	*b*	0.03	0.01	0.01	0.06
19	*b*	0.04	0.01	0.02	0.08
20	N	4,367	1,532	2,293	9,491

Note: At each sampling period (*t*), these parameters generate an abundance estimate (*B*). N is the population estimate derived from the sum of *B* values generated for sampling periods. In this best-fitting model, the probability of capture (0.05) and survival (0.36) were fixed and only values for *b* were estimated by the model function.

of probability of entrance over time (*b*) (Table 1). The resulting population estimate of migratory eel at the CGS using this model was N = 4,367 (95% C.I. 2,293–9,491; Table 2).

4 | DISCUSSION

Our POPAN model generated an estimate of 4,367 (95% C.I. 2,293–9,491) migrating juvenile American Eel present at the CGS barrier. The model used estimates of survival over the study period (36%) that were believed to be highly conservative, meaning that actual survival over the study period would likely be higher. For example, there was a survival rate of 100% and a retention rate of 78% in eels translocated from the St. Lawrence River to the reach upstream of the Carillon Generating Station on the Ottawa River (Twardek, Stoot, Cooke, Lapointe, & Browne, 2021). A low survival estimate (such as the one used here) could apply, however, if high passage failure rates led to high rates of emigration. Despite the conservative nature of these estimates, and despite the confidence intervals of our estimate being wide, the model results indicate that thousands of juvenile eels attempted to migrate past the CGS in 2010. If probability of survival can reasonably be expected to be higher, such as when high passage failure does not lead to high emigration, then these results should be considered a lower limit for the number of migrating juveniles in the area.

4.1 | Implications for local management and broader applications

Estimating the number of migrating eels approaching a barrier can inform mitigation actions. Our model results suggest that even at the lowest end of the confidence range, at least 2000 juveniles were attempting to migrate above the CGS in 2010.

Our estimate is higher than the number of trapped eels from the Guindon and Desrochers (2010) study used to generate the model, however, it aligns reasonably with upstream passage counts from other barriers when we consider that abundance past additional barriers would decrease sharply (MacGregor et al., 2009). For example, the eel ladder counts at Moses-Saunders Dam in 2017 show that over 15,000 eels ascended the ladder that year (Guillemette, Guindon, & Desrochers, 2017). While population estimates of American Eel at the

Carillon Dam are sparse, studies from 2009 suggested a density of 3.6 eels/ha below the dam and densities ranging from 0.2–1.2 eels/ha above the dam (Casselman & Marcogliese, 2010). The trap and transfer program that traps juvenile American Eels at Moses-Saunders and transports them above the Carillon Dam (400 eels in 2017) may also play a role in increasing the eel population since that study was conducted.

Given the lack of facilitated passage at the Carillon location, it is unlikely that there were high rates of passage success for this group. This lack of passage success has an impact on upstream reaches. Studies in the Ottawa River from 1997–2004 show a marked decrease in relative abundance beyond the CGS, decreasing from a catch-per-unit-effort of 0.4 at Lac Dollard des Ormeaux to 0.3 at Lac Deschenes before reaching 0 at Lac Rocher du Fendu and reaches beyond (MacGregor et al., 2011). Eels are now almost extirpated from the Mississippi River, a tributary of the Ottawa (Casselman & Marcogliese, 2010) as well as other local rivers like the Bonnecherre, Madawaska, and Petawawa rivers.

Of course, while facilitating upstream passage would be beneficial, such installations need to be combined with measures to facilitate successful out-migration if survival rates of downstream migration are to increase in *A. rostrata*. Additionally, it is important to note that actions for mitigation such as fishway installation should also take into account the possibility of developing an ecological trap (McLaughlin et al., 2013) or providing facilitated passage for invasive species, particularly if downstream migration is not equally supported. Thus, the decision on suitable mitigation actions should depend on the value of the action to the population as a whole and approach the issue from ecosystem and life cycle perspectives.

A recovery strategy is in place in Ontario that includes methods such as commercial license buy-back (MRNF, 2009), closures of sport and commercial fisheries (MacGregor et al., 2009), and stocking of glass eels (MacGregor et al., 2011). Much of these efforts are focused on the St. Lawrence River and Lake Ontario region. Additional measures have been called for within the Ottawa River watershed (e.g., MacGregor et al., 2011) and some progress has been made. An eel ladder has been installed at the Chaudière Falls site, the next major barrier on the main stem river, and a downstream turbine bypass was also installed at a Chaudière facility.

Increasing upstream passage at the CGS through the construction of a permanent eel ladder would serve a number of purposes. It would potentially allow the re-establishment of a permanent population in the Ottawa River system, it would provide a second population monitoring station to complement the existing monitoring on St. Lawrence, and since these two are the only entry points for the entire province, would serve as an index for Ontario as well.

4.2 | Suggestions for future use of this approach

Ideally, future studies would manage to reduce the size of the confidence intervals associated with the estimate. To improve precision in future studies, a robust design (Pollock, 1982) could be implemented, and/or greater effort should be applied to collect a large sample size of marked and recaptured individuals. Robust designs can result in a more accurate estimate of population size and can improve estimation for small sample sizes (Kendall, 2017; Rankin et al., 2016; Schwarz & Seber, 1999). However, robust designs use closed and open population models in tandem.

The issue in using a robust model for American Eel is designing a study to have periods of time where a closed model can be used. The prolonged juvenile eel migration season makes this unlikely and combined with the low numbers attempting to migrate upstream, it may backfire and result in an estimate with higher confidence intervals than an open population model with high capture probabilities. However, the exact dates of upstream migration at the Moses-Saunders fishways are known each year and could help narrow down the migration window dramatically.

Given that American Eels are endangered in Ontario and few individuals are available for capture, an alternative recommendation could be to conduct an additional study to compare methods for increasing capture probability, such as by using different sampling gear or evaluating the effectiveness of attractants. Ideally, capture probabilities would be increased to 0.1 for closed models (Otis, Burnham, White, & Anderson, 1978) and 0.3 for open models (Pollock, Nichols, Brownie, & Hines, 1990) to improve the precision of the population size estimate (Rosenberger & Dunham, 2005).

A final suggestion is to consider alternate forms of estimation that do not share the same constraints. For example, Lyons et al. (2016) used a Bayesian state-space approach to a Jolly-Seber mark-recapture model and found that their approach showed very low (never above 2%) relative estimate bias. The study focused on a population of migratory sandpipers, but the authors suggest that the applicability of this approach is high given its flexibility and could be applied to fish as well.

5 | CONCLUSION

Though the confidence interval in our estimate of migrating population of juvenile American Eel is larger than is desirable, this estimate is still the first available for this population on the Ottawa River, offers a reasonable starting point for comparison with current data, and can inform conservation actions such as installation of an eel ladder at CGS. As discussed, confidence intervals may be reduced with additional sampling and adjustments to the protocol, especially considering that the original study wasn't designed to estimate eel numbers and had low recapture rates.

The results of this study are not expected to shift the current opinion of an industry that installing a fish ladder in this location would not be warranted economically. Since locally relevant estimates are lacking, there is nothing to compare the results of our model to. That is the benefit of this study—there is now a number to work with, support, or refute with further research, and to serve as an estimate for modelling management options. In short, our result provides a missing contemporary baseline that can be used to inform such management decisions and localized contributions to recovery strategies, which are sorely needed for this species.

ACKNOWLEDGEMENTS

The authors wish to thank Hydro Quebec for providing the data supporting this analysis. We also thank Guy Verreault and Laurent Beaulaton for providing independent reviews of an earlier version of this manuscript along with our two anonymous reviewers for sharing their expertise with such helpful and effective suggestions. Finally, we thank our colleagues David Browne, Sean Brillant, and David Callaghan for their advice and assistance.

DATA AVAILABILITY STATEMENT

Data Availability Statement Data used to generate the population estimate in this study were shared under a data-sharing agreement with Hydro Québec. Parties interested in accessing the raw data from the Guindon and Desrochers (2010) study are therefore asked to contact Hydro Québec. Parties interested in accessing the POPAN model data for the population estimates are welcome to contact the authors at the Canadian Wildlife Federation.

ORCID

Steven M. Woods https://orcid.org/0000-0002-8653-1173

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How to cite this article: Woods, S. M., Bower, S. D., & Lapointe, N. W. R. (2022). A contemporary estimate for the abundance of juvenile American Eel *Anguilla rostrata* attempting to migrate past a barrier in the Ottawa River. *River Research and Applications*, 1–8. https://doi.org/10.1002/rra.4034