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Conservation value to assisting live-stranded neonates and entrapped juvenile beluga (*Delphinapterus leucas*) from the St. Lawrence Estuary population

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#### Foreword

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

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### ABSTRACT

The potential benefits of relocating live stranded or entrapped beluga to recovery of the St. Lawrence Estuary (SLE) beluga population were examined using a demographic model. The population is currently declining and information on reproduction rates are lacking. It is not possible to determine if the current decline is due to low reproductive rates, elevated mortality among neonates, or both. Adult survival is already guite high and there is unlikely much room for further improvement. Efforts to improve survival of neonates and juveniles are most likely to benefit population recovery. However, considerable numbers of animals must be assisted and successfully relocated each year to improve survival and halt the decline in population trend. Reports of live-stranded neonates or entrapped juveniles are infrequent. Therefore, the benefit of relocating these few individuals to population recovery is nil, and does not meet the objectives of Conservation Translocation. The occurrences of entrapped juveniles are rare, but if in good health these animals are more likely to survive when relocated compared to abandoned neonates. From a conservation perspective, the benefit of relocating entrapped juveniles to the population as a whole is likely nil given the rarity of these events. However, relocating these individuals may be considered on other grounds. Some of the factors that need to be considered have been identified (e.g. DFO Release and Rehabilitation Criteria).

### INTRODUCTION

Some marine mammal populations have shown remarkable recoveries after severe depletions, while others have remained at low abundance levels, or continued to decline and become extinct or extirpated (Magera et al. 2013). Among different groupings, pinnipeds and other marine mammals (sirenians, polar bears and otters) have shown the highest proportion of recovering populations, likely benefiting from life history strategies favouring higher reproduction rates, and occupancy of nearshore habitats that provide visibility and protective management measures. However, recovery has been less frequent among cetaceans, particularly for offshore populations (Magera et al. 2013).

Wildlife conservation as formed in the early years of the 20<sup>th</sup> Century, focused on the exploitation of natural resources, where surplus production was removed while at the same time leaving some of the resource protected for future use (Lavigne 2006). At the time, wildlife were considered commodities, and the focus was on the necessary population size to ensure adequate resources for exploitation. Beginning in the 1960s, the focus broadened to include animal welfare concerns and the maintenance of functioning ecosystems, which over time has led to a shift from concern for a resource to one where terrestrial and aquatic ecosystems and all that lived in them were not viewed as not simply resources to be developed, but as having inherent rights to exist (Lavigne 2006; Harrington et al. 2013). This new approach has set up competing visions of natural resources, with one camp concerned with sustainable development and the other concerned with resources having rights to persist because they exist. Inherent within this right to exist have been concerns for the welfare of individuals, which has led to the increased efforts to save individual animals, rehabilitate them and return them to the wild (Harrington et al. 2013). For individual animals that have found themselves compromised due to some human element, it has been suggested that humans must attempt to right the wrong, and make every reasonable effort to assist the animal to return it to its natural environment (Moore et al. 2007; Grogan and Kelly 2013). Rescuing compromised individuals can require considerable resources, may pose risk to the individuals themselves and human handlers, or the population through potential disease transfer (Quakenbush et al. 2009). From a conservation perspective, the value of returning a compromised individual to the population will depend in part on the current status of the population, the probability that the animal will survive and its life expectancy, the number of offspring that it is likely to produce (reproductive value), and the number of animals that are assisted and released. Generally, conservation and reproductive values are considered higher for females than males because females produce offspring and, in many cases, only a few males are needed to fertilize many females. However, males may be more important if they contribute to offspring care or if large numbers of males are needed to induce ovulation among females.

Marine mammals are charismatic megafauna, whose fate at the individual level provokes particularly strong emotions among the public. This is especially true when an individual is in difficulty as a result of some human activity. Emergency response networks to rescue compromised marine mammals exist in different countries, including Canada. Responses vary from pushing the individual back into the water, to euthanasia, to a full rehabilitation in specialized facilities where the objective is to improve health status to levels that favour its survival in the wild. In some cases there are calls to rehabilitate individuals belonging to abundant healthy species where there is little conservation concern, e.g., harp seals (*Pagophilus groenlandicus*) or harbour seals (*Phoca vitulina*) on the east and west coasts, respectively. However, for species of conservation concern, efforts to assist individual animals may be beneficial to population recovery. The benefit will, however, depend on the age and sex of the individual, and chances of survival considering its health status when released.

The St. Lawrence Estuary (SLE) beluga (*Delphinapterus leucas*) is considered a relic population found at the southernmost limit of the species range (Mosnier et al. 2010; COSEWIC 2014). Severely depleted by commercial hunting, this population now numbers less than 900 animals (Mosnier et al. 2015; DFO 2017). The SLE beluga is currently listed as "Endangered" under Canada's Species at Risk Act. After a period of stability or even a slight increase (1988-1998), the population appears to have declined from 1999 through to 2012 (Mosnier et al. 2015). A recovery plan has been developed and elements favouring recovery of the population have been proposed.

As part of its monitoring efforts, the Department of Fisheries and Oceans Canada (DFO) maintains a carcass recovery program to monitor numbers, age and sex composition of beluga that are found dead along the shores of the SLE. Over the last 25+ years of the program, the number of carcasses reported each year has remained relatively constant (median = 15 individuals per year; Lesage et al. 2014; Gosselin et al. 2017). Since initiation of the program in 1983, 13 neonate beluga, days to a few weeks old, have stranded alive, including five individuals over the past four years (DFO Québec, unpublished data).

In recent years, three relatively vigorous neonates that had stranded alive within the limits of the normal distribution range of the SLE population were assisted. They were relocated within a herd of adults and young, in the hope they would be adopted, or would eventually find their mother. There was recognition that the odds of this happening and chances of survival were likely nil, and the decision to proceed was motivated by efforts to save individual animals of this declining population. Release was done after taking a skin sample for DNA analysis and future cross-reference with biopsied or dead beluga. The fate of the three relocated neonates is unknown.

Over the same period, there have also been a few cases where independent juveniles became entrapped and needed assistance to return to their normal habitat. In 2001, one juvenile later identified as originating from the Arctic, was entrapped in a river near the Strait of Belle-Isle, i.e., well outside of the SLE distribution range. The animal was relocated to the entrance of the river and survived the operation; it was resignted during the following months in a port in Newfoundland, but eventually died by being hit by a propeller. No treatment was applied prior to relocation; only a skin sample was taken. In June 2017, a juvenile beluga (approximately 3-4 years old) of unknown sex was observed in the Nepisiguit river near Bathurst, NB, i.e., outside the normal distribution range of the population at that time of year (Mosnier et al. 2010). Again in the context of the known decline of the population, it was decided to capture and transport the animal from the river to the SLE where it was released in an area known to be occupied by conspecifics. During transport, the animal was rehydrated, blood samples were taken and a satellite transmitter was deployed on the animal for monitoring. The signal was lost after 19 days. The Bathurst animal was captured and released within 5-6 h, but its fate is unknown. It is unclear how long the animal had been in the river, but its poor condition upon capture suggested that it had spent some time in this limited area. This raised concerns that perhaps the animal could have benefitted from additional rehabilitation efforts before release.

Under SARA there are no provisions for the euthanasia or rehabilitation of animals unless there is scientific value. The SARA directorate has requested advice from Science on the potential benefits to recovery of the SLE beluga population of assisting live-stranded neonates and entrapped juveniles, and the animal welfare concerns related to rehabilitation, euthanasia, or no human intervention.

In this study, we examine the potential benefits of assisting individual beluga of different age classes to SLE population recovery. We use a demographic approach to estimate and compare the effects of changes in survival, or reproduction of particular age classes, as well as the

proportional contribution of different aspects of the life cycle to population growth rate and recovery (Heppell et al. 2000; Caswell 2001). This analysis responds to point 1 and part of points 2 and 3 of the request for advice concerning the following questions:

- 1. Is attempting to rehabilitate or relocate a live-stranded newborn beluga or juvenile likely to contribute to the recovery of the SLE beluga population?
- 2. What are the chances of survival of a stranded newborn beluga calf, and how should we assess the health status of newborn beluga?
- 3. In the case of a live-stranded SLE beluga, what factors should be considered in the decision to rehabilitate, re-locate, or leave the animal where it is? What practical steps should be undertaken to minimize animal welfare concerns?
- 4. In the event that a stranded animal is relocated, what scientific information should be collected during the relocation?

### MATERIALS AND METHODS

In humans a neonate is a newborn up to 0-28 days of age, but there does not appear to be any clear definition for a neonate mammal, although it has been suggested that an animal in its first week seems appropriate (Saunders Comprehensive Veterinary Dictionary, 3 ed. © 2007 Elsevier, Inc.). Here, we assume a neonate is a few days to a few weeks old. For beluga, the duration of lactation is 1.5 to 2 years (Brodie 1971; Doidge 1990), therefore, left alone, these animals will die. We define a calf as a suckling animal (age=0 to 2y).We define a juvenile as an animal that has been weaned, but has not yet reached sexual maturity, which occurs at around 8 years old in females (and a few years later for males), at which point the animals become adults.

The data consist of age frequency distribution and fecundity data from a healthy population of beluga in Alaska, and from the record of dead beluga recovered as part of the carcass monitoring program in the SLE between 1983 and 2017 (Table 1) (Burns and Seaman 1985; Lesage et al. 2014). The age frequency distributions from the two populations were used to construct a life-table, under the assumption that they were obtained from populations with stable age distributions. The two types of data are slightly different, with the data from Alaska consisting of harvested data, where the age frequencies represent multiples of the survival schedule, referred to as an  $I_x$  type life table by Caughley (1977). The data from the SLE consist of beach cast carcasses. The age frequencies of the beach cast animals are multiples of the d<sub>x</sub> schedule, which involves a slightly different approach to reconstruct the life table (Caughley 1977).

Table 1. Age frequencies of harvested animals from Alaska (Burns and Seaman 1985), and beach cast carcasses from the St. Lawrence Estuary for the period 1983-2017 (Lesage et al. 2014; DFO, Unpublished Data). Births for both populations are estimated from reproductive rates of Alaskan beluga (Table 2) (Burns and Seaman1985). The original data for Alaska were presented assuming 2 growth layer groups (GLG) per year, which was the convention when the data were collected, but were converted to recognize that 1 GLG represents one year (Stewart et al. 2006). For SLE beluga it is assumed that 1 GLG equals one year.

Age		Alaska		St. Lawrence Estuary			
(GLG)	Count	Deaths	Births	Count	Deaths	Birts	
0	50	15	0	466	96	0	
1	-	-	-	370	7	0	
2	35	4	0	363	9	0	
3	-	-	-	354	11	0	
4	31	2	0	343	2	0	
5	-	-	-	341	5	0	
6	29	3	0	336	2	0	
/	-	-	-	334	1	0	
0	20	I	0	321 222	4	0	
9	- 25	-	-	323	1	0	
10	25	2	0	310	4	51	
12	23	-	4	307	7	50	
13	-	-	-	300	2	49	
14	22	1	4	298	4	49	
15		-	-	294	3	48	
16	21	2	3	291	3	47	
17	-	-	-	288	3	47	
18	19	1	3	285	9	46	
19	-	-	-	276	3	45	
20	18	1	3	273	2	44	
21	-	-	-	271	10	45	
22	17	1	3	261	3	43	
23	-	-	-	258	3	43	
24	16	0	3	255	2	42	
25	-	-	-	253	3	42	
26	16	1	3	250	4	42	
27	-	-	-	246	4	41	
28	15	1	2	242	4	40	
29	-	-	-	238	5	40	
30	14	1	2	233	1	39	
31	-	-	-	226	6	38	
ు∠ ??	15	I	2	220	5 6	31 26	
33	- 12	-	-	215	0	30	
35	12	0	2	209	2 8	34	
36	- 12	-	- 2	108	6	33	
37	-	-	-	190	3	32	
38	11	1	2	189	12	31	
39	-	-	-	177	9	29	
40	10	0	2	168	11	28	
41	-	-	-	157	9	26	
42	10	1	2	148	12	25	
43	-	-	-	136	5	23	

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$ \begin{array}{c c c c c c c c c c c c c c c c c c c $	Age		Alaska		St. Lawrence Estuary		
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	(GLG) -	Count	Deaths	Births	Count	Deaths	Birts
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	44	9	0	1	131	6	22
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	45	-	-	-	125	16	17
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	46	9	1	1	109	6	15
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	47	-	-	-	103	6	14
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	48	8	1	1	97	13	13
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	49	-	-	-	84	12	12
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	50	7	0	1	72	14	10
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	51	-	-	-	58	7	5
$\begin{array}{cccccccccccccccccccccccccccccccccccc$	52	7	1	1	51	8	5
54       6       0       1       37       4       3         55       -       -       -       33       6       3         56       6       1       1       27       7       2         57       -       -       -       20       5       1         58       5       0       0       15       2       1	53	-	-	-	43	6	4
55       -       -       33       6       3         56       6       1       1       27       7       2         57       -       -       -       20       5       1         58       5       0       0       15       2       1	54	6	0	1	37	4	3
56       6       1       1       27       7       2         57       -       -       -       20       5       1         58       5       0       0       15       2       1	55	-	-	-	33	6	3
57 20 5 1 58 5 0 0 15 2 1	56	6	1	1	27	7	2
58 5 0 0 15 2 1	57	-	-	-	20	5	1
	58	5	0	0	15	2	1
59 13 3 1	59	-	-	-	13	3	1
60 5 1 0 10 4 1	60	5	1	0	10	4	1
61 6 4 0	61	-	-	-	6	4	0
62 4 0 0 2 2 0	62	4	0	0	2	2	0

Table 2. Age-specific birth rate data from beluga harvested in Alaska, based on females with term fetuses or neonates based on reported age of maturity and late and early pregnancies (Burns and Seaman 1985). The original data were presented assuming 2 growth layer groups (GLG) per year, which was the convention when the data were collected, but were modified to recognize that 1 GLG represents one year (Stewart et al. 2006).

Age	Age	Birth rate
(2 GLG per year)	(1GLG per year)	
0-5	0-9	0
6-10	10-20	0.326
11-22	21-44	0.333
23-25	45-50	0.278
26-28	51-56	0.182
29-35	57-70	0.125

The information from the life tables was used to examine the dynamics of the population using a Leslie Matrix approach:

$$N(t+1) = A^*n(t),$$

(1)

where n is an age-structured vector at time t and t+1, and **A** is a population projection matrix defined by age-specific fertility and survival rates (Leslie 1945, 1948; Caswell 2001).

$$\mathbf{A} = \begin{vmatrix} 0_i & F_{i+1} & \cdots F_{i+z-1} & F_z \\ P_i & 0 & \cdots & 0 & 0 \\ 0 & P_{i+1} & \cdots & 0 & 0 \\ 0 & 0 & \cdots & P_{i+z} & 0 \end{vmatrix}$$

P<sub>i</sub> is the probability of an individual in age class i, surviving to age class i+1, F<sub>i</sub> is the fertility for individuals in age class i, which here is the number of young per individual at time i that are alive at time i+1. Parameter i varied from 1 to x, where x was the maximum age. The projection matrix is assumed to be irreducible and non-negative. That is all age classes in the model contribute to producing the younger age classes, i.e., post-reproductive age-classes are not included in the model and all matrix elements are equal to or greater than zero.

We assumed a 1:1 sex ratio in the population, females give birth to a single calf, and calves are all born at the same time (birth-pulse). We examined only the female component of the population. In both models, age-specific fertility information was taken from the Alaska population (Burns and Seaman 1985; Table 2). The assumption of a 2 GLG deposition per year in Burns and Seaman (1985) introduced some imprecision in age identification for sexual maturity and first births. Animals were mature at age 8, and first birth occurred at age 9. There appears to be a decline in calving rates among older animals suggesting onset of senescence and few beluga reproduce after age 70 (based on Burns and Seaman 1985, but converted assuming deposition of one GLG per year). However, the maximum age for successful reproduction is difficult to determine due to the loss of dentinal layers among old animals (Burns and Seaman 1985). To respect the requirement for irreducibility, the maximum number of age classes was truncated when the number of calves produced by an age class fell to zero.

We examined the two populations in terms of their rate of increase ( $\lambda$ ), stable age structure, age-specific and net reproductive values and elasticities. The  $\lambda$  provides a single measure of the dynamics of the population, summarizing the combined effects of reproduction, age at maturity, and survival. If the matrix is irreducible and non-negative, then the population rate of increase is the dominant eigenvalue of the transition matrix **A**, and can be estimated using the characteristic equation. The eigenvalues are the solution to the characteristic equation:

 $det(\mathbf{A} - \lambda \mathbf{I})=0$ ,

where det is the determinant, **I** is the identity matrix and  $\lambda$ , the rate of population increase, is the dominant eigenvalue.

The age-specific reproductive value is the expected number of future offspring produced by an animal aged *i*. This parameter combines the influences of reproduction, survival and age into a single value. Typically, reproductive value is low at birth, increases to a peak near the age of first reproduction, then declines to zero for post-reproductive age classes. The reproductive value of particular age classes can thus be seen as their relative potential for contributing to future generations. Age-specific reproductive values are normalized so that the reproductive value of the first age class, here newborn calves, is unity. The low value for this age class relative to older animals reflects the probability of a neonate dying before reproducing and the delay until reaching maturity (Caswell 2001).

The discrete equation for Reproductive value  $(v_x)$  for an animal age x can be estimated in matrix form from:

$$\nu_x = \sum_{j=x}^{s} \left( \coprod_{h=i}^{j-1} P_h \right) F_j \ \lambda^{i-j-1}$$

The net reproductive rate ( $R_0$ ) is the mean number of offspring by which a newborn will be replaced by the end of its life. It provides an indication of the rate by which the population will increase from one generation to the next. The discrete equivalent can be estimated from (Caswell 2001):

$$R_0 = \sum_i F_i \left( \prod_{j=1}^{i-1} P_j \right)$$

The Generation Time (T) is the time for the population to increase by a factor of  $R_0$ . It can be estimated as:

$$T = \frac{\log R_0}{\log \lambda_1}$$

It is important to understand what factors contribute most to population growth. This is complicated by the different scales on which demographic parameters operate. For example, the probability of survival can only vary between 0 and 1, whereas the reproductive rate can vary from 0 to several offspring depending on litter size. Elasticity analysis decomposes the population growth rate into the contributions made by the life cycle transitions. It is a perturbation measure in matrix projection models that quantifies the proportional change in population growth rate as a function of a proportional change in survival or fertility of a given age class. Elasticities thus indicate the proportional sensitivity or relative "importance" of life cycle transitions for population growth and maintenance (Caswell 2001; de Kroon et al. 2000). Elasticities ( $e_{ij}$ ) were developed by Caswell (1978):

$$\mathbf{e}_{ij} = \frac{a_{ij}}{\lambda} \frac{\partial \lambda}{\partial a_{ij}}$$

where  $\lambda$  is the population rate of increase,  $a_{ij}$  is the proportional change in the matrix element and  $\partial$  is the partial derivative.

All analyses were completed using the construction and analysis of age-structured demographic models package 'demogR' written in R (Jones 2007; R Development Core Team 2007).

The finite rate of increase ( $\lambda$ ) is the change in abundance over a period of time (t) and can be estimated from:

$$\lambda = \frac{N_{t+1}}{N_t}.$$

If  $\lambda$  and N<sub>t</sub> are known, then it is possible to estimate N<sub>t+1</sub>. In the case of a population decline, the difference in abundance divided by the probability of survival provides an estimate of the number of animals that must be re-introduced or returned to the population to allow the population to stabilize or to increase.

Expected population size at time t+1 ( $N_1$ ) is calculated from population size at time t ( $N_0$ ) and  $\lambda$ :

$$N_1 = N_0 \cdot \lambda$$

where  $\lambda$  was set at 0.0987 to reflect the modelling results of Mosnier et al. (2015) and at the  $\lambda$  estimated by the current model projections.

The minimal number of individuals (*n*) that need to be rescued to stabilize the population, i.e.,  $\lambda = 1$ , can be calculated as follows:

$$n = \frac{N_{0-N_{t+1}}}{P_i}$$

Where  $P_i$  is the survival rate of the age class of the rescued animals.  $N_0$  is the population at time=0. In the most recent assessment the population estimate for SLE beluga is approximately 900 individuals (Mosnier et al. 2015; DFO 2017). However, here we are more interested in

comparing how sensitive the dynamics of two different beluga populations might be to different human interventions, rather than the actual size. Therefore, in both cases,  $N_0$  was set to 1,000 animals.

The model was constructed initially using age frequency and reproductive rate data collected from beluga in Alaska. We applied a similar approach to the age frequency data from the SLE beluga carcasses. Since there are no reproductive rate data for SLE beluga, we developed the SLE model using the reproductive rate data from Alaska. However, there are concerns that reproductive rates may not be normal in this population therefore, we also simulated the effects of lower reproductive rates on the dynamics of the population as well (DFO 2014). The SLE model is based on the age frequency distribution of beluga carcasses that have been detected on the beach or at sea. Beluga calves are much smaller, and may not be detected as easily as older animals. Mosnier et al. (2015) estimated that adults were 2-7 times more likely to be detected than calves. Therefore we also examined the impact of varying detection probability of newborn carcasses. This resulted in a total of five beluga models, one for Alaska and four for SLE beluga.

### RESULTS

The projection for beluga from Alaska shows a population increasing very slowly at a rate of about  $0.8\% \cdot y^{-1}$  (Table 4, Figure 1). The SLE beluga population model based on the age distribution frequencies from the recovered carcasses and the reproductive rate data from Alaska indicated that the population is increasing at a rate of  $1.7\% \cdot y^{-1}$ . This estimate is much higher than the changes estimated using a recently developed integrated population model that combined several sources of data, and which estimated a slightly positive trend at a rate of  $0.13\% \cdot y^{-1}$  between 1983 and 2002 followed by a declining trend at a rate of  $-1.13\% \cdot y^{-1}$  between 2003 and 2012 (Mosnier et al. 2015). The lower (compared to the Alaska) positive trend observed during 1983-2002 could be simulated by reducing reproductive rates by 32%, or by assuming that the carcasses of only 30% of the calves that had died (compared to adults) were detected. The declining trend of the SLE population observed during 2003-2012 could be simulated assuming that only 17% of the calf carcasses (compared to adults) were recovered (Figure 1, Table 4).

Net reproductive rate or the mean number of offspring by which a female neonate will be replaced at the end of its life, was the highest for the initial SLE beluga model, followed by Alaska, then the modified SLE models (Table 3). The time in years for the population to increase by a factor equivalent to the estimated net reproductive rate, i.e., generation time, was shorter for Alaska beluga than SLE beluga indicating a higher population turnover (Table 3).



Figure 1. Model trajectories of beluga populations based on survival and fertility schedules identified in Tables 1 and 2 for beluga from Alaska and the St Lawrence Estuary (SLE). In all simulations, the populations were scaled to a start population of 1,000 animals. Only the first 50 years of the 100 year projection are shown to show how the trends deviate. The 'Re-scaled Mosnier et al.' run uses trend data from Mosnier et al. (2015) where the 1960-2010 SLE beluga trend has been re-scaled to a starting population of 1,000 animals and setting 1960 as year zero. The 'SLE reduced reproduction' and '30% of calves detected compared to adult' runs simulate the slowly increasing population observed during the first 40 years by Mosnier et al (2015), while the '17% of calves detected compared to adult' run represents the declining trajectory observed over the last decade (Mosnier et al. 2015). Table 3. Population parameters for the Alaska and the St. Lawrence Estuary (SLE) populations estimated using a Leslie matrix approach, and projections over 100 years. Net reproductive rate ( $R_0$ ) represents the mean number of offspring by which a newborn female will be replaced at the end of its life. Generation Time (T) is the time in years for the population to increase by a factor of  $R_0$ .

		St. Lawrence Estuary					
		This study			Mosnier et al. 2015		
Parameter	Alaska	Initial model	reprod. reduced by 32%	30% of calves detected	17% of calves detected	1983-2002	2003-2012
Population growth rate $(\lambda)$	1.008	1.0172	1.0013	1.0018	0.9859	1.0013	0.987
Net reproductive rate (R <sub>0</sub> )	1.11	1.531	1.034	1.048	0.683	-	-
Generation time (T)	72	125	130	130	135	-	-

The stable age distribution in Alaska model comprised twice as many young calves and a greater number of juveniles than the SLE beluga models, while the SLE models comprised a greater number of younger adults (Table 4). Probability of survival of young calves from Alaska beluga was intermediate to estimated values for SLE beluga (Table 4).

Parameter	Population or Scenario	Young calf Age 0 SLE Age 0-1 Alaska	Old calf/Juvenile Age 1-8	Younger adults Age 9-44	Older adults Age 45+
	Alaska	10	35	47	8
	SLE	5	29	67	4
Stable Age	SLE rpd reduced 32%	4	23	70	6
(%)	SLE 30% of calves detected	4	23	70	6
	SLE 17% of calves detected	4	17	73	9
	Alaska	0.71	0.93	0.94	0.92
	SLE	0.86	0.98	0.97	0.83
Probability of	SLE rpd reduced 32%	0.86	0.98	0.97	0.83
survival	SLE 30% of calves detected	0.70	0.98	0.97	0.83
	SLE 17% of calves detected	0.54	0.98	0.97	0.83
	Alaska	1.0	1.7	1.4	0.3
	SLE	1.0	1.34	1.0	0.15
Reproductive	SLE rpd reduced 32%	1.0	1.25	0.8	0.12
value	SLE 30% of calves detected	1.0	1.54	1	0.13
	SLE 17% of calves detected	1.0	1.83	1	0.12

 Table 4. Differences in various descriptors of population dynamics between Alaskan and St. Lawrence

 Estuary beluga. Age is based on deposition of 1 GLG per year (Stewart et al. 2006).

Survival of young calves for the Alaska and SLE model where only 30% of the neonate carcasses are detected was intermediate between survival rates estimated for the basic SLE model and the model where only 17% of the young calf carcasses were detected (Table 4, Figure 2). Survival rates improved rapidly from birth to weaning, then increased slowly until maturity. Survival rates among adults remained high until age 40, but dropped off more quickly for animals aged 40+ years (Figure 2). Generally, survival rates among adult SLE beluga were slightly higher than for beluga from Alaska.



Figure 2. Estimated changes in age specific survival rates from different models of beluga from Alaska and the St. Lawrence Estuary. The y axis has been offset to clearly show survival at age 0 years. Survival values overlap for the SLE scenarios starting at age 1 year.

Reproductive values for beluga from Alaska increased from birth to age 10 y, then maintained a relatively high plateau until the age of 18 y, then declined over time (Table 4, Figure 3). The SLE model where only 17% of the calves were detected increased rapidly in the first year, remained at a plateau until maturity then declined. Among the remaining SLE models the increase in reproductive value was more gradual, peaking at around 9-10 years old, then declining gradually with no definitive plateau compared to Alaska high (Table 4; Figure 3).



Figure 3. Change in reproductive values with age in Alaska and the St. Lawrence Estuary (SLE) beluga.

Elasticities provide a measure of sensitivity of proportional changes in  $\lambda$ , to proportional changes in survival rates and fertility. The elasticity analysis indicated that, for any age classes, an increase in the probability of survival has a greater impact on population growth rate than an increase in fertility, and that changes in survival are likely to have a much greater impact on Alaska beluga than among SLE beluga (Table 5, Figure 4). This analysis also demonstrates that among age classes, the greatest impact on the population rate of increase would be achieved by improving the relative survival of immature individuals and young adults (Table 6, Figure 4). Changing fertility rates (here meaning the combination of births and calf survival) has a greater impact on  $\lambda$  among beluga from Alaska than from the SLE. For Alaska, the fertility elasticity increases rapidly to peak at age 12 and then declines (Table 5, Figure 4). In the SLE populations, the elasticities for fertility increases rapidly from birth to age 10, then maintains a plateau for many years (ages 9-32 y), but never reach the high level observed for the Alaska population.

Parameter	Population or Scenario	Young calf Age 0 SLE Age 0-1 Alaska	Old calf/Juvenile Age 1-8	Younger adults Age 9-44	Older adults Age 45+
	Alaska	0.07	0.07	0.03	0.002
	SLE	0.042	0.042	0.016	0.00
Elasticity of $\lambda$	SLE rpd reduced 32%	0.039	0.039	0.017	0
to survival	SLE 30% of calves detected	0.039	0.039	0.017	0.007
	SLE 17% of calves detected	0.036	0.036	0.018	0.001
	Alaska	0	0.0005	0.004	0.001
	SLE	0	0.001	0.001	0.000
Elasticity of $\lambda$	SLE rpd reduced 32%	0	0.0001	0.001	0.0001
to Fertility	SLE 30% of calves 0 detected		0	0.001	0.0001
	SLE 17% of calves detected	0	0	0.001	0.0001

Table 5. Differences in elasticities between Alaskan and St. Lawrence Estuary beluga. Age is based on deposition of 1 GLG per year (Stewart et al. 2006).



Figure 4. Elasticity estimates for survival (left vertical axis) and fertility (right vertical axis) for models of beluga from Alaska and the St. Lawrence Estuary (SLE). The elasticities for the models that examined a 32% reduction in reproduction rates and the model where 30% of the calf carcasses were detected lie on top of each other.

# RECOVERY

The analyses above indicate that improving the survival and productivity of juveniles is likely to be the most beneficial to the SLE population. A second part of the analysis is to determine how many animals need to survive to contribute to halting the decline and recovery of the population. Currently the population is declining at a rate of  $1.13\% \cdot y^{-1}$  (i.e.,  $\lambda = 0.987$ ). If the population is to be stabilized, then 10 to 19 belugas per year will have to be assisted and returned to the population depending on their age class and probability of survival (Table 6). Assuming the probability of survival is unaffected by the stranding event, 10 juveniles or 12 - 19 newborn calves, would need to be assisted and successfully re-introduced to halt the decline in the population. This number would increase if the stranding event had a negative impact on an animal's chances of survival. If the normal probability of survival declined by 50% or more, then the number of animals that must be relocated successfully exceeds the maximum possible production estimates of 108-135 calves for an estimated population of 900 individuals (Lesage and Kingsley 1995), i.e., 108-135 neonates for an estimated population of 900 individuals].

Table 6. Minimum number of individuals of SLE beluga that need to be relocated each year (n) to reverse the current decline, i.e., achieve a population growth rate ( $\lambda$ ) of 1.0. Results are presented for a declining population ( $\lambda$  = 0.9887)(Mosnier et al. 2015). Calculations were made assuming probability of survival P is unaffected by the animal having stranded alive, as well as for various percent reductions of this probability of survival (in %). Note that if n > 100, then it exceeds the estimated the annual calf production for this population.

%	<i>P for</i> Calf Initial model 17% of calves detected model		Minimum n of calves to rescue	<i>P for</i> Juvenile	Minimum n of juveniles to rescue
in P			λ =0.9887	Any model	λ =0.9887
0	0.86	0.54	12-19	0.98	10
10	0.774	0.486	13-21	0.882	11
20	0.6192	0.3888	16-26	0.7056	14
30	0.43344	0.27216	23-37	0.49392	20
40	0.260064	0.163296	38-61	0.296352	34
50	0.130032	0.081648	77-122	0.148176	67
60	0.0520128	0.0326592	192-306	0.0592704	169
70	0.01560384	0.0097978	641-1021	0.01778112	562
80	0.00312077	0.0019596	3204-5103	0.00355622	2812
90	0.00031208	0.000196	32043-51032	0.00035562	28120

## DISCUSSION

The demographic vigour of a population is defined as its well-being in terms of fecundity and survival. It does not necessarily provide information on future growth, but it does indicate how a population is coping with current conditions (Caughley 1977; Hammill et al. 2007). Information derived from such analyses is useful not only in understanding life history theory, but also in deriving management and conservation strategies (Doidge 1990).

The model developed using age frequency and reproductive rate data from Alaska, represents that of an apparently healthy beluga population compared to SLE beluga (Burns and Seaman 1985). This is reflected in their slightly shorter generation time and higher net productivity compared to the SLE beluga models. The modeled Alaskan adult survival rates were slightly lower than estimates from the SLE beach-cast samples, while calf survival was intermediate. The lower survival rates of adults in the Alaskan model compared to the SLE likely reflect the additional mortality resulting from harvesting, which is not a factor for the protected SLE beluga. The adult survival for SLE beluga estimated here were similar to those derived using an integrated population model and taken together, indicate that they are likely to be reasonable

parameters for this species. Unfortunately, information on age-specific reproductive rates are lacking for SLE beluga, therefore in developing the SLE models, it was also assumed that reproductive rates were similar to data from whales harvested in Alaska. Sergeant (1986) indicated that ovulation rates were similar between Arctic and SLE beluga, but ovulations rates, tend to over-estimate reproductive rates, because not all animals that ovulate become pregnant and not all pregnant animals carry to term. Burns and Seaman (1985) noted slight declines in reproductive rates depending on whether estimates were based on young or near-term fetuses. Aerial surveys have shown that the proportion of calves in the surveys are lower in the SLE population compared to Arctic populations, but some of these differences could be attributed to changes in methods in the surveys flown after 1999 (Brodie 1967; Sergeant and Hoek 1988). However, when surveys flown using similar methods are compared there has been a noticeable decline in the proportion of calves detected on photographs from 15-18% between 1988 and 1997, to three to eight percent between 2000 and 2008 (Gosselin et al. 2014). This change has been accompanied by an apparent change in breeding cycle with animals moving from a three year breeding cycle to a two year cycle associated with a decline in reproductive rates, or an increase in very early neonate mortality, freeing up females to reproduce in the following year (Mosnier et al. 2015).

Since afforded protection in 1979, the SLE beluga population has shown little sign of significant recovery. Instead, various studies have suggested that the population was declining slightly (pre-1987) (Béland et al. 1988), increasing slightly (1988-1997) (Kingsley 2002); or has changed little (1985-2006) (Hammill et al. 2007). A more recent analysis, which combined information from photographic aerial surveys (1990-2009) and beach cast carcass samples (1983-2012) into a single integrated population model, concluded that the population had been stable or increasing at a slow rate since the end of hunting in the 1960s up until the early 2000s, but that starting in 1999, the population entered a period of instability initiated with an increase in calf mortality in 1999, and subsequent steady decline (Mosnier et al. 2015).

The population dynamics models developed in this study assumed a stable age structure. This is a fair assumption for the Alaska population (Burns and Seaman 1985), and was likely true for SLE beluga between 1983 and 1999-2002, a period when the population was relatively stable. After 2002, the population appears to have declined, likely due to an increase in neonate mortality, and considerable oscillation in apparent productivity meant that this assumption was no longer true (Michaud 2014; Mosnier et al. 2015). These changes prevented us from using the age-frequency distribution information to model the 1983-2002 and 2003-2016 period trends separately. Unfortunately reproductive data specific to this population are not available. Earlier work suggests that the productivity of SLE beluga may be lower than among Arctic beluga, and that calves are less likely to be detected than adults in the beach-cast stranding data (Gosselin et al. 2014; Mosnier et al. 2015). We found that the relatively stable trend observed prior to 2002, could be simulated by assuming that reproductive rates were 32% lower than indicated by the data from Alaska, or by assuming that only 30% of the calves that had died, compared to adults were found (i.e. adult carcasses were roughly 3 times more likely to be detected than calf carcasses). Unfortunately, we are unable to distinguish between either scenarios, but both scenarios suggest that a major factor limiting recovery of this population points to a lack of recruitment, rather than elevated mortality among older animals. Similarly, the declining trend observed after 2002 could be duplicated assuming that calf mortality was even higher than represented by the number of carcasses recovered, again pointing to the need to improve our understanding around factors affecting recruitment among SLE beluga.

An elasticity analysis can be used as a tool to understand the response of population growth to perturbations that affect vital rates. The elasticity pattern is composed of the relative contributions of survival and fertility to population growth that are grouped in biologically

meaningful ways for comparative analysis (Heppel et al 2000). Among long-lived species with low reproductive rates, adult survival rates may have among the highest elasticities but, the rates among adults might already be so high that no management alternative is likely to improve them (Heppel 2007). Within this context trying to increase survival among younger animals is likely to be more important in contributing to population growth. At the same time, efforts to assist compromised young juveniles are not likely to be as beneficial as for older juveniles or young adults from a conservation perspective because of their lower probability of survival.

Conservation translocation is the deliberate movement of organisms from one site, for release in another. It must be intended to yield a measurable conservation benefit at the levels of a population, species or ecosystem, and not only provide benefit to translocated individuals. Often translocation is considered to be the movement of several animals from an area where animals are abundant, to an area where populations are trying to re-establish themselves (IUCN/SSC. 2013). The return of animals in difficulty, from areas that they are normally not seen, to areas where they are abundant shares some characteristics with translocation in the sense that the objective is to release animals into areas and under conditions favouring their survival, and where, in the case of SLE beluga their survival will contribute to population recovery. Guidelines developed for use in translocation operations associated with the capture, care, release and subsequent monitoring of released animals to evaluate survival and success could also provide guidance to situations dealing with animals in difficulty.

Our analysis indicates that the highest reproductive value and population rate of increase elasticity lies with weaned juveniles, and that to stabilize the population, improvements in survivorship among juveniles are likely to be the most effective. However, considering the current rate of decline of the SLE population, a large number of animals would need to be successfully re-introduced into the population each year to stabilize it, and more animals would need to be rescued each year to reverse the current trend and lead to population growth. Our estimates are likely to be optimistic because they assumed that survival of compromised neonates and juveniles that are returned to their population were the same as healthy animals in the wild. However, survival of a neonate found hours after stranding alive is likely to be lower than healthy conspecifics.

The rescue of nursing neonates has no apparent population benefit. Animals in this age class already have a low probability of survival, which reduces to nil if animals are not able to rapidly consume milk (Brodie 1971; Matthews and Ferguson 2015). For animals in the wild, adoption by another female is highly unlikely, and this would need to occur in the few hours following reintroduction for the neonate to survive.

From an animal welfare perspective, relocating neonates or letting nature to take its course might result in increased distress and suffering, and ultimately mortality (Singer 2003). Under these circumstances, actions to reduce unnecessary animal suffering such as euthanasia should be considered (Harrington et al. 2013; DFO 2015), and followed by a necropsy to improve our understanding of potential causes for stranding. An alternative consideration would be to take live-stranded neonates into captivity. However, abandoned calves are considered unreleasable once in captivity under the new DFO guidelines for release and rehabilitation (DFO 2018).

Recognizing that multiple actions are needed to mitigate threats to stabilize the SLE population, relocating entrapped juveniles may be considered on other grounds than to meet Conservation Translocation objectives, given the higher likelihood of survival of these individuals following relocation. Effective translocation programs require thorough pre-release planning, and post-release monitoring such as conducting health screening, assessing habitat quality at the release location and tracking of individuals to monitor health and survival (Norris et al. 2017). For

animals in difficulty, opportunities for pre-release planning may be limited, or may require extensive adjustment depending on the health status of the animal. An evaluation by a veterinarian will provide insights into the general health status of the animals and probability of survival. If rehabilitation is considered as an option, then adequate resources must be identified for transport and care of the animal until health status has improved to permit release. At the same time, consideration must also be given to the expected timeline for rehabilitation, since the chances for release decrease with increasing time in captivity (DFO 2018).

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