Status of the belugas of the St Lawrence estuary, Canada.

Michael C.S. Kingsley

Greenland Institute of Natural Resources, P.O. Box 570, DK–3900 Nuuk, Greenland.

ABSTRACT

A population of belugas (*Delphinapterus leucas*) inhabiting the estuary of the St Lawrence river in Quebec, Canada, was depleted by unregulated hunting, not closed until 1979. Surveys in 1977 showed only a few hundred in the population. Surveys since then have produced increasing estimates of population indices. An estimate of the population, fully corrected for diving animals, was 1,238 (SE 119) in September 1997. The population was estimated to have increased from 1988 through 1997 by 31.4 belugas/yr (SE 13.1). Observations of population age structure, as well as data on age at death obtained from beach-cast carcasses, do not indicate serious problems at the population level, although there are indications that mortality of the oldest animals may be elevated. Few animals appear to live much over 30 years. From examination of beach-cast carcasses, it appears that most deaths are due to old age and disease; hunting is illegal, ship strikes and entrapments in fishing gear are rare, ice entrapments and predation are unknown. Among beach-cast carcasses recovered and necropsied, about 23% of the adults have malignant cancers, while most of the juveniles have pneumonia; other pathological conditions are diverse. No factors are known to be limiting numbers of this population. Habitat quality factors, including persistent contaminants, boat traffic and harassment, may affect the population's rate of increase, but these effects have not been quantitatively evaluated. Comprehensive legislation exists with powers to protect the population and the environment of which it is a component, but application and enforcement of the laws is not without problems.

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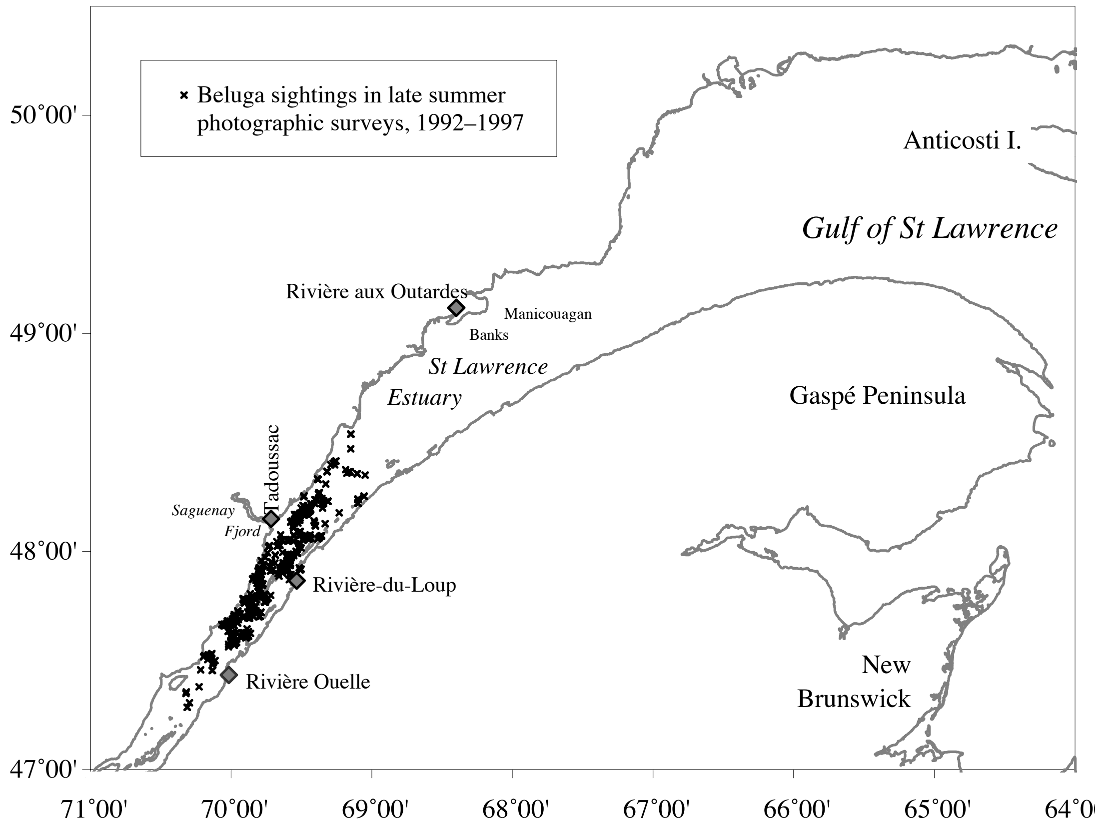
INTRODUCTION

A population of beluga whales summers in the estuary of the St Lawrence River. The zone principally frequented by this population in mid-summer lies between about 47° 30' N and 48° 20' N in the central estuary (Pippard 1985, Sergeant and Hoek 1988, Kingsley and Hammill 1991, Michaud 1993, Kingsley 1993, 1996, 1999) (Fig. 1). The summering area also includes the lower 15 miles or so of the Saguenay river, a tributary to the St Lawrence estuary which enters through a deep, oligotrophic fjord on the north shore (Pippard 1985). The population winters further downstream in the estuary and in the Gulf of St Lawrence; but winter surveys have not succeeded in defining the extent and location of the wintering areas (Boivin and INESL 1990).

The pristine distribution of this population is also uncertain. Pippard (1985) following Vladykov (1944) shows the original distribution as including the lower estuary and the north shore of the Gulf of St Lawrence, but the documentary evidence for this distribution is not

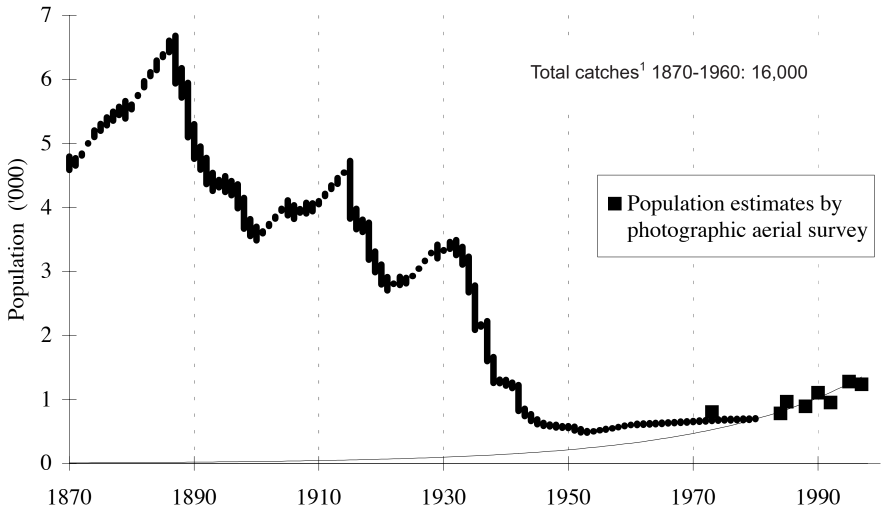
Fig. 1. The lower St Lawrence estuary and northwestern Gulf of St Lawrence, showing the summer distribution of beluga sightings.



stated, and the sparse settlement along that coast provided few records. de Charlevoix (1744) distinguished porpoises in salt water from those in fresh water, which were white and the size of a cow. Early writers concurred that the distribution extended as far upstream as Quebec City (de Charlevoix 1744, St-Cyr 1886), but all agreed that the main distribution was in the central estuary (Lescarbot 1914) with major fisheries from the Rivière aux Outardes to the Rivière Ouelle. According to de Charlevoix (1744) belugas were numerous on the coasts of Acadia, corresponding to the western and southern Gulf of St Lawrence.

The beluga was long regarded as a major exploitable resource of the central estuary. For example, in 1723 it was prohibited to fish eels near beluga weirs: "the king wants as many fisheries of this kind as possible" (St-Cyr 1886). Unregulated commercial and recreational hunting, as well as control measures intended to protect fisheries, diminished the population (Pippard and Malcolm 1978, Laurin 1982, Reeves and Mitchell 1984). Studies in the early and mid- 1970s reported remaining numbers in the low hundreds and all hunting was closed in 1979. In 1983, the Committee on the Status of Endangered Wildlife in Canada classified the population as 'Endangered' (Campbell 1985) on the basis of an assessment that it numbered no more than 350 and was still rapidly declining (Pippard 1985). In the mid-1980s, the size of the population, its trend, and the perceived negative effect of anthropogenic contaminants and disturbance on the population, were controversial. At one extreme the population was said to number possibly as few as 250, to be in poor health, and to be declining at 6%/yr (Béland and Martineau 1985, 1986, Béland *et al.* 1987). At the other, a series of surveys had given photographic index estimates—only partly corrected for visibility—of about 500, with no discernible trend in numbers (Sergeant and Hoek 1988).

Pathological conditions were found in beach-cast carcasses in the early 1970s. Contaminants, notably lead, mercury, and persistent chlorinated organic compounds of various kinds, were



[1]**Catch data:** 1870-1960 Reeves & Mitchell (1984); 1961-1997 Laurin (1982) and Pippard & Malcolm (1978). No catches after 1980.

[2] **Assumptions:** A closed population, with no immigration or emigration, has had a density independent growth rate of 4%/yr. The harvest estimates in the referenced publications have been considered reliable. Recent data on the visibility of belugas (Kingsley and Gauthier 2002) has been used to correct aerial survey data.

[3] **Method:** A population trajectory was fitted to recent population estimates. For each year from 1980 back to 1870, the year's catches (shown by vertical bars) were added to the population, and the year's population increase (at 4%/yr) subtracted, to estimate the previous year's population.

Fig. 2. *Estimated trajectory*[2,3] *of the population of belugas inhabiting the St Lawrence estuary, back-calculated from estimated commercial catches.*

found at high levels in tissues (Martineau *et al.* 1987), and were said to be a principal cause both of the pathological conditions found in dead belugas and of their deaths (Béland and Martineau 1986). The mean age at death apparently remained high at about 18 years. The life expectancy deduced from stranded carcasses was so high that the population could not have been stationary unless birth rates had been about 50% of those estimated for an Alaskan stock (Béland *et al.* 1988). Although there was not, in fact, good evidence that the population was stationary, lowered birth rates were considered a probable effect of contaminants.

The population has been monitored since 1982 with a sustained programme of aerial surveys flown on average about every 2 years through 1997. Carcasses, mostly beach-cast, a few drifting at sea, were reported at an average rate of about 15/yr. As many as possible were recovered and examined, some undergoing a full laboratory necropsy. Age data for beach-cast carcasses is nearly complete, but is biased by tooth wear and by the loss of teeth with advancing age, which prevents estimating the age of the oldest animals (cf Vladykov 1944, Plate 20).

This review of the status of St Lawrence belugas considers population size, trend, structure and dynamics. It discusses factors that could potentially reduce population growth rate, and summarises protective measures now in place.

POPULATION SIZE AND TREND

The original size of the stock is unknown; its size was first estimated in 1973 (Sergeant and Hoek 1988). Although it was heavily exploited, there was no licensing or fishery control, and therefore no fishery statistics. From commercial records of catches and sales of beluga oil and hides, it has been estimated that about 16,000 belugas were taken from the St Lawrence between about 1870 and 1960, and possible trajectories can be back-calculated to a population possibly in the low thousands (Fig. 2). When the exploiting companies ceased operations between about 1940 and 1950, even this data source disappeared.

Table 1. Published indices of population size for St Lawrence belugas, 1977–1997.

Year	Published estimate (SE[1])	Index or Limit	Visual or Photo	Correction included	Source
1977	300	I	V	No	Pippard 1985
1982	512	I	V	No	Sergeant and Hoek
1988					
1984	431	I	P	18%	”
1985	530	I	P	18%	”
1985	<340	L[2]	V	Y	Béland *et al*. 1987
1988	491 (69)	I	P	15%	Kingsley and Hammill
1991					
1990	607 (308)	I	P	15%	”
1992	525 (71)	I	P	15%	Kingsley 1993
1995	705 (108)	I	P	15%	Kingsley 1996
1997	681 (91)	I	P	15%	Kingsley 1999

[1]tabulated values are estimated sampling standard errors, probably biased upward;
[2]this estimate was published as an upper bound on the absolute size of the population; all others were published as population indices, partly corrected for visibility.

The size of the population has recently been estimated or monitored by aerial sample surveys, flown in calm, clear weather usually in late summer. Visual surveys were conducted in 1977 and 1982, but all others used vertical aerial photography. Large-format mapping cameras were used, that took 9-inch-square (228.6 mm-square) frames (Sergeant 1986, Sergeant and Hoek 1988, Kingsley 1998, 1999). Positive colour transparency film was used. Surveys were initially flown at 3,000 ft (915 m), giving a photographed strip 1,371 m in width. In 1992 the target altitude was raised to 4,000 ft (1,220 m), giving a photographic scale of 1:8,000 and a wider photographed strip. Belugas were so easy to detect on photographs at this scale that

Table 2. Corrections to photographic population estimates for belugas in the estuary of the St Lawrence, 1984–1997 (from Kingsley 1999).

Year	Published index value		Estuary estimate	Remove partial correction factor of	Surface visible estimate	corrected by a factor of 209%[1]	Resulting corrected estimate for estuary
	Estimate (see Table 1 for source)	Includes Saguenay count					
1984	431	30	401	18%	339.8	370.4	710.2
1985	530		530	18%	449.2	489.5	938.7
1988	491		491	15%	427.0	465.3	892.3
1990	607		607	15%	527.8	575.4	1103.2
1992	525	3	522	15%	453.9	494.8	948.7
1995	705	50	655	15%	569.6	620.8	1,190.4
1997	681	20	661	15%	574.8	626.5	1,201.3

[1] Gauthier (1999); Kingsley and Gauthier (2002)

surveys in 1995 and 1997 were also flown at 4,000 ft.

The sampling design for photographic surveys since 1992 used transects aligned across the St Lawrence estuary and systematically spaced every 2 nautical miles (3.704 km) over an area that much exceeded the usual summer range (Kingsley 1993, 1996, 1999). Earlier surveys used a variety of designs, including transects oriented parallel to the estuary instead of across it (in 1984 and 1985); survey areas more closely tailored to the known range and (in 1985) to the previously determined locations of major concentrations; and (in 1985, 1988 and 1990) stratified sampling that was less intensive in marginal areas. In some years, a visual survey of the Saguenay fjord was included, in others not. For visual surveys, sample-survey estimates of surface-visible belugas were published, but photographic surveys were adjusted by adding 15% or 18% to the counts as a partial correction for visibility (Table 1, Sergeant and Hoek 1988, Kingsley 1996, 1998). The resulting published index estimates increased from about 300 in 1977 (Pippard 1985) to 680 – 700 in 1995–97 (Kingsley 1996, 1999) (Table 1).

A full correction factor for diving animals which could be applied to photographic counts to produce a population estimate was estimated at 209% (SE 16%) (Gauthier 1999, Kingsley and Gauthier 2002). The photographic surveys since 1984 (Table 1) used two different partial correction factors, both much smaller than the value that Gauthier (1999) showed to be appropriate, and some of them included visual counts in the Saguenay. To produce a consistent series of fully-corrected estimates, the following procedure was used (Kingsley 1999):

1. If the Saguenay fjord had been surveyed, the count in the Saguenay was subtracted from the total;
2. any partial visibility correction factor—18% added in 1984 and 1895 (Sergeant and Hoek 1988), or 15% in 1988 through 1997 (Kingsley and Hammill 1991)—was removed;
3. the resulting index of surface-visible belugas in the estuary was multiplied by 209% (Gauthier 1999, Kingsley and Gauthier 2002) to produce a fully corrected estimate for the St Lawrence estuary (Table 2);
4. these estimates were taken in sets of 5 consecutive surveys, and a linear model of population change (Zar 1996) was fitted to each set to generate a smoothed estimate and standard error for the central year of the set (and for the last two and first two years of the entire series) and a mean rate of change for the period (Table 3);
5. a single linear model was fitted to the four surveys made in the Saguenay (1984, 1992, 1995 and 1997) and used to generate (and interpolate) smoothed estimates and standard errors for all survey years, as well as a rate of change for the numbers sighted in the Saguenay (Table 3);
6. for each survey year, the smoothed corrected estimate for the estuary was added to the smoothed estimate for the Saguenay fjord to give an estimate for the entire population;
7. for each set of 5 consecutive surveys the rate of change for the estuary was added to the rate of change for the Saguenay fjord to produce a total average rate of change for the population for that period.

The most recent sequence of 5 surveys covered the years 1988 to 1997. The average population estimate over this period was 1,096 and the average linear rate of increase over the same period was 31.4/year (SE 13.1); the average relative rate of growth is about 2.9% (Kingsley 1999) (Fig. 3). For the most recent survey year, 1997, the smoothed estimate for the population was 1238, with SE 119. These SEs included most variability factors, including sampling and detection error, between-survey variations in weather, tide, etc., uncertainty in the correction factor, and errors due to fitting a linear model to the population trend.

These estimates of numbers apply only to the main summer range in the Saguenay fjord and the central estuary of the St Lawrence. They ignore belugas occasionally sighted, at about survey time, in the northern Gulf of St Lawrence and elsewhere outside the normal estuary summer range (Kingsley and Reeves 1998, Kingsley, unpublished data). Such sightings do not represent a significant proportion of the population.

Table 3. Growth models fitted to survey estimates of population size

St Lawrence estuary				Saguenay fjord			
Smoothing Period	Fitted population model	Resulting estimates		Smoothing Period	Fitted population model	Resulting estimates	Total (SE)
1984–1992	811.5 in 1984 + 28.2/yr	1984	812	1984–1997	24.9 in 1984 + 0.1/yr	25	836 (109)
"	"	1985	840	"	"	25	865 (99)
"	"	1988	924	"	"	25	950 (90)
1985–1995	1,014.7 in 1990 + 23.6/yr	1990	1,015	"	"	26	1,040 (91)
1988–1997	1,054.6 in 1992 + 31.3/yr	1992	1,055	"	"	26	1,080 (92)
"	"	1995	1,149	"	"	26	1,175 (104)
"	"	1997	1,211	"	"	26	1,238 (119)

POPULATION STRUCTURE AND DYNAMICS

Collections from this population have not been possible, and the principal source of information has been beach-cast carcasses of naturally dead belugas (Béland *et al*. 1988, Department of Fisheries and Oceans unpublished data). These have furnished data on age at death and on levels of reproductive activity. The standard for comparison has been hunted or netted samples from Arctic populations. The two sampling methods have different biases: sampling of beach-cast carcasses is biased toward the sick and the old, while hunted samples are biased away from small juvenile animals (Burns and Seaman 1986), but probably toward healthy animals. The different sampling biases should be borne in mind in making comparisons. The proportion of young animals counted in surveys provided additional information on population structure, but is biased by the relative difficulty of seeing small darker juveniles compared with large white adults (Richard *et al*. 1994, Gauthier 1999, Kingsley and Gauthier 2002).

Mature female belugas in prime condition normally have a 3-year reproductive cycle, although 2-year intervals may occasionally occur, but pregnancy rate decreases in older animals (Brodie 1971, Sergeant 1973, Burns and Seaman 1986). In beach-cast females in the St Lawrence in the 1980s, evidence of recent reproductive activity was less frequent than would be consistent with this rate (Béland *et al*. 1988, 1992, 1993). However, the beach-cast sample probably gives a negatively biased estimate of the overall average birth rate in the live population (Béland *et al*. 1993).

Grey juveniles were counted from boats (Michaud 1993), and small belugas were counted on aerial photographs (Kingsley 1996); both studies showed proportions of juveniles that were near normal expectations.

On average, about 12.9 beach-cast carcasses were aged per year from 1988 through 1998 and a further 1.8 were classified as adult, but not aged (Fig. 4). The estimated average population over that period was 1,112 (Kingsley 1999). A life table for an Alaskan population was constructed from adult age structure and reproductive rates, corrected for hunters' bias away from young animals (Burns and Seaman 1986). Deaths were approximately 29% first-year, 24% older juveniles, and 47% adults (Table 4).

Applying the Alaskan population model to an average population of 1,112 belugas in the St Lawrence would estimate 105 neonates in the population, 30.6 of them dying each year; 307 older juveniles with 24.7 deaths/yr, and 701 adults, with 49.0 deaths/yr. In the St Lawrence strandings record between 1988 and 1998 (Fig. 4) there were on average 1.2 first-year animals beach-cast per year, 1.2 older juveniles, and about 12.4 adults. The ages at which beach-cast adults died (mean 21.4 years) showed a life expectancy at age 6 of 15.4 years, implying an average adult death rate of the order of 6.5%/yr, not different from Burns and Seaman's estimate.

The numbers at all ages in the stranding record were less than those predicted by the Alaskan model for a population of the size of that in the St Lawrence (Table 4). Either the population estimate was too high, death rates in the St Lawrence were much lower than the model, or a high proportion of carcasses were not recovered. It is unlikely that population estimates were much too high, as surveys presented a consistent series of estimates (Kingsley 1999) and the visibility correction factor estimated from direct observation in the St Lawrence was consistent with the values obtained from tagging studies on Arctic populations (Kingsley and Gauthier 2002). It was unlikely that adult death rates in the St Lawrence were much lower than the Alaskan model, because the life expectancy at maturity in the St Lawrence was similar to that from Alaska. However, it is not unlikely that many, or most, carcasses are lost. Losses would not be proportionally distributed over all age classes or all causes of death and the recovered sample might likely be biased.

The age structure of the strandings record is different from that predicted by the Alaskan model, with a smaller proportion of neonates and ju-

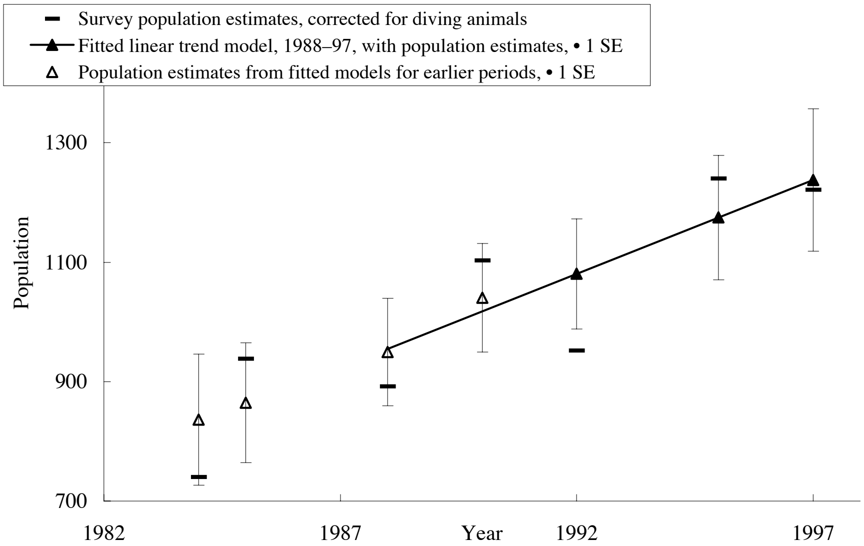


Fig. 3. *Survey population estimates for the belugas of the St Lawrence estuary corrected for diving, with fitted model values and linear trend 1988–97 (after Kingsley 1999).*

Fig. 4. *Age distribution of beach-cast carcasses of St Lawrence belugas.*

Table 4. Age distribution of beach-cast carcasses in the St Lawrence compared with an Alaskan population model.

	Alaskan population model			St Lawrence observations
	Age class as % of standing stock	Annual death rate	Deaths/yr/'000	Strandings/yr/'000
First year	9.4	29.4%	27.5 (29.4%)	1.1 (8.1%)
Age 1–5	27.6	8.0%	22.2 (23.7%)	1.1 (8.1%)
Adult	63.0	7.0%	43.9 (53.1%)	11.1 (83.8%)
Total			93.6	13.2

veniles and more adults (Table 4). All sub-adult deaths (neonates and older juveniles combined) were 53% of total deaths in the Alaskan model, but composed only 16.2% of the age-classified carcasses in the St Lawrence. Birth rates in the St Lawrence may have been lower than the Alaskan model so few young were born relative to the number of adults; alternatively, or as well, juvenile death rates may have been low. Proportions of juveniles in the live population are near normal (see above). To reconcile this observation with a small number of juvenile deaths would require both a low birth rate and very high juvenile survival. A more probable reason for the lack of juvenile carcasses, however, is sampling bias, as neonate and juvenile carcasses would be more likely than adults to sink before stranding or disappear before being found (Béland *et al*. 1988).

At ages over 19 years death rates in the St Lawrence appeared to increase rapidly resulting in a mode in the death schedule at about 25 years, whereas in the Alaskan hunted sample death rate was estimated to remain constant until about 30 years' age was reached (Fig. 5), and the death rate was still only 27%/yr at the age of 37. This difference may have been partly due to different incidence of tooth wear, apparent age of St Lawrence belugas being possibly limited by wear, but Béland *et al*. (1988) found the effect of tooth wear to be small. The Alaskan population profile showed no peak in adult deaths, and little evidence of age-related mortality at any adult age. The St Lawrence strandings data were probably subject to sampling bias, but did not show a large number of deaths relative to the size of the population, nor a high proportion of deaths occurring at young ages. However, the population was hunted until 1979, and the catch composition was never recorded and is unknown. Effects of hunting can persist (Whitehead *et al*. 1997), and generation length (Krebs 1994) for belugas is about 15 years, so the population structure and dynamics of this population may have been changing for much of the period since hunting was closed, and may still not have reached a steady state.

LIMITING FACTORS

There are several factors that have been suggested as possibly affecting population dynamics. Among them are habitat modification by alteration of river discharges; inadequate food supplies; mortality increases or birth-rate reductions due to contaminants; and mortality or disturbance due to marine traffic. However, the natural maximum rate of increase of monodontid populations is thought to be only about 5% (Sergeant 1981) or less (Kingsley 1989), or perhaps as low as 2% (Beland *et al*. 1988). It is therefore difficult to measure, and changes due to human activity are hard to detect.

Food stocks

From analysis of stomach contents of hunter-killed belugas in summer, important summer foods were found to be capelin (*Mallotus villosus*), sand-lance (*Ammodytes* spp.), Atlantic cod (*Gadus morhua*), tomcod (*Microgadus tomcod*), decapod and amphipod crustaceans, and polychaete worms (*Nereis* spp.), as well as several other species. However, the collection season was short, and local ecological observations indicated that herring (*Clupea harengus*) or smelt (*Osmerus mordax*) could also be important, at other seasons (Vladykov 1946). At Churchill, Manitoba—another sub-Arctic area—belugas also fed heavily on capelin, and *Nereis* were thought to be important and squid a staple food (Doan and Douglas 1953). Recent

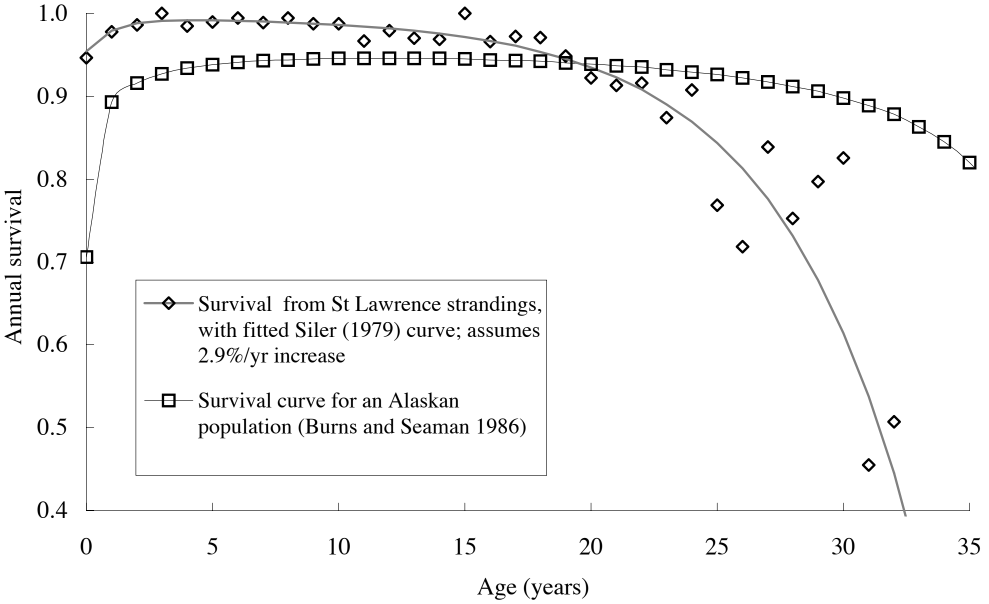


Fig. 5. *Survival rates inferred from ages of beach-cast St Lawrence belugas assuming no sampling bias, compared with an Alaskan live-sampled population.*

information from the St Lawrence is scanty, as the stomachs of beach-cast carcasses are almost always empty.

Combining the population age structure suggested by Burns and Seaman (1986), an age-length relationship (Kingsley 1996) and a length-weight relationship (Doidge 1990), belugas are estimated to average 495 kg in weight. A population of 1,240, eating (say) 2% of body weight per day, would require about 4,500 t of food per yr. However, it is not known how food requirements are distributed in space and time.

Biomass estimates are not available for all the species identified as important to the beluga, even as aggregated estimates for the entire Gulf and estuary of the St Lawrence. Low-trophic-level fishes of little commercial importance such as capelin and sand-lance abound, but total biomass has not been estimated. In the ecosystem of the Gulf and estuary of the St Lawrence, biomass and annual production are measured in hundreds of thousands to millions of tons per species (Cairns *et al*. 1991, DFO 1999, 2000a, b, c, d). The small population of belugas is probably not constrained by a shortfall in the few thousand tons per year it requires, and no observations have suggested that it is. However, competition for food, as well as standing biomass, is measured on these large scales: other predators may take as much as one million tons per year from the capelin stock (DFO 1999). Informative, quantitative studies on beluga feeding ecology have not been carried out, and interpreting results of such studies in terms of a population limit would be difficult in view of the small niche occupied by the population. Competition from other predators might increase if a climatic shift were to extend the season during which the St Lawrence ecosystem is open to seabirds and non-ice-adapted marine mammals.

River discharges

Between the 1950s and the 1970s the St Lawrence belugas apparently abandoned the Manicouagan banks, a shallow-water area near the downstream limit of the then summering grounds (Fig. 1), where in the 1940s they were numerous enough to support a fishery (Vladykov 1944). It was suggested that this change in distribution could have been due to the harnessing of these rivers with hydro-electric dams in 1965–70 with consequent alteration of their discharge regimes, including reduction of peak flows and, possibly, lower peak summer temperatures (Sergeant and Brodie 1975, Pippard 1985). Similar questions have been raised for Arctic populations: whether flow-regime alterations due to hydroelectric development on rivers whose estuaries are beluga summer habitat might affect summer distribution, movement, or population dynamics (Lawrence *et al*.

1992). However, the reduction in population numbers by hunting (Pippard and Malcolm 1978, Laurin 1982, Reeves and Mitchell 1984) was another possible cause of this change in distribution, and the possible effects of persistent contaminants captured public concern more insistently. Even in Arctic situations, with fewer confounding influences, it proved difficult to design definitive research studies on the downstream effects of hydro dams on estuary use by belugas (Lawrence *et al*. 1992). So for various reasons, this remains an unresolved question—but the Manicouagan banks are still unfrequented.

Contaminants

The effects of persistent contaminants from agriculture and other industries on the belugas of the St Lawrence was a concern in the 1970s and continued to be so in the 1980s. A juvenile beluga found floating dead in the early 1970s had about 800 ppm each of DDT and total PCBs in the blubber (Sergeant 1986). The presence of high levels of organochlorines and of lead and mercury has been associated with the deaths of belugas in the St Lawrence and with the presence of various pathological conditions in beach-cast carcasses, notably cancers (Béland *et al*. 1992, 1993, Béland and Martineau 1986, de Guise *et al*. 1995, Martineau *et al*. 1987, 1999). Examination seldom assigns a cause of death, and pathological conditions are diverse (Table 5), with few recurring syndromes, but may be summarised thus:

1. juveniles found dead are usually suffering from pneumonia, which is often the cause of death and is often a verminous form;
2. malignant neoplasms are found in about 19% of carcasses or 23% of adults (mean age 23.4 yrs); other conditions are diverse (Béland *et al*. 1992, 1993).

It is not clear to what extent the pathological conditions found in belugas are abnormal, as there is little basis for defining a normal condition for aged cetaceans.

In the 1980s and 1990s, organochlorine levels were measured in blubber samples taken from beach-cast carcasses (Martineau *et al*. 1987, Muir *et al*. 1990, 1996a, 1996b). PCB and DDT levels were variable between individuals, from low tens into the low hundreds of ppm wet weight in blubber Levels were not extremely high for long-lived cetaceans from industrialised waters, but similar to levels that have caused physiological stress in laboratory animals. Samples from the early 1990s showed little change in mean levels in belugas, although other fauna from the St Lawrence generally had lower levels than in the 1970s (Muir *et al*. 1996a, b). Lead and mercury have been measured in kidney, liver, and muscle, and were also high (Wagemann *et al*. 1990), while cadmium is lower in the St Lawrence belugas than in Arctic belugas. The presence of contaminants is not in doubt, and *in vitro* studies have shown that some of these contaminants affect cellular components of the immune system of Arctic belugas (de Guise *et al*. 1995, 1998).

Polycyclic aromatic hydrocarbons (PAHs), which are strongly carcinogenic, are present in abnormal concentrations in sediments in some peripheral habitat areas, notably in the upper reaches of the Saguenay (Cossa 1990, DFO 1997). It has proved difficult to demonstrate that belugas are excessively exposed to these compounds, partly because they are quickly metabolised by mammals.

The population-level effects of contaminants are unknown. St Lawrence belugas are not known to die often from being hit by ships, caught in fishing gear, or entrapped in ice (Martineau *et al* 1994). Hunting is banned, and predation is unknown. Beach-cast carcasses of belugas in the St Lawrence are in general old, and have suffered from various pathological conditions, many of which might be associated with or aggravated by toxic contaminants or their immune-suppressive effects. However, existing information is hard to interpret without studies relating contaminants to immune system perturbations in material from the St Lawrence, immune system perturbations to population dynamics, and contaminant levels to pathologies within the St Lawrence population. There is also little information on pathological conditions in naturally-dead belugas of similar age from uncontaminated populations.

Controlled dose-response and exposure-response studies are essentially non-existent for cetaceans, and most of the inferences of contaminant effects on St Lawrence belugas have been drawn from a weight-of-evidence approach which is often of debatable validity (Addison 1989). Inferences tend to rely on '*post hoc, ergo propter hoc*' reasoning, comparing features of the beach-cast St Lawrence sample with observations from other populations sampled differently.

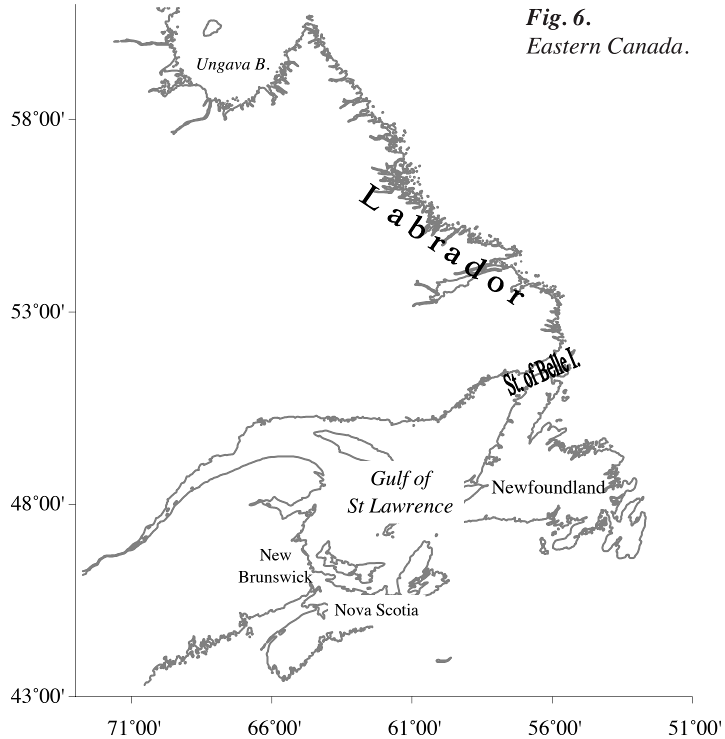


Fig. 6.
Eastern Canada.

A salient feature of the pathological record from beach-cast carcasses is the presence of cancers. Fifteen cancers were found in a variety of organs in 14 animals necropsied between 1983 and 1994, representing about 14% of those necropsied (Table 4; Martineau *et al*. 1999). Over this period the population averaged about 970 (Table 3). This indicates a minimum annual incidence per 100,000 of about 120, but rigorous quantitative interpretation is not possible inasmuch as strandings are a partial record of deaths and skewed towards the older animals, which are more likely to develop cancers. This rate would not be exceptional in humans and some domestic animals (Martineau 1999), but cancer is seldom observed in wild mammals. It has been suggested that cancer in St Lawrence belugas may be caused by exposure to PAHs, operating in conjunction with an immunosuppressive effect of persistent organochlorines (de Guise *et al*. 1995, Martineau *et al*. 1999).

Table 5. Malignant neoplasms in 14 of 73 necropsied carcasses of beluga whales beach-cast in the St Lawrence estuary between 1983 and 1994 (Martineau *et al*. 1999 Table 2).

Age	Sex	Year	Organ	Cancer type
17	M	83	Urinary bladder	Transitional cell carcinoma
29+	M	89	Intestine	Adenocarcinoma
20+	M	89	Intestine	Adenocarcinoma
25+	M	93	Intestine	Adenocarcinoma
27+	M	94	Intestine	Adenocarcinoma
27+	F	94	Intestine	Adenocarcinoma
21+	F	88	Stomach	Adenocarcinoma
27+	M	94	Stomach	Adenocarcinoma
24	M	86	Salivary gland	Adenocarcinoma
22+	F	88	Liver	Adenocarcinoma
same animal as above			Mammary gland	Adenocarcinoma
24.5	F	85	Ovary	Granulosa cell tumour
21+	F	88	Ovary	Granulosa cell tumour
25+	F	89	Ovary	Dysgerminoma
18+	M	90	Mediastinum	Poorly differentiated malignant neoplasm

Marine traffic and harassment

The St Lawrence is heavily travelled by heavy freight traffic, crossed by ferries, and visited by cruise liners. It is a popular waterway for recreational boating, and home to a signifcant whale-watching industry. Within the belugas' summering area, freight traffic mostly follows the north channel, and whale-watching is concentrated in the north-eastern part of the summering area. Recreational boating tends to be concentrated near marinas at Rivière-du-Loup and Tadoussac. The creation of a Marine Park centred on Tadoussac has provided a measure of protection, but has also increased tourist traffic. The effect on belugas of these diverse kinds of marine traffic is not easily measurable (Blane 1990). Fewer belugas were reported to pass into the Saguenay Fjord in the 1980s, presumed a result of increased traffic at the mouth of the fjord (Caron and Sergeant 1988). Deliberate harassment occurs, as does unintentional disturbance, but the frequency of such events is hard to measure. Belugas appear well adapted to slow, noisy, and regular traffic, but are at risk from fast-moving small boats including inflatables. Propeller strikes and hull strikes occur (Martineau and Mikaelian 1997). They are rare in the stranding record (2 propeller strikes and one possible hull strike, all in recent years).

Immigration and emigration

The St Lawrence population is considered distinct from others of the species. Belugas are sighted outside, or in peripheral areas of, the main range, for example in the north-eastern Gulf of St Lawrence, the Strait of Belle Isle, or off the coasts of Labrador, Newfoundland or Nova Scotia (Curren and Lien 1998, Kingsley unpublished data) (Fig. 6). Curren and Lien (1998) tracked down 37 sightings in the 13 years 1979 – 1991, of which 16% were on the Labrador, 40% on the east coast of Newfoundland, 14% on the west coast of Newfoundland and 5% in the southern Gulf. The source of most sightings is unknown, but 2 beach-cast carcasses on the east coast of Newfoundland were identified as Arctic animals by their contaminant signatures (Béland *et al*. 1992), and a juvenile which took up residence on the east coast of Nova Scotia (and probably went there with its mother) was genetically identified as a St Lawrence animal. The continuous string of sightings (Curren and Lien 1998) along the eastern coasts of Labrador and Newfoundland probably implies that they are from stocks in the eastern Canadian Arctic, especially if the chemical identification of Newfoundland strandings as Arctic animals is taken into account. There is no reason to suppose that they ever join the St Lawrence population, but no other fate is known for them either. Animals seen in northern New Brunswick and other parts of the south-western and southern Gulf, on the other hand, are probably from the St Lawrence population, but not necessarily permanently lost to it. Occasional sightings in the north-eastern Gulf, the Strait of Belle Isle and on the west coast of Newfoundland could derive from either source. Permanent emigration from the population is probably no more than 1 to 2 animals per year on average, if so much, which would be a small proportion of the estimated net annual increase.

The nearest summering stocks are those in Ungava Bay in the northeastern Canadian mainland, depleted almost to non-existence (Kingsley 2000) and in southern Baffin Island, also small (NAMMCO 2000). However, any migrants to the St Lawrence would more probably come from a failure of some few among the several tens of thousands that winter in Hudson Strait to find their way back to their summering area. Observations of belugas on the Labrador coast and in the Strait of Belle Isle are few and do not support the thesis that there is any significant immigration.

PROTECTION

Marine mammals are a Federal responsibility under the Canadian constitution, and a number of pieces of Federal legislation are available to protect the St Lawrence belugas and their habitat. Under the *Fisheries Act* the Minister of Fisheries and Oceans has comprehensive powers to regulate fisheries, including fisheries for marine mammals, and to protect habitat for (by implication) fishable resources. The *Marine Mammal Regulations* under the Fisheries Act forbid disturbance of marine mammals in general and hunting of belugas in the St Lawrence in particular. The population was reduced mainly by uncontrolled hunting, and these Acts have removed that threat. The *Oceans Act* gives the Minister of Fisheries and Oceans responsibilities for environmental protection in the marine environment and powers to protect marine areas for reasons not connected with fisheries. The *Canada Wildlife Act* gives the Minister of the Environment powers to protect marine areas for species conservation.

Contaminant threats to beluga habitat have in the past not been well controlled; toxic substances find their way into fish habitat without being discharged there. Other Federal laws regulate toxic substances. The *Canadian Environmental Protection Act* is concerned with regulating toxic substances. It regulates the preparation, use and disposal of toxic products, and requires that information on toxicity should be acquired, or created by testing and analysis. The *Canada Shipping Act* regulates *inter alia* discharges of waste, and construction standards for ships, including special provisions for ships carrying toxic or dangerous substances. Although the marine environment is a federal domain, the control of important influences on the St Lawrence such as farming practices and municipal water treatment are largely under provincial jurisdiction.

Marine traffic may be deleterious to the belugas' welfare. The St Lawrence is a major commercial waterway, but large commercial vessels are not thought to be a major hazard. The central estuary is also a scenic area and a popular recreational waterway. A large whale-watching industry based mainly at Tadoussac is focussed on fin whales that congregate to feed on krill concentrated by currents off the mouth of the Saguenay. Belugas are frequently found in this area, and may be hit by fast-moving small boats or plagued by spectators. The creation of the Saguenay-St-Lawrence Marine Park (Saguenay-St-Lawrence Marine Park Ac*t* 1998) has placed about half the current summer habitat of the St Lawrence belugas under protection (Kingsley 1999). Park regulations control the behaviour of small vessels, forbid seeking belugas for whale-watching, and can limit the number of boats in a particular area.

It has taken time for belugas to be protected under endangered-species legislation, and at the time of this writing the process is still not complete. Under the Canadian constitution there are difficulties in framing such legislation comprehensively at the federal level as most terrestrial wildlife and its habitat are under provincial jurisdiction; while federal endangered-species legislation that has been drafted within the constitutional limits on federal competence has been perceived as weak and inadequate (*e.g.* Austen 1997, Kondro 2002), and therefore delayed.

While legislation is in theory quite comprehensive, its practical enforcement is not always easy. If actions are defined as illegal in terms of their effects, it is often difficult to prove that such effects have occurred: 'harassment' may be declared illegal, but yet remain difficult to identify. Substances that have legitimate uses may find their way into aquatic habitats, where they are deleterious, without being intentionally discharged there.

CONCLUSIONS

1. The best estimate of population size in the most recent survey year, 1997, is 1,238 (SE 119). The best estimate of population rate of change over the most recent 5 surveys, which were all carried out with similar methods, is 31.4 per year with SE 13.1. Both the population estimate and the estimate of growth rate are biased downward.

2. Population dynamics information is not conclusive, but does not show any large or unusual mortality factors, nor serious failures in reproduction. Mortality of old animals appears higher than in an Alaskan population. The St Lawrence population appears to be in condition to maintain its numbers if restrained by a limiting resource, or to increase its numbers if it is not limited by resources.

3. There are no factors known to be limiting numbers. No resource needed by the population has been identified as limiting its growth. There are some habitat factors, including marine traffic and pollution, that are probably detrimental to population growth, but their effects are difficult to quantify and have not been quantified.

4. Legal protection is in theory comprehensive. In practice, enforcement is sometimes difficult because it is difficult to prove the effects of some actions on the population.

5. The population does not meet IUCN (1996) criteria for 'Endangered' status, which are less than 250 mature animals or decreasing at 50% in 3 generations; for belugas about 1.25%/year. It is not small enough, and is not decreasing. It does meet IUCN criteria for 'Vulnerable' status.

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