STOCK STATUS REVIEWS

Contents

Annex 1: Western Okhotsk, or Sakhalin-Shantar, beluga population (Russia):
Sakhalin-Amur, Ulbansky, Tugursky and Udskaya summer stocks

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Note:
Below is provided the assessment of a biological population of the beluga, which in summer occupies the bays of the Western Okhotsk Sea.

*At the GROM meeting, it was agreed that the reoccurring "**summer aggregations"** should be considered separate "**stocks**" when geographic or genetic isolation was demonstrated and, in future, should be assessed and managed as independent units.*

The Western-Okhotsk beluga population assessment initially prepared for the GROM meeting (below) is supplemented with the summaries of the summer stocks as designated by the GROM expert joint agreement: 1) Sakhalin-Amur (presumably, includes Nikolaya bay); 2) Ulbansky; 3) Tugursky (a putative summer stock, genetic isolation of which from Udskaya stock is not confirmed); 4) Udskaya.

1. Distribution and stock identity

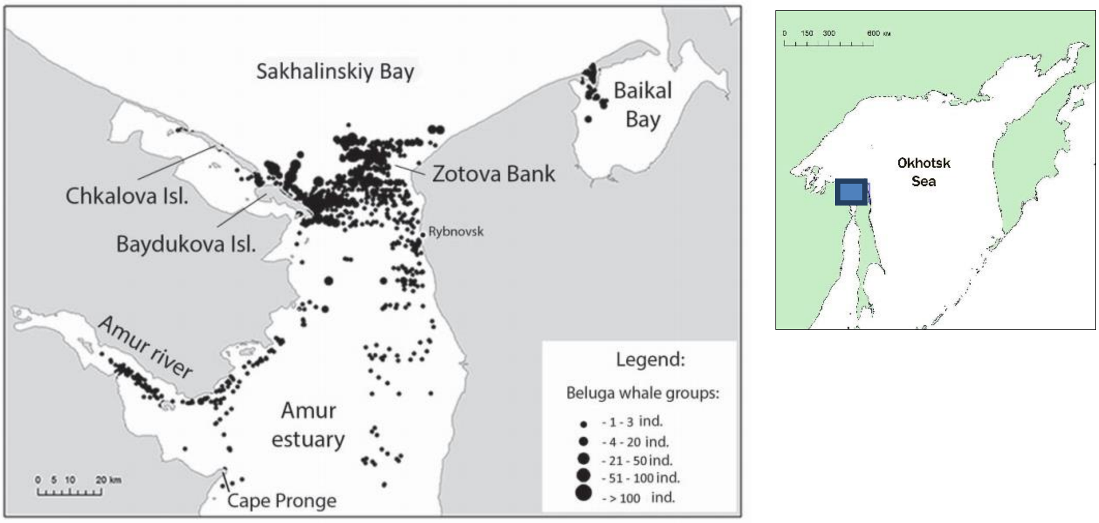
The Western-Okhotsk beluga population, previously thought to consist of two – Sakhalin-Amur and Shantar – stocks (IWC, 2000), has been extensively studied starting 2007. Based on aerial surveys and coastal observations (Solovyev et al. 2015), the population in summer is divided onto several nursery aggregations, which concentrate in shallow waters of Sakhalin-Amur and Shantar regions (Figure 1: 1 and 2, respectively). The population identity was confirmed by genetic studies (see below), according to which all belugas summering in the western Okhotsk Sea share a single nuclear gene pool and thus represent a single population. Within this population, belugas summering in different areas may be preliminary subdivided into 3 demographic units: 1) Sakhalin-Amur and Nikolaya bays, 2) Ulbansky Bay, and 3) Tugursky-Udskaya Bays (Yazykova et al. 2012, Meschersky et al. 2013). The status of belugas observed in Nikolaya and Tugursky bays remains to be confirmed: increasing genetic sample size in both areas and satellite tracking are required.

Summer distribution was studied in 2007-2013 (Figure 1). In Sakhalinsky Bay, same individuals were re-sighted in different years and within seasons (Shpak et al., 2014). In other Shantar bays we also re-sighted belugas both, intra- and inter- seasonally (our data, unpubl.). Together with results of genetic analysis, our observations suggest that in summer belugas form at least semi-residential aggregations in different bays of the western Okhotsk Sea, and at least some individuals return to the same bays summer after summer. Exclusion was when in the middle of summer in Nikolaya Bay we found two belugas previously tagged in Sakhalinsky Bay (Shpak et al., 2014). It is known that starting September belugas may move from Sakhalinskiy Bay westward – to Nikolaya and Ulbansky Bays (Shpak et al. 2010, 2012), and it is possible that some individuals do so in summer. Genetic analysis of a limited sample from Nikolaya Bay (n=8) collected in summer (July) has not revealed differences between Sakhalin-Amur and Nikolaya Bay belugas (Meschersky et al. 2013). If belugas observed in Nikolaya Bay in summer form a separate aggregation remains to be discovered.

Single "migrants" or small groups, some – with immature individuals, were sometimes observed outside the places of major concentrations – between the bays or near the Shantar islands.

Russian-Japanese ship-based surveys conducted in July-August 2009-2010 mostly in pelagic waters of the Okhotsk Sea resulted in no sightings of beluga whales (Istomin et al. 2013), further confirming the hypothesis that in summer beluga distribution is limited to coastal waters and bays.

1.



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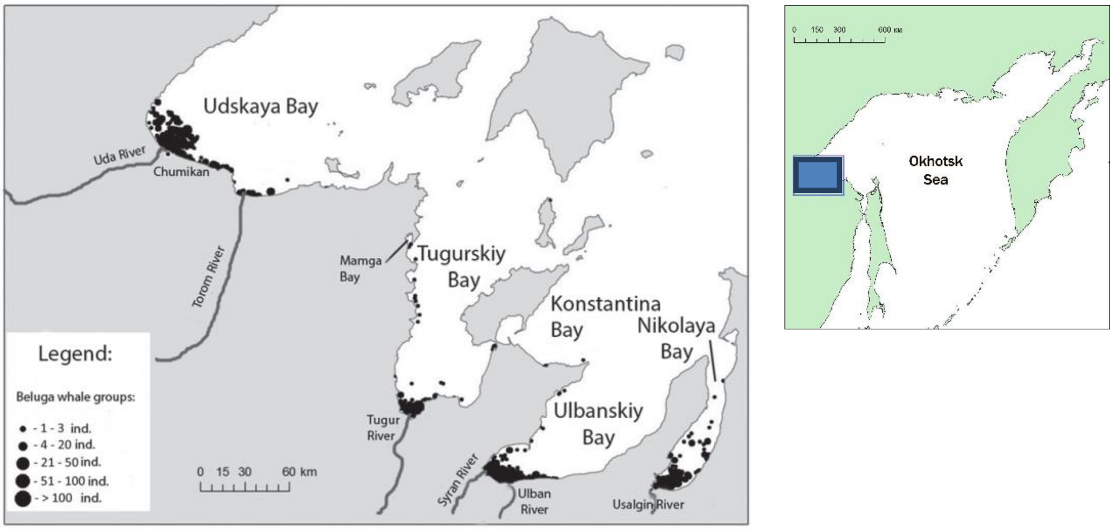

Figure 1. Summer sightings of belugas in Sakhalin-Amur (1) and Shantar (2) regions, visual observations, boat and aerial surveys, 2007-2013 (from Solovyev et al., 2015).

In winter, according to results of satellite tracking (Shpak et al. 2010, 2011, 2012, 2013), belugas move offshore to N and NE, but remain in the Sea of Okhotsk throughout the year (Figure 2).

Beluga whales – singletons or small groups (incl. mom-calf pair) – can be found in Sea of Japan (Sato and Ichimura 2011, Melnikov and Seredkin 2014, our interview data), but such cases are rare and should be considered as sightings outside the normal range.

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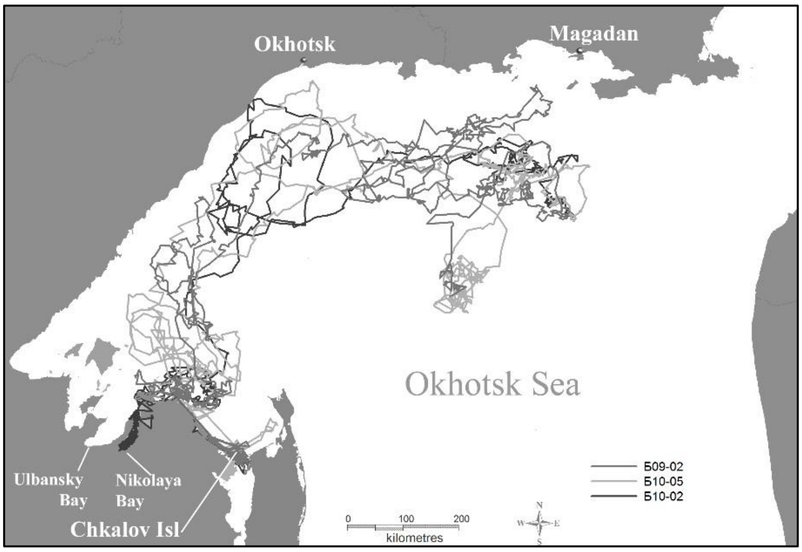

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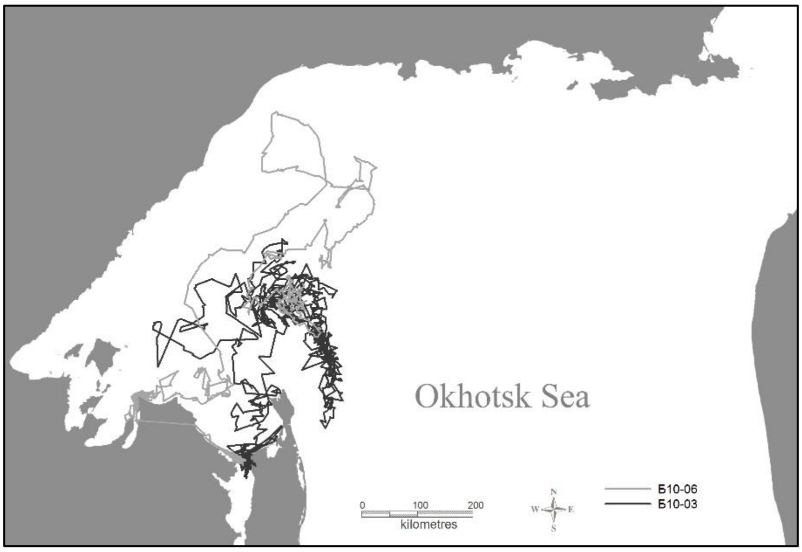

Figure 2. Satellite tracking of belugas from Sakhalin-Amur summer aggregation, 2009 ("Б09" whales) and 2010 ("Б10" whales). 1. Tracks of belugas, which migrated to "eastern" winter grounds. 2. Tracks of belugas, which migrated to "western" winter grounds (Shpak et al. 2012).

<u>Genetic studies</u>

Earlier published data (Meschersky et al., 2012 and Yazykova et al., 2012: analysis of ca. 130 samples from Sakhalinsky Bay and about 180 individuals biopsied in 4 bays of Shantar region; Meschersky et al., 2013: ca.100 individuals from Sakhalinsky Bay and about 40 individuals from Udskaya Bay of Shantar region) allowed to state a high level of genetic unity of these belugas and consider them as a single biological population (reproductive unit). At the same time, significant difference in mtDNA lineages (haplotypes) frequency is found between certain bays. This fact allowed to conclude that in summer period the population is divided into geographically isolated groups (spatial or demographic units). It was shown that the Western Okhotsk beluga population is significantly isolated from the other studied North Pacific beluga stocks and, probably, isolated from belugas summering off western coast of Kamchatka peninsula.

The statements presented here are based on analysis of the following samples:

- 184(141)[1] individuals from Sakhalinsky Bay (2004-2014, 99 males, 82 females, for 3 individuals the sex was unknown);
- 9(8) individuals from Nikolaya Bay (2009-2012, 8 males, 1 female);
- 90(86) individuals from Ulbansky Bay (2010-2012, 35 males, 53 females, for 2 individuals the sex was unknown);
- 32(27) individuals from Tugursky Bay (2010-2013, 25 males, 7 females);
- and 90(78) individuals from Udskaya Bay (2008-2015, 61 males, 29 females).
  (Samples were provided by O. Shpak, D. Glazov, D. Litovka, L. Mukhametov).

As genetic markers we used allelic composition of 17 microsatellite loci (Cb1, Cb2, Cb4, Cb5, Cb8, Cb10, Cb11, Cb13, Cb14, Cb16, Cb17 – Buchanan et al., 1996; Ev37, Ev94 – Valsecchi, Amos, 1996; 415/416, 417/418, 464/465, 468/469 – Schlötterer et al., 1991) and 559 bp sequence of mtDNA control region.

For comparative analysis, in addition to our data for other Russian beluga stocks, were used the data of analysis of 8(5) individuals from Norton Sound, Eastern Bering Sea (the samples were kindly provided by the Mammal Genomic Resources Collection, University of Alaska Museum of the North) and published data on frequency of mtDNA control region (409 bp) haplotypes (409 bp) known for Norton Sound (66 individuals, O'Corry-Crowe et al., 1997).

We used the whole sample of Sakhalinsky Bay and the whole sample of the entire Shantar region as two independent sets for inter-population analysis, and the samples representing separate bays – for analysis of intra-population structure.

The analysis of 17 microsatellite loci allele frequencies (Fst criterion, Arlequin 3.1 Software) showed that both groups of belugas – from Sakhalinsky Bay and Shantar region bays – are significantly reproductively isolated from:

- belugas of Shelikhov Bay (North-Eastern Okhotsk Sea) population: Fst = 0.04483 and Fst = 0.03250, respectively;
- belugas of Anadyr Estuary (Western Bering Sea) population: Fst= 0.05361 and Fst = 0.04161, respectively;
- belugas of Norton Sound (Eastern Bering Sea) population: Fst= 0.09527 and Fst = 0.08099, respectively;

all values are statistically significant at $p< 0.0001$ level.

The Bayesian clustering approach (Structure v. 2.3.4 software) also demonstrate doubtless reproductive isolation of the Western Okhotsk belugas from the other studied populations (Figure - A, B, C)

The level of differences in mtDNA lineages occurrence (Fst criterion - haplotype frequencies only, Arlequin 3.1 Software) showed that both Western Okhotsk groups of belugas – from Sakhalinsky Bay and Shantar region bays – are also geographically (spatially) isolated from:

- belugas of Shelikhov Bay (North-Eastern Okhotsk Sea) population: Fst = 0.34017 and Fst = 0.35433, respectively;
- belugas of Anadyr Estuary (Western Bering Sea) population: Fst = 0.31797 and Fst = 0.36062, respectively;

all values are statistically significant at $p< 0.0000$ level, and- from belugas of Norton Sound (Eastern Bering Sea) population: Fst=0.12811 ($p= 0.01158$) and Fst = 0.15468 ($p= 0.00634$), respectively, based on 559 bp control region fragment; and Fst =0.11557 ($p=0.00000$), Fst = 0.06088 ($p= 0.00020$), respectively, based on 409 bp fragment.

[1] here and below the first number is quantity of individuals analyzed for mtDNA sequence and the second (given in parenthesis) is number of specimens used for microsatellite loci allele analysis

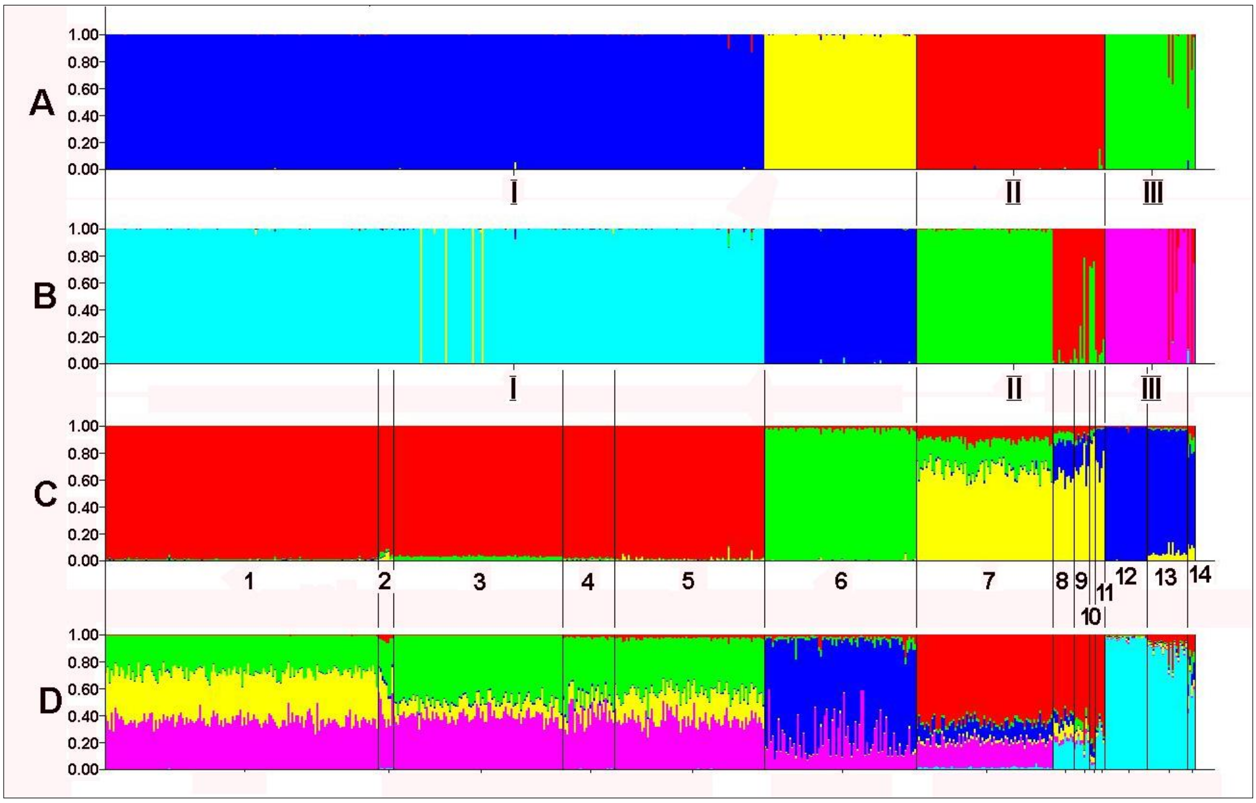

Figure 3. The results of clustering analysis.
A - locprior no admixture model for the layout where all individuals from each sea (I - the Okhotsk Sea, II - Bering-Chukchi-Beaufort Seas region, III - the White Sea) were "assigned" to a single population, resulted with K=4 as optimal (Evanno method, Structure Harvester online analysis).
B - the same model as A, resulted with K=6 in accordance with minimal Mean LnP(K) value.
C - locprior admixture model for the layout where the individuals from each location were «assigned» to a separate population, resulted with K=4 in accordance with minimal Mean LnP(K) value. Locations: 1 - Sakhalinsky Bay, 2 - Nikolaya Bay, 3 - Ulbansky Bay, 4 - Tugursky Bay, 5 - Udskaya Bay, 6 - Shelikhov Bay, 7 – Anadyr Liman, 8 – Chukotka peninsula coast together with 3 samples from Little Diomede Island, 9 - Point Lay, 10 - Beaufort Sea, 11 - Norton Sound, 12-14 - White Sea
D - the same model as C, for K=6

Thus, the independent (both reproductively and geographically) status of the Western Okhotsk population may be stated.

The difference in 17 loci alleles frequencies between Sakhalinsky Bay sample and the entire Shantar region sample (Fst= 0.00578) is statistically significant at p< 0.0000 level, but is essentially smaller than that found in inter-population comparisons. Particularly, this difference is caused by difference found between Sakhalinsky and Ulbansky bays (Fst = 0.00836, p< 0.0000) and between Sakhalinsky and Udskaya bays (Fst = 0.00392, p= 0.00297), whereas no difference was found between Sakhalinsky and the two other Shantar region bays (Nikolaya and Tugursky) as well as for any compared pair of the bays within the Shantar region.

The situation is illustrated by clustering method, when a slight difference between Sakhalinsky and Shantar samples appeared for K=6 (Figure - D).

On the other hand, for mtDNA haplotypes frequencies, no statistically significant differences were revealed only for Sakhalinsky vs. Nikolaya and Tugursky vs. Udskaya bays. For all other pairs of comparison, the "inter-bay" difference in mt-lineages occurrence is evident (Fst = 0.09507-0.32011 at p-level varied between 0.00000 - 0.01396).

This fact confirms the conclusion that, in summer, belugas of Western Okhotsk population form geographically isolated groups, or summer local aggregations. Taking into account the fact that low, but statistically significant, differences were also found between some bays, we can conclude that the population may be subdivided into separate demographic units. Three of these units are Sakhalin-Amur group, Ulbansky Bay group and Tugursky-Udskaya bays group. To define a status of belugas of Nikolaya bay, the sample size must be increased.

2. Abundance

Most recent abundance aerial surveys were conducted in 2009 and 2010 (Glazov et al. 2012, Shpak and Glazov 2013a, 2013b). The methods and results were presented by Shpak (Shpak et al. 2011) and explicitly discussed at the IUCN Expert meeting in Chicago, March 2011. Sakhalin-Amur and Shantar regions were surveyed each year twice. Due to the large size, the surveyed area was divided onto the survey regions that corresponded to geographic features of the coast line. Abundance estimate was conducted separately for each survey region (Table 1). For the southern part of Sakhalinsky Bay and the Amur Estuary, which were surveyed in parallel line-transects, beluga abundance was calculated in the program ‘BELUKHA 2’ with the extrapolation method (Chelintsev, 2010a; 2010b; 2012). For the other regions, covered with the single-line coastal survey, beluga abundance was taken to be equal to the number of visually detected animals; and in cases of large aggregations, the visually estimated number was corrected with photographs. For the reasons of poor weather conditions and incomplete area coverage, the results of some sections of the surveys in 2009 and 2010 were considered unsatisfying. The first of the 2010 surveys (2010A) was chosen for beluga abundance estimate in the western section of the sea (N=4780, Table 1).

Table 1. Results of the Western-Okhotsk aerial survey 2010A, Sakhalin-Amur and Shantar regions (Shpak and Glazov, 2013a, excl. last column).

Date of survey 2010a	Part of region	Method of count	Estimated beluga number (Ni)	Relative statistical error (cv)	Corrected for availability (Ncorr)
aug 8	Amur mouth	direct	35	0.000	70
aug 8	Amur estuary	sample	108	0.453	216
aug 8	Sakhalinsky Bay	sample	1305	0.318	2610
aug 8	Baikal Bay	direct	126	0.000	252
aug 7	Tugursky Bay	direct	753	0.000	1506
aug 7	Nikolaya Bay	direct	54	0.000	108
aug 7	Ulbansky Bay	direct	1167	0.000	2334
aug 7	Udskaya Bay	direct	1232	0.000	2464
Western OS, total			**4780**	**0.087**	**9560**

All estimations did not take into account belugas invisible to observers due to being underwater (no availability correction was applied) and reflected minimal abundance.

As suggested by Shpak et al. (2011) and supported by IUCN expert panel review (Reeves et al., 2011), ca. 50% of whales may had remained unseen to the observers. Thus, the results of the survey were multiplied by 2 (Table 1, last column in blue font), and the corrected abundance of the **Western-Okhotsk stock Ncorr = 9560 belugas**. Sakhalin-Amur aggregation, roughly, constituted one-third of the population, 3148 belugas (August 8, Table 1).

In a different document, Shpak and Glazov (2013b) have separately presented an average abundance estimate for the Sakhalin-Amur aggregation, based on 3 surveys of this area conducted in 2009 and 2010: 1977 – average estimate uncorrected for availability bias (3954 – corrected) (Table 3, below in PBR chapter).

3. Anthropogenic removals

According to the specialists from Russian Fisheries Institutes, marine mammal resources, including beluga, are under-exploited; their growing number leads to a disbalance in marine ecosystems and an increase of conflict with fishermen (Myasnikov et al. 2011; Boltnev et al. 2011, 2016).

Total Allowed Take (TAT) volumes are issued in corresponding Orders by the Ministry of Agriculture and are available publically. Sometimes, Russian Agency of Fisheries corrects TATs in the beginning of the year, and these corrected figures are not easily accessible. The factual capture numbers (traditional harvest together with live-captures) are not available publically, and it is possible they are not reported properly to the corresponding state organs.

To our knowledge, the captures of beluga whales in the Okhotsk Sea starting 2000 were as follows (Table 2):

Table 2. The annual beluga Total Allowed Takes (TAT), for North-Okhotsk/West-Kamchatka subzones, and *actual* permanent removals by live-capture (LC, # of whales) from Sakhalinsky Bay, North-Okhotsk subzone (from Shpak *et al*. 2011, amended from Shpak & Glazov 2014, and publically available documents).

year	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
TAT	n/a	n/a	n/a	n/a	n/a	n/a	1000/0	400/400	100/100	300/300
LC	10	22	10	26	25	31	20	0	25	24

year	2010	2011	2012	2013	2014	2015	2016	2017
TAT	300/300	150/150	360/50	360/50	150/0	150/25	0/0	150/25
LC	30	33	44	81[1]	?	?	0	n/a [2]

[1] the number reported by Shpak and Glazov (2014) was estimated based on direct observations and reports by capture teams.

[2] For the first time, it is recommended to distribute the TAT among different summer aggregations according to their sizes, and to limit the take in Sakhalin-Amur region to 40 belugas.

A. Boltnev with co-authors (2016) mentions that in average, live-captures of Sakhalin-Amur belugas do not exceed 30 whales annually (apparently, in recent years, but no later than 2013), but that maximum take was “over 100 animals”. In 2013, live-captures in Sakhalinsky Bay were the highest compared to the past years (Shpak and Glazov, 2014). According to our knowledge (our observations and reports by capture teams), 81 beluga were taken, and minimum 32 drowned at capture or died soon after and remained unreported, but these numbers were not included in any of the official documents to which A. Boltnev with co-authors referred when prepared their paper; neither our report was cited. This makes us think that in 2013 the total take amount available to us (81 beluga) was lower than a real (officially reported) take, which was “over 100” belugas. This is excluding unreported belugas that were killed (drowned) during captures.

All (or absolute majority) live-captured belugas are immature individuals of 2-3 (rarely, 1-5) years old. Usually, the preferred sex for takes are females.

Quotas for beluga harvest are seldom requested. To our knowledge, a quota for traditional harvest in the North-Okhotsk subzone for 90 beluga whales was issued in 2012, but no whales were harvested under this permit (Shpak & Glazov, 2013).

Separately should be considered poaching (illegal harvest). Based on interviews with local people, we suggest that in Shantar region they take “several” belugas per settlement annually. Most people reported feeding dogs in winter as a purpose for take. Locals from one village, mostly inhabited by evenks, reported they take beluga as man-food. There are totally 3 settlements in the region. In Sakhalin-Amur region we did not hear about harvesting belugas for food, but there were mentioned several cases of shooting belugas by fishermen when the whales entered salmon traps or approached nets.

4. Incidental mortality

Human-caused beluga incidental mortality – by-catch in salmon traps or gillnets and poachers' sturgeon nets and ship-strikes – is nearly impossible to estimate in the study regions due to rejection to report, the vast scarcely populated area and impossibility to arrange regular coastal patrols (Shpak et al., 2011). We are aware of several cases of beluga by-catch in Sakhalin-Amur and Shantar regions. Ship strikes were not recorded/reported. The analysis of photos collected in Sakhalin-Amur, Shantar and western Kamchatka regions revealed very few whales with scars/injuries that may be potentially caused by boat engines (Shpak et al., 2011, Russkova et al., 2012; Tarasyan et al., 2012, 2013).

5. Population trajectory

In last 20 years, two surveys in 2009 and 2010 were conducted for the Western-Okhotsk population. Reports on previous surveys do not contain enough information on survey design and analysis methods as well as area coverage to enable comparison of the results for assessing the population trend.

6. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

Although Sakhalin-Amur and Shantar belugas share the same nuclear gene pool, there is a clear evidence of a strong philopatry among discrete summer aggregations. For management purposes, the independent demographic units (Sakhalin-Amur, Ulbansky, Tugursky and Udskaya Bay summer aggregations) should be considered as **separate units**, i.e. TAT should be calculated for each aggregation separately. Insufficiently studied Nikolaya Bay is occupied by a small summer stock or, possibly, visited by several groups from Sakhalin-Amur aggregation and at present should not be considered as a place for captures at all (Shpak and Glazov, 2013a).

In Shpak *et al*. (2011) and Reeves *et al.* (2011), PBR-method was recommended to estimate a sustainable quota for beluga live-captures in the OS (in absence of traditional harvest). Upon recommendations of the IUCN review panel, we re-calculated the *PBR* presented in Shpak *et al.* (2011) as $PBR_{mean}=f(N_{mean}, cv(N_{mean}))$, *where* N_{mean} is arithmetic mean of abundance estimates of all aerial surveys used for calculation.

Arithmetic mean of the three successful abundance estimates of Sakhalin-Amur area was obtained and further corrected for availability (50%) to obtain corrected abundance estimate: $N_{cor} = N_{mean}/0.5$. Two values of recovery factor F_r =0.5 and F_r =0.65 were used, the first one suggested by Reeves et al. (2011) and the second – based on our assessment of the status of the aggregation (Table 3, Shpak and Glazov, 2013). Having suggested a recover factor value of 0.65, we accepted it would likely have to be reduced after the final data on 2013-catch become available.

In the Order by the Ministry of Agriculture #445 from October 10 2016 on approving Total Allowed Takes of Water Biological Resources…, for the first time beluga TAT of 150 whales for the North-Okhotsk fishing subzone was subdivided into:

40 belugas in Sakhalin-Amur region, 40 belugas in Nikolaya and Ulbansky Bays, 20 - in Tugursky Bay, 40 – in Udskaya Bay, and 10 – along the northern coast to the east of Okhotsk town.

Thus, in 2017 the use of beluga resource in the North-Okhotsk subzone will be distributed among different summer aggregations (in contrast to previous years, at least up to 2013, when all whales were taken in Sakhalinsky Bay). This is a definite advantage in management of the Western-Okhotsk population. However, beluga captures were reported to be conducted in Nikolaya Bay in 2014-2015 (no numbers available, interview data). In the quota distribution for 2017, Nikolaya Bay is lumped with Ulbansky Bay into one catch region, and the allowed removal of 40 belugas in Nikolaya-Ulbansky is likely to take place only in Nikolaya Bay, where the killer whale/beluga capture base already exists. Such takes may not be considered sustainable.

Table 3. Sakhalin-Amur 2009-2010 aerial survey results and calculation of PBR.

Year of survey	Region	Estimation of beluga number, N	Relative statistical error, cv
2009	Sakhalin-Amur	2293	0.355
2010a	Sakhalin-Amur	1574	0.266
2010b	Sakhalin-Amur	2064	0.538
N_{mean}		**1977**	**0.242**
N_{cor}		**3954**	
N_{min}		**3233**	
$PBR_{mean\ (0.5)}$		**32**	
$PBR_{mean(0.65)}$		**42**	

7. Habitat and other concerns

Sakhalinsky Bay and Amur River estuary are the areas intensively exploited by salmon fishery. This type of industry has also developed in all bays of the Shantar region. The major concerns are:

- Conflict with fishermen
- Carrying capacity of Sakhalinsky Bay (no studies)
- Amur and Uda River floods: washing down human/pet/livestock waste and chemicals
- Ice cover reduction

8. Status of the stock

It is hard to assess the population trend, since there were no reliable abundance surveys in the past. Nonetheless, the general expert opinion in Russia is that the population may be considered stable. Uncontrolled removals though may quickly exhaust Sakhalin-Amur summer aggregation, first of all – by ageing it, since only juvenile (mostly, 2-3 y.o.) belugas are taken. Limiting take from this aggregation and establishing “rest-years” with zero quota, if implemented, would keep the use of this unit at sustainable level.

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SUPPLEMENT to the Western Okhotsk (Sakhalin-Shantar) population assessment: Sakhalin-Amur, Ulbansky, Tugursky and Udskaya summer stocks

1. Sakhalin-Amur Summer Stock

Sakhalin-Amur stock is the largest and best studied of all Western-Okhotsk summer aggregations. Although all belugas summering in the western part of the Okhotsk Sea share a common nuclear gene pool and belong to a single population, the composition and frequencies of the maternal lineages represented in Sakhalin-Amur region are different from those in Ulbansky (Fst=14%), Tugursky (Fst=9.5%) and Udskaya (Fst=11%) bays. An average abundance estimate of Sakhalin-Amur stock based on 3 line-transect surveys conducted in 2009 and 2010 was 1977 (CV=0.24), and correction for availability bias (50%) resulted in estimate of 3954 belugas. The stock was harvested until 1950s. Starting mid1980s, live-captures have been conducted in the southern part of Sakhalinsky Bay. PBR was estimated 42 with recovery factor 0.65 taken into account past exploitation. Major concerns include conflict with coastal fisheries (disturbance, potential entanglement, shooting); water contamination – the Amur River discharge of industrial and agricultural pollutants and transfer of livestock deceases; potential local depletion if quota regulation and capture supervision are not further developed. Carrying capacity of the area has not been estimated, but likely diminished in recent decade due to a high increase in salmon fishery. Status of stock: stable. Level of concern: moderate.

Nikolaya Bay is occupied with a relatively low number of belugas (usually, less than 1 hundred), and it is unclear whether this aggregation is residential, or different groups visit the bay in summer. Independent status of beluga aggregation in Nikolaya Bay is not supported with the data available to-date. A pairwise analysis of haplotype frequencies resulted in no differences between Sakhalin-Amur and Nikolaya Bay samples (Fst=3.6%). At the same time, the difference between Nikolaya and adjoining Ulbansky Bay proved to be the highest (32%) of all compared pairs of bays in the Western Okhotsk Sea. Results of genetic analysis should be interpreted with caution due to a very small size and male-skewed sample from Nikolaya Bay (8 males and 1 female). Nonetheless, further evidence of relatedness of Nikolaya belugas to Sakhalin-Amur summer stock was obtained from photo-identification studies and behavioural observations. Until the status of Nikolaya Bay belugas is confirmed with the data of sufficient power, the animals observed in Nikolaya Bay may be assigned to Sakhalin-Amur stock. No takes from Nikolaya Bay are sustainable due to the remaining uncertainty regarding the status.

2. Ulbansky Summer Stock

The identity of the Ulbansky beluga summer stock as a separate demographic unit within the Western-Okhotsk population is based on the multi-year summer and autumn observations in Ulbansky bay and genetic analysis. In September-October, some belugas from Sakhalinsky Bay move to Nikolaya Bay and may also visit Ulbansky, but overall beluga numbers in the inner part of Ulbansky Bay seem to decrease in autumn. Winter migratory routes and feeding grounds are unknown. In August 2010, 1167 belugas were counted during direct count aerial survey (estimated 2334 belugas, when corrected for availability bias). All belugas summering in the western part of the Okhotsk Sea share a common nuclear gene pool and thus belong to a single biological population. However, composition and frequencies of the maternal lineages represented in Ulbansky Bay significantly differ from those in the other bays: pairwise Fst values are 17% for Udskaya Bay, 14% for Sakhalinsky and 18% for Tugursky bays ($p < 0.0001$ for all pairs). For geographically closest Nikolaya Bay (small sample size: n=8), this difference is the highest and reaches 32%. Belugas in Ulbansky Bay, to our knowledge, have never been harvested, and no live captures have been conducted. The stock is likely a subject to killer whale predation: beluga kills have not been observed, but numerous observations showed panic escape reactions of the entire aggregation upon approach of mammal-eating killer whale groups. There is a fishing plant, which deploys salmon nets along the coast and in the Ulban river mouth, and a coastal gold-mining company machinery and fuel terminal. A mining site is located on a tributary of the Ulban River. Major concerns for this stock are entanglement and shooting associated with fisheries and contamination of habitat in case of toxic discharge from gold mining. Status of stock: presumably stable. Level of concern: low/moderate.

3. Tugursky Summer Stock

The identity of the Tugursky summer stock as a separate demographic unit within the Western-Okhotsk population is based on historical information and opportunistic observations of beluga summer aggregation in the bay. Genetic analysis supports its geographic isolation from Sakhalinsky and the other Shantar bays, except Udskaya Bay. In summer, belugas are regularly seen in the inner part of Tugursky Bay and sometimes along the western coast, but no belugas have been observed travelling *between* Tugursky and Udskaya bays. Small groups have been reported near the south coast of the Big Shantar Island and along the northeast coast of Tugursky Bay. Behaviour differences (attitude to a boat) were noted between beluga groups in Tugursky and Udskaya bays. Winter migratory routes and feeding grounds are unknown. In August 2010, 753 belugas were counted during direct count aerial survey (estimated 1506 belugas, when corrected for availability bias). Although all belugas summering in the western part of the Okhotsk Sea share a common nuclear gene pool and thus belong to a single biological population, the composition and frequencies of the maternal lineages represented in Tugursky Bay differ from those in Sakhalinsky (Fst = 9.5%, $p < 0.0001$) and Ulbansky (Fst = 18%, $p < 0.0001$) bays. However, no significant genetic difference was found between belugas in Tugursky and Udskaya bays (32 and 90 specimens, respectively). A larger sample from Tugursky Bay is required to determine whether Tugursky belugas are demographically isolated from those in Udskaya Bay. Belugas were harvested in Tugursky Bay by both locals and commercial hunters starting in the late 1800s and until the 1950s. At present, they are occasionally taken by locals, either as a result of by-catch in salmon nets or by shooting. No live-captures from this stock have been conducted. There is one settlement, one fishing plant, and a coastal gold-mining company base in the bay. Major concerns are conflict with fishermen, potential habitat contamination caused by gold ore mining (heap leaching), and the river discharge with human and livestock waste. Status of stock: unknown. Level of concern: low/moderate.

4. Udskaya Summer Stock

The identity of the Udskaya summer stock as a separate demographic unit within the Western-Okhotsk population is based on historical information, multi-year observations of beluga summer aggregation in the bay, and genetic analysis. Belugas are present in the estuarine area from June to October and often enter the Uda River. Belugas are also known to concentrate in the estuary of the Torom River. There are no genetic samples from the second concentration area, but regular beluga sightings between the two rivers (approx., 40 km distance) suggest that all animals belong to the same stock. Upon ice formation in the Uda estuary, belugas move along the entire south coast of the bay, but keep near the coastline. Winter migratory routes and feeding grounds are unknown. In August 2010, 1232 belugas were counted during direct count aerial survey (estimated 2464 belugas, when corrected for availability bias). Although all belugas summering in the western part of the Okhotsk Sea share a common nuclear gene pool and thus belong to a single biological population, the composition and frequencies of the maternal lineages represented in Udskaya Bay strongly differ from those in Sakhalinsky, Nikolaya, and Ulbansky bays: pairwise Fst values are 11-17%, $p < 0.0045$-0.0001). However, no difference was found between belugas from Udskaya Bay and those from Tugursky Bay. A larger sample from Tugursky Bay and sampling in the Torom River estuary in Udskaya Bay are required to better understand the stock structure of Tugursky and Udskaya belugas. Differences in behaviour responses to presence of a boat were noted between Tugursky and Udskaya beluga groups. Belugas were harvested in Udskaya Bay by both locals and commercial hunters until the 1950s. At present, belugas are occasionally taken by locals, either as a result of by-catch in salmon nets or by shooting. No live-captures from this summer stock have been attempted. There are two settlements, three fishing plants with multiple fishing camps, three coastal gold-mining bases, and one gold ore loading terminal in the gulf. Diesel fuel is being unloaded in at least four locations. Major concerns are conflict with fishermen, habitat contamination by toxic river discharge (gold-mining), discharge of human and livestock waste, ship traffic / noise, and leaks during diesel fuel transport and unloading. Status of stock: presumably stable. Level of concern: moderate.

Annex 2: Shelikhov Bay (North-Eastern Okhotsk Sea) Beluga Stock for the Global Review of Monodontids, 13-16 March 2017, Copenhagen

By Olga Shpak and Ilya Meschersky

1. Distribution and stock identity

The modern (since 2000) data on Shelikhov Bay (and the entire north-east of the Okhotsk Sea, OS, Fig. 1) beluga abundance, distribution and population status are mostly limited to the data collected by A.N. Severtsov Institute, and in 2016 – in collaboration with Dr. O. Filatova. (PEW-fellowship).

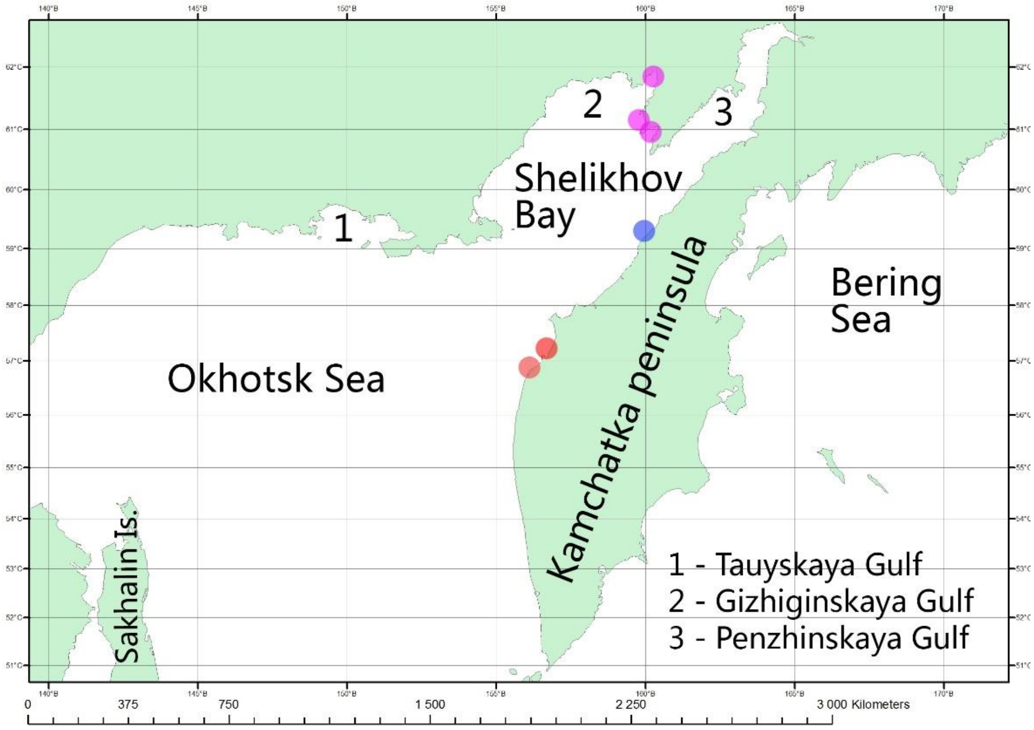

***Figure 1**. Map of north-eastern Okhotsk Sea. The colored dots – spots of biopsy collection: red – the mouths of Khayruzova and Moroshechnaya rivers, blue – the Palana river mouth, pink – Gizhiginskaya Gulf.*

Summer distribution

Most recent summer distribution data for the coastal waters of the entire northern OS were collected during the aerial surveys in 2009 and 2010 and during coastal observations on the western coast of Kamchatka peninsula (2010-2012). The results were presented at several conferences and published in regional journals in 2012-2014, but recently have been summarized by Solovyev with co-authors (2015).

Numerous Soviet literature sources described beluga distribution patterns in the northern Okhotsk Sea. In the first half of the 20th century, belugas aggregated in Tauyskaya Gulf (Fig. 1), where they were commercially harvested in 1930s. In the second half of the 21st century belugas were not observed in this area. The group was either extirpated or abandoned this summer ground. Unusual sightings of belugas in Tauyskaya Gulf were reported by A.I. Grachev from MagadanNIRO (pers. comm.), when in late spring ca. 1500 beluga whales entered Tauyskaya Gulf. Same year, in late June, 100–150 beluga whales again entered the Gulf. Apart from these two observations, there were very few sightings of singletons or small groups reported from Tauyskaya gulf.

In Shelikhov Bay, belugas are known to approach river estuaries during herring, smelt and salmon runs. Larger concentrations were observed in the bottoms of the gulfs (Solovyev et al. 2015), but in this region beluga distribution was less confined to the bottoms of the bays (Fig. 2) as compared to Sakhalin-Amur region in the western part of the OS.

In June-July 2016 during a ship-based survey in Tauyskaya and Gizhiginskaya gulfs, a large aggregation of over 400 belugas was observed feeding on salmon in the bottom of Gizhiginskaya gulf, and smaller groups of up to 20, mostly adult, individuals – along its eastern coast (Filatova et al. 2017, in press), similarly to the results of aerial surveys as shown on Fig. 2.

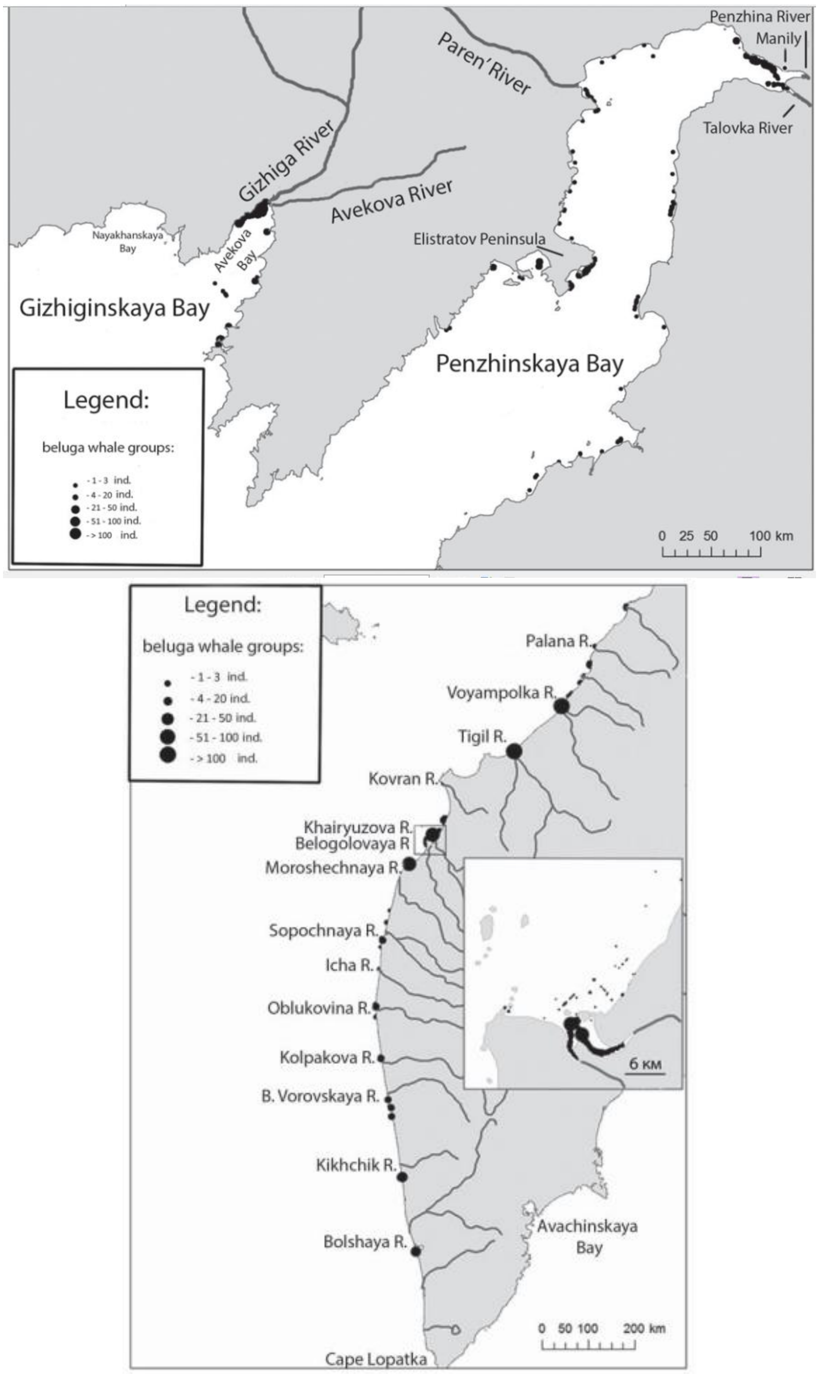

Figure 2. *Summer sightings of belugas in 2009-2010: top – the northern part of Shelikhov Bay, aerial surveys; bottom – West Kamchatka, boat and aerial surveys (from Solovyev et al. 2015).*

Winter distribution

Little is known on winter distribution of Shelikhov belugas. Vladimirov and Melnikov (1987) proved that belugas do not leave the northern part of the OS in winter when presented results of 2 aerial surveys conducted in January 1982 and February 1983: in two surveys combined, they saw 109 belugas along the ice edge in Shelikhov Bay and along the west coast of Kamchatka. Satellite transmitters were deployed on 3 beluga whales in 2010 (West Kamchatka, our data – unpubl.). Tracking the whale, whose tag transmitted till late December, showed that this beluga did not leave Shelikhov Bay. Variation of ice edge extent largely varies inter-annually, and often stretches south from Shelikhov Bay. We suppose, Shelikhov belugas remain near the ice edge, to the south of Shelikhov Bay and along the coast of Kamchatka. As soon as the ice condition allows, belugas return to the bay. Fedoseev (1984) encountered belugas in Shelikhov Bay in April.

Genetic studies

Earlier published results of analysis of 30-35 individuals (Borisova et al., 2012; Meschersky et al., 2012) and 14 individuals (Meschersky et al., 2013) biopsied off western coast of Kamchatka peninsula showed the significant level of reproductive isolation of this group from Anadyr Liman (Western Bering Sea) population and allowed to suppose that this group is also isolated from the Western-Okhotsk beluga population.

The statements presented here are based on analysis of samples from 80(79)[2] individuals (60 males, 18 females, for 2 individuals the sex was unknown) collected from (Fig. 1 above):

- the mouths of Khayruzova and Moroshechnaya rivers (2010-2012, 54 individuals, our data),
- the Palana river mouth (2009, 4 individuals, provided by A. Burdin)
- Gizhiginskaya Gulf (2016, 22 individuals, collected in collaboration with O. Filatova)

As genetic markers we used allelic composition of 17 microsatellite loci (Cb1, Cb2, Cb4, Cb5, Cb8, Cb10, Cb11, Cb13, Cb14, Cb16, Cb17 – Buchanan et al., 1996; Ev37, Ev94 – Valsecchi, Amos, 1996; 415/416, 417/418, 464/465, 468/469 – Schlötterer et al., 1991) and 559 bp sequence of mtDNA control region.

For comparative analysis, in addition to our data for other Russian waters beluga stocks, we used the data of analysis of 8(5) individuals from Norton Sound, Eastern Bering Sea (the samples were kindly provided by the Mammal Genomic Resources Collection, University of Alaska Museum of the North), as well as published data on frequency of mtDNA control region (409 bp) haplotypes known for Norton Sound (66 individuals, O'Corry-Crowe et al., 1997).

The analysis of 17 microsatellite loci allele frequencies (Fst criterion, Arlequin 3.1 Software) showed that belugas of Shelikhov Bay are significantly reproductively isolated from:

- belugas of Sakhalinsky Bay (Fst = 0.04483) and the bays of Shantar region (Fst = 0.03250) in the western part of the Okhotsk Sea,
- belugas of Anadyr Liman (Western Bering Sea) population: Fst= 0.06336,
- belugas of Norton Sound (Eastern Bering Sea) population: Fst= 0.08501,

all values statistically significant at $p< 0.0000$ level.

The Bayesian clustering approach (Structure v. 2.3.4 software) also demonstrates apparent reproductive isolation of Shelikhov Bay belugas from the Western-Okhotsk and the other studied populations (Fig. 3 – A, B, C).

The level of differences in mtDNA lineages occurrence (Fst criterion - haplotype frequencies only, Arlequin 3.1 Software) showed that belugas of Shelikhov Bay are also geographically (spatially) isolated from:

[2] *the first number is quantity of individuals analyzed for mtDNA sequence and the second (given in parenthesis) is number of specimens used for microsatellite loci alleles analysis.*

- belugas of Sakhalinsky Bay (Fst = 0.34017) and the bays of Shantar region (Fst = 0.35433) in the western part of the Okhotsk Sea,
- belugas of Anadyr Liman (Western Bering Sea) population: Fst = 0.38545, all values statistically significant at $p < 0.0000$ level, and
- belugas of Norton Sound (Eastern Bering Sea) population: Fst= 0.39211 ($p= 0.00010$) based on 559 bp control region fragment, and Fst =0.48499 ($p=0.00000$) based on 409 bp fragment.

Thus, the independent (both reproductively and geographically) status of Shelikhov, or North-Eastern Okhotsk, beluga population may be stated.

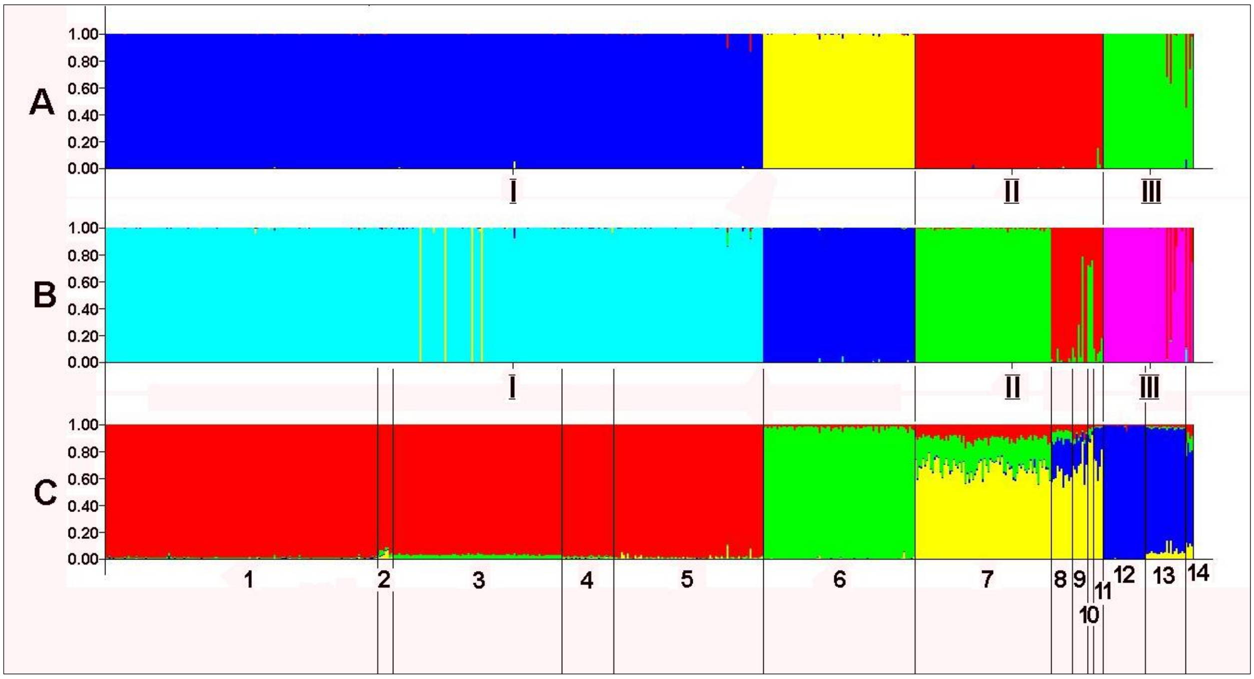

Figure 3. *The results of clustering analysis.*
A *- locprior no admixture model for the layout where all individuals from each sea (I - the Okhotsk Sea, II - Bering-Chukchi-Beaufort Seas region, III - the White Sea) were "assigned" to a single population, resulted with K=4 as optimal (Evanno method, Structure Harvester online analysis).*
B *- the same model as A, resulted with K=6 in accordance with minimal Mean LnP(K) value.*
C *- locprior admixture model for the layout where the individuals from each location were «assigned» to a separate population, resulted with K=4 in accordance with minimal Mean LnP(K) value. Locations: 1 - Sakhalinsky Bay, 2-5 - Shantar region bays, 6 – Shelikhov Bay, 7 – Anadyr Liman, 8 – Chukotka peninsula coast together with 3 samples from Little Diomede Island, 9 - Point Lay, 10 - Beaufort Sea, 11 - Norton Sound, 12-14 - White Sea.*

2. Abundance

In 1980-1990s belugas abundance in the northern OS varied from 3,000 to 10-20,000 (multiple sources).The estimates were based both on expert opinions and aerial surveys regularly conducted in 1980s.

In 2009-2010, we conducted aerial surveys of the OS (Glazov et al. 2012, Shpak and Glazov, 2013, see Table 1). For the northeastern part, the results of 2010 were chosen for abundance estimate due to better weather conditions during the flights. The survey was conducted as a coastal single line (Fig. 4) with a direct count (no extrapolation), i.e. minimal estimate.

Table 1. *Results of August 2010 aerial survey in the northeastern Okhotsk Sea (from Shpak and Glazov, 2013)*

Date	**Surveyed region**	**Number of belugas**
10.08.2010	Tauyskaya Gulf	0
19.08.2010	Gizhiginskaya Gulf	370
18.08.2010	Penzhinskaya Gulf	312
13-14.08.2010	West coast Kamchatka, north	638
14.08.2010	West coast Kamchatka, south	13
North-Eastern Okhotsk Sea, total		**1333**

Figure 4. *Flight routes and areas where beluga whales were encountered during aerial surveys, August 2010. Black lines – flight routes, regions where belugas were sighted – grey ovals. 1- Udskaya Bay; 2- Tugursky Bay, 3- Ulbansky Bay; 4 – Nikolaya Bay, 5- Sakhalinskiy Bay, 6 – Amur estuary; 7- Baikal Bay; 8 – Tauiskaya Bay; 9 – Gizhiginskaya Bay; 10 – Penzhinskaya Bay; 11 – r. Moroshechnaya (from Glazov et al 2012).*

Corrected for availability (belugas below surface, 50%), the abundance of Shelikhov population was calculated as 2,666 (Shpak and Glazov, 2013).

3. Anthropogenic removals

To our knowledge, no beluga quotas have been requested by local hunters from the region. Total allowed takes are issued as follows:

Table 2. *The annual beluga Total Allowed Takes (TAT), West-Kamchatka subzone.*

Year	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
TAT	0	400	100	300	300	150	50	50	0	25	0	25

There are no live-capture operations in the subzone. No recent information is available on the size of illegal takes by local people. For the western Kamchatka, Krupnik and Bogoslovskaya (2000) refer to communication with an ethnographer, who suggested that beluga takes in the northern part of the OS are occasional and, probably, do not exceed 10 belugas per year.

4. Incidental mortality

Same as Western Okhotsk

5. Population trajectory

Unknown

6. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

N/A

7. Habitat and other concerns

Competition with fishermen, climate change

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Annex 3: Anadyr Gulf Beluga Whale Stock (Western Bering Sea, Russia) for the Global Review of Monodontids, 13-16 March 2017, Copenhagen

By: Dennis I. Litovka, Ilya G. Meshchersky, Olga V. Shpak

1. Distribution and stock identity

The Anadyr Gulf beluga stock, previously thought to be a part of the Far Eastern Russian population (Kleinenberg et al, 1964; Vladimirov, 1994), has been a target for numerous studies starting 1998.

Based on coastal observations (Litovka, 2002), the stock consists of a single summer aggregation, which concentrates in the Anadyr Liman shallow waters (Figure 1). The stock identity was confirmed by genetic studies. Earlier published data based on analysis of about 75 individuals (Borisova et al., 2012; Meschersky et al., 2012) or on analysis of 37 individuals (Meschersky et al., 2013) biopsied in the Anadyr Estuary showed a significant level of reproductive isolation of this group from belugas of Western Okhotsk population and of geographic isolation of this group from belugas of both Western Okhotsk population and populations of Eastern North Pacific.

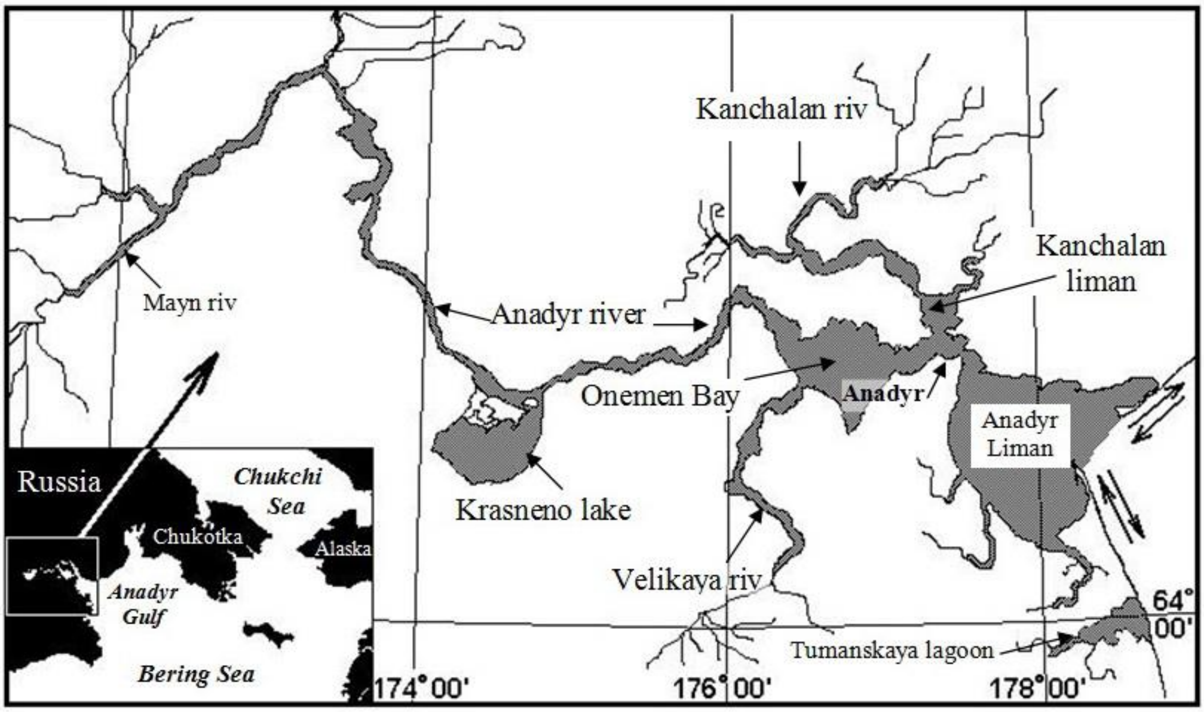

Figure 1. *Summer sightings of belugas in the Anadyr liman by visual observations, boat and aerial surveys (from Litovka, 2002)*

The statements presented below are based on analysis of samples representing 76(71)[3] individuals from the Anadyr Liman (2010-2011, 58 males, 18 females).

As genetic markers we used allelic composition of 17 microsatellite loci (Cb1, Cb2, Cb4, Cb5, Cb8, Cb10, Cb11, Cb13, Cb14, Cb16, Cb17 – Buchanan et al., 1996; Ev37, Ev94 – Valsecchi, Amos, 1996; 415/416, 417/418, 464/465, 468/469 – Schlötterer et al., 1991) and 559 bp sequence of of mtDNA control region.

For comparative analysis, we used the results of analysis:

[3] *Here and below the first number is quantity of individuals analyzed for mtDNA sequence and the second (given in parenthesis) is number of specimens used for microsatellite loci alleles analysis*

1) of beluga samples from other stocks in Russian waters (our data),
2) of the samples kindly provided by Mammal Genomic Resources Collection, University of Alaska Museum of the North: 10(8) individuals from the Eastern Chukchi Sea (Point Lay), 8(5) – from the Eastern Bering Sea (Norton Sound), 3(3) – off Little Diomede Island and 3(3) – from the Beaufort Sea (microsatellite data for the two latter samples for clustering analysis only),

   and additionally,
3) published data (O'Corry-Crowe et al., 1997) on frequency of mtDNA control region (409 bp) haplotypes known for the Eastern Chukchi Sea (103 individuals), the Eastern Beaufort Sea (97 individuals) and Norton Sound (66 individuals).

The analysis of 17 microsatellite loci alleles frequencies (Fst criterion, Arlequin 3.1 Software) showed that Anadyr Liman belugas are significantly reproductively isolated

- from belugas of Shelikhov population (North-Eastern Okhotsk Sea): Fst = 0.03938
- from both groups of Western-Okhotsk population: Sakhalinsky Bay (Fst = 0.05361) and Shantar region bays (Fst = 0.04161), all the values are statistically significant at p< 0.0000 level.

At the same time, no statistically significant difference was found between Anadyr Liman belugas and belugas

- from Chukotka peninsula coastal waters: Fst = 0.01270, p = 0.07148,
- from Point Lay (the Eastern Chukchi Sea): Fst = 0.00474, p= 0.24681, and
- from Norton Sound (the Eastern Bering Sea): Fst = 0.00869, p= 0.21414.

Meanwhile, the Bayesian clustering approach (Structure v. 2.3.4 software) demonstrated reduced level of genetic unity of Anadyr Liman belugas and belugas from other Bering-Chukchi-Beaufort Sea (B-C-B) regions (Figure 2: B, C).

These discrepant results may be consequence of the small size of samples from Chukotka peninsula coast (8), Point Lay (8) and Norton Sound (5) in our analysis.

The level of differences in occurrence of mtDNA lineages (Fst criterion - haplotype frequencies only, Arlequin 3.1 Software, 559 bp sequences) showed that belugas of Anadyr Liman are geographically (spatially) isolated not only

- from belugas of Shelikhov (North-Eastern Okhotsk Sea) population: Fst = 0.38545
- from both groups of Western-Okhotsk belugas: Sakhalinsky Bay (Fst = 0.31797) and Shantar region bays (Fst = 0.36062), all the values are statistically significant at p< 0.0000 level,

but, as well,

- from belugas of Point Lay (the Eastern Chukchi Sea): Fst = 0.34309, p< 0.0000, and
- from belugas of Norton Sound (the Eastern Bering Sea): Fst = 0.34490 p = 0.00010.

However, no significant differences in 559 bp haplotypes frequency was found for Anadyr Liman sample and the sample from the Chukotka peninsula coast: Fst = 0.05068 p = 0.09316.

In case of using 409 bp sequence (and essentially larger sample sizes), the values of Fst criterion (haplotype frequencies only) proved the significant geographic isolation between Anadyr Liman and groups from the eastern part of BCB region:

- for the Eastern Bering Sea (Norton Sound) Fst = 0.44019, p = 0.00000, and
- for the Eastern Chukchi Sea Fst = 0.31353, p = 0.00000

For the Beaufort Sea the value was essentially smaller but also statistically significant: Fst = 0.04941 p = 0.00129.

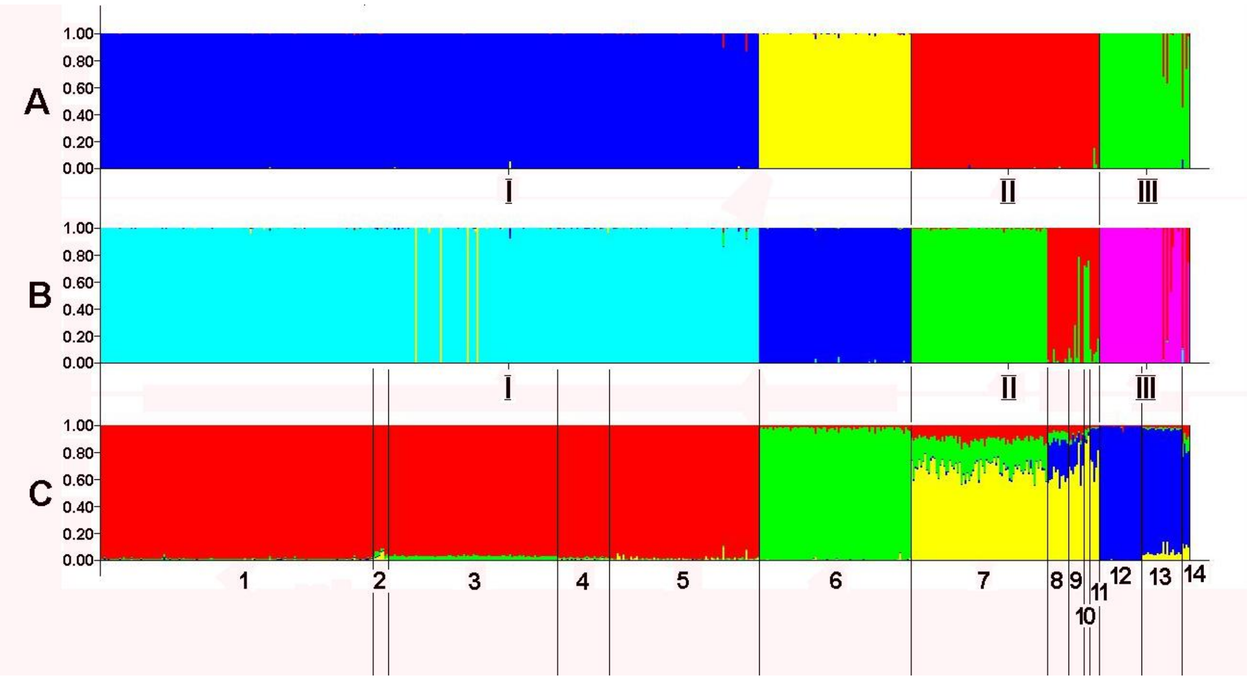

Figure 2. *The results of clustering analysis.*

> ***A*** *- locprior no admixture model for the layout where all individuals from each sea (I - the Okhotsk Sea, II - Bering-Chukchi-Beaufort Seas region, III - the White Sea) supposedly belong to a single population, resulted with K=4 as optimal (Evanno method, Structure Harvester online analysis).*
>
> ***B*** *- the same model as A, resulted with K=6 in accordance with minimal* Mean LnP(K) *value.*
>
> ***C*** *- locprior admixture model for the layout where the individuals from each location supposedly belong to a separate population, resulted with K=4 in accordance with minimal* Mean LnP(K) *value.*
>
> *1-5 - Western Okhotsk Sea groups, 6 – Shelikhov Bay, 7 – Anadyr Liman, 8 – Chukotka peninsula coast together with 3 samples from Little Diomede Island, 9 - Point Lay, 10 - Beaufort Sea, 11 - Norton Sound, 12-14 - White Sea.*

Thus, at present, it is clear that Anadyr Gulf belugas are seasonally geographically isolated from the other stocks recognized in the B-C-B region and should be managed as a separate demographic unit. However, in order to define its population status, more studies are required to estimate the level of reproductive isolation of Anadyr Gulf stock from the other B-C-B region recognized stocks.

Summer distribution was studied in 2001-2016 (Hobbs et al, 2007; Litovka, 2013; Litovka et al, 2013; Citta et al., 2016). In Anadyr Liman, same individuals were re-sighted in different years and within seasons (Prasolova et al., 2014; Prasolova et al., unpubl.). Together with results of genetic analysis, our observations suggest that in summer belugas form a residential aggregation in the Anadyr Liman of the western Bering Sea, and may return to the same water areas summer after summer.

Belopol'sky (1931) and Pikharev (1943) described the belugas movement into the Anadyr Liman after ice breakup and stated that the whales were commonly seen in groups from couple dozens to couple thousands.

In the Anadyr liman (Figure 1) in the ice-free period whales are present in all reachable areas, in river deltas, and it is known they can move by 200 miles up the Anadyr liman (Litovka, 2001; Litovka, 2002; Smirnov and Litovka, 2001).

Telemetry study of Anadyr belugas (Figure 3a) in 2001-2010 (Litovka et.al., 2002, 2004, 2013; Hobbs et al., 2007; Citta et al., 2016) has confirmed and clarified distribution and movement patterns in the Anadyr Liman and Anadyr Gulf, which previously were based on coastal and aerial counts of belugas (Figure 3b) in different years (Smirnov, Litovka, 2001).

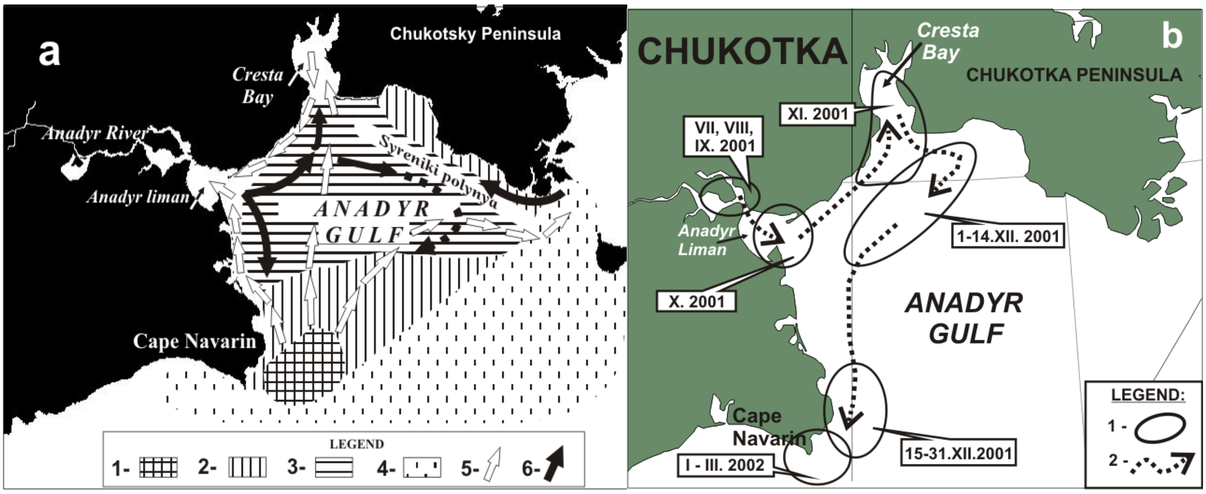

Figure 3. *A – Anadyr Gulf beluga distribution and movement scheme (from Smirnov and Litovka, 2001) based on coastal observations and aerial survey counts. Beluga density: 1) high; 2) medium; 3) low and 4) unknown. Movement patterns: 5) spring; 6) autumn. B – Anadyr Gulf beluga distribution and movement scheme based on telemetry study: 1) concentration areas; 2) movements (from Litovka et al., 2002)*

Belugas spend summer-autumn feeding period (total about 5-6 months) in the Anadyr Liman with the latest sighting in late November. Ice forming in the liman forces belugas to leave Anadyr River mouth. They move northeast – to the Kresta Bay, probably, to feed on smelt until they migrate to the middle and southern part of the Anadyr Gulf (Litovka et. al., 2013).

Telemetry along with aerial survey data (Litovka et al. 2002, 2006, 2013; Citta et al., 2017) show the majority of Anadyr belugas spend winter (December-March) around Cape Navarin (Figure 4). Modeling has shown the same result with the maximum sightings in regions with ice concentration of 80-90% (Litovka, 2013).

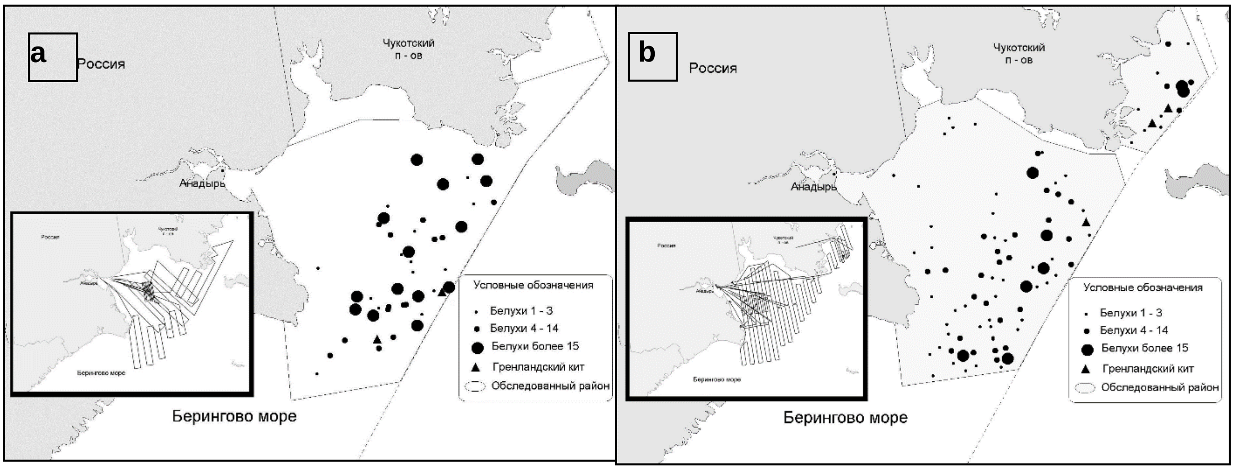

Figure 4. *Aerial survey transects and beluga sightings in April, 2005 (a) and 2006 (b) in the Western Bering Sea (from Litovka et al., 2006)*

Results of telemetry study (Citta et al., 2017 - Fig. 2) and aerial surveys (Litovka et al., 2006) shown that at winter-spring (December-April) feeding areas off Cape Navarin the Anadyr belugas may mix with some of B-C-B region stocks. Limited data on tagging suggest that mixing – if it takes place – is more likely to occur with the Eastern Beaufort Sea belugas than with other stocks, which is also supported by the results of genetic analysis presented above.

2. Abundance

Coastal observations near the city of Anadyr show that belugas appear in the Anadyr Liman on the third day of ice breakup and after that they remain here the whole ice-free period (Litovka, 2002). Usually, two peaks of sightings are observed: 1) from the end of June till the beginning of July, and 2) in the beginning of August, which are both directly connected to salmon spawning (Litovka, 2006). Maximum of whales (241 animals) were recorded on the third decade of June (Litovka, 2002).

No summer aerial counts of Anadyr Gulf belugas have been conducted. Pacific walrus aerial surveys with opportunistic beluga counts were conducted in April 2005 and 2006 (Litovka, 2013; Laidre et al, 2015). The surveys did not cover the area south of Navarin Cape, where according to satellite tracking data, part of Anadyr belugas may have remained at the time of flying. The availability correction factor calculated for Anadyr belugas in Anadyr liman and Anadyr Gulf during telemetry study was 2.86±0.76 (Litovka et al., 2004, 2006). The total beluga abundance in the area of the western Bering Sea covered by survey in April 2006 (Fig. 4b), with the availability coefficient applied, was 15127, lim=7447 ÷ 30741 (Litovka, 2013; Laidre et al, 2015). Beluga whales counted during aerial surveys in 2005 and 2006 might have belonged to several B-C-B region stocks, and the total abundance may not be applied to any of the currently recognized units.

An expert estimate of the Anadyr Gulf stock is ca. 3000 belugas (Litovka 2002).

3. Anthropogenic removals

See Annex 4: Bering-Chukchi-Beaufort pool, Russia.

4. Incidental mortality

Human-caused beluga incidental mortality (by-catch in salmon nets and ship-strikes) has not been estimated in the study region.

Ship strikes of belugas in Anadyr liman and Gulf were not recorded/reported. The analysis of photos collected in this area in 2013-2016 revealed very few whales with scars/injuries that may be potentially caused by boat engines (Prasolova et al, unpubl.).

For about 20 years, there were three cases of beluga entanglement in fishing gear.

5. Population trajectory

Unknown.

6. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

See Annex 4: Bering-Chukchi-Beaufort pool, Russia.

7. Habitat and other concerns

Anadyr liman and Anadyr Gulf are the areas extensively exploited by salmon fishery. This type of industry has also developed in all bays of the Chukotka region. The major concerns are:

- Competition with fishermen
- Increasing ship traffic
- Ice period reduction

8. Status of the stock

It is hard to assess the population trend, since there were no reliable abundance surveys in the past. Nonetheless, the general expert opinion is that the population may be considered stable (Litovka, 2013; Litovka and Khitzova, 2014).

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Annex 4: Bering-Chukchi-Beaufort Beluga Whale pool, Chukotka Peninsula, Russia for the Global Review of Monodontids, 13-16 March 2017, Copenhagen

By: Olga V. Shpak, Denis I. Litovka, Ilya G. Meshchersky

1. Distribution and stock identity

The beluga stock assessment conducted in 1999 (IWC, 2000) has recognized the following stocks, which at different seasons may be present in Russian waters: Eastern Bering Sea, Eastern Chukchi Sea, Beaufort Sea (or Eastern Beaufort), West Chukchi – East Siberian Seas, Anadyr Gulf stocks (i.e. stocks # 3, 4, 5, 25, 26). The status of Anadyr Gulf stock is largely defined, and its assessment is presented in a separate document (see Annex 3: Anadyr). The status of other belugas observed along Chukotka peninsula in the Chukchi and Bering Seas remains to be confirmed. In the Soviet and Russian multiple literature sources, belugas from Bering-Chukchi-Beaufort (B-C-B) region and the East Siberian (ES) Sea were sometimes grouped / subdivided into one or multiple stocks (populations), but either classification lacked sufficient grounding. Below we will consider only recent Russian sources, since the available data on the B-C-B pool stocks published in English are reviewed by the US and Canadian colleagues in corresponding assessments.

Overall, beluga sightings along Chukotka peninsula are rare in summer time. Most whales concentrate in Chukchi Sea in autumn, and in the Bering Strait and Being Sea – in winter and early spring. The sightings in the ES Sea are rare and limited to the eastern part of the Sea (Kochnev, 2003). In autumn, belugas briefly enter these waters. The most recent westernmost sighting of an unknown number of belugas (“many”) ca. 80 km east from the Kolyma River mouth (Fig. 1) in late September 2002 was provided by Kochnev (2003) as a pers. comm. with a local hunter. Kochnev (pers. comm.) has also noted that the beluga approaches to the coast of ES Sea are irregular and undulating in time and may be linked to the ice conditions: they were frequent in 1950-1960s, and then – in 1990s. Whether belugas enter the ES Sea from Chukchi Sea, following coast along the Chukotka peninsula, or from the north – remains unclear, but available observations support the second route (Belikov et al. 2002, Melnikov 2014).

Belikov and Boltunov (2002) reviewed the data from aerial ice-reconnaissance surveys and the data from the Soviet polar stations. Although, they assume that the Ayon ice-massif in the central part of the ES Sea may be a barrier to beluga westward distribution (which is not in recent time), they accept that in warm years beluga population exchange is possible. The aerial survey data show beluga presence in the western Chukchi and the eastern part of the ES Sea well above 75N. In summer, marine mammal observers on the polar stations recorded belugas above the latitude of 80 degrees N with the northernmost record of 86N (Belikov and Boltunov 2002, Belikov et al. 2002). The circumpolar distance at this latitude is ca. 480 nautical miles, and the distances between belugas from different Arctic populations, if they travel so far north, are “erased”.

B. Solovyev with co-authors (2013) created a representative picture of beluga seasonal distribution based on the coastal observations conducted in villages along Chukotka peninsula coast (Figure 2).

Kochnev (2003) denies hypothesis according to which large numbers of belugas summer in the WC Sea along the northern coast of Chukotka peninsula and the waters around Wrangel island. According to him, belugas are absent near Wrangel island in summer, and along the northern coast of the peninsula – most of the year, except for autumn. The lack of historic traditional beluga harvest to the west of 172W further supports his opinion (Bogoslovskaya and Krupnik 2000, Kochnev 2003). Melnikov (2014) also states that in summer belugas are rarely observed along Chukotka coast, both in Chukchi and Bering Seas.

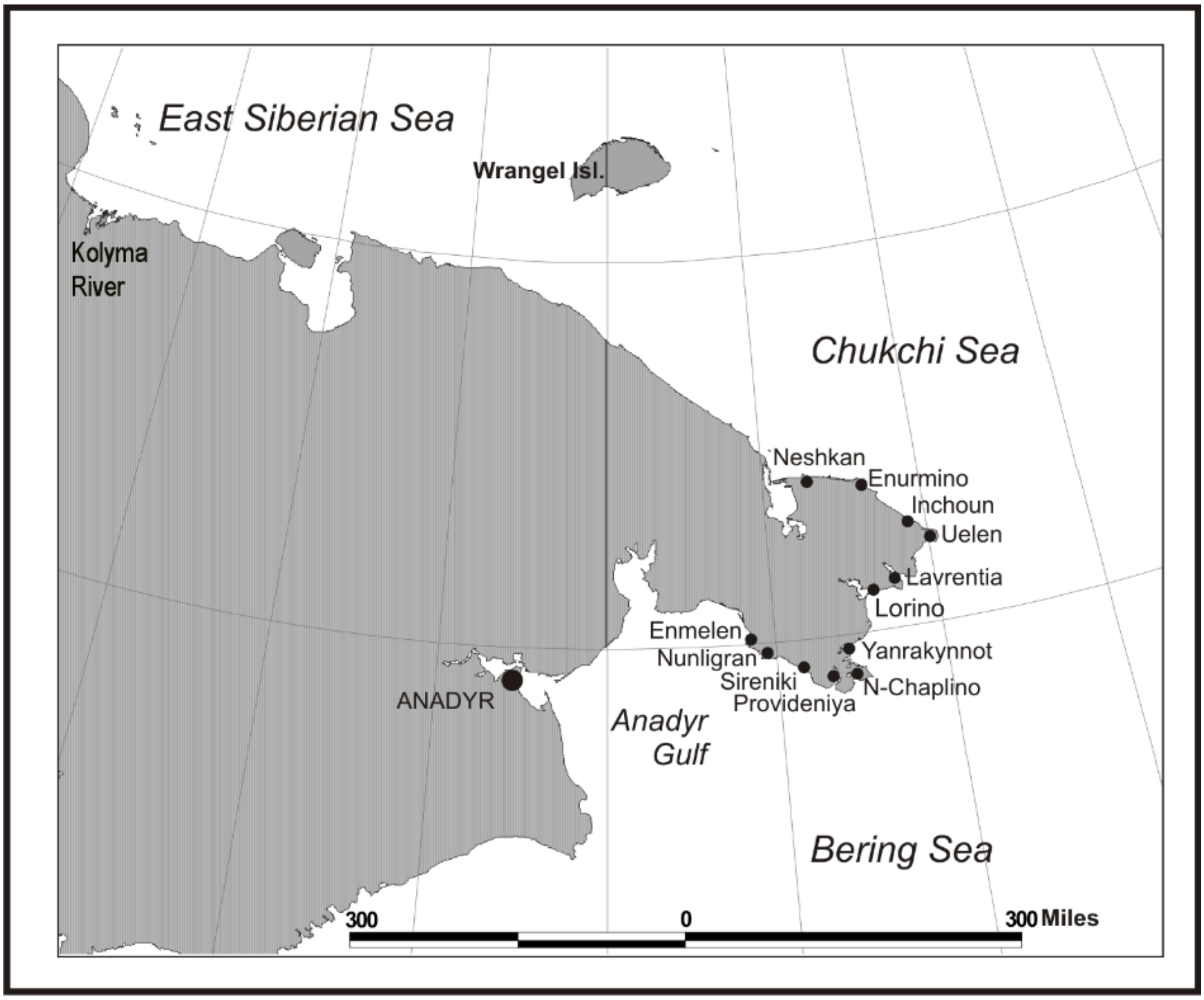

Figure 1. *Chukotka peninsula with geographic names used in current document.*

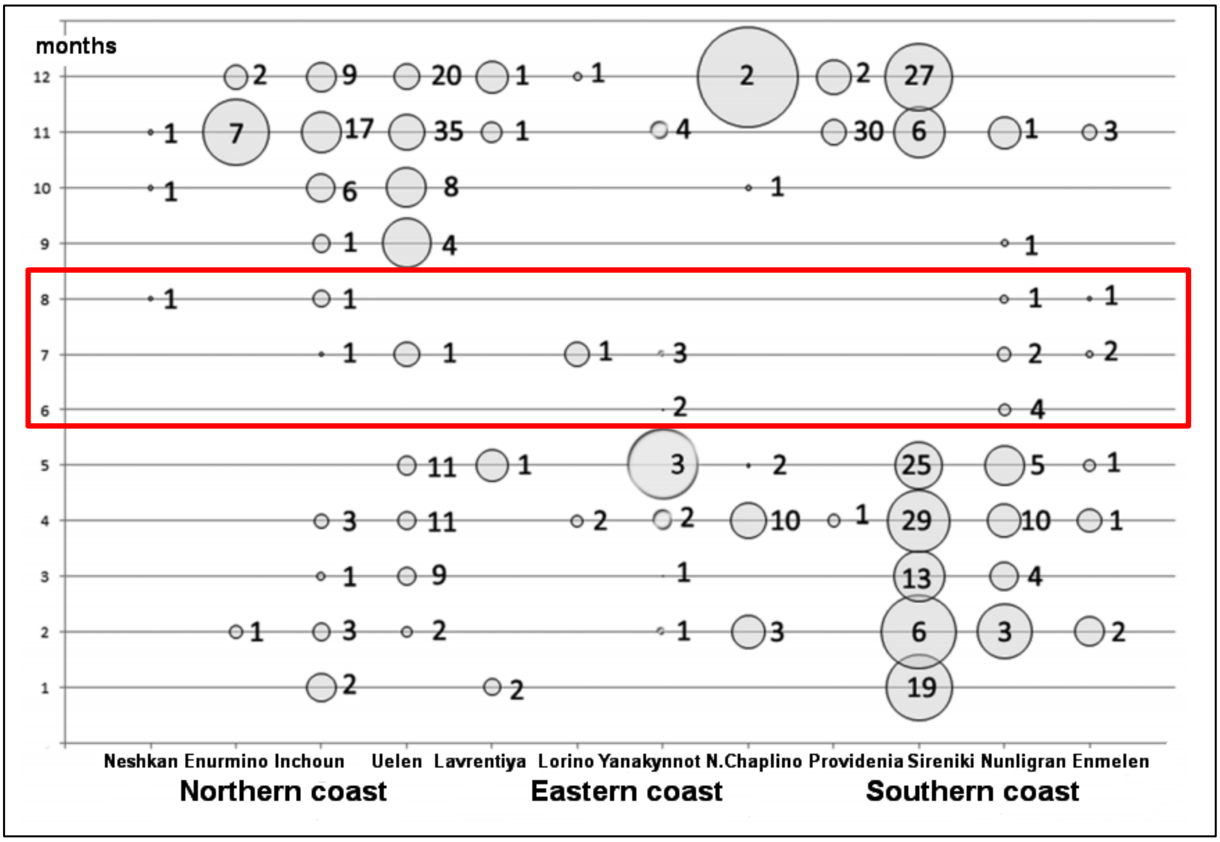

Figure 2. *Number of the beluga whale sightings from the observation points in 1999–2012, by months and mean size of the beluga groups. Circle diameters are proportional to the group sizes in the range 1–2000 whales. Number of the sightings in each point is shown by figure near or inside the circle. From Solovyev et al., 2013. Note low presence of belugas in months 6, 7 and 8 (in red rectangle).*

From all reviewed sources it is clear that most frequent / abundant beluga sightings fall on the time of spring / autumn migration. This is illustrated on the picture shown above (Figure 2). According to Melnikov (2014), spring migration along the southern and eastern peninsula coasts starts in April, and most observations suggest that belugas approach the SE-peninsula from south rather than from west (i.e. from the Bering Sea vs. Anadyr Gulf). These observations correspond well with the satellite tracking analysis (Citta et al. 2017).

NOTE: The interviews suggest presence of calves-of-the-year (Melnikov 2014) and “very-very small” (possibly, newborns) (Solovyev et al. 2013) as early as April (!). This information needs to be confirmed with the photographs or by professional observers before taken into account. If confirmed and such early-in-the-year birth indeed takes place, the supposed time and place of mating of belugas from B-C-B pool should be re-considered.

It is important to note that most data in the papers by Solovyev et al. (2013) and Melnikov (2014) were collected not by the authors personally but by “Beringia” National Park local employees and hired local residents. Although remoteness of the study area and lack of comprehensive research make such data invaluable, they should be treated with caution, and cross-checking of information should be applied whenever possible. In addition, in paper by Melnikov it is not always obvious (either in the text or on the figures) when the author uses personal data and when – the information from earlier published sources.

When discussing the possibility of beluga wintering in the leads and polynias in the southern Chukchi Sea, the authors present different opinions. Solovyev with co-authors acknowledges this possibility noting the lack of “the high level of herding during migration activity” (from Matishov and Ognetov 2006) among belugas along the northern coast of the peninsula in late autumn, and quick beluga emergence following Arctic cod approach in the opening waters in winter months. On the contrast, Kochnev (2003) and Melnikov (2014) do not support this idea, and Kochnev specifically notes that from December to June belugas are absent along the northern coast of peninsula. The sightings are limited to the NW cape of the mainland – Dezhneva Cape.

Neither of recent publications reviewed here conclude the existence of a separate Western Chukchi (or WC-ES) population. Still, the possibility exists that some belugas from B-C-B pool, instead of migrating south through the Bering Strait in late autumn, may remain in the polynias beyond the shorefast ice and / or the leads of consolidated ice further north in the Chukchi Sea or the Arctic Ocean. However, in this case, belugas wintering in the high Arctic would be isolated from the other stocks of the pool, which winter in the Bering Sea and remain there during spring (period of mating), and thus would represent a reproductively isolated unit. Unfortunately, this question will likely remain unresolved for a long time.

No genetic data on belugas of Russian part of the B-C-B region (except for belugas from Anadyr Gulf summer aggregation, Meschersky et al., 2013) have been published to-date.
Here, we present the data for only 8(8)[4] samples (7 males, 1 female) collected in 2011-2012 from belugas harvested by local people along Chukotka peninsula (Nunligran, 2-Sep-2011 - 1 male; Uelen, 9-Oct-2011 - 1 male; Lorino, 15-Nov-2011 - 1 male; Sireniki, "2011/2012 winter season" - 1 female; Seniavin strait, 12-Jan-2012 - 4 males (samples were provided by B. Solovyev and I. Zagrebin; the samples in Seniavin Strait were collected from a group of belugas trapped in ice, see below).

As genetic markers we used allelic composition of 17 microsatellite loci (Cb1, Cb2, Cb4, Cb5, Cb8, Cb10, Cb11, Cb13, Cb14, Cb16, Cb17 – Buchanan et al., 1996; Ev37, Ev94 – Valsecchi, Amos, 1996; 415/416, 417/418, 464/465, 468/469 – Schlötterer et al., 1991) and 559 bp sequence of mtDNA control region.

For comparative analysis, we used the results of analysis:

4) of beluga samples from other stocks in Russian waters (our data),
5) of the samples kindly provided by Mammal Genomic Resources Collection, University of Alaska Museum of the North: 10(8) individuals from the Eastern Chukchi Sea (Point Lay), 8(5) – from the Eastern Bering Sea (Norton Sound), 3(3) – off Little Diomede Island and 3(3) – from the Beaufort Sea (microsatellite data for the two latter samples for clustering analysis only),

and additionally,

6) published data (O'Corry-Crowe et al., 1997) on frequency of mtDNA control region (409 bp) haplotypes known for the Eastern Chukchi Sea (103 individuals), the Eastern Beaufort Sea (97 individuals) and Norton Sound (66 individuals).

The analysis of 17 microsatellite loci alleles frequencies (Fst criterion, Arlequin 3.1 Software) did not revealed significant differences between our Chukotka peninsula sample and belugas

- from Anadyr Liman: Fst = 0.01270 p = 0.07148,
- from Point Lay: Fst = 0.00026 p = 0.60974,
- from Norton Sound: Fst = 0.01295 p =0.33086.

The Bayesian clustering approach (Structure v. 2.3.4 software) demonstrated reduced level of genetic unity between Chukotka peninsula belugas and belugas from Anadyr Liman; however, no differences were found between individuals from the western coast of the Bering Strait and whales from the other Bering-Chukchi-Beaufort Seas (B-C-B) regions (Figure 3, B and C).

[4] *Here and below the first number is quantity of individuals analyzed for mtDNA sequence and the second (given in parenthesis) is number of specimens used for microsatellite loci alleles analysis*

The level of differences in occurrence of mtDNA lineages (Fst criterion - haplotype frequencies only, Arlequin 3.1 Software, 559 bp sequences) showed that our sample of Chukotka peninsula belugas differs from samples of Point Lay (Fst = 0.22431 p= 0.00436) and Norton Sound (Fst = 0.26786 p= 0.00188), but no difference was found between Chukotka peninsula belugas and belugas of Anadyr Liman (Fst = 0.00000 p= 0.38996).

In case of using 409 bp sequence (and essentially larger sample sizes), the values of Fst criterion (haplotype frequencies only) proved geographical isolation between Chukotka belugas and belugas of Point Lay (Fst=0.21932 p=0.00010) and Norton Sound (Fst=0.42565, p= 0.00000),
whereas no difference was found when compared to the Beaufort Sea sample (Fst = 0.00000, p= 0.87665).

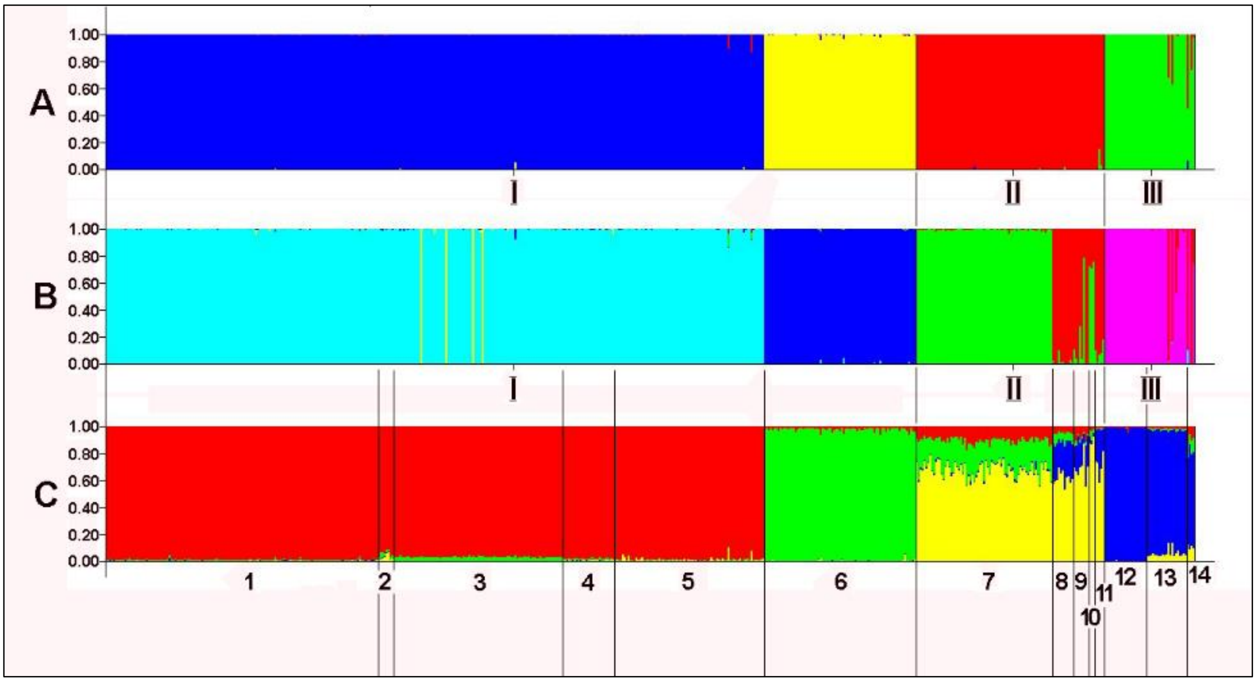

Figure 3. *The results of clustering analysis.*

A - locprior no admixture model for the layout where all individuals from each sea (I - the Okhotsk Sea, II - Bering-Chukchi-Beaufort Seas region, III - the White Sea) were "assigned" to a single population, resulted with K=4 as optimal (Evanno method, Structure Harvester online analysis).

B - the same model as A, resulted with K=6 in accordance with minimal Mean LnP(K) value.

C - locprior admixture model for the layout where the individuals from each location were «assigned» to a separate population, resulted with K=4 in accordance with minimal Mean LnP(K) value. Locations: 1-5 - Western Okhotsk Sea groups, 6 – Shelikhov Bay, 7 – Anadyr Liman, 8 – Chukotka peninsula coast together with 3 samples from Little Diomede Island, 9 - Point Lay, 10 - Beaufort Sea, 11 - Norton Sound, 12-14 - White Sea.

Summarizing, no definitive conclusions can be made regarding the status of belugas sampled along Chukotka peninsula in autumn-winter. Generally, they belong to the Bering-Chukchi-Beaufort pool in its broad definition, but whether they form a separate subunit or represent a part of more widely spread group can not be resolved due to a small sample size.

2. Abundance

No reliable abundance estimates for belugas appearing along Chukotka peninsula are available. General understanding of beluga presence in different months may be obtained from Figure 2 (above).

Pacific walrus aerial surveys were conducted in and around Anadyr Gulf (Figure 4) with opportunistic count of other marine mammal species, including beluga whales. In April 2005 resulted in counting 162 groups of 410 whales total; in April 2006 – 195 groups of 403 individuals total (Litovka et al. 2006). The number presented in Laidre et al (2015) should be taken with caution, because calculation, which resulted in abundance of over 15,000 beluga whales in Anadyr Gulf, was made as "theoretical abundance calculated as direct extrapolation of the estimated mean density to the unsurveyed areas" (Litovka 2013).

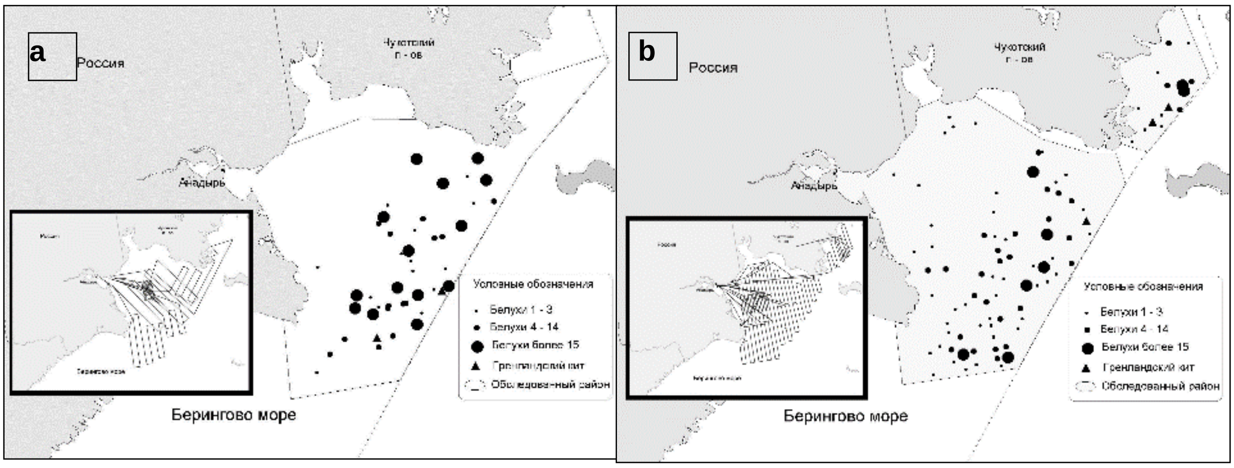

Figure 4. *Aerial survey transects and beluga sightings (circles) in April, 2005 (a) and 2006 (b) in the Western Bering Sea (Litovka et al., 2006)*

3. Anthropogenic removals

Total Allowed Takes (TAT) are issued by the Ministry of Agriculture based on current available data on resource abundance. TATs do not represent actual takes, but rather maximum theoretically sustainable volumes. The captures of beluga whales in the Western Bering Sea, in the Western Chukchi and in the Eastern ES Seas starting 2000 are summarized in Table 1.

Thus, in the whole Chukotka region Native hunters harvested 173 belugas in 17 years, i.e. 10 animals per year (Litovka, 2013). The majority (>82%) of whales were harvested in the Bering Straight area (CFZ) and alaong the Arctic coast of the Chukchi Peninsula (CSFZ). They were taken only during spring and fall migrations. The beluga harvest significantly decreased after technical re-orientation of the Chukotka Native harvest to larger species of whales (bowhead and gray whales) and walruses in 2008-2010. Among other marine mammal species harvested on Chukotks coast, "possible" harvest of beluga whales takes 3.1% (Datsky et al. 2006).

The illegal harvest of belugas in Chukotka is considered insignificantly small, because it requires Native skin boats, special skills and equipment. But no more than a quarter of marine mammal hunters possess them.

According to other sources, in 2006 Chukotka hunters landed 13 belugas, and none were taken in 2007 (Zdor and Mymrin, 2008). In 2009, according to the same authors, 6 belugas were landed in the region (Mymrin and Zdor, 2010). These numbers do not coincide with the numbers available to us (Table 1). Despite the differences in te numbers of landed whales, it is clear that beluga harvest in Chukotka is far below the TAT set by the Ministry of Agriculture.

Table 1. *The annual beluga Total Allowed Takes (TAT) for Chukotka Region and total actual landings in each of the 4 Chukotka fishing zones, 2000-2016 (from Boltnev et al., 2016 updated by D. Litovka).*

Year	TAT	WBSZ[1]	CFZ[2]	CSFZ[3]	ESSFZ[3]	Total take
2000	200	2	0	4	0	6
2001	200	3	0	4	0	7
2002	200	1	3	2	0	6
2003	200	0	0	0	0	0
2004	200	12	2	12	0	26
2005	200	10	0	10	0	20
2006	200	1	1	0	0	2
2007	200	0	3	0	0	3
2008	200	0	6	2	0	8
2009	200	0	50	0	0	50
2010	200	0	8	0	0	8
2011	200	0	0	0	0	0
2012	200	1	9	8	0	18
2013	200	0	11	3	0	14
2014	200	0	0	0	0	0
2015	200	0	3	0	0	3
2016	200	0	2	0	0	2

[1] - WBSZ - Western Bering Sea Fisheries Zone (from Koryak coast to 175°W);
[2]- CFZ - Chukotskaya Fisheries Zone (from 175°W to C. Dezhnev);
[3] - CSFZ - Chukchi Sea Fisheries Zone (from C. Dezhnev to Longa Straight);
[4] - ESSFZ - East-Siberian Sea Fisheries Zone (from Longa Straight to Kolyma River)

4. Incidental mortality

Belugas sometimes become entrapped in ice in Seniavin Strait (Figure 5, also see Yanrakynnot and N.Chaplino on Figure 1).

The most dramatic occasion happened in December 1984 when ca. 3000 beluga whales got entrapped. Mymrin (2006) described the entrapment and actions taken by locals and authorities from December 13, when belugas were first spotted by a local hunter, until June 5 when few belugas were seen the last time. Over 500 belugas from the entrapped aggregation were harvested by locals during winter.

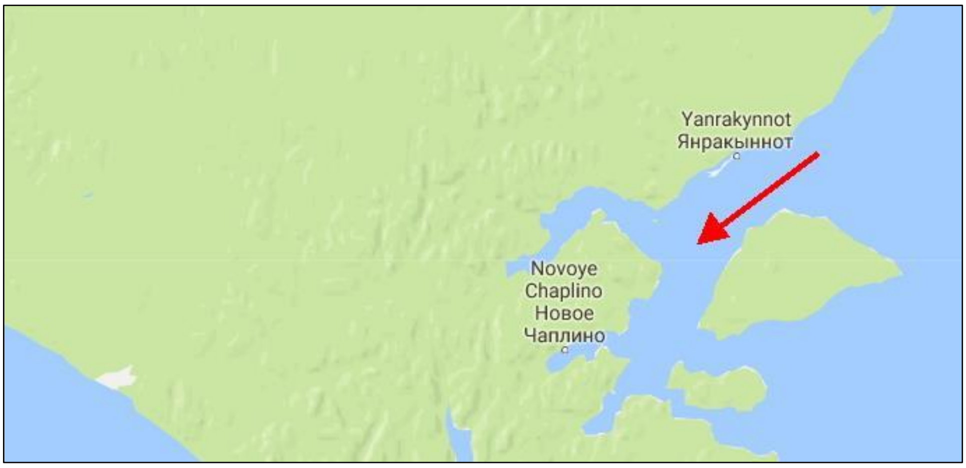

Figure 5. Seniavin Strait (marked with red arrow) - a place of mass beluga entrapments in 1984-1985 and 2011-2012.

Another case of beluga ice entrapment in Seniavin Strait took place in late autumn 2011. Approximately 100 beluga whales spent winter, and most – if not all of them – died in polynias of Seniavin Strait (Zagrebin, 2012). Observartions were conducted from December 10 until early April.

Figure 6. Belugas in ice trap in the Seniavin Strait, Jan 12, 2012 (from Zagrebin, 2012). According to the author, in this "breathing hole" there were 50-80 belugas.

On January 12, 2012 (Figure 6), 7 belugas were harvested, and 1 was found dead. All harvested whales had empty stomachs. On January 16, one dying whale was taken. On February 10, 2 whales were found dead, 2 were harvested; there were still ca. 30 whales in the opening. In early march and in early April, there were 3-5 belugas remaining. Zagrebin suggested that a fast air temperature drop has caused a quick formation of shorefast ice in the northern part of the Seniavin Strait. The author also mentioned that neither Arctic cod nor Saffron cod were observed in the strait during the winter months, and that probable cause of death of belugas whales was starvation.

4. Population trajectory

N/A, since this document describes a mix of stocks on their migration routes and wintering grounds

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

In the absence of published data on current abundance, the theoretical growth rate of 4% and precaution coefficient 0.5 were used to calculate the TAT for Chukotka. The last published abundance estimates of 10000 animals for the western Bering Sea and 4000 whales for the western Chukchi Sea were presented in 1994 (Vladimirov, 1994). These estimates were based on different sources and should be considered as an “expert opinion”. In the view of absence of exact figures and errors, the abundance of 10,000 belugas (Boltnev et al. 2016 based on Vladimirov 1994) was used in calculation for the TAT for Chukotka waters, and was estimated as 10000 x 0.04 x 0.5 = 200 whales (Table 1, above).

6. Habitat and other concerns

Seniavin strait (see above Incidental mortality) may be considered a place of concern. Belugas enter the Seniavin Strait on the way to wintering grounds in late November-early December following Arctic and Saffron cod. A combination of nature factors (air temperature drop, change of wind direction) may lead to a quick ice formation, when belugas get entrapped in an ice belt 20-25 km wide and remain there as late as early June. A short operation by an ice-breaker on making an ice-free corridor appeared ineffective, and fish-supply to the trapped whales, together with keeping the breathing holes open, may be the only solution to save the belugas in such cases (Mymrin, 2006).

There are few cases of killer whale predation on beluga whales. For 10 years of observations, Melnikov (2012) recorded two cases.

Future climate change and sea ice reduction will extend the period and increase the flow of marine traffic in the Bering Strait. Whether this will affect belugas is unknown, since their migration routes and time may also change.

Seismic and military activities in the B-C-B region, including the eastern part of the Eastern-Siberian Sea, conducted in coastal waters and on the continental shelf are also of major concern.

7. Status of the stock

N/A, since this document describes a mix of stocks on their migration routes and wintering grounds.

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Annex 5: Cook Inlet Stock of Beluga Whales — Assessment for the Global Review of Monodontids. 13-16 March 2017

Prepared by Roderick Hobbs, from Beluga Whale (*Delphinapterus leucas*): Cook Inlet Stock *In* Alaska Marine Mammal Stock Assessments, 2016 (Muto et al. 2017). Revised 30 September 2017 to include results from the June 2016 abundance survey.

Marine Mammal Laboratory, Alaska Fisheries Science Center, US National Marine Fisheries Service, 7600 Sand Point Way NE, Seattle, WA 98115

1. Distribution and stock identity

During ice-free months, Cook Inlet beluga whales are typically concentrated near river mouths in upper Cook Inlet (Rugh et al. 2010). The fall-winter-spring distribution of this stock is not fully determined; however, there is evidence that most whales in this population inhabit upper Cook Inlet year-round (Hansen and Hubbard 1999, Rugh et al. 2004, Hobbs et al. 2005, Lammers et al. 2013, Shelden et al. 2015a, Castellote et al. 2015). During summers from 1999 to 2002, satellite tags were attached to a total of 18 beluga whales to determine their distribution through the fall and winter months (Hobbs et al. 2005, Goetz et al. 2012a, Shelden et al. 2015a). Ten tags transmitted whale locations through November and, of those, three transmitted into January, three into March, and one into late May. All tagged beluga whales remained in Cook Inlet, primarily in upper inlet waters (Hobbs et al. 2005, Goetz et al. 2012a, Shelden et al. 2015a).

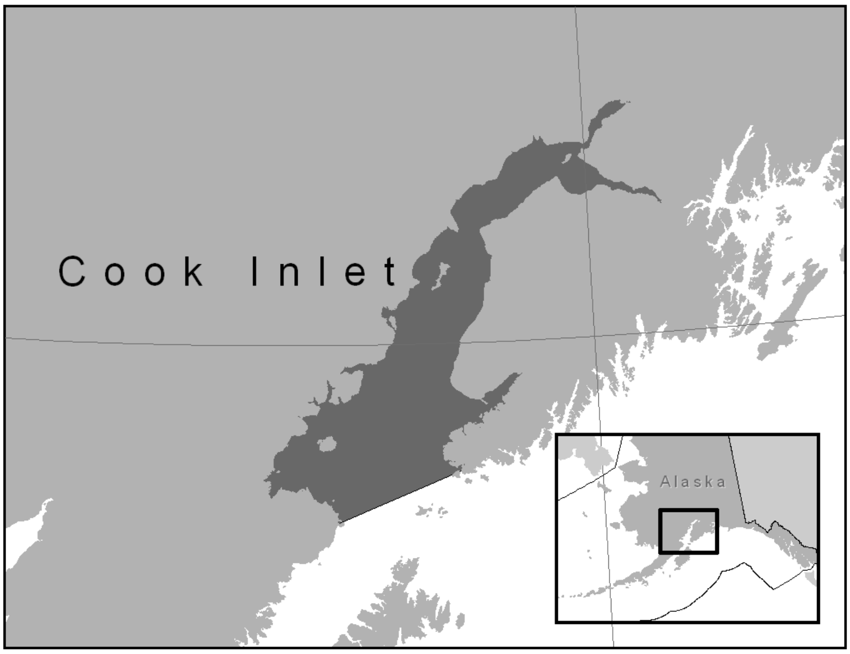

Figure 1. Approximate distribution of beluga whales in Cook Inlet.

A review of all marine mammal surveys and anecdotal sightings in the northern Gulf of Alaska between 1936 and 2000 found only 28 beluga whale sightings outside of Cook Inlet, indicating that very few beluga whales occurred in the Gulf of Alaska outside Cook Inlet (Laidre et al. 2000). A small number of beluga whales (fewer than 20 animals: Laidre et al. 2000, Lucey et al. 2015, O'Corry-Crowe et al.,2015) are regularly observed in Yakutat Bay. Based on genetic analyses, traditional ecological knowledge (TEK), and observations by fishers and others reported year-round, the Yakutat beluga whales likely represent a small, resident group that is reproductively separated from Cook Inlet (Lucey et al. 2015, O'Corry-Crowe et al. 2015). Furthermore, this group in Yakutat appears to be showing signs of inbreeding and low diversity due to their isolation and small numbers (O'Corry-Crowe et al. 2015). Although the beluga whales in Yakutat Bay are not included in the Cook Inlet Distinct Population Segment (DPS) of beluga whales under the Endangered Species Act (ESA), they are considered part of the depleted Cook Inlet stock under the Marine Mammal Protection Act (MMPA) (50 CFR 216.15; 75 FR 12498, 16 March 2010). Notice-and-comment rulemaking procedures would be required to change the NMFS regulatory definition under the MMPA. Thus, Yakutat Bay beluga whales remain designated as "depleted" and part of the Cook Inlet stock.

In Alaska, depending on season and region, beluga whales may occur in both offshore and coastal waters, with summer concentrations in upper Cook Inlet (north of the East and West Forelands), Bristol Bay, the eastern Bering Sea (i.e., Yukon Delta, and Norton Sound), eastern Chukchi Sea (including Kotzobue Sound), and Beaufort Sea (Mackenzie River Delta) (Gurevich 1980, Hazard 1988). Seasonal distribution is affected by ice cover, tidal conditions, access to prey, temperature, and human interaction (Lowry 1985). Beluga whales satellite-tagged in Cook Inlet (Hobbs et al. 2005, Goetz et al. 2012a, Shelden et al. 2015a) remained in Cook Inlet throughout the year, i.e., they are non-migratory and do not interact with other Alaska beluga populations.

Beluga whale stock structure in Alaska was based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution discontinuous (Frost and Lowry 1990); 2) Population response data: possible extirpation of local populations, distinct population trends among regions occupied in summer; 3) Phenotypic data: unknown; and 4) Genotypic data: mitochondrial DNA analyses indicate distinct differences among populations in summering areas (O'Corry-Crowe et al. 2002). Based on this information, five beluga whale stocks are recognized within U.S. waters: 1) Cook Inlet (Fig. 1), 2) Bristol Bay, 3) Eastern Bering Sea, 4) Eastern Chukchi Sea, and 5) Beaufort Sea.

2. Abundance

Aerial surveys during June documenting the early summer distribution and abundance of beluga whales in Cook Inlet were conducted by NMFS each year from 1993 to 2012 (Rugh et al. 2000, 2005; Shelden et al. 2013), after which NMFS began biennial surveys in 2014 (Shelden et al. 2015b) (Fig. 2). NMFS changed to a biennial survey schedule after detailed analysis showed that there would be little reduction in assessment quality (Hobbs 2013).

The abundance estimate for beluga whales in Cook Inlet is based on counts by aerial observers and video analysis of whale groups. Paired, independent observers count each whale group while video is collected during each counting pass. Each count is corrected for subsurface animals (availability correction) and animals at the surface that were missed (sightability correction) based on an analysis of the video tapes (Hobbs et al. 2000). When video counts are not available, observers' counts are corrected for availability and sightability using a regression of counts and an interaction term with an encounter rate against the video count estimates (Hobbs et al. 2000). The variance estimate of the abundance equation was revised using the squared standard error of the average for the abundance estimates in place of the abundance estimate variance and the measurement error (Hobbs et al. 2015a). This reduced the CVs by almost half. The estimate of abundance from the June 2016 survey was 328 belugas (CV = 0.08, 95% CI: 279 to 386, Nmin = 306; Shelden et al. 2017). The 10-year trend (2006-2016) was -0.5% /year (SE = 1.0%, probability of a declining trend: $P(< 0.0)$ = 70%). The trend during the period since management of the hunt began in 1999, (1999-2016), the trend was -0.4% /year (SE = 0.6%, probability of a declining trend: $P(< 0.0)$ = 73%). The June 2016 estimate falls between the June 2014 estimate of 340 whales (CV = 0.08) (Shelden et al. 2015b) and the estimate of 312 beluga whales for 2012 and falls within the statistical variation of both estimates. Annual abundance estimates based on aerial surveys of Cook Inlet beluga whales during the most recent 3 survey period were 312 (2012), 340 (2014), and 328 (2016) resulting in an average abundance estimate for this stock of 327 (CV = 0.06) beluga whales.

3. Anthropogenic removals

Fisheries Information

The estimated minimum average annual mortality and serious injury rate incidental to U.S. commercial fisheries is unknown, although probably low, because only one known beluga whale mortality due to fishery interaction has been reported in the past 10 years. There are no observers on fisheries in Cook Inlet and there have been no voluntary reports of beluga whale mortalities in U.S. commercial fisheries in Cook Inlet. The incompleteness of the data for commercial fisheries operating within the range of

Cook Inlet beluga whales is a concern for this small population. One entanglement in a subsistence fishery was reported to the NMFS Alaska Region on May 7, 2012 by a fisherman reporting a juvenile beluga whale entangled in his salmon fishing net near Kenai, Alaska. The beluga whale was dead and necropsy findings indicated that it was in poor health prior to entanglement and the cause of death was drowning. However, it was not determined whether the beluga whale died before or after the net entanglement.

Alaska Native Subsistence/Harvest Information

Subsistence harvest of beluga whales in Cook Inlet is important to one local village (Tyonek) and the Alaska Native subsistence hunter community in Anchorage. Between 1993 and 1998, the annual recorded subsistence take ranged from 17 to 123 animals (Fig. 2), including beluga whales struck and lost (NMFS 2015).

Following a significant decline in Cook Inlet beluga whale abundance estimates between 1994 and 1998, the Federal government took actions to conserve, protect, and prevent further declines in the abundance of these whales. In 1999 and 2000, Public Laws 106-31 and 106-553 established a moratorium on Cook Inlet beluga whale harvests except for subsistence hunts conducted under cooperative agreements between NMFS and affected Alaska Native organizations. A cooperative agreement, also referred to as a co-management agreement, were was not signed in 1999, so harvest was not authorized in 1999 and 2000. Harvests from 2001 through 2004 were conducted under harvest regulations (69 FR 17973, 6 April 2004) following an interim harvest management plan developed through an administrative hearing. Three beluga whales were harvested in Cook Inlet under this interim harvest plan. In August 2004, an administrative hearing was held to create a long-term harvest plan. An interim plan would have allowed up to eight whales to be harvested between 2005 and 2009 (https://alaskafisheries.noaa.gov/pr/interim-harvest-plan, accessed June 2016). Two whales were taken in 2005 and no takes were authorized in 2006 and later under this agreement. A long-term harvest plan (https://alaskafisheries.noaa.gov/pr/cib-long-term-harvest-management, accessed June 2016), established allowable harvest levels for a 5-year period, based on the average abundance in the previous 5-year period and the growth rate during the previous 10-year period. A harvest is not allowed if the previous 5-year average abundance is less than 350 beluga whales. Under the long-term harvest plan, the 5-year average abundance during the first review period 2003-2007 was 336 whales, harvest would not have been allowed during the subsequent 5-year period 2008-2012 (73 FR 60976; 15 October 2008), so the cooperative agreement was not signed and no hunt occurred. The average abundance of Cook Inlet beluga whales remained below 350 whales during the second review period 2008-2012; therefore, a harvest is not allowed for the current 5- year period 2013-2017.

4. Population trajectory

Current Population Trend

The corrected annual abundance estimates for the period 1994-2014 are shown in Figure 2. From 1999 to 2014, the rate of decline was -1.3% (SE = 0.7%) per year, with a 97% probability that the growth rate is declining (i.e., less than zero), while the 10-year trend (2004-2014) is -0.4% per year (with a 76% probability of declining) (Hobbs et al. 2015, Shelden et al. 2015b).

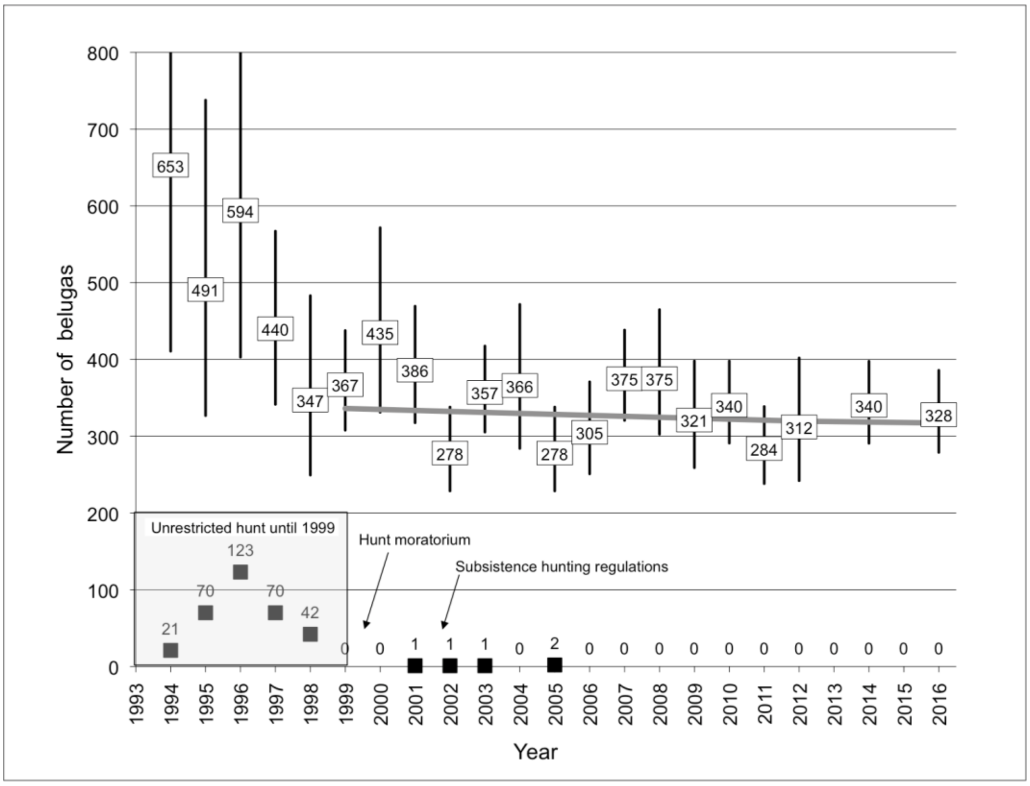

Figure 2. Annual abundance estimates of beluga whales in Cook Inlet, Alaska, 1994-2016 (Hobbs et al. 2015a, Shelden et al. 2015b, Shelden et al. 2017). Black squares show reported removals (landed plus struck and lost) during the Alaska Native subsistence hunt. A struck and lost average was calculated by the Cook Inlet Marine Mammal Council (CIMMC) and hunters for 1996, 1997, and 1998. Black vertical bars depict plus and minus one standard error for each abundance estimate (box label). From 1999 to 2016, the rate of decline (gray trend line) is -0.4% per year (with a 73% probability that the growth rate is declining), while the 10-year trend (2006-2016) is -0.5% per year (with a 70% probability of declining).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

Minimum Population Estimate

The minimum population estimate (N_{MIN}) is calculated according to Equation 1 from the potential biological removal (PBR) guidelines (Wade and Angliss 1997). Thus, $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the 3 survey average population estimate (N) of 327 whales and an associated CV(N) of 0.06, N_{MIN} for the Cook Inlet beluga whale stock is 310 beluga whales.

Current and maximum net productivity rates

A reliable estimate of the maximum net productivity rate is currently not available for the Cook Inlet beluga whale stock. Hence, until additional data become available, the cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% is recommended to be employed for this stock (Wade and Angliss 1997). This figure is similar to the 4.8% annual increase that has been documented for the Bristol Bay beluga whale stock (Lowry et al. 2008).

Potential Biological Removal

Under the 1994 reauthorized MMPA, the PBR was defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. In past Stock Assessment Reports for this stock, from 1998 through 2005, NMFS calculated a value for PBR. Given the low abundance relative to historical estimates and low known levels of human-caused mortality since 1999, this stock should have begun to grow at or near its maximum productivity rate (2-6%), but for unknown reasons the Cook Inlet beluga whale stock is not increasing. Because this stock does not meet the assumptions inherent to the use of the PBR, NMFS has decided it would not be appropriate to calculate a maximum number that may be removed while allowing the population to achieve its Optimum Sustainable Population. Thus, the PBR for this stock is undetermined.

6. Habitat and other concerns

Other Mortality

Mortality related to live stranding events, where a group of beluga whales becomes stranded as the tide recedes has been reported in Cook Inlet (Table 1). Improved record-keeping was initiated in 1994, and reports have since included the number of beachcast carcasses and live stranded beluga whales (NMFS 2016). Most whales involved in a live stranding event survive, although some deaths may be missed by observers if whales die later from live stranding-related injuries (Vos and Shelden 2005, Burek-Huntington et al. 2015). Between 2009 and 2014, there were approximately 300 whales involved in 10 known live stranding events, with four deaths reported (Table 1). In 2014, necropsy results from two dead whales found in Turnagain Arm suggested the whales had recently live stranded, and that the live stranding may have contributed to their deaths. No live stranding events were reported to NMFS in the period prior to the discovery of these whales suggesting that not all strandings are observed (Table 1). Most live strandings occur in Knik Arm or Turnagain Arm, both of which are shallow and dangerous waterways. Turnagain Arm has the largest tidal range in the U.S., with a mean of 9.2 m (30 ft).

Table 1. Cook Inlet beluga whale strandings investigated by NMFS during 2009-2014 (NMFS~~,~~ 2015).

Year	Beachcast carcasses	Number of beluga whales per live stranding event (number of associated known or suspected resulting deaths)
2009	4	16-21 (0)
2010	5	11(0), 2(0)
2011	3	2(0)
2012	3	12(0), 23(0), 3(0)
2013	5	0
2014	10	76 (0), unknown (2)
Total	30	145-150 (2)

Another source of beluga whale mortality in Cook Inlet is predation by mammal-eating killer whales. Killer whale sightings were not well documented and were likely rare in the upper inlet prior to the mid-1980s. From 1982 through 2014, 29 killer whale sightings in upper Cook Inlet (north of East and West Foreland) were reported to NMFS. It is not known which of these were mammal-eating killer whales (i.e., transient killer whales) that might prey on beluga whales and which were fish-eating killer whales (i.e., resident killer whales) that would not prey on beluga whales. Between 9 and 12 beluga whale deaths during this time were suspected to be a direct result of killer whale predation (NMFS 2016). The last confirmed killer whale predation of a beluga whale in Cook Inlet occurred in 2008 in Turnagain Arm. In June 2010, a beluga whale carcass found near Point Possession was speculated to have injuries associated with killer whale predation; however, the poor condition of the beluga whale carcass prevented a positive determination of cause of death. From 2011 through 2014, NMFS has received no reports of killer whale sightings in upper Cook Inlet or possible predation attempts.

A photo-identification study (Kaplan et al. 2009) did not find any instances where Cook Inlet beluga whales appeared to have been entangled in, or to have otherwise interacted with, fishing gear. However, in 2010, a beluga whale with a rope entangled around its girth was observed and photo-documented during the period of May through August. The same whale was photographed in July and August 2011, August 2012, and July 2013, still entangled in the rope line (McGuire et al. 2014). This whale is currently considered to have a non-serious injury (Helker et al., 2016).

Between 1998 and 2013, 38 necropsies were performed on beluga whale carcasses (23% of the known stranded carcasses during this time period) (Burek-Huntington et al. 2015). The sample included adults (n = 25), juveniles (n = 6), calves (n = 3), and aborted fetuses (n = 4). When possible, a primary cause of death was noted along with contributing factors. Cause of death was unknown for 29% of the necropsied carcasses. Cause of death in the others was attributed to various types of trauma (18%), perinatal mortality (13%), mass stranding (13%), single stranding (11%), malnutrition (8%), or disease (8%). Several animals had mild to moderate pneumonia, kidney disease, and/or stomach ulcers that likely contributed to their cause of death.

Habitat Concerns
Beluga whale critical habitat includes two geographic areas of marine habitat in Cook Inlet that comprise 7,800 km2 (3,013 mi2), excluding waters by the Port of Anchorage (76 20180, 11 April 2011). Based on available information from aerial surveys, tagged whales, and opportunistic sightings, beluga whales remain within the inlet year-round. Since 2000, most whales have been found in the upper inlet north of East and West Foreland not only during the summer months (Rugh et al. 2010) and in the fall as well (Rugh et al. 2004), with tagged whales travelling between the lower and upper inlet and offshore waters >10 m deep during the winter (Hobbs et al. 2005, Goetz et al. 2012a, Shelden et al. 2015a, Castellote et al. 2015). Whether this contracted distribution is a result of changing habitat (Moore et al. 2000), prey concentration, or predator avoidance (Shelden et al. 2003) or can simply be explained as the contraction of a reduced population into a small number of preferred habitat areas (Goetz et al. 2007, 2012b) is unknown.

With the limited range of this stock, Cook Inlet beluga whales are vulnerable to human-induced or natural perturbations within their preferred habitat. Goetz et al. (2012b) modeled habitat preferences using NMFS' 1994-2008 abundance survey data. In large areas, such as the Susitna Delta and Knik Arm, they found a high probability of beluga whale presence in larger group sizes. Beluga whale presence also increased closer to rivers with Chinook salmon (*Oncorhynchus tshawytscha*) runs, such as the Susitna River. The Susitna Delta also supports two major spawning migrations of a small, schooling smelt (eulachon, *Thaleichthys pacificus*) in May and July. Threats that have the potential to impact this stock and its habitat include the following: changes in prey availability due to natural environmental variability, ocean acidification, and commercial fisheries; climatic changes affecting habitat; predation by killer whales; contaminants; noise; ship strikes; waste management; urban and airport runoff; construction projects; and physical habitat modifications that may occur as Cook Inlet becomes increasingly urbanized (Moore et al. 2000, Lowry et al. 2006, Hobbs et al. 2015b, Norman et al. 2015). Planned projects that may alter the physical habitat of Cook Inlet include highway improvements; mine construction and operation; oil and gas exploration and development; and expansion and improvements to ports.

Threats
The recovery plan for Cook Inlet belugas lists ten potential threat types and ranks them as having a low, medium, or high level of relative concern for affecting the CI beluga population (NMFS 2016). The identified threat types and their level of relative concern are: catastrophic events (relative concern: high); cumulative effects of multiple stressors (relative concern: high); anthropogenic noise (relative concern: high); disease agents (relative concern: medium); habitat loss or degradation (relative concern: medium); reduction in prey (relative concern: medium); unauthorized take (relative concern: medium); pollution (relative concern: low); predation (relative concern: low); and subsistence hunting (relative concern: low).

7. Status of the stock
The Cook Inlet stock of beluga whales is small, less than 350 whales, and stable or declining, 17 year (1999-2016) trend -0.4% per year. This beluga whale stock was designated as "depleted" under the MMPA in May 2000 (65 FR 34590, May 21, 2000), and on October 22, 2008, NMFS listed Cook Inlet beluga whales as "endangered" under the ESA (73 FR 62919, October 22, 2008). Therefore, the Cook Inlet beluga whale stock is considered a MMPA strategic stock. NMFS completed a Recovery Plan for Cook Inlet beluga whales in December 2016 (NMFS 2016).

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Annex 6: Eastern Bering Sea Beluga Whale Stock Status Review

for the NAMMCO Global Review of Monodontids

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1. Distribution and stock identity

Belugas of the eastern Bering Sea (EBS) stock are found in summer near the Yukon Delta and throughout Norton Sound (Lowry *et al*. in press). As ice forms in the late autumn these whales move offshore and south as far as St. Lawrence Island to the west and Togiak Bay to the south, generally remaining in ice covered waters (Citta *et al*. 2017).

The non-uniform distribution of beluga whales in coastal waters of the Bering, Chukchi, and Beaufort Seas in summer is indicative of likely population subdivision and formed the basis for original, but provisional, stock designations (Frost and Lowry 1990). It was recognized at the time that identification of more biologically meaningful stocks would require genetic studies to elucidate the underlying patterns of demographic and reproductive relationships among seasonal groupings (O'Corry-Crowe and Lowry 1997). Over the past two decades several genetic studies have been conducted on seasonal groupings that occur adjacent to Alaska and Chukotka (Russian Federation), primarily summering and migrating groups, to resolve patterns of dispersal and gene flow. The studies revealed substantial mitochondrial DNA (mtDNA) differentiation among summering groups in Bristol Bay, Norton Sound, and Anadyr Gulf in the Bering Sea, in nearshore waters along Kasegaluk Lagoon in the Chukchi Sea, and in the Mackenzie Delta-Amundsen Gulf region in Beaufort Sea, that likely reflects long-established patterns of female-mediated philopatry and demographic isolation (O'Corry-Crowe *et al*. 1997, 2002; Brown-Gladden *et al*. 1997, Meschersky *et al*. 2008; Fig. 1). This has led to their identification as the following five demographically distinct management stocks: 1) Bristol Bay, 2) EBS, 3) Gulf of Anadyr, 4) Eastern Chukchi Sea, and 5) Beaufort Sea (Laidre *et al*. 2015, Muto *et al*. 2016). A few studies have documented lower levels of nuclear DNA (microsatellite) heterogeneity among geographic strata compared to mtDNA. This has been taken as evidence of male-mediated gene flow among summering groups, possibly in shared wintering areas (Brown-Gladden *et al*. 1999, Meschersky *et al*. 2013), or it could reflect a slower rate of drift in markers with higher effective population size (O'Corry-Crowe *et al*. 2010). More recent studies question the common wintering area hypothesis (Citta *et al*. 2017) and whether gene flow is extensive among stocks in the Bering, Chukchi, and Beaufort seas (O'Corry-Crowe *et al*. in prep.).

Beluga whales can occur in the waters of the northeastern Bering Sea, from the Yukon and Kuskokwim deltas to Norton Sound, in all seasons. Whales from more than one stock likely migrate through this region in spring and autumn between summering grounds in the northeastern Bering, and the eastern Chukchi and Beaufort Seas and wintering grounds in the central and southern Bering Sea (O'Corry-Crowe *et al*. 1997, Citta *et al*. 2017). Only one of these groupings, the EBS stock, occupies nearshore waters in the northeastern Bering in summer (Fig. 1).

The occurrence of belugas in Norton Sound in the 1840s was described by Zagoskin (1967). He noted that beginning in July "the beluga appear in great numbers with their young as they follow the fish outside the mouths of the Yukon." He described large organized hunts that occurred in mid-July in Pastol Bay, where as many as 100 animals were taken in a single drive. According to Nelson (1887), belugas usually appeared in the southern Sound between the 5th and 10th of June, and schools of 20 to over 100 animals were frequently seen in the bay nearby. He documented the summer occurrence of belugas at the mouth of the Yukon River, and as much as 800 km upstream.

A compilation of all available observations, including both scientific and traditional knowledge, showed that belugas occur throughout the coastal zone of the EBS from the mouth of the Yukon River to northern Norton Sound near Nome, with relatively few sightings made far offshore (Frost and Lowry 1990). Whales were seen from shortly after breakup (usually May) until freezeup (usually November). A further confirmation that belugas have occurred regularly in the EBS region comes from records of harvests by Alaska Native hunters at 9 villages in southern, eastern, and northern Norton Sound, and 13 villages in the Yukon delta (Lowry *et al*. 1989, Frost and Suydam 2010, Alaska Beluga Whale Committee (ABWC) unpublished).

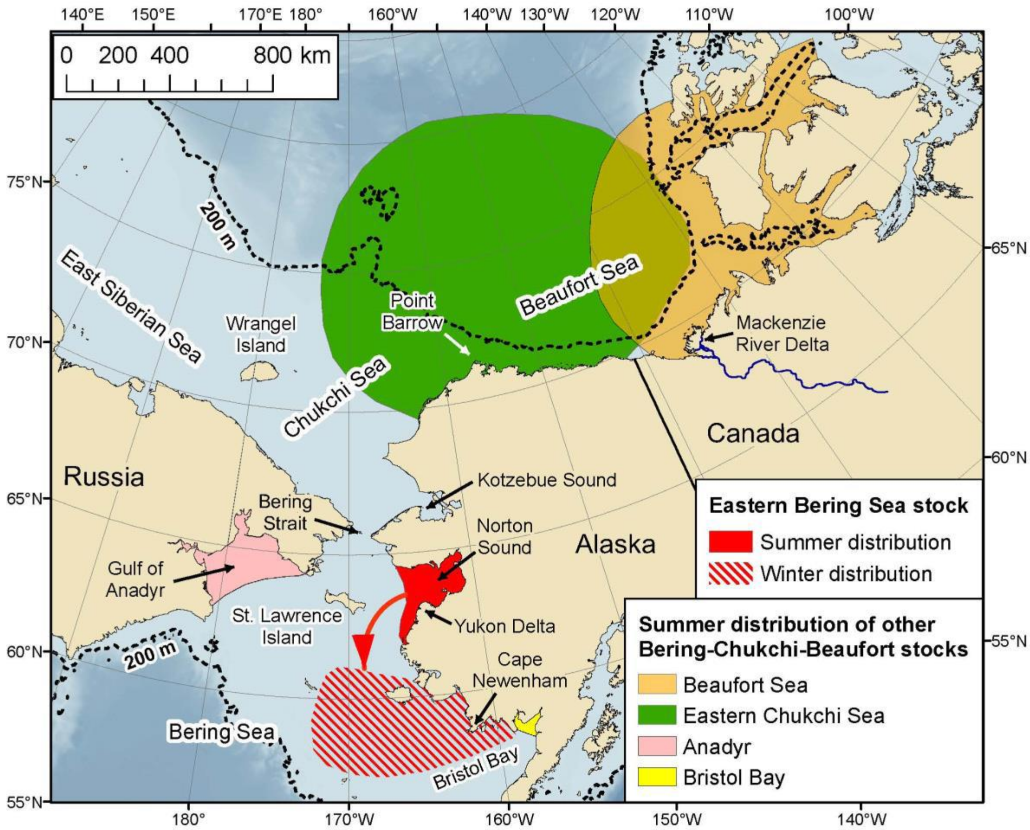

Figure 1. Map of the Bering-Chuckhi-Beaufort sea region showing summer distribution of all beluga stocks in the region and the winter distribution of the eastern Bering Sea stock.

The ABWC began flying aerial surveys for beluga whales in the EBS in 1992. Most of those surveys were flown in June when belugas were concentrated off the mouths of the Yukon River and in southern Norton Sound (Fig. 2, Lowry *et al*. 1999, Lowry *et al*. in press). Satellite depth recorders (SDRs) were attached to two beluga whales in northern Norton Sound in autumn of 2012 (Citta *et al*. 2017). Those whales remained in Norton Sound in October and early November, then with advancing sea ice cover they shifted their distribution southward but still remained in the EBS region (Fig. 1). The tagged animals were both back in Norton Sound by mid-June. Another beluga was tagged in northern Norton Sound in November 2016. That animal spent November, December, and January in the western Sound and adjacent waters of the EBS (http://www.north-slope.org/departments/wildlife-management/co-management-organizations/alaska-beluga-whale-committee/abwc-research-projects/satellite-maps-of-tagged-alaskan-beluga-stocks/satellite-tagging-maps-nov-2016).

Studies on patterns of mtDNA variation revealed that the summer beluga concentration in Norton Sound is demographically distinct from the near-resident population in Bristol Bay and groups with summering areas in the eastern Chukchi and Beaufort seas (O'Corry-Crowe *et al*. 1997, 2002; Brown-Gladden *et al*. 1997). Whales from the Yukon and Kuskokwim deltas were similar to Norton Sound but sample sizes were too small to definitively assign them to the Norton Sound subpopulation. However, the three belugas that have been SDR tagged in northern Norton Sound all spent time in the Yukon Delta. Similarly, no clear distinction has been observed between early and late summer whales in Norton Sound. The summering groups in Norton Sound were subsequently identified as the EBS population (Laidre *et al*. 2015, Muto *et al*. 2016). As with a recent 1996 event in Kotzebue Sound (see eastern Chukchi Sea assessment), analyses of mtDNA and microsatellite loci detected an anomalous occurrence of whales from another stock in Norton Sound in 1996. This atypical year most likely involved whales from the Beaufort Sea stock and the anomalous events coincided with anomalous ice years in the Bering-Chukchi-Beaufort region (O'Corry-Crowe *et al*. 2016). Recent genetic analysis of nuclear DNA in conjunction with the mtDNA work has determined that belugas of the EBS stock may interbreed with other stocks in the Bering-Chukchi-Beaufort region, possibly during winter or early spring (O'Corry-Crowe *et. al* in prep.).

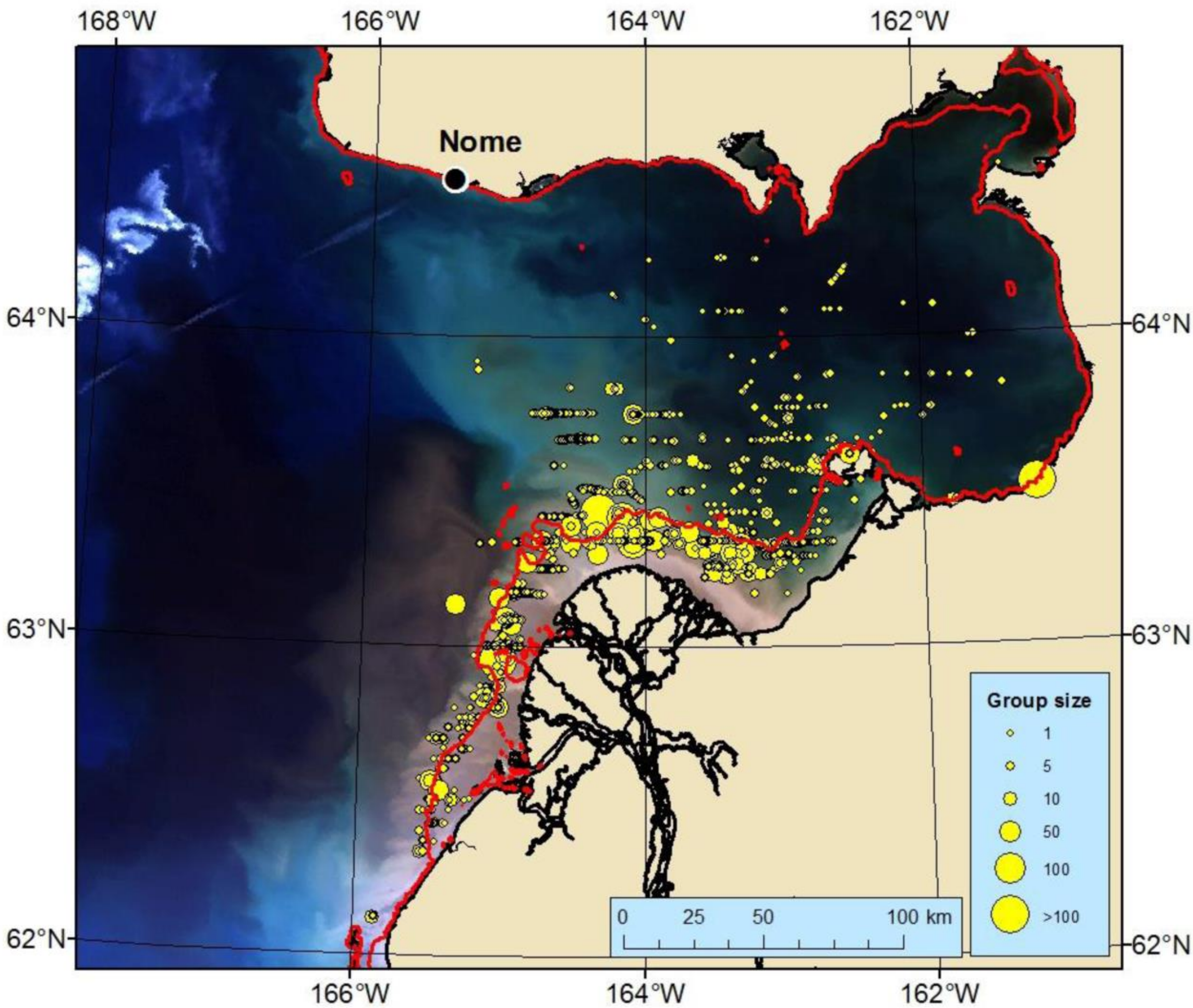

Figure 2. MODIS image of Norton Sound and the Yukon River Delta taken from the Terra satellite on 17 June 2002. Yellow dots are sightings of beluga whales made during aerial surveys 1995-2000. Red line indicates the 5m isobath. The discharge plume of the Yukon River shows as gray/brown.

2. Abundance

The ABWC has worked to develop a population estimate for the EBS stock beginning with the first systematic aerial surveys of beluga whales in the Norton Sound/Yukon Delta region flown during May, June, and September 1992, and June 1993-1995 (Lowry *et al*. 1999). Preliminary abundance estimates confirmed that the EBS stock was quite large but the estimates were not at that time considered ready to use for calculation of removable levels. Additional surveys were flown in June of 1999 and 2000. Density and abundance were estimated from the 2000 survey because it represented the most recent data and had the most complete and systematic coverage of the area (Lowry *et al*. in press). In 2000, belugas were rare in the northern portion of Norton Sound, thus the study area was reduced to central and southern Norton Sound and the Yukon Delta and divided into four strata by latitude. Density estimated with the model that received most Akaike Information Criterion support was 0.121 belugas/km2 and the number of belugas at the surface in the study area was estimated to be 3,497 (coefficient of variation (CV) = 0.37). A generally accepted correction factor for availability of 2.0 was applied, resulting in an abundance estimate of 6,994 (95% confidence interval 3,162-15,472).

3. Anthropogenic removals

Subsistence harvest

The ABWC has collected data on Alaska Native subsistence harvests of EBS belugas since 1987 (Fig. 3a). Harvest data for 1987-2006 were reported by Frost and Suydam (2010). Here, we report EBS harvest data for 2007-2016 (ABWC, unpublished data).

Twenty-two villages harvest belugas from the EBS stock, 9 from Norton Sound and 13 from the Yukon delta (some almost 150 km from the ocean). Harvest levels have been variable, ranging from 31 in 1987 to 281 in 2002. The average annual reported harvest from this stock increased from 152 during 1987-2006 to 190 during 2007-2016. This increase was not statistically significant and is almost certainly due to better data being collected from more villages. When monitoring began in 1987, only 4 villages reported their harvest (Frost and Suydam 2010) but by 2016, 21 villages were reporting (ABWC, unpublished data). During 2007-2016 there was a small and non-significant ($p = 0.55$) increasing trend in the number of belugas harvested (Fig. 3b).

Reporting of struck and lost belugas is sporadic. Intermittent struck and lost data are available for the EBS stock for 17 villages during the last five years. During those years, the number of belugas struck and lost averaged 13% of the landed harvest (ABWC, unpublished data). Frost and Suydam (2010) did not report a struck and lost rate for the EBS stock.

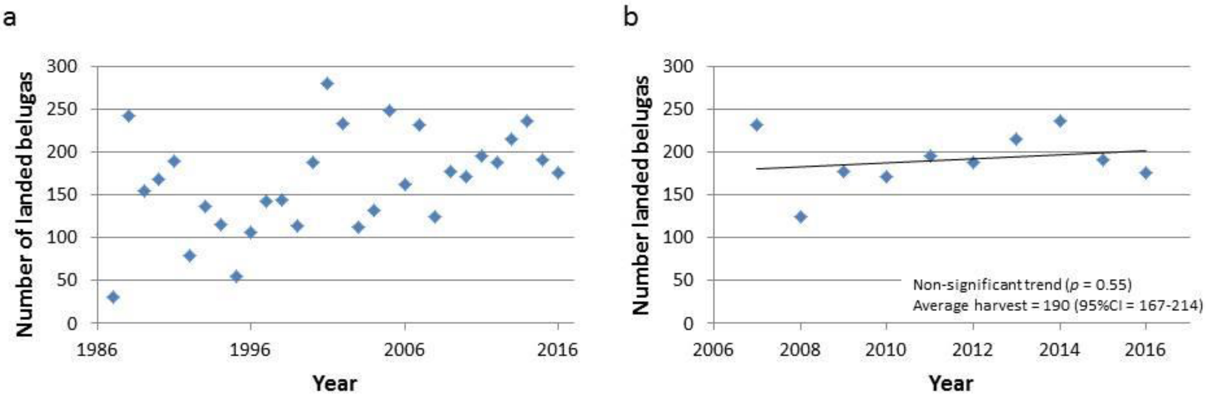

Figure 3. The number of EBS belugas landed by Alaska Native subsistence hunters during 1987–2016 (a), and the trend in the number of belugas landed during 2007–2016 (b). For more information on how harvest is documented, see Frost and Suydam (2010).

Bycatch

In the USA, some commercial fisheries that operate in federal waters (3-200 nm offshore) and may take marine mammals as bycatch are regularly monitored. In Alaska, three commercial fisheries that could have interacted with beluga whales from the EBS stock have been monitored: Bering Sea and Aleutian Islands groundfish trawl, longline, and pot fisheries. No mortality or serious injury to beluga whales was reported in those fisheries. State-managed commercial, personal use, and subsistence gillnet fisheries occur in nearshore waters of the EBS. While they are a potential source of bycatch mortality and bycatch is not systematically monitored, only one beluga whale take has been reported in a subsistence salmon gillnet, and there is no reliable estimate of total fisheries bycatch for this stock (Muto *et al*. 2016).

4. Population trajectory

There are no data on maximum net productivity for EBS belugas. For the Bristol Bay beluga stock the estimated rate of increase over the 12-year period 1992-2005 was 4.8%/year (95% confidence interval = 2.1%-7.5%; Lowry *et al*. 2008), but that may not be the maximum rate. The value measured for Bristol Bay is close to the 4%/year that is used by the National Marine Fisheries Service (NMFS) as the default maximum net productivity rate for cetaceans (Wade1988).

Because there has been only one population estimate, the trend in abundance of the EBS stock is unknown (Laidre *et al*. 2015, Muto *et al*. 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The U.S. Marine Mammal Protection Act defines the potential biological removal (PBR) as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = NMIN × 0.5RMAX × FR. However, because the most recent abundance estimate available for the EBS at the time of the last NMFS Stock Assessment Report was more than eight years old the PBR for the stock was considered to be "undetermined" (Muto *et al*. 2016).

A PBR can be calculated using the abundance estimate provided in Lowry *et al*. (in press) as follows: NBEST = 6,994; CV = 0.37; NMIN = 5,173, RMAX = 0.04; FR = 1.0; PBR = 103. It should be noted that this estimate includes an arbitrary correction factor that has no associated CV.

6. Habitat and other concerns

Because they are an ice-associated species there is concern about the possible effects on belugas of climate warming and associated loss of sea ice habitat. Laidre *et al*. (2015) found little change in the duration of the reduced ice (summer) period in the Bering Sea from 1979 to 2013. In a long-term study of belugas off West Greenland, Heide-Jørgensen *et al*. (2010) found that belugas responded to changing sea ice by shifting their distribution and that abundance increased during a period of generally declining ice cover. They stated that "Global warming and sea-ice declines may pose less of a problem for belugas than to other Arctic marine mammals." Laidre *et al*. (2008) concluded that on a rangewide basis the beluga would be the arctic cetacean least sensitive to climate change because of their wide distribution and flexible habits.

O'Corry-Crowe *et al*. (2016) analyzed long-term sighting and genetic data on belugas in the Bering, Chukchi, and Beaufort seas in conjunction with multi-decadal patterns of sea ice to investigate the influence of sea ice on spring migration and summer residency patterns. While substantial variations in sea ice conditions were found across seasons, years, and sub-regions, the pattern of beluga migration and residency was quite consistent. Those results suggest that belugas can accommodate widely varying sea ice conditions to perpetuate philopatry to traditionally used areas.

With climate warming and decreases in sea ice there will be increased human activity in northern waters and especially in the Arctic (Reeves *et al*. 2014, Laidre *et al*. 2015). In addition to oil and gas exploration and production, shipping, tourism, and other commercial development have the potential to impact belugas and their habitat. However, predicting the type and magnitude of likely impacts is difficult at this time (Muto *et al*. 2016).

Belugas that summer in the Yukon Delta region very likely feed on Pacific salmon (*Oncorhynchus* spp.). They may consume a substantial portion of some Yukon River salmon runs, thereby affecting trophic structure of the ecosystem and potentially impacting catches in commercial and subsistence fisheries (Lowry *et al*. in press).

7. Status of the stock

The EBS stock of beluga whales is one of four stocks in western Alaska that is co-managed by NMFS and the ABWC (Adams *et al*. 1993, Fernandez-Gimenez *et al*. 2006). Two of the agreed upon objectives of the management plan are to "conserve the Western Alaska beluga whale population" and to "protect Alaska Native beluga whale subsistence hunting traditions and culture" (ABWC 1999). The average harvest for the past 10 years (190) is considerably higher than the PBR calculated based on abundance surveys conducted in 2000 (103). However, the estimate of PBR is almost certainly low because the 2000 survey did not include all potential beluga habitat (e.g., the Yukon River itself), dark gray animals were particularly hard to see in muddy water coming from the Yukon, and the analysis did not account for perception bias (Lowry *et al*. in press).

The EBS beluga stock is quite large, and every June they concentrate off the mouths of the Yukon River and in Norton Sound. They are widely spread throughout the area and in essence form a single school of whales approximately 200 km long (Fig. 2). The most recent estimate of about 7,000 is based on data collected in 2000 and relies on an arbitrary correction factor to account for availability bias. A repeat of this survey is being planned for June 2017 to better estimate abundance and PBR. Additional work (e.g., tagging) is needed to develop better correction factors. Of particular concern is the effect of turbid Yukon River water on beluga sightability.

While available scientific data do not allow an estimation of population trend, local and traditional knowledge indicates that there has not been any decrease in abundance or availability of EBS belugas in recent years (ABWC, unpublished).

EBS beluga whales are not designated as "depleted" or "strategic" under the MMPA nor are they listed as "threatened" or "endangered" under the U.S. Endangered Species Act (Muto *et al.* 2016). In an assessment done in 2008, the International Union for the Conservation of Nature listed belugas as a species as "Near Threatened" and also noted that the various subpopulations should be assessed separately (Jefferson *et al*. 2012).

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Annex 7: Bristol Bay Beluga Whale Stock — Status Review for the NAMMCO Global Review of Monodontids

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1. Distribution and stock identity

Belugas of the Bristol Bay stock are typically found in Nushagak and Kvichak Bays and tributaries during the summer and ranging widely in the northeast region of Bristol Bay in the winter (Fig. 1). The Bristol Bay stock of beluga whales is probably the most studied beluga stock in Alaskan waters. This is largely because Bristol Bay contains the largest commercial sockeye salmon (*Oncorhynchus nerka*) fishery in the world (Jones *et al.* 2013). Studies of belugas in Bristol Bay began in the 1950s because there was concern that they were consuming too many smolt and limiting salmon populations (e.g., Brooks 1955; Lensink 1961; Fish and Vania 1972). Since then, researchers have studied the diet (e.g., Brooks 1955; Lowry *et al.* 1986; Quakenbush *et al.* 2015), distribution (e.g., Frost *et al.* 1984, 1985; Frost and Lowry 1990; Lowry *et al*. 2008; Citta *et al.* 2016, 2017), abundance (e.g., Frost and Lowry 1990; Lowry *et al.* 2008), behavior (e.g., Frost *et al.* 1985), health (e.g., Norman *et al.* 2012; Cornick *et al.* 2016), and subsistence harvest (Frost and Suydam 2010) of belugas in Bristol Bay.

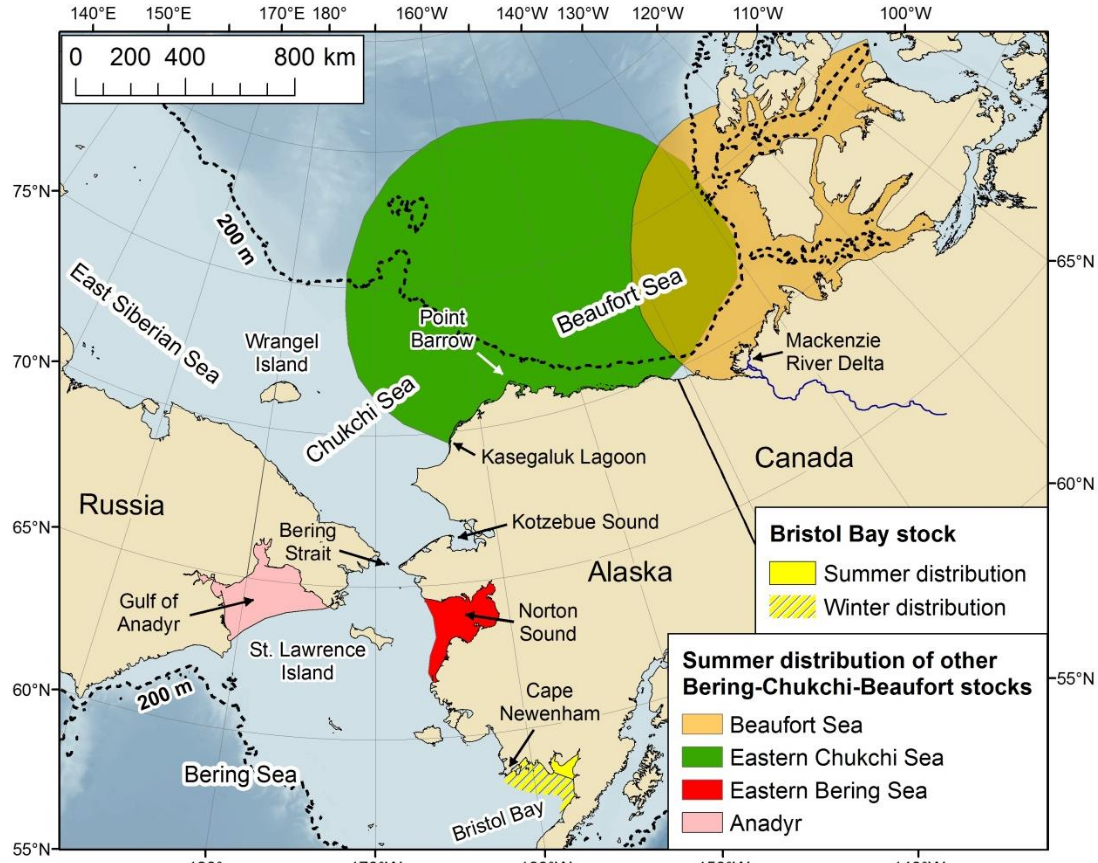

Figure 1. The annual range of belugas in the Bristol Bay stock and the summer distribution of other known beluga stocks in the Bering, Chukchi, and Beaufort seas.

Satellite telemetry studies indicate that Bristol Bay belugas remain in the greater Bristol Bay region throughout the year (e.g., Citta *et al.* 2016, 2017). In spring and summer (Fig. 2a and 2b), their distribution is largely restricted to Nushagak and Kvichak bays (Frost *et al.* 1984, 1985; Lowry *et al.* 2008; Citta *et al.* 2016), which are in the northeast of Bristol Bay. Here, belugas are known to feed on a variety of fish, including salmonids, and invertebrates (e.g., Brooks 1955; Lowry *et al.* 1986). After the salmon runs end in late summer (Fig. 2c), their distribution widens, but is still contained mostly within Nushagak and Kvichak bays (Citta *et al.* 2016). In winter, Bristol Bay belugas range into outer Bristol Bay, frequenting the inner bays less often, perhaps because they are covered in ice and pose a risk of entrapment or because there are few prey available there. However, even in winter, Bristol Bay belugas tagged with satellite depth recorders (SDRs) have not passed west of Cape Newenham (Fig. 2d; see also Citta *et al.* 2016). The nearest stock of belugas is the Eastern Bering Sea stock; the ranges of these two stocks overlap in winter, at least in space if not time (Fig. 3). Belugas in both the Bristol Bay and Eastern Bering Sea populations were tagged with SDRs in 2013. Although a beluga from the Eastern Bering Sea stock moved into the range of Bristol Bay belugas in January 2013, this occurred when Bristol Bay belugas were within the inner bays and there was no evidence that the two populations were in the same place at the same time (Citta *et al.* 2017).

Studies examining patterns in mitochondrial DNA (mtDNA) support the idea that Bristol Bay belugas are distinct from other stocks that summer and winter in the Bering Sea (O'Corry-Crowe et al. 1997, 2002; Muto *et al*. 2016). More recent analyses of microsatellite (nDNA) variation has found a lower but still discernable level of differentiation compared to mtDNA, indicating that there is only limited exchange among beluga stocks in the Bering Sea (O'Corry-Crowe *et. al* In prep).

Furthermore, the Bristol Bay stock is a single stock and is not composed of distinct sub-populations within Bristol Bay. Satellite tagging studies show that belugas commonly move between their summer concentration areas in Nushagak and Kvichak bays (Citta *et al.* 2016) and a comparison of mitochondrial DNA from whales in Nushagak and Kvichak bays found no genetic differentiation (O'Corry-Crowe, unpublished data).

2. Abundance

Aerial surveys were conducted in Bristol Bay periodically between 1993 and 2016 (Lowry *et al.* 2008; Alaska Beluga Whale Committee (ABWC; unpublished data). Within each survey year, multiple flights covered the entire area where belugas have been observed during the survey period in late June and early July. Weather permitting, two flights were flown each day; only data from flights with good viewing conditions were considered (See Lowry *et al.* 2008 for more information). The count of belugas varied greatly between individual flights and population inference was typically made using the maximum count within a year, as this was the minimum number of belugas in the population.

Counts from aerial surveys are typically corrected for the number of belugas that are diving and not available to be sampled and/or for the number that are available but missed by the observer. Because beluga calves are small and gray colored and are typically not spotted in the silty (i.e., gray-colored) water, a separate correction is used for calves (e.g., Brodie 1971). In Bristol Bay, however, correction factors have only been developed to correct for the number of adults at the surface (i.e., availability correction). Frost *et al.* (1985) used VHF transmitters to estimate an availability correction factor of 2.75. This estimate was later revised to 2.62 by Frost and Lowry (1995). Citta *et al.* (ABWC unpublished data) used satellite transmitters to estimate a correction factor of 3.3 (SD=4.52). The estimate of abundance for Bristol Bay belugas in the most current National Marine Fisheries Service Stock Assessment Reports is 2,877 (Muto *et al.* 2016) and was derived by multiplying the average of the maximum count from surveys in 2004 (794) and 2005 (1,067) by an availability correction factor (2.62) and by a correction for the number of calves (1.18) from a study of belugas in Cumberland Sound, Baffin Island, Canada (Brodie 1971).

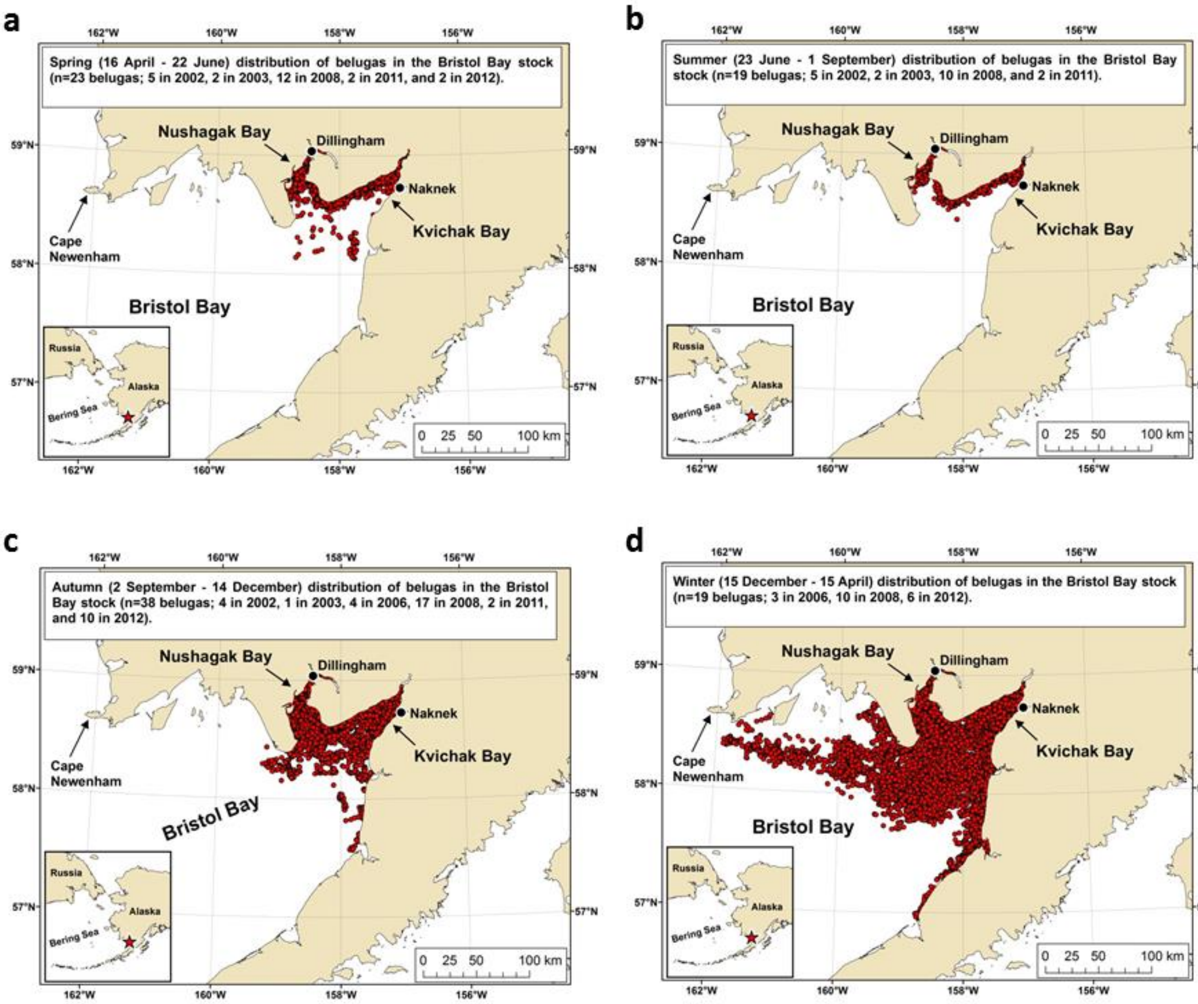

Figure 2. Locations for satellite tagged beluga whales in the Bristol Bay stock for (a) the spring (16 April – 22 June), when salmon smolt (*Oncorhynchus* spp.) and rainbow smelt (*Osmerus mordax*) are migrating; (b) the summer (23 June – 1 September), when adult salmon are migrating; (c) the autumn, after the salmon migrations are complete (2 September – 14 December); and (d) the winter (15 December – 15 April), when sea ice is typically present. Data include those presented in Citta *et al.* (2016).

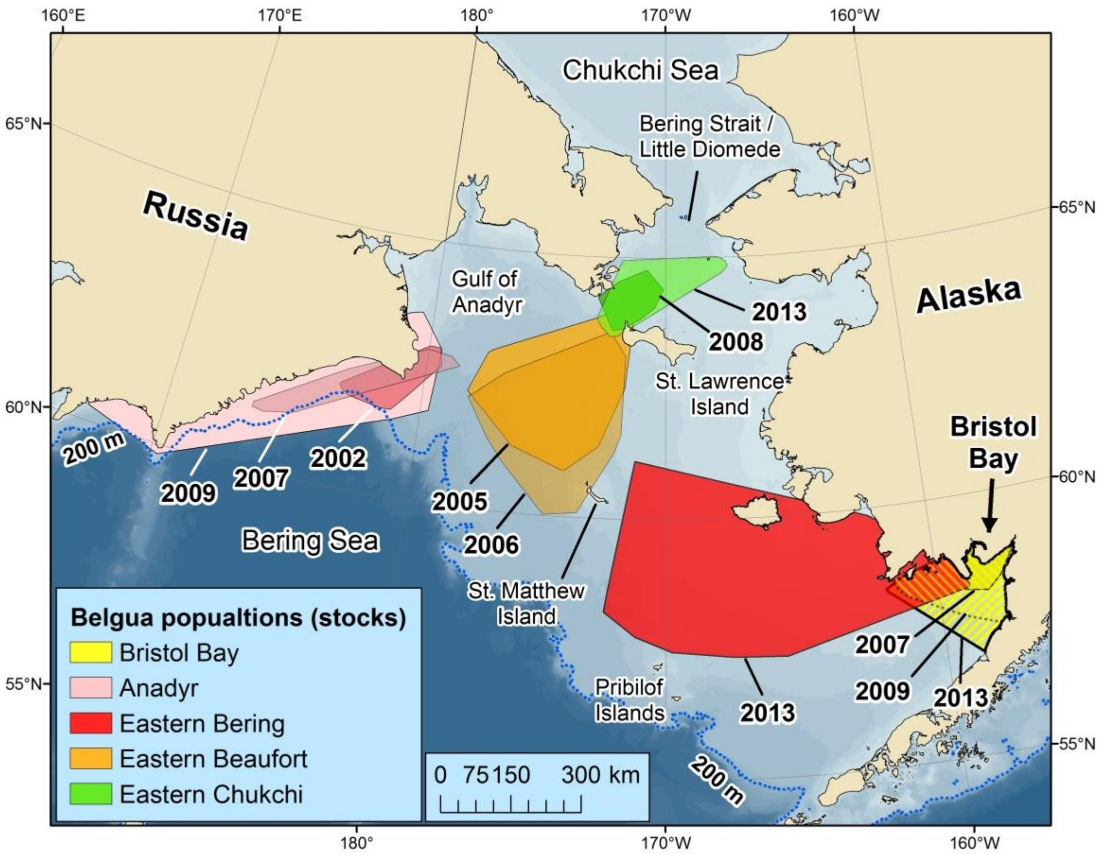

Figure 3. Winter ranges (minimum convex polygons of beluga satellite tag locations) of beluga stocks that winter in the Bering Sea. Polygons are drawn using January-March locations and years are denoted by the degree of shading. Figure reproduced from Citta *et al.* (2017).

The Alaska Beluga Whale Committee conducted aerial surveys again in 2016 (ABWC unpublished data). The survey methods followed the methods used for the 2004-2005 surveys. However, estimation methods were modified to calculate the updated abundance from the 2016 surveys, instead of using the maximum count the average count from all of the surveys was used with the same correction factors as the 2004-2005 estimate. This method allowed an estimate of the abundance estimate using the CV of the average count and a CV for the correction developed below. The average count from eight complete surveys of Bristol Bay in 2016 was 660 (CV=0.09, standard error = 61). Using the correction that has been applied in the past, 2.62 X 1.18, yields an estimated abundance of 2040 for 2016.

Estimating a CV for this abundance estimate is somewhat problematic because there is no CV given for the correction factor. We assume a CV of 0.2. This is supported by results from Cook Inlet belugas presented in Lerszak et al. 2000 where the standard deviation of individual average dive interval was 6.4 seconds and the average dive interval was 24.1 seconds. Assuming that this is dive behavior typical for belugas in an estuarine environment then we can apply these values to the two animals that were used to estimate the submerged animal correction of 2.62 for Bristol Bay yielding an estimated CV of 0.19 (=6.4/24.1/sqrt(2)). The value of 1.18 is a correction for perception of small gray dependent calves either young of the year or yearling calves. The actual number of calves will depend on the success during the survey year and the previous year. If we consider the variation in calves from back to back good reproductive years of 15% calves each to back to back poor years of 5% calves each, a CV of 6% for the 1.18 value covers this range. For the combined correction of 3.09 we have a CV of 0.20 = sqrt(0.19^2 + 0.06^2). With the CV for the average of counts of 0.09 yields a CV for the abundance estimate of 0.22 and a 95% CI of (1,541-2,702).

Counts of belugas often vary widely, even when surveys are conducted on the same day and cover the exact same area. In 2016, replicate counts ranged from 484 to 1,024 on days with good viewing conditions. In fact, these two counts were collected on the same day, within a few hours of each other, thus it is important to conduct replicate counts. This also suggests that average beluga behavior in a population can vary substantially with changes in conditions, tides or other phenomenon in a short time interval. In 2002, the ABWC began a genetic mark-recapture project in Bristol Bay as an alternative method for population estimation and to provide a correction factor that was not constructed based on limited sample of dive behavior and assumptions about perception bias. Abundance estimated by mark-recapture methods are not reliant on estimating correction factors and provide an independent estimate of abundance. During 2002–2011, the Alaska Department of Fish and Game (ADFG) worked with Alaska Native beluga hunters and collected skin samples with biopsy tips mounted on jab-sticks. Unique genotypes were determined by PCR amplification of mtDNA and eight microsatellite loci. Matching of genetic samples was accomplished using program CERVUS (Kalinowski *et al.* 2007). During the study, we identified 516 individual belugas in two inner bays, 468 from Kvichak Bay and 48 from Nushagak Bay, and recaptured 75 belugas in separate years. Using a POPAN Jolly-Seber model, abundance was estimated at 1,928 belugas (95% CI = 1,611 to 2,337), not including calves, which were not sampled. Most belugas were sampled in Kvichak Bay at a time when belugas are also known to occur in Nushagak Bay. The pattern of genetic recaptures and data from belugas with satellite transmitters suggested that belugas in the two bays regularly mix. Hence, the estimate of abundance likely applies to all belugas within Bristol Bay. Simulations suggested that POPAN estimates of abundance are robust to most forms of emigration, but that emigration causes negative bias in both capture and survival probabilities. Because it is likely that some belugas do not enter the sampling area during sampling, our estimate of abundance is best considered a minimum population size.

In summary, the genetic mark-recapture study supports the estimate of 2040 belugas from the 2016 aerial surveys.

3. Anthropogenic removals

Subsistence harvest

The ABWC and the Bristol Bay Native Association have collected data on Alaska Native subsistence harvests within Bristol Bay since 1987. Harvest data during 1987–2006 are presented by Frost and Suydam (2010). Here, we show the harvest record through 2016 (Fig. 4a; Frost *et al.* in prep.).

Over the last ten years, the annual harvest has averaged 23 belugas (95% CL = 21–25). Although there is a slight positive trend in the harvest (an increase of 0.15 belugas per year), this trend is neither statistically significant (p=0.64) nor biologically important (Fig. 4b). The current potential biological removal (PBR) is almost twice this value (see Section 5, below).

Reporting of struck and lost belugas is sporadic. A struck and lost beluga is reported once every few years (ABWC, unpublished data) and the true rate is likely higher. Frost and Suydam (2010) did not report struck and lost rates as they were inconsistently reported for most communities in Alaska, including those in Bristol Bay.

Bycatch

Fishery observers monitored the groundfish trawl, longline, and pot fisheries within greater Bristol Bay during 1990–1997 and no incidental mortalities or injuries were observed (Muto *et al*. 2016). Aerial surveys occur in late June and early July, during the sockeye fishery, and belugas are observed swimming around gillnet sets suggesting belugas could be caught in the commercial salmon set gillnet and drift gillnet fisheries that occur in the inner bays (i.e., Nushagak and Kvichak bays). During May-July 1983, Frost *et al*. (1984) conducted beach surveys in the inner bays from airplanes and boats and found 27 dead belugas, at least 12 of which were clearly attributed to fisheries. The commercial gillnet fisheries have never been monitored for bycatch and there are no current, reliable data on incidental take. There is also a large subsistence gillnet fishery for salmon in Bristol Bay in which four belugas were reported taken during 2005–2012 (Allen and Angliss 2011; Muto *et al*. 2016). Some belugas caught in subsistence gillnet fisheries are reported as harvest because they are consumed by Alaska

Natives, however, the proportion of bycatch that is reported as harvest is unknown. Bycatch would have to be at least 20 belugas per year, after accounting for an average annual harvest of 23 belugas, to exceed PBR (See Section 5). Documenting the current level of bycatch is warranted.

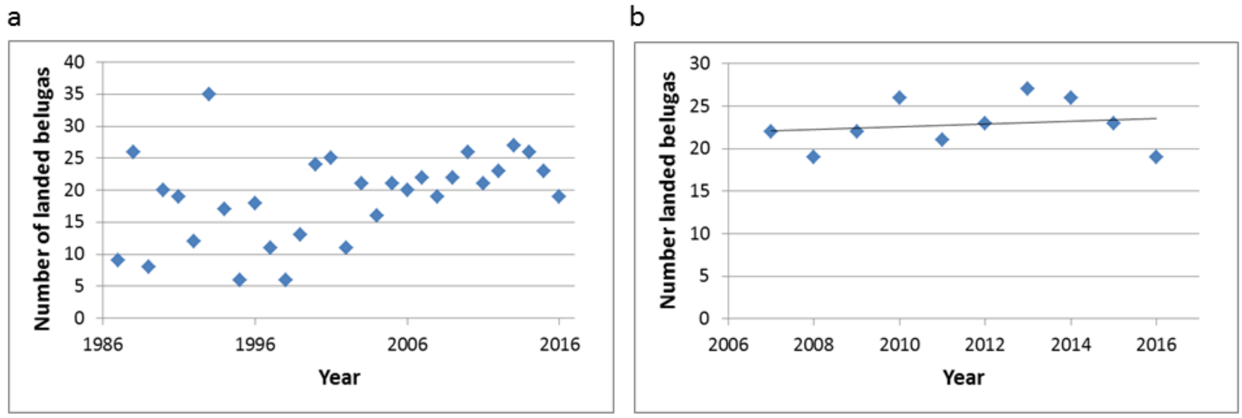

Figure 4. (a) Number of belugas landed by subsistence hunters in Bristol Bay, Alaska, 1987–2016, and (b) the trend in the number of belugas landed during the last ten years (2007–2016). For more information on how harvest is documented, see Frost and Suydam (2010).

4. Population trajectory

As described above (See Section 2), aerial surveys have been conducted in Bristol Bay periodically between 1993 and 2016 and results from 1993 to 2005 are reported by Lowry *et al.* (2008). Using the trend in the number of belugas counted over time, they estimated the Bristol Bay stock increased 4.8% per year over the 12-year period. Although this value is higher than the maximum net productivity rate (4%) that has been used as a default for cetaceans (Wade 1998); the value estimated by Lowry *et al.* (2008) had a confidence interval (95% CI = 2.1–7.5%) that includes 4%. Lowry *et al.* (2008) speculated that the high net productivity rate indicated the population may have been recovering from research harvests in the 1950s and 1960s (e.g., Brooks 1955), a decline in subsistence harvest, or a delayed response to increases in salmon abundance in the 1980s.

The Alaska Beluga Whale Committee conducted aerial surveys again in 2016 (ABWC unpublished data). Compared to the last survey in 2005, the average count in 2016 increased by 3.7%. Given the variability in the proportion of belugas that are available to be counted during any given survey, these changes are minor and it appears that the population growth observed in during 1993–2005 has slowed or ceased. Although more surveys will be necessary to conclusively determine the current trend, the average count data show that there is approximately the same number of belugas in Bristol Bay in 2016 ($\bar{x}$ =660, CV=0.09) as there were in 2004 ($\bar{x}$ =637, CV=0.21) and 2005 ($\bar{x}$ =640, CV=0.13) (Fig. 5).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

Under the 1994 reauthorized Marine Mammal Protection Act (MMPA), the PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$ (Wade and Angliss 1997). N_{MIN} is the lower 20th percentile of a log-normal distribution that represents the minimum number of whales after accounting for uncertainty in the estimates. Most counts of belugas do not include reliable estimates of variability. Because of this, Muto *et al.* (2016) used a default coefficient of variation (CV) of 0.2, resulting in a minimum population size of 2,467 belugas. Here we use the abundance estimate from the 2016 survey of 2,040 (CV 0.09) which results in an Nmin of 1,809. R_{MAX} is the maximum net productivity rate (4.8%; Lowry *et al.* 2008) and F_R is the "recovery factor" and this is equal to 1.0 when a population is stable or increasing. Muto *et al.* (2016) used the average of the maximum counts from aerial surveys in 2004 and 2005 to calculate a PBR of 59 belugas (2,467 * 0.024 * 1.0) in Bristol Bay. Applying the same methods to the estimate from the 2016 aerial survey yields a PBR of 43 belugas (1,809 * 0.024 * 1.0).

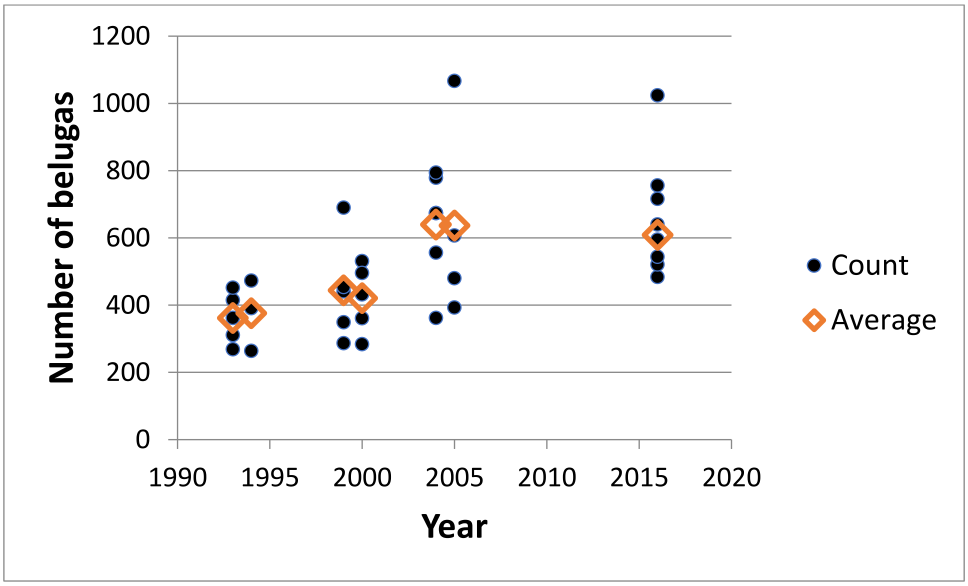

Figure 5. Number of beluga whales observed during aerial surveys in Bristol Bay, 1993–2016. Black dots are the number of belugas counted during replicate flights and red diamonds are the annual averages. For more information on aerial survey methods, see Lowry *et al.* (2008).

6. Habitat and other concerns

Sea ice and climate warming

Sea ice is declining in most of the Arctic, however, Bering Sea ice is largely disconnected from trends in most other Arctic regions (e.g., Douglas 2010; Laidre *et al.* 2015). Bristol Bay is at the southern boundary of seasonal sea ice extent and multiyear ice has never been present (Neibauer and Schell 1993). Rather, sea ice is highly fragmented within Bristol Bay and winds from the north may create open water within the inner bays at any time in winter. Citta *et al.* (2016) documented how belugas with SDRs will move into the inner bays when north winds create open water. Although belugas never traveled south of the ice edge, they were also never located in the inner bays when there was no open water, perhaps due to risk of entrapment. Sea ice in Bristol Bay will likely form later and melt earlier as the climate warms and this may allow belugas more access to the inner bays in winter. Unfortunately, virtually nothing is known about the winter diet of belugas or what habitats they prefer in winter. If climate warming has an effect on Bristol Bay belugas, it will likely be through the expansion of new prey species into their range (Watt *et al.* 2016), the introduction of new pathogens or parasites that could affect belugas or their prey, or the loss of feeding habitat if sea ice provides a refuge from killer whales.

Fisheries bycatch

As mentioned above (see Section 3), no incidental mortalities or injuries to beluga whales were reported by fishery observers that monitored the groundfish trawl, longline, and pot fisheries during 1990–1997 (Muto *et al.* 2016). Other observations show that belugas have been caught in the commercial and subsistence salmon fisheries that occur in the inner bays but overall there are no reliable data on incidental take. Although beluga mortalities due to fisheries occur, they did not prevent the population from growing between 1993 and 2005 (Lowry *et al.* 2008). We suspect that unless there is a change in how or where commercial gillnet fisheries occur, these fisheries will not be a threat to the long-term sustainability of belugas in Bristol Bay. However, assessing current levels of bycatch is warranted.

Oil and gas development

In 2014, then U.S. President Obama used his authority under the Outer Continental Shelf Lands Act to permanently withdraw Bristol Bay from petroleum development. The withdrawal area contains all of Bristol Bay outside of State of Alaska territorial seas and contains most of the winter range of Bristol Bay belugas. The remaining range of Bristol Bay belugas is contained within state waters in Nushagak and Kvichak bays. Although oil and gas leases are periodically offered for sale by the State of Alaska, there are currently no oil or gas wells and no active leases in state waters within Bristol Bay (Alaska Department of Natural Resources 2014; http://dog.dnr.alaska.gov/Publications/OGInventory.htm).

Mining
A large copper, gold, and molybdenum mine is proposed for an area that includes the headwaters of both Nushagak and Kvichak rivers. This mine, named the Pebble Mine, would process ore using a cyanide solution and mine effluents would be toxic to fish if leaked into the river systems. All mine shares are currently owned by the Northern Dynasty Partnership and, at the moment, plans to develop the mine are on hold. There is political opposition to developing the mine and most of Northern Dynasty's funding partners backed out of the project between 2011 and 2014. In 2014, the U.S. Environmental Protection Agency also issued rules unfavorable for the development of this mine. At this time, it is unclear when or if the mine will be developed.

7. Status of the stock

The Bristol Bay stock of beluga whales is one of three stocks in western Alaska that is co-managed by NMFS and the ABWC (Adams *et al*. 1993; Fernandez-Gimenez *et al*. 2006). Two of the agreed upon objectives of the management plan are to "conserve the Western Alaska beluga whale population" and to "protect Alaska Native beluga whale subsistence hunting traditions and culture" (ABWC 1999). Bristol Bay beluga whales are not designated as "depleted" or "strategic" under the MMPA nor are they listed as "threatened" or "endangered" under the Endangered Species Act. In an assessment done in 2008, the IUCN listed belugas as a species as "Near Threatened" and also noted that the various subpopulations should be assessed separately (Jefferson *et al*. 2012).

The Bristol Bay population is relatively small (~2,000); however, the abundance and trend of this stock are periodically monitored and the stock appears to be stable. The potential biological removal (PBR) for this population is at least 43 belugas/year. Annual subsistence harvest over the last decade has been less than half this number ($\bar{x}$=23/yr). Although there is little information regarding incidental take or struck and lost rates, the fact that the population has increased in recent decades suggests that these sources of mortality are insignificant (Lowry *et al*. 2008). There are currently few threats to population persistence, although changes in resource development or the invasion of novel species or pathogens due to climate warming could pose challenges in the future.

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Annex 8: Eastern Chukchi Sea Beluga Whale Stock
Status Review for the NAMMCO Global Review of Monodontids
Submitted 18 February 2017

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1. Distribution and stock identity

The eastern Chukchi Sea (ECS) beluga stock occurs in the lagoons and adjacent waters of the ECS in late spring and early summer (Frost *et al*. 1993). Individuals of this stock range widely throughout the ECS and Beaufort Sea and into the Arctic Ocean during summer and early fall (Suydam 2009, Hauser *et al*. 2014) and then move through the Bering Strait into the Bering Sea in the winter, returning to the Chukchi Sea the following spring (Citta *et al*. 2017).

The non-uniform distribution of beluga whales in coastal waters of the Bering, Chukchi, and Beaufort Seas in summer is indicative of likely population subdivision and formed the basis for original, but provisional, stock designations (Frost and Lowry 1990). It was recognized at the time that identification of more biologically meaningful stocks would require genetic studies to elucidate the underlying patterns of demographic and reproductive relationships among seasonal groupings (O'Corry-Crowe and Lowry 1997). Over the past two decades several genetic studies have been conducted on seasonal groupings that occur adjacent to Alaska and Chukotka (Russian Federation) primarily summering and migrating groups, to resolve patterns of dispersal and gene flow. The studies revealed substantial mitochondrial DNA (mtDNA) differentiation among summering groups in Bristol Bay, Norton Sound, and Anadyr Gulf in the Bering Sea, in nearshore waters along Kasegaluk Lagoon in the Chukchi Sea, and in the Mackenzie Delta-Amundsen Gulf region in Beaufort Sea that likely reflects long-established patterns of female-mediated philopatry and demographic isolation (O'Corry-Crowe *et al*. 1997, 2002; Brown-Gladden *et al*. 1997, Meschersky *et al*. 2008; Fig. 1). This has led to their identification as the following five demographically distinct management stocks: 1) Bristol Bay, 2) eastern Bering Sea, 3) Gulf of Anadyr, 4) ECS, and 5) eastern Beaufort Sea (Muto *et al*. 2016, Laidre *et al*. 2015). A few studies have documented lower levels of nuclear DNA (microsatellite) heterogeneity among geographic strata compared to mtDNA. This has been taken as evidence of male-mediated gene flow among summering groups, possibly in shared wintering areas (Brown-Gladden *et al*. 1999, Meschersky *et al*. 2013), or it could reflect a slower rate of drift in markers with higher effective population size (O'Corry-Crowe *et al*. 2010). Recent studies question the common wintering area hypothesis (Citta *et al*. 2017) and whether gene flow is extensive among stocks in the Bering, Chukchi, and Beaufort seas (O'Corry-Crowe *et al*. in prep.).

Beluga whales in the ECS have traditionally occupied two geographically distinct coastal concentration areas, Kotzebue Sound and the nearshore waters along Kasegaluk Lagoon (Fig. 1). Studies conducted in the 1970s and early 1980s reported beluga whales entering Kotzebue Sound in mid- to late-June each year with or following ice breakup, while whales began to congregate in nearshore waters and passes near Kasegaluk Lagoon typically in late June (Seaman *et al*. 1988, Frost and Lowry 1990). The whales tended to remain in these nearshore locations for periods of weeks to a month or so before moving on, presumably to areas further north and/or offshore. Traditional knowledge of the local Inuit confirmed that these were long established migration routes and summer concentration areas (Huntington *et al.* 1999). The pattern of beluga whales returning to these two traditional locations, however, has diverged dramatically since the mid-1980s. Numbers of whales returning to Kotzebue Sound declined dramatically after 1983 and have not recovered, despite a few years when large numbers of whales briefly entered the Sound in summer (Frost and Lowry 1990, Seaman *et al*. 2015). By contrast, the return of belugas to the Kasegaluk Lagoon area has been very consistent throughout much of the past three decades (Suydam 2009).

Other than the annual return to the Kasegaluk Lagoon area, essentially nothing was known about distribution of this stock until belugas were tagged with satellite depth recorders (SDRs). During 1998-2012, 29 belugas were captured in conjunction with the annual subsistence hunt at Point Lay and equipped with SDRs that provided location data for 5-522 days (Suydam 2009, Hauser *et al.* 2014).

Results showed that after leaving Point Lay in July, whales moved northward into the northern Chukchi and Beaufort seas and into the Arctic Ocean with some animals penetrating heavy ice cover to north of 80° N latitude (Suydam *et al*. 2001). During summer, they ranged widely, but belugas of all ages and both sexes were most often found in water deeper than 200 m, along and beyond the continental shelf break and into very deep waters. They rarely used inshore waters of the Beaufort Sea (Suydam 2009). Hauser *et al*. (2014) used these same data to describe beluga distributions and home ranges for July through November, by which time the whales had moved southward through the Chukchi Sea to the Bering Strait region. The six whales whose tags transmitted long enough passed through Bering Strait in November-December then remained in the northern Bering Sea, between Bering Strait and St. Lawrence Island, into May. One tag lasted long enough to re-enter the Chukchi Sea in late May and another stopped transmitting in early May, just south of Bering Strait (Citta *et al*. 2017).

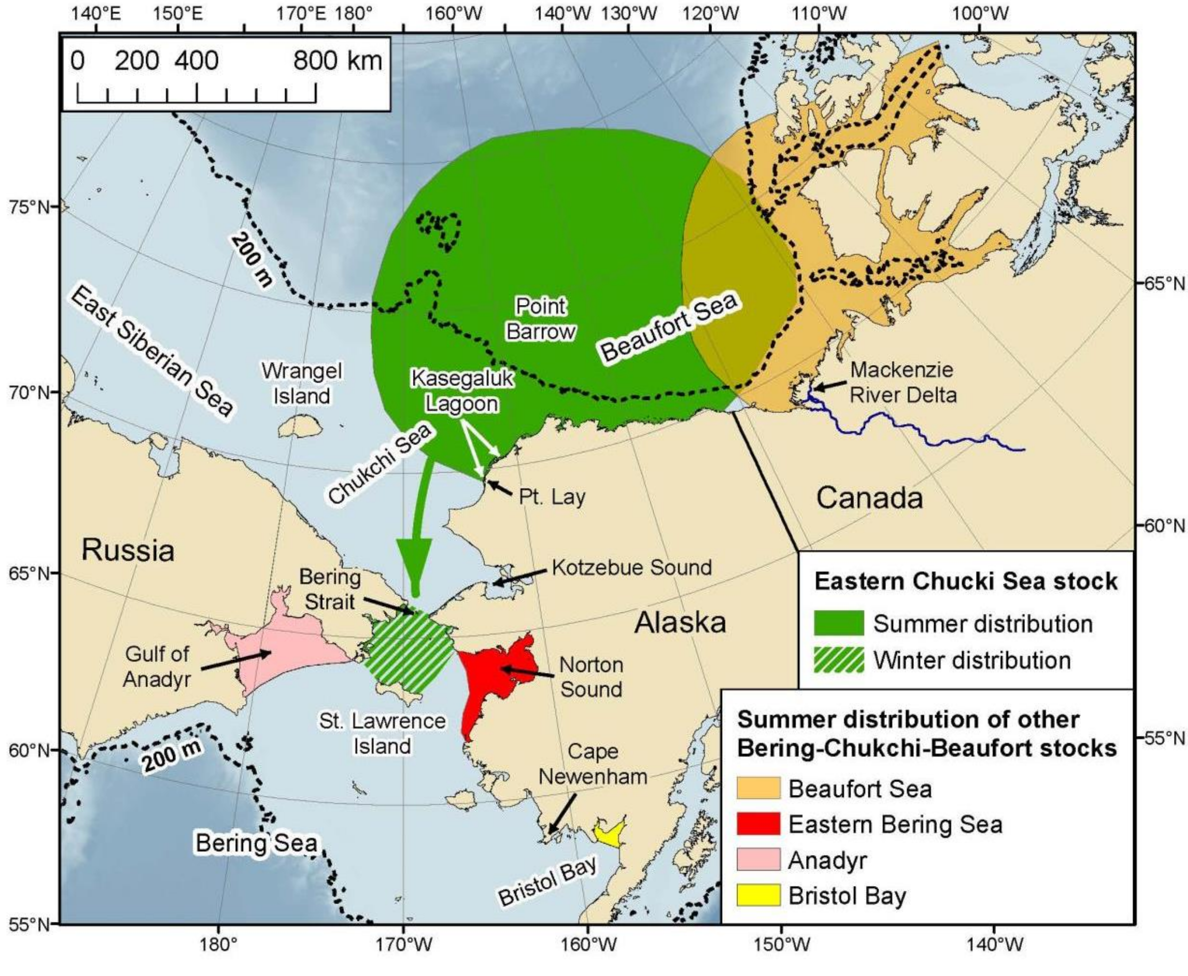

Figure 1. Map of the Bering-Chukchi-Beaufort sea region showing summer distribution of all beluga stocks in the region and the winter distribution of the eastern Chukchi Sea stock.

Studies on patterns of mtDNA variation revealed that the summering concentration along Kasegaluk Lagoon was demographically distinct from other summering groups in the Beaufort and Bering seas and these whales were subsequently identified as the ECS stock (O'Corry-Crowe and Lowry 1997, O'Corry-Crowe *et al*. 1997, 2002, Brown-Gladden *et al*. 1997, Muto *et al*. 2016). Based on the pattern of annual return, it was initially hypothesized that the original Kotzebue and Kasegaluk summering groups were part of the same demographically distinct subpopulation and thus the same stock. A series of genetic studies, however, have revealed that beluga whales from the pre-decline era in Kotzebue Sound were genetically distinct from the ECS stock (i.e., those that use Kasegaluk Lagoon). Additionally, whales from two subsequent anomalous years (1996 and 2007), when large numbers of animals entered the Sound, were also genetically distinct from the pre-1983 Kotzebue Sound beluga and from the ECS stock (O'Corry-Crowe *et al*. 2001, 2016). Those atypical years most likely involved whales from the Beaufort Sea stock and the anomalous events coincided with anomalous ice years in the Bering-Chukchi-Beaufort region (O'Corry-Crowe *et al*. 2016).

2. Abundance

Sightings of beluga whales in the ECS in summer occur mostly in June-July in Kotzebue Sound and off Kasegaluk Lagoon (Seaman *et al*. 1988, Frost and Lowry 1990, Lowry *et al*. 1999), and initial abundance surveys were focused in those areas. At that time it was thought that belugas in those two areas belonged to the same stock, but genetic evidence now shows that they are different (see above). Distribution, abundance, and movements of the potential "Kotzebue stock" are essentially unknown and it will not be further considered in this assessment.

The first efforts to assess abundance of the ECS beluga stock were made in the late 1970s by Seaman *et al*. (1988). They took photographs of belugas concentrated at Kasegaluk Lagoon passes, and estimated that there were 2,282 animals there on 15 July 1979. The estimate included correction factors for whales outside the concentration area (+10%), whales too deep to be seen on the photographs (+20%), and dark colored yearlings that are difficult to see (+8%). Frost and Lowry (1990) flew an aerial strip transect survey over a large concentration of belugas off Point Lay on 8 July 1987. They counted 723 whales, and suggested that there may have been 1,400-2,100 animals in that group (using correction factors of 2 and 3 to account for animals missed because they were diving in relatively deep water).

Frost *et al*. (1993) conducted aerial surveys of ECS coastal waters during 1989-1991. Survey effort was concentrated along the shore near Kasegaluk Lagoon, an area regularly used by belugas during the open-water season. They made numerous sightings of beluga whales in that region with the highest single day count of 1,200 whales. Offshore waters where belugas also occur were not surveyed. If this minimum count is corrected using radio tag data for the proportion of animals that were diving and thus not visible at the surface (2.62; Frost and Lowry 1995), and for the proportion of newborns and yearlings not seen due to small size and dark coloration (1.18; Brodie 1971), the total abundance of the eastern Chukchi stock was estimated as 3,710 (1,200 × 2.62 × 1.18) whales. This is the figure that has been used in National Marine Fisheries Service (NMFS) Stock Assessment Reports (Muto *et al*. 2016) and elsewhere (e.g., Laidre *et al*. 2015).

The Alaska Beluga Whale Committee (ABWC) conducted additional surveys in the Kasegaluk Lagoon region in 1996-98 and found belugas in the nearshore areas previously surveyed but also detected groups of whales further offshore (Lowry *et al*., 1999). Subsequent survey efforts in 2001-03 included more offshore flight lines, but while belugas were occasionally sighted more than 50 km offshore, sightings were very infrequent (Lowry and Frost 2002, 2003). Also, data from whales equipped with satellite depth recorders (SDRs) at Kasegaluk Lagoon showed that many whales were outside of the area surveyed during the survey period (Suydam *et al*. 2001). Because of the high cost of aerial surveys and the relatively low value of results for population assessment, beluga-specific surveys in the ECS were suspended by the ABWC after 2003.

An analysis of data from SDRs attached to belugas in coastal concentration areas of the ECS and Beaufort Sea stocks (Hauser *et al*. 2014) provided an overview of distribution and movements of the stocks and allowed the identification of an area (140°W to 157°W in the Beaufort Sea) and time period (19 July-20 August) when the distributions of the two stocks do not overlap (Lowry *et al*. in prep.). Aerial survey data collected in 2012 in that region during those dates by the Aerial Surveys of Arctic Marine Mammals (ASAMM) project (Clarke *et al*. 2013) were used in a line transect analysis that estimated there were 5,547 (coefficient of variation (CV) = 0.22) surface-visible belugas in the study area. Data from SDRs were used to develop correction factors to account for animals that were missed because they were outside of the study area or diving too deep to be seen, resulting in a total abundance estimate of 20,675 (CV = 0.66; Lowry *et al*. in prep.). Additional survey data were collected in that region in 2013-2016 and a full analysis of ECS beluga abundance using all available ASAMM data is anticipated.

3. Anthropogenic removals

Subsistence harvest

The ABWC and the North Slope Borough Department of Wildlife Management have collected data since 1987 on Alaska Native subsistence harvests by villages harvesting from the ECS. Harvest data through 2006 were reported by Frost and Suydam (2010). However, in that publication data for Kotzebue Sound were included in the ECS harvest. Here, we report revised 1987-2006 ECS harvest data, as well as data for 2007-2016 (Fig. 2; ABWC, unpublished data). Harvest data for Kotzebue Sound are not reported here since the stock from which belugas have been harvested is not known for all years.

Harvest of the ECS stock occurs mainly at two communities, Point Lay and Wainwright. The revised average annual harvest for 1987-2006 was 48 belugas (range 0-86; 95% CL = 37-59). During 2007-2016, the average annual harvest increased to 57 belugas (range 14-121; 95% CL = 35-79; Fig. 2a). The increase in average harvest is almost certainly due to improved reporting for the village of Wainwright. Annual variation in the harvest is high and can differ more than tenfold. During 2007-2016, there was a slight negative trend in harvest (Fig. 2b) that was statistically insignificant (p = 0.15). The current potential biological removal (PBR) is more than four times the average harvest during the last 10 years (see Section 5, below).

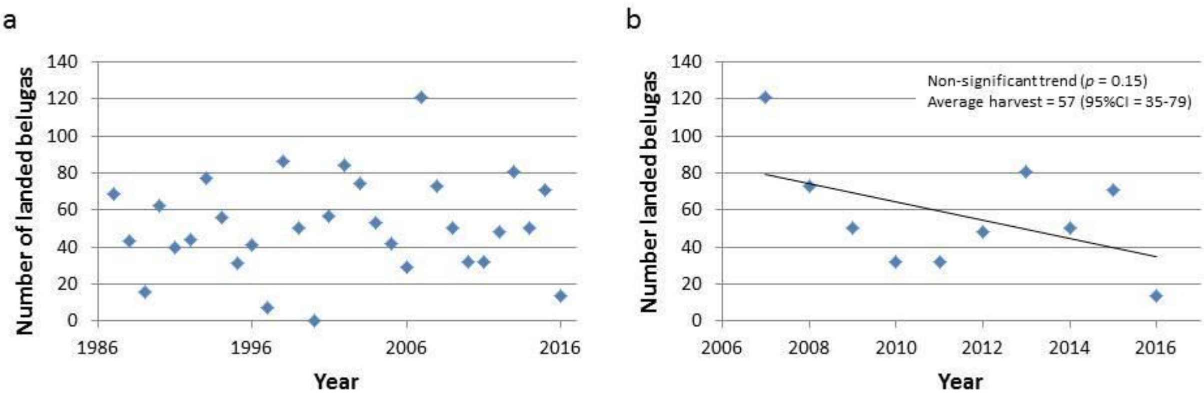

Figure 2. The number of ECS belugas landed by Alaska Native subsistence hunters during 1987–2016 (a), and trend in the number of belugas landed during 2007–2016 (b). For more information on how harvest is documented, see Frost and Suydam (2010).

Reporting of struck and lost belugas has been sporadic but because the hunts at Point Lay and Wainwright are drive hunts, the number of whales struck and lost is low. There were some struck and lost whales reported for the ECS stock in 3 of the last 10 years, although more animals may have been lost. During those years, the number of belugas struck and lost averaged 7% of the landed harvest (ABWC, unpublished data). Frost and Suydam (2010) also reported a struck and lost rate of 7% for the ECS stock.

Bycatch

In the USA, some commercial fisheries that operate in federal waters (3-200 nm offshore) and may take marine mammals as bycatch are regularly monitored. In Alaska, three commercial fisheries that could have interacted with beluga whales from the ECS beluga stock have been monitored: Bering Sea and Aleutian Islands groundfish trawl, longline, and pot fisheries. No mortality or serious injury to beluga whales was reported in those fisheries. State-managed commercial, personal use, and subsistence gillnet fisheries occur in nearshore waters of the eastern Chukchi Sea. While they are a potential source of bycatch mortality and bycatch is not systematically monitored, no beluga whale takes have been reported in those fisheries (Muto *et al*. 2016). Low numbers of belugas have been entangled and killed in subsistence fishing nets at Barrow, Alaska. Those animals were reported and are included as subsistence harvests for the Beaufort Sea stock (ABWC, unpublished data) but may have been from the ECS stock.

4. Population trajectory

There are no data on maximum growth rate (RMAX) for ECS belugas. For the Bristol Bay beluga stock the estimated rate of increase over the 12-year period 1992-2005 was 4.8%/year (95% CI = 2.1%-7.5%; Lowry *et al*. 2008). The measured value for Bristol Bay is close to the 4%/year that is used by NMFS as the default RMAX for cetaceans (Wade1988).

Peak counts made at Kasegaluk Lagoon during 1978-2003 have varied considerably but do not give any clear indication of changes in abundance over that period (Table 1). The trend in abundance of ECS belugas is considered unknown (Laidre *et al*. 2015, Muto *et al*. 2016).

Table 1. Results of counts of ECS beluga whales in the Kasegaluk Lagoon region, 1978-2003.

Year	Maximum count	Date	Number of surveys	Comments
1978[1]	703	10-Jul	5	nearshore, count from photos
1979[1]	1,601	15-Jul	5	nearshore, count from photos
1981[1]	670	8-Jul	5	nearshore, visual count
1987[2]	724	8-Jul	1	offshore, visual count
1990[3]	1,212	5-Jul	12	nearshore, visual count
1991[3]	938	6-Jul	12	nearshore, visual count
1996[4]	1,035	30-Jun	10	nearshore and offshore, visual count
1997[4]	130	7-Jul	4	mostly poor survey conditions
1998[4]	1,172	6-Jul	5	nearshore and offshore, visual count
2001[4]	667	6-Jul	5	nearshore and offshore, visual count
2002[4]	582	6-Jul	7	nearshore and offshore, visual count
2003[4]	369	5-Jul	6	early spring, counts not comparable to previous years

1 Seaman *et al*. 1988
2 Frost and Lowry 1990
3 Frost *et al*. 1993
4 Alaska Beluga Whale Committee, unpublished data

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The U.S. Marine Mammal Protection Act (MMPA) defines the PBR as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = NMIN × 0.5RMAX × FR. However, because the most recent abundance estimate available at the time of the last NMFS Stock Assessment Report was more than eight years old the PBR for the stock was considered to be “undetermined” (Muto *et al*. 2016).

A PBR can be calculated using the abundance estimate provided in Lowry *et al*. (in prep) as follows: NBEST = 20,675; CV = 0.66; NMIN = 12,461, RMAX = 0.04; FR = 1.0; PBR = 249. The average annual Alaska Native subsistence harvest from the ECS stock for the last 10 years (57 belugas) is about 0.3% of the population estimate (Lowry *et al.* in prep.). Although coastal fisheries are not regularly monitored for incidental take, all indications are that anthropogenic removals from the ECS beluga stock are sustainable.

6. Habitat and other concerns

Because they are an ice-associated species there is concern about the possible effects on belugas of climate warming and associated loss of sea ice habitat. Laidre *et al*. (2015) showed that the duration of the reduced ice (summer) period increased by 44 days in the Chukchi Sea and 52 days in the Beaufort Sea from 1979 to 2013. In a long-term study of belugas off West Greenland, Heide-Jørgensen *et al*. (2010) found that belugas responded to changing sea ice by shifting their distribution but that abundance increased during a period of generally declining ice cover. They stated that "Global warming and sea-ice declines may pose less of a problem for belugas than to other Arctic marine mammals." Laidre *et al*. (2008) concluded that on a rangewide basis the beluga would be the arctic cetacean least sensitive to climate change because of their wide distribution and flexible habits.

There have been two studies that specifically address the potential influence of changes in ice conditions on Pacific Arctic belugas. O'Corry-Crowe *et al*. (2016) analyzed long-term sighting and genetic data on belugas in the Bering, Chukchi, and Beaufort seas in conjunction with multi-decadal patterns of sea-ice to investigate the influence of sea-ice on spring migration and summer residency patterns. While substantial variations in sea-ice conditions were found across seasons, years, and sub-regions, the pattern of beluga migration and residency was quite consistent. Those results suggest that belugas can accommodate to varying sea-ice conditions to perpetuate philopatry to traditionally used areas. Hauser *et al*. (2016) compared the timing of the autumn migration of ECS and Beaufort Sea belugas during the periods 1993-2002 and 2004-2012. They found that in the later period ECS beluga migration from the Beaufort and Chukchi seas was delayed by 2 to >4 weeks, but that Beaufort Sea belugas did not shift migration timing between periods. Although some stocks may focus on certain prey, such as Beaufort Sea belugas specializing on arctic cod, *Boreogadus saida* (Loseto *et al.* 2009), belugas are capable of consuming a wide variety of prey and are best classified as generalist predators. For example, examination of stomach contents from harvested ECS belugas found 5 species of fish from 4 families and 15 species of invertebrates (Quakenbush *et al.* 2015). Belugas clearly show flexibility and adaptive capacity which makes it particularly difficult to predict how they may be affected by climate change.

An increase in the duration of the open water season and the decline in multi-year sea ice has generated concern that increases in oil and gas exploration and development and shipping may have negative consequences for belugas (e.g., Moore *et al.* 2000, Lowry *et al.* 2012, Reeves *et al*. 2014). Most oil and gas activity within the range of ECS belugas currently occurs over the continental shelf in the Beaufort Sea, although from 2006 to 2015 there was also considerable activity in Chukchi Sea. In the Beaufort Sea, the distribution of ECS belugas is predominantly limited to offshore areas, near the shelf break and within the Arctic Basin. At present, oil and gas activity in the Alaskan portion of the Beaufort Sea is far inshore of where belugas typically range (Suydam *et al*. 2005). Oil and gas activity in the Canadian portion of the Beaufort Sea is largely limited to shallow shelf waters northeast of the Mackenzie River Delta (Fig. 1) and is outside of the range of ECS belugas. In 2016, President Obama used the Outer Continental Shelf Lands Act of 1953 to remove most of the U.S. portion of the Chukchi Sea from future leasing. However, there are still active oil and gas leases in the Camden Bay area of the Alaskan Beaufort Sea and in the Russian portion of the Chukchi Sea. In the summer of 2016 hydrophones detected active seismic surveys near Wrangel Island (Catherine Berchock, pers. comm.). Russian lease areas are largely outside the range of ECS belugas, however, the effects of oil and gas development (e.g., noise or oil spills) could extend into their range.

Although shipping is increasing with declining sea ice (Eguíluz *et al.* 2016, Pizzolato *et al.* 2016), belugas are not known to be particularly susceptible to ship strikes, even in congested areas such as the Saint Lawrence River (Kingsley 2002). Furthermore, factors in addition to sea ice, such as where resources are being developed and commodity pricing, determine shipping trends (e.g., Brigham 2011, Bensassi *et al*. 2016, Pizzolato *et al*. 2016). As such, predicting how patterns in shipping will change is difficult, as is how belugas will respond to those changes. Impacts to belugas in the far north from sounds associated with shipping, including ice breaking, may be more of a concern than ship strikes. There is scant information about how belugas respond to sounds associated with shipping. Dedicated studies are needed that 1) overlay shipping routes with the temporal distribution of ECS belugas, and 2) investigate the response of belugas to shipping activity.

7. Status of the stock

The ECS stock of beluga whales is one of four stocks in western Alaska that is co-managed by NMFS and the ABWC (Adams *et al*. 1993, Fernandez-Gimenez *et al*. 2006). Two of the agreed upon objectives of the management plan are to “conserve the Western Alaska beluga whale population” and to “protect Alaska Native beluga whale subsistence hunting traditions and culture” (ABWC 1999).

ECS beluga whales are not designated as “depleted” or “strategic” under the MMPA nor are they listed as “threatened” or “endangered” under the U.S. Endangered Species Act. In an assessment done in 2008, the IUCN listed belugas as a species as “Near Threatened” and also noted that the various subpopulations should be assessed separately (Jefferson *et al*. 2012). The population estimate from 2012 of approximately 20,000 belugas (Lowry *et al*. in prep) and the relatively low subsistence harvest suggests that ECS belugas are not at immediate risk from anthropogenic activities or climate change. However, additional monitoring of population size and trend, subsistence harvest, and health of belugas is warranted.

Biological samples have been collected from ECS belugas since the 1980s (Suydam 2009). One objective of that study was to examine reproduction, including pregnancy rates. From 1987 to 2005, the pregnancy rate for adult females was 0.41, which indicates a calving interval of between 2 to 3 years. That pregnancy rate appears to be somewhat higher than other studies (e.g., Burns and Seaman 1988, Heide-Jørgensen and Teilmann 1994) suggesting that ECS belugas are reproductively healthy and producing many calves. Data collections have recently focused on assessing the health status of ECS belugas by monitoring body condition, exposure to contaminants, disease, and other measures.

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Annex 9: Eastern Beaufort Sea Beluga Stock

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1. Distribution and stock identity

Distribution:

Belugas (*Delphinapterus leucas*) of the Eastern Beaufort Sea (EBS) stock arrive in the southeast Beaufort Sea in late May and June (Fraker, 1979, Richard et al. 2001a, Citta et al. 2017). During July, the belugas aggregate mainly in the warm, shallow waters of the Mackenzie River estuary (Norton and Harwood, 1986) where whales are significantly clustered in space and time (Harwood et al. 2014) (Figure 1).

From late July through August, their distribution shifts offshore (Norton and Harwood, 1985; Harwood et al., 1996, Harwood and Kingsley 2013), but the extent of their range beyond the estuary is less well known. Satellite tracking studies have confirmed that belugas of this stock use the offshore Beaufort Sea extensively and also that they travel in August to even more distant summer ranges, including Amundsen Gulf and Viscount Melville Sound (Figure 2) (Richard et al. 1997, Richard et al. 2001a and 2001b, Paulic et al. 2012).

Their return fall migration to wintering areas in the Bering Sea, which begins in August and continues into September, occurs far offshore and sometimes under heavy pack ice conditions, seaward of the continental shelf (Richard et al. 2001a, Hauser et al. 2016). Other stocks of belugas also use the Bering Sea as a wintering area, and recent studies using satellite telemetry have revealed that each of the stocks generally winter in traditional and mostly exclusive parts of the Bering Sea (Citta et al. 2017). EBS belugas arrive in the Bering Sea late November to early/mid-December and leave sometime in April to begin their spring migration back to the Beaufort Sea summering range.

Stock Identity:

Mitochondrial DNA analyses of harvested samples have identified EBS belugas as a distinct summering stock from western Arctic stocks (Alaska and Russia), and from central and eastern Canadian Arctic stocks, most likely due to maternally directed annual philopatry to the Beaufort Sea area (O'Corry-Crowe et al. 1997, Brown Gladden et al. 1997, O'Corry-Crowe et al. 2002). However, recent mtDNA genetic analyses of EBS belugas have shown that the migration patterns of this stock and the dedicated use of specific habitats can be altered during years of unusual ice patterns (O'Corry-Crowe et al. 2016).

Additional analyses of nuclear DNA microsatellite loci indicate this stock is not a distinct biological population, but instead is part of the Bering Sea beluga population that also includes the Bristol Bay, eastern Bering (Norton Sound), and eastern Chukchi (Point Lay) stocks around Alaska (Brown Galdden et al. 1999). Breeding in this population is thought to occur in March and April while whales are still in the Bering Sea (Suydam 2009). The ranges of winter area use by the different beluga summer stocks, including EBS belugas, do have patterns of juxtaposition and overlap that would allow for genetic exchange during the mating season, especially with the eastern Chukchi Sea beluga stock (Citta et al. 2017).

More recent analyses of larger numbers of EBS beluga samples from nearshore and entrapment harvests (n=858), and an increased amount of genetic information (709bp of mtDNA sequence; 16 microsatellite loci) have not revealed finer scale spatial or temporal genetic structure within the nearshore portion of the stock (Figure 3) (Postma and Frasier, in prep.). However, Discriminant Analysis of Principal Components (DAPC) of microsatellite data does support the hypothesis that related groups of females are returning to the overall nearshore EBS area each summer (Figure 4) (Postma and Frasier, in prep.)

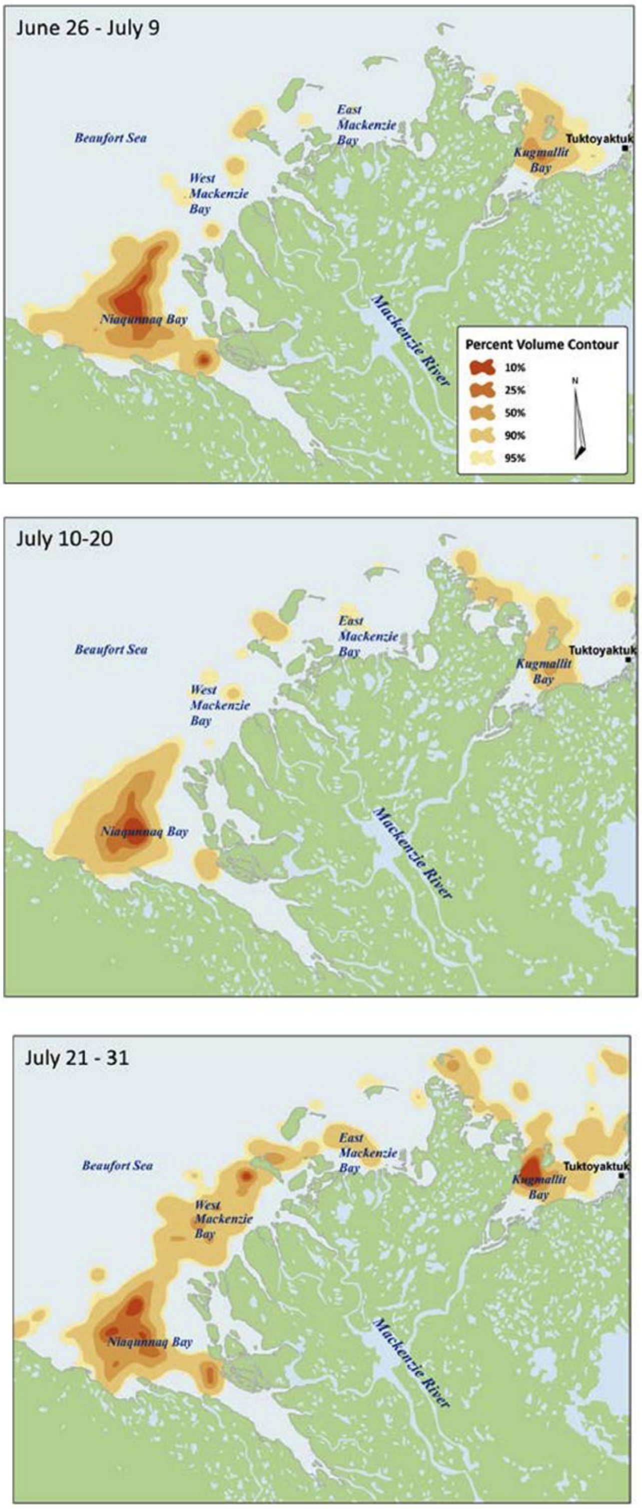

Figure 1. Percent volume contours of beluga sightings made during systematic aerial surveys during early, mid, and late July time periods, 1977-1985 and 1992. Figure from Harwood et al. (2014)

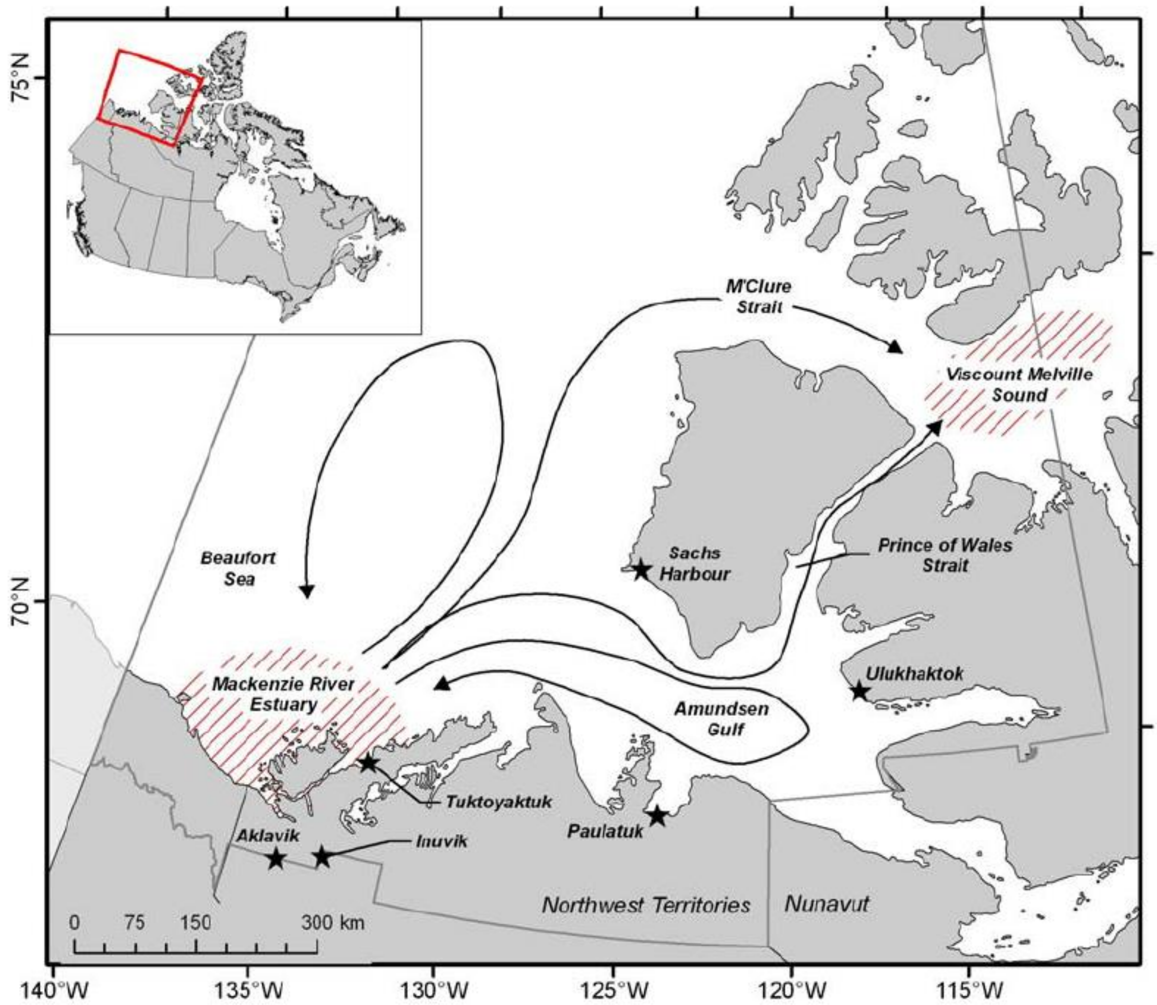

Figure 2. Map indicating the distribution of the Eastern Beaufort Sea beluga stock during spring, summer and fall movements. Summer aggregations of high densities occur in areas of red hatchings. Figure from Paulic et al. (2012).

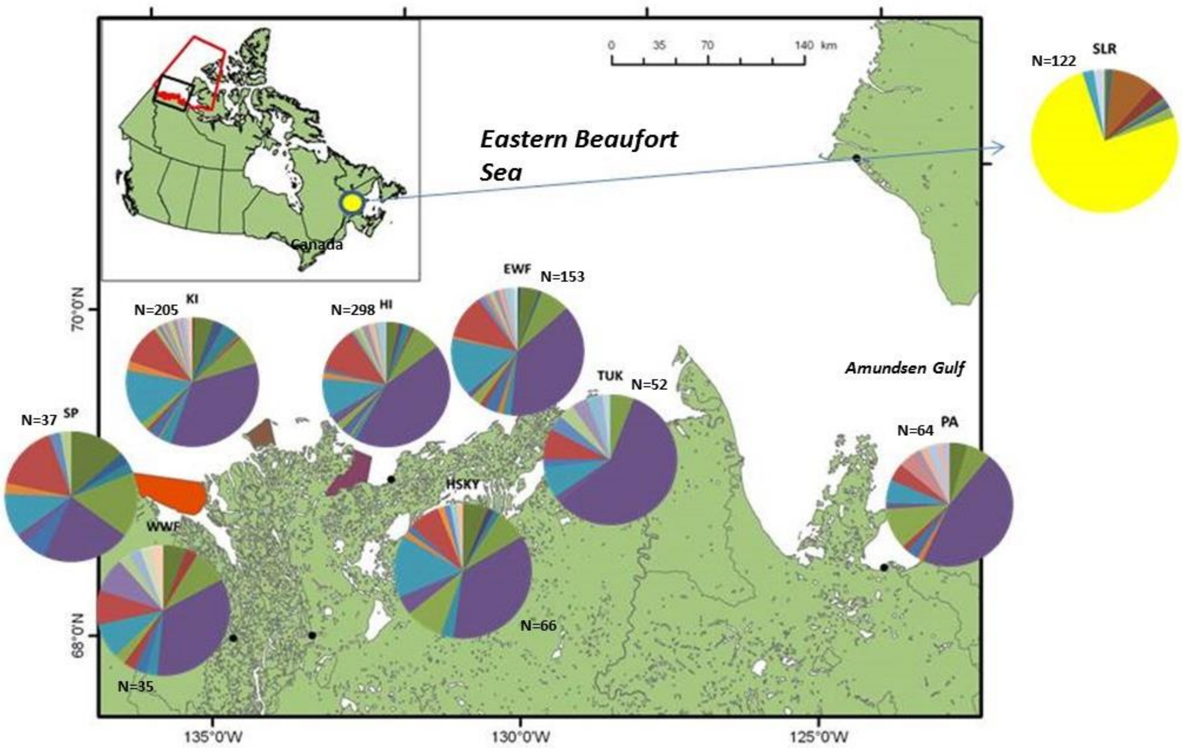

Figure 3. Distribution of mtDNA haplotypes among Beaufort Sea harvest sampling locations with comparison to the St. Lawrence Estuary population. Each colored slice of the pie represents a unique haplotype (n=55) and the size of the pie slice indicates the relative frequency of that haplotype in the total sample at each location. Abbreviations: SP, Shingle Point; WWF, West Whitefish Station; KI, Kendall Island; HI, Hendrickson Island; HSKY, Husky Lakes; EWF, East Whitefish Station; TUK, Tuktoyaktuk; PA, Paulatuk; SLR, St. Lawrence Estuary.

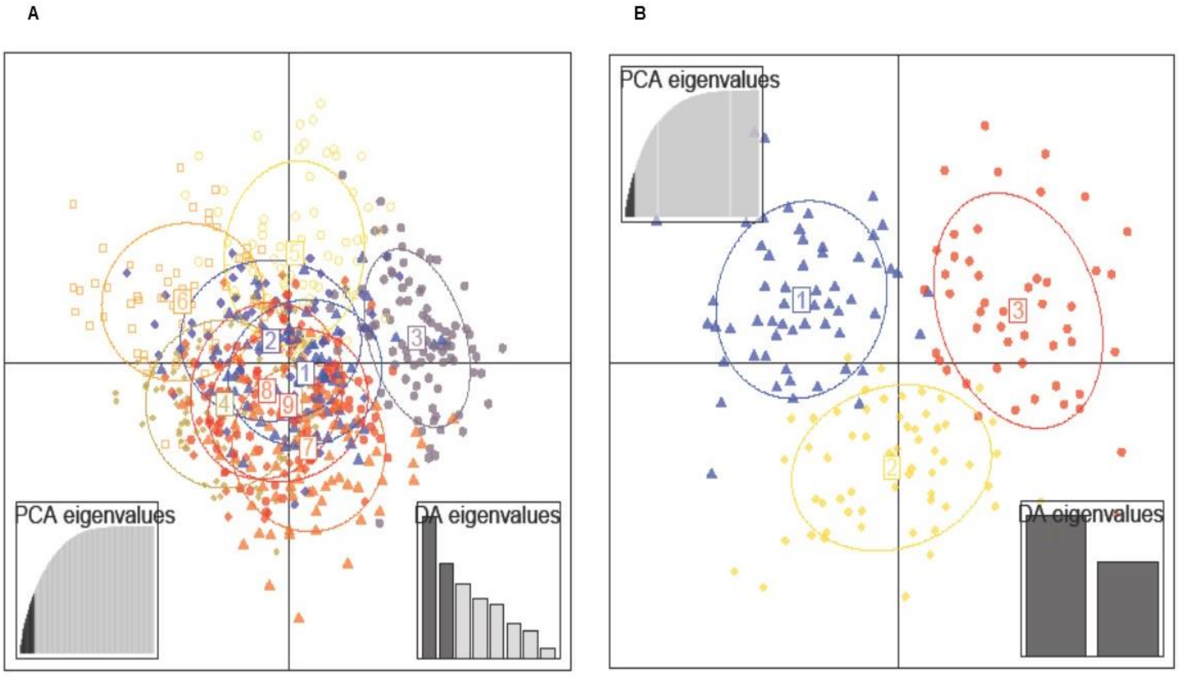

Figure 4. Discriminant Analysis of Principal Components (DAPC) clustering of male (A) and female (B) beluga samples from Beaufort Sea sampling locations (Shallow Bay, East Mackenzie Bay, Kugmallit Bay, Paulatuk and Husky Lakes), coloured based on group assignment.

2. Abundance

The most recent abundance estimate for this stock (based on July 1992 aerial surveys) is 19,628 (CV 0.229) (Harwood et al. 1996) which was corrected for belugas not visible to observers (submerged whales) to 39, 258 (Allen and Angliss 2015).

During these surveys, three Twin Otter aircraft, each with three or four observers, were used to conduct a systematic survey of the southeast Beaufort Sea (4.5-6.3% coverage), Mackenzie estuary (15-29% coverage), and west Amundsen Gulf (2.9% coverage) over a 55-h period on 23-25 July 1992 (Harwood et al. 1996).

In the estuary stratum, a strip-transect method was used. Standard transect lines established by Fraker (1977) were flown between 12:00 and 19:00 on 23 July 1992 in four substrata of the Mackenzie estuary: Kugmallit Bay, west Mackenzie Bay, east Mackenzie Bay, and Shallow Bay/Niakunak Bay (Harwood et al. 1996). The density of beluga in the offshore stratum was estimated using a line-transect method. This method was applicable to the clear offshore stratum, where fewer sightings were expected, so the time spent obtaining the perpendicular angle was unlikely to result in missed sightings. The four offshore substrata, west Beaufort Sea, middle Beaufort Sea, east Beaufort Sea, and west Amundsen Gulf, were surveyed between 14:00 on 24 July and 19:00 on 25 July 1992 (Harwood et al. 1996) (Figure 5).

The 1992 surveys provided an initial estimate of abundance using only data collected by the primary observers and yielded an overall, uncorrected estimate of 15,307 (95% CI 12,305 – 18,309) visible beluga whales in the Mackenzie estuary, southeast Beaufort Sea, and west Amundsen Gulf (Harwood et al. 1996). Incorporating data collected by the secondary observers, the estimate for the study area was adjusted to 19,629 (95% CI 15,134 – 24,125) surfaced, visible beluga (though calves were under-represented). This estimate includes an adjustment for missed-at-surface whales (missed by the primary observer but detected at the surface by the secondary observer) and for about-to-surface whales (i.e., those that surface during the short time separating the observation periods of the primary and secondary observers) (Harwood et al. 1996).

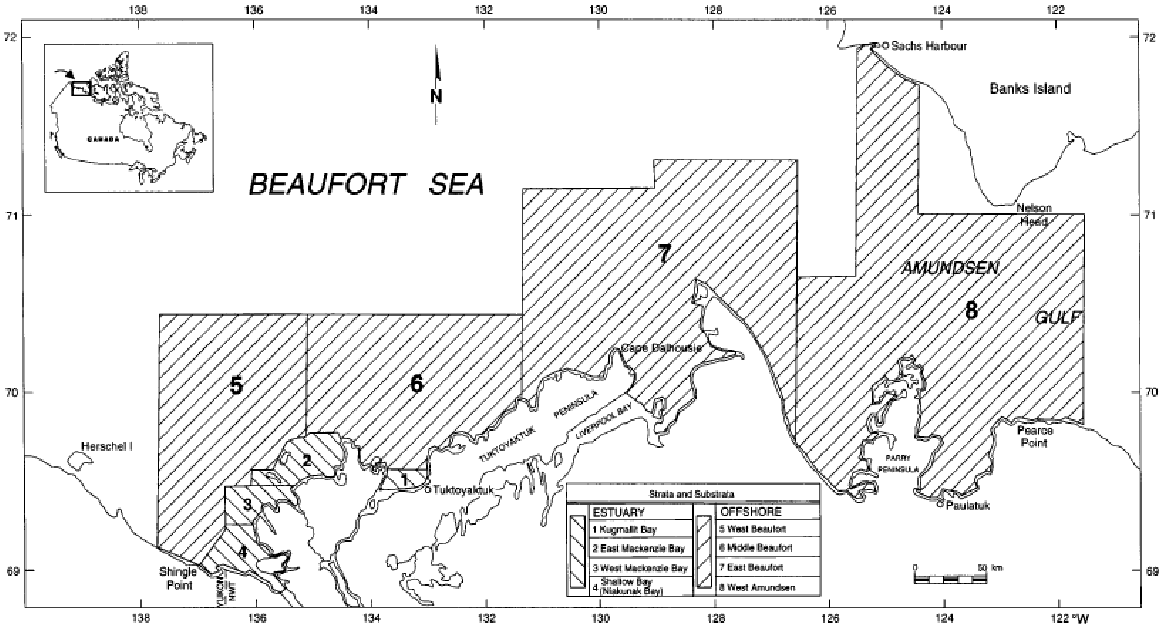

Figure 5. Stratum and substratum boundaries for the Beaufort Sea, Mackenzie estuary, and Amundsen Gulf aerial survey, 23-25 July 1992. Figure from Harwood et al. 1996.

However, these 1992 surveys did not sample the complete summer range of the stock and a considerable (but yet undetermined) number of whales were underwater during the aerial counts. Thus, this index was considered a low number in comparison to the undetermined estimate of the total size of the EBS stock (DFO 2000).

The NOAA 2014 Stock Status Report for Beaufort Sea belugas (Allen and Angliss, 2015), used correction factors for this July 1992 index of 19,629 (CV = 0.229) to provide an updated estimate. To account for availability bias, a correction factor (CF) of 2, which was not based on data, has been recommended for the Beaufort Sea beluga whale stock by a group of experts at a workshop (Duval 1993). This led to a calculation of a population estimate of 39,258 (19,629 × 2) animals. A coefficient of variation (CV) for the CF is not available; however, this CF was considered negatively biased by the Alaska Scientific Review Group, considering that aerial survey CFs for this species have been estimated to be between 2.5 and 3.27 (Frost and Lowry 1995). It still remains that the 1992 surveys did not encompass the entire summer range of Beaufort Sea belugas (Richard et al. 2001a and 2001b) and thus are negatively biased.

Though the 1992 survey was the most recent survey designed and timed to yield an adjusted estimate of abundance for the EBS beluga stock, systematic strip-transect aerial surveys for bowheads were also used to examine the distribution and relative abundance of surfaced belugas in the offshore Beaufort Sea in late August of 1982, 1984 – 85, and 2007 – 09 (Harwood and Kingsley 2013). Belugas were seen throughout the offshore area in both survey series, on 114 of 149 transects (76.5%). They were particularly common over the continental shelf offshore of the Tuktoyaktuk Peninsula and within 30 km seaward of the Mackenzie River estuary, but they were also seen singly or in small groups in most other offshore habitats surveyed (Figures 6 and 7).

The distribution of belugas had a similar pattern in both series, but the number of surfaced belugas counted was higher in the 2000s than in the 1980s (Harwood and Kingsley 2013). In total, 305 belugas (145 sightings, mean group size 2.1) were observed on-transect in 20 858 km^2 of surveying in the 1980s, and more than three times that number (1061) were observed in a similar area (19 829 km^2) during the 2007 – 09 survey series (378 sightings; mean group size 2.6) (Figure 8).

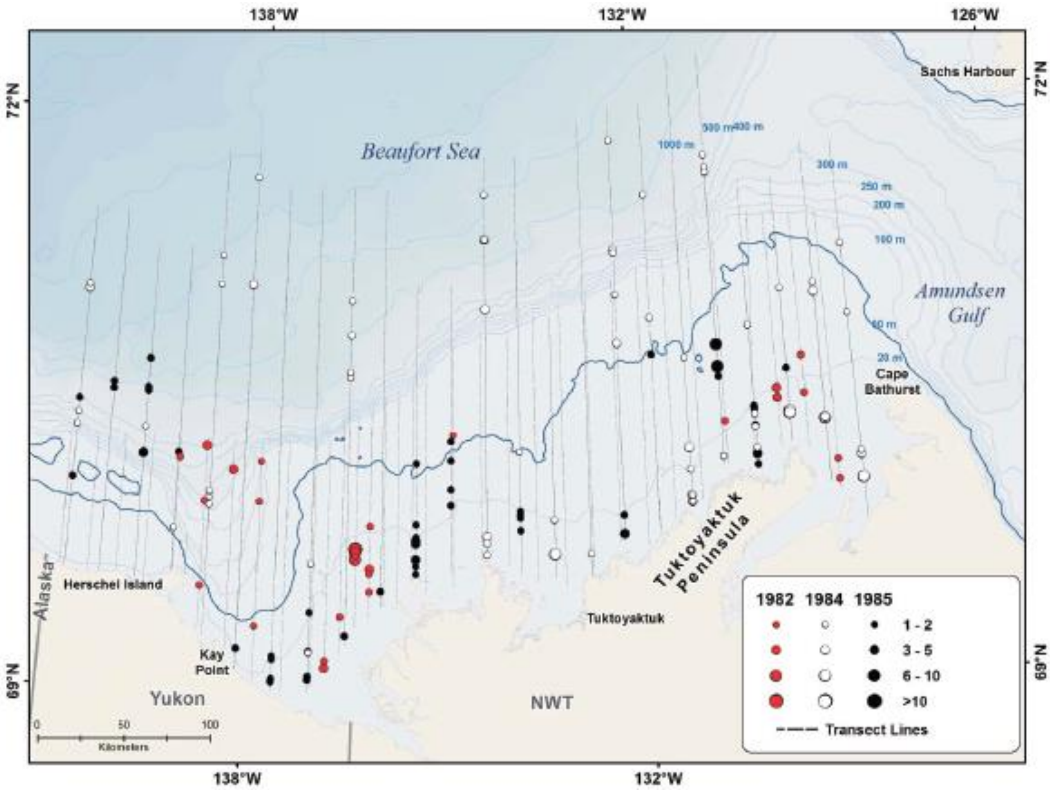

Figure 6. Location of transects (numbered from west to east) and numbers of surfaced belugas sighted in the offshore Beaufort Sea during aerial surveys in late August 1982, 1984, and 1985. Figure from Harwood and Kingsley (2013).

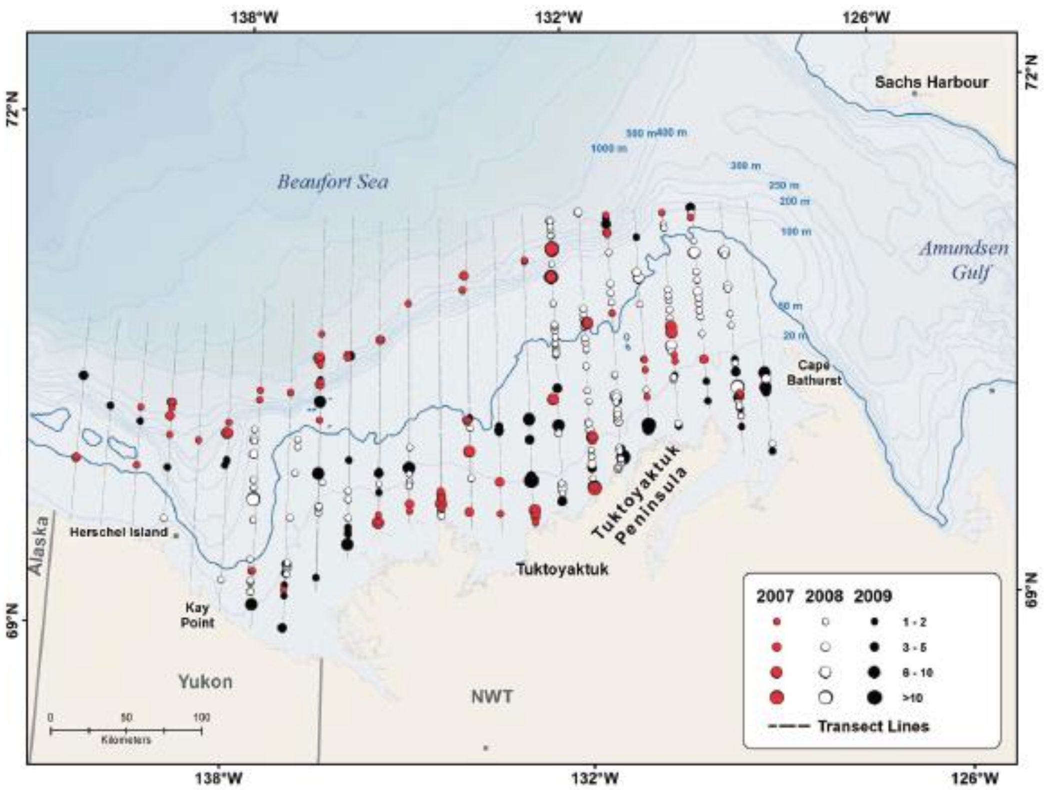

Figure 7. Location of transects (numbered from west to east) and numbers of surfaced belugas sighted in the offshore Beaufort Sea during aerial surveys in late August in 2007 to 2009. Figure from Harwood and Kingsley (2013).

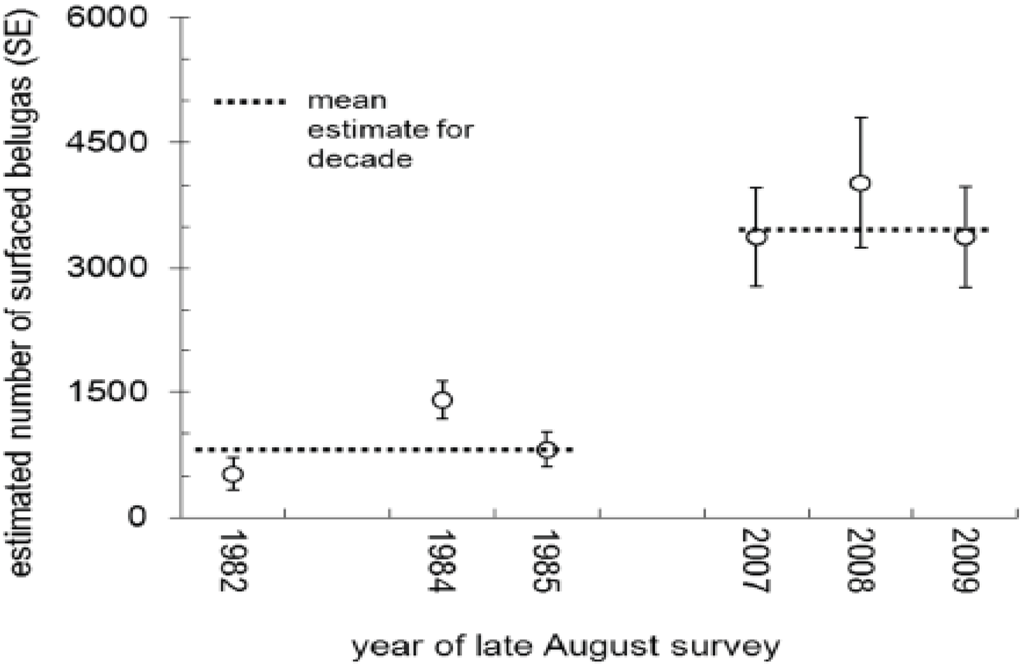

Figure 8. Estimated number of surfaced, visible belugas (and SE) in the southeastern Beaufort Sea (1980s vs 2000s), extrapolated for un-surveyed areas but not corrected for subsurface belugas or belugas outside the study area at the time of the survey. Figure from Harwood and Kinsley (2013).

The comparison of beluga counts in the 1980s vs the 2000s was not statistically analyzed to determine the significance of a trend, as it was not intended for this purpose. However, population growth alone, though probably not sufficient to explain the changes observed in relative abundance between decades, could be partly responsible for the apparent increase in belugas (Harwood and Kingsley 2013). The most plausible explanation is that the offshore became more attractive to belugas in the 2000s, because of either a decrease in the intensity or extent of industrial activity, or changes to the marine ecosystem related to climate warming, or both (Harwood and Kingsley 2013). These changes in numbers and distribution are echoed by observations of hunters in communities throughout the Beaufort Sea area.

3. Anthropogenic removals

There has been a long history of beluga hunting by the Inuvialuit and their ancestors in the Western Arctic, mainly from traditional whaling camps while whales are concentrated in the Mackenzie River estuary and distributed near the coast and communities (Harwood et al. 2002). Formal harvest monitoring programs have been in place in the Mackenzie Delta since 1973 and have resulted in the collection of data on the number of whales harvested, the efficiency of the hunts, and biological data for the animals sampled since the 1980 (Norton and Harwood 1986, Harwood et al. 2002). The most recent compilation of harvest data are presented in Table 1.

Whales from the EBS beluga stock are also taken by hunters in Alaska. Based on annual harvest numbers collected by each country, the mean estimated subsistence take in Canadian (2005-2009) and U.S. (2008-2012) waters from the Beaufort Sea beluga stock is 166 (100 + 65.6) whales (Allen and Angliss, 2015). This number of total removal from the stock (landed, struck and lost) for Alaska and Canada combined is less that the number used in the last Fisheries and Oceans Canada EBS beluga stock status assessment of 186 whales annually (DFO 2000). This assessment concluded that annual harvest was considered to be far below the level which might negatively affect the population.

Table 1. Known and estimated removals of Beaufort Sea belugas, 1987 – 2015.

Year		Canada		Alaska		Total
	Struck	Landed	Lost	Landed	Est. of 20% or unreported[2]	Struck (Can) + total (AK)
1987	174	144	30	50	10	234
1988	139	116	23	67	13	219
1989	156	117	39	26	5	187
1990	106	87	19	34	7	147
1991	144	116	28	43	9	196
1992	130	121	9	28	6	164
1993	120	110	10	85	17	222
1994	149	141	8	62	12	223
1995	143	129	14	4	1	148
1996	139	120	19	24	5	168
1997	123	114	9	43	9	175
1998	93	86	7	59	12	164
1999	102	86	16	35	7	144
2000	84	78	6	66	13	163
2001	32	91	1	25	5	122
2002	85	83	2	24	5	114
2003	123	111	12	43	9	175
2004	143	133	10	32	6	181
2005	108	106	2	20	4	132
2006	126	121	5	5	1	132
2007	82	82	0	62	12	156
2008	81	75	6	50	10	141
2009	102	96	6	13	3	118
2010[3]	93	90	3	71	14	178
2011	102	98	4	42	8	152
2012	75	73	2	92	18	185
2013	92	90	2	35	7	134
2014	106	104	2	24	5	135
2015	83	82	1	43	9	135
					Mean 1987 - 2015	163.6
					SD	32.9

[1]Data sources: Strong 1989; Weaver 1991; DFO, Fisheries Joint Management Committee unpubl. data; Harwood et al. 2002, 2015; Frost and Suydam 2010; ABWC unpubl. data).

[2] Added proportion of annual harvest based on Frost and Suydam (2010).

[3]Data for 2010-2015 is preliminary.

Annual harvest, by community and overall for the region are presented (Table 1). However, traditional knowledge from eastern Beaufort Sea beluga hunters offers explanations as to the reasons for declines in the harvest over time. There is diminishing interest in traditional subsistence hunting activities in the younger generation, changes in the availability of animals near their communities (reduces access and increases costs related to reaching animals further away for hunting), escalating costs of hunting (gas, equipment, etc…), and poorer weather conditions when whales are available for harvesting. These are all thought to be contributing factors.

The harvest of EBS belugas is gender biased because hunters select larger and older males. Harwood et al. (2014) examined patterns for data collected from standardized hunter-based sampling of harvested whales. Sex was determined for 95.2% (N= 3026 out of 3179 harvest samples) of the belugas landed in the Mackenzie Delta and Paulatuk harvests between 1980 and 2009. In 1980 – 89, males outnumbered females in the harvest by 2.0 to 1; in 1990 – 99, by 3.0 to 1 (Harwood et al., 2002); and in 2000 – 09, by 3.6 to 1. Trend tests revealed that this diminishing proportion of females in the harvest was statistically significant over the time series (Kendall's tau-b $\tau 0.281$, $p = 0.033$). Over the 1980-2009 study period, the proportion of females landed ranged from a low of 19.6% in Paulatuk to a high of 32.4% in the Kendall Island area.

Six ice entrapments, or savssets, are on recent record for Canada's Western Arctic (1966-2015) that have occurred in the area of the Husky Lakes (L. Harwood, pers. comm.). Husky Lakes is a series of four progressively less saline lakes connected by a set of "fingers" between each set, that are linked to the Beaufort Sea by Liverpool Bay. The Lakes are narrow, and relatively deep; at the narrowest point, which is in the second set of fingers, the opposing shorelines are only 38 m apart. The waters are ice-covered for eight to nine months of the year. Ice usually forms first in the fingers. Belugas are presumable attracted to these lakes for unique foraging opportunities as the Husky Lakes are rich in a number of fish species (e.g. lake trout, whitefish, cod) (Inuvialuit Land Administration 2011, Kocho-Schellenberg 2010). Entrapments have mostly involved small numbers of whales, with a known total of 250 belugas overall that were drowned or removed for humane reasons, between 1966 and 2015 (L. Harwood, pers. comm.).

4. Population trajectory

Given the lack of comparable abundance estimates for this stock, the population trajectory for this stock is unknown.

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The PBR for this stock has been calculated as follows (DFOa, in Prep., DFOb in prep.):

The Potential Biological Removal (PBR; Wade 1998) is calculated as:

$$PBR = N_{min} * 0.5 * R_{max} * F_R$$

where N_{min} is the estimated population size using the 20-percentile of the lognormal distribution ($N/[\exp(z20 * \mathrm{sqrt}[\ln(1+CV^2)])]$), R_{max} is the maximum rate of population increase (unknown for belugas and assumed to be 0.04, the default for cetaceans), and F_R is a recovery factor (between 0.1 and 1).

N_{min} for this stock, based on aerial survey data from the 1992 (Harwood et al. 1996) and determined by Wade and Angliss (1997), is 32,453.

A reliable estimate of the maximum net productivity rate (R_{max}) is not available for the EBS beluga stock, mainly due to age and gender biases of the harvests and thus samples are not available to calculate this or examine temporal shifts in reproductive rates. It was recommended that the default maximum theoretical net productivity rate for cetaceans of 4% be employed for this stock (as per Wade and Angliss 1997).

Due to the dated age of the most recent survey (1992) for the estimate of abundance, and that the population trajectory is unknown, a recovery factor of 0.75 was used (DFOa, in prep.)

Thus, using this information, PBR was calculated 487 animals (DFOa in prep.).

6. Habitat and other concerns

In the spring, entry of belugas into the Mackenzie estuary is linked to the timing of ice-break up seaward of the Estuary (Norton and Harwood 1986). Hornby et al. (2016) report spring distributions in 2011 to 2013 along the land-fast ice edge, which forms out to a winter depth of 20m in this area, is dependent on sea ice, bathymetry and turbidity habitat classes (Hornby et al. 2016). Even in years when belugas had access to a wide range of open water, turbidity and depth classes beyond the ice edge, whales were primarily found close to the ice edge (< 50 m deep), where fresh turbid water were present (Hornby et al. 2016). This is consistent with what has been seen and documented in the past from 1972-1985 (Norton and Harwood 1986).

Summer and fall habitat patterns have been investigated using resource selection function analysis of satellite telemetry data (Richard et al. 2001a). The objective was to better understand beluga habitat use of sea ice and bathymetry. The late summer to early fall habitat usage differed among size and sex classes, demonstrating sexual segregation on the summer range (Loseto et al. 2006). Within the Beaufort Sea area, three beluga habitat use groups were defined in relation to length, sex and reproductive status of the whales:

1) females with and without calves and small males (< 4 m) selected shallow open-water near the mainland;
2) medium length males (3.8 – 4.3 m) and a few females (>3.4 m) without neonates selected the sea ice edge; and,
3) the largest males (4 – 4.6 m) selected heavy sea ice concentrations in deep, offshore waters.

These divisions of summer and fall habitat use among beluga size and sex classes are thought to support a balance of access to prey, as well as the avoidance of high risk areas (Loseto et al. 2006, 2009).

There was a declining temporal trend in size-at-age of belugas landed in the Mackenzie Delta and Paulatuk noted between 1989 and 2008 (Harwood et al. 2014) (Figure 9). This temporal trend in size-at-age was investigated by fitting a linear trend with time to all size parameters—asymptotic lengths for both sexes and length and growth rate at age zero, and the standard deviations—using a Gompertz model. Year-to-year differences were small, ranging from +2% in 1994 to −2% in 2003, and were not statistically significant among the different years tested. The linear trend was a decline of 0.08% (SE 0.038%) per year over the 19 year series up to 2008 (Figure 9). The linear-trend model was selected by the information criterion and its gross change in likelihood was statistically significant at 5%, although not when tested against the residual variation between years.

The subtle changes in growth of belugas over the time series up to 2008 may reflect ecosystem changes that have reduced the availability or quality and quantity of their prey in recent years, a finding which was paralleled in other upper trophic species during the corresponding period (ringed seals, Harwood et al. 2012).

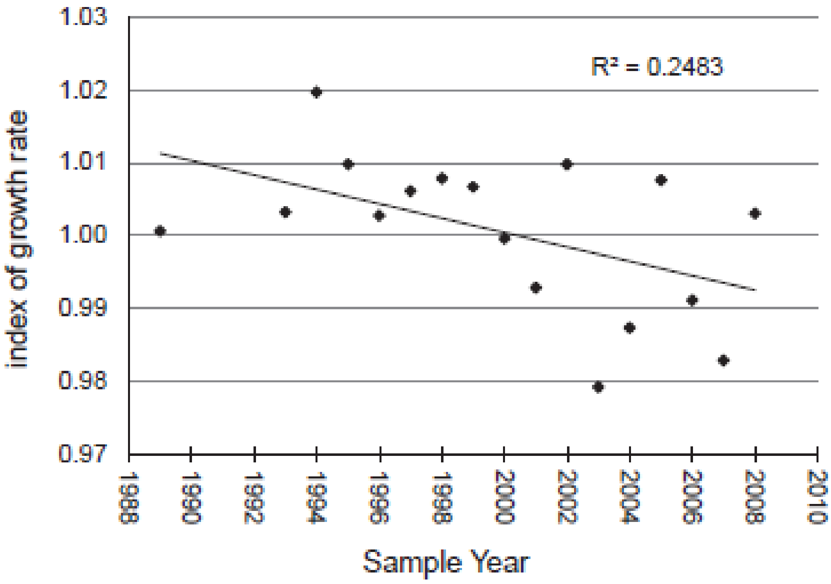

Figure 9. Temporal trend in size-at-age of EBS belugas landed in the Mackenzie Delta and Paulatuk subsistence harvests in 1989 and 1993-2008. Figure from Harwood et al. (2014).

7. Status of the stock.

Currently, the EBS beluga stock is considered “Not at Risk” by the Committee On the Status of Endangered Wildlife in Canada (COSEWIC 2015). At the time the stock status was last evaluated by the committee (2004), the designation was based on information demonstrating that the stock was large and hunted at sustainable levels under an international agreement.

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Annex 10: Eastern High Arctic – Baffin Bay and West Greenland Beluga Whale Stock

Steven Ferguson and Rikke Guldborg Hansen

1. Stock Definition and Distribution

Beluga whales, although circumpolar in their distribution, display seasonal migrations with a strong fidelity to summer aggregation areas. Stock delineation is based on these summer aggregation areas as there is strong genetic evidence for spatial segregation based on matrilineal subpopulation structure (Brown Gladden et al. 1997, de March and Postma 2003, Turgeon et al. 2012, Colbeck et al. 2012). Within the eastern Canadian Arctic, three populations have been identified: Hudson Bay, Cumberland Sound, and High Arctic-Baffin Bay (HA-BB) (Brown Galdden et al. 1999, de March et al. 2002, de March and Postma 2003). The HA-BB population summers largely in and close to bays, inlets, and estuaries around Somerset Island in the Canadian Arctic Archipelago (Koski and Davis, 1980; Smith and Martin, 1994). Notable aggregation areas include Radstock Bay, Maxwell Bay, and Crocker Bay on Devon Island; Cunningham Inlet, Creswell Bay, and Elwin Bay on Somerset Island, and Coningham Bay on east Prince of Wales Island. Some belugas from the HA-BB population winter in the North Water (northwest Baffin Bay and Smith Sound) with the majority spending winter in West Greenland (Doidge and Finley, 1993) in pack ice near the ice extent (Heide-Jorgensen and Laidre 2004).

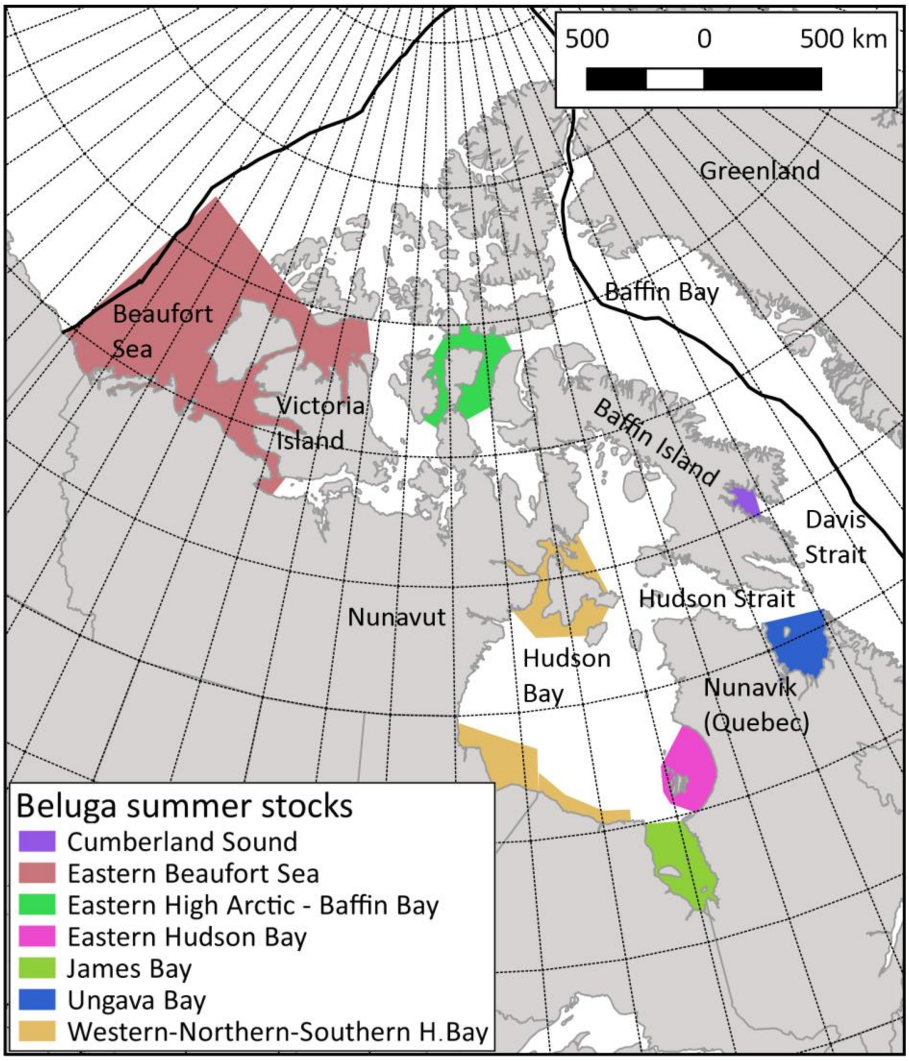

Figure 1. Beluga whale stocks in the Canadian Arctic. Dark green shows core summer aggregation range of High Arctic-Baffin Bay population.

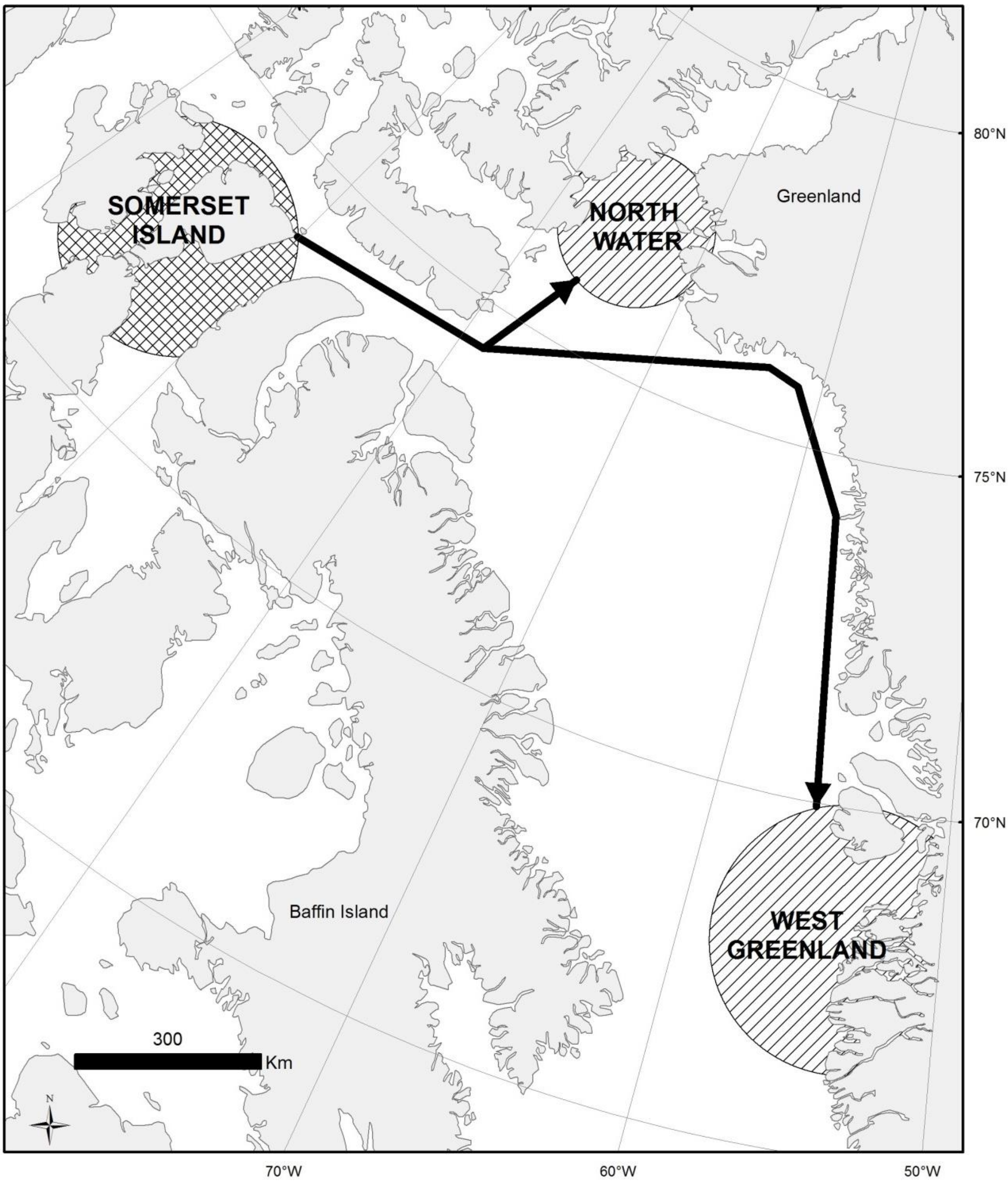

Figure 2. Movements of belugas from summering grounds (double hatched areas) around Somerset Island in Northeast Canada to wintering grounds (hatched area) either in the North Water or along West Greenland south of Disko Island.

2. Abundance

The estimate of HA-BB population size from an aerial survey in the Canadian summer of 1996 was 21,213 belugas (95% CI 10,985 to 32,619) (Innes et al. 2002).

In 2012, a survey off West Greenland estimated 7456 belugas (95% CI: 3293–16,987) (Heide-Jorgensen et al. 2016). A mark–recapture distance analysis that corrects for perception and availability bias for the 2012 survey estimated the abundance to be 9072 whales (cv= 0.32; 95% CI 4895-16,815).

3. Occurrence of belugas on the wintering grounds in West Greenland

As described in more detail in Heide-Jørgensen et al. 2016, a visual double-observer aerial line transect survey was conducted using a fixed-winged aircraft (DeHavilland Twin Otter) equipped with four bubble windows flying at a target altitude and speed of 213 m and 166 km·h-1 respectively. The front (observer 1) and rear (observer 2) observers acted independently of each other, recording declination angles to sightings (using a Suunto inclinometer) as well as species and group size when the animals passed abeam. Time-in-view was recorded as the difference between the time at first sighting and time when the sighting passed abeam. Beaufort sea state and glare were recorded at the start of the day and whenever they changed. Decisions about duplicate sightings (animals seen by both observer 1 and 2) were based on coincidence in timing and positions, group size and direction of movement. Declination angles (□) measured when animals were abeam were converted to perpendicular distances (x) using the following equation from Buckland et al. (2001): x= vR*Rtan(90-□) where v is the altitude of the airplane. Forward distance (y) to each sighting was calculated based on time of first sighting, time when passing abeam and speed of aircraft.

The survey was conducted from 24 March to 15 April 2012 covering the area between 65°40’N and 75°30’N (~243,000 kmP2P, ~7,800 km ‘on effort’, Fig. 2, Table 1) and observers recorded sightings of all marine mammals. Sixteen strata with 116 transect lines were identified and these lines were systematically placed so that east-west density gradients would be crossed.

A modification of the hidden Markov line transect model (hmltm) of Borchers et al. (2013) and Rekdal et al. (2015) was used to estimate the detection probability, density and abundance of belugas. This involved first estimating the parameters of a hidden Markov model (HMM) for whale availability and then integrating these with the line transect data, using both perpendicular and forward distances to detected whales to estimate detection probability (see Borchers et al. 2013 for details).

No data exist on the dive cycle of belugas in their wintering ground in West Greenland and logistical difficulties prevent such data from being collected with currently available techniques. Instead dive cycle observations from Martin and Smith (1999) were incorporated into a two-state Markov model for the time series of states, and Bernoulli random variables with the parameters Pr(avail|state 1) and Pr(avail|state 2) for availability were used given the hidden states.

Estimation methods for detection function parameters, group size, group abundance and animal abundance are described in Supplementary Material. Model selection was based on AIC and goodness-of-fit p-values for all models that converged. Coefficients of variation (cv’s) were obtained by bootstrapping mean durations of dive cycle and time available from the Markov model for availability, and bootstrapping transects within strata. 1,000 bootstrap resamples were drawn and confidence intervals were obtained from the point estimate and cv, assuming log-normality.

Both a conventional distance sampling (cds) and a mark-recapture distance sampling (MRDS) analysis that accounts for the so-called 'perception bias' for animals available at distance 0 but missed by the observers were applied to the survey data. Encounter rate and cluster size were estimated by stratum, with detection probability pooled across all strata. Variances and confidence intervals were estimated as above.

In 2012 a total of 7,800km was flown over 75 transects in 16 strata covering a total area of 242,650 km2. Belugas were observed in 5 out of 16 strata and they were found primarily in coastal areas along

West Greenland and in shallow water (<200 m deep, Fig. 2). No belugas were observed south of 67.7°N or north of 71°N and none were detected in Disko Bay. Belugas were seen in the highest densities at the northern edge of Store Hellefiske Bank, southwest of Disko Bay. They were also found in the northern opening of Vaigat and off Uummannaq.

When including the previous nine surveys of belugas in West Greenland, a significant correlation was detected between the longitude of sightings (i.e. distance from the coast) and the extent of sea ice, i.e. the more pack ice in Baffin Bay the closer to the coast belugas were observed at the time of the survey (ANOVA, p=0.002, Fig. 3). However, this correlation is strongly driven by the observations in 2006 when little open water was present. If 2006 is excluded from the analysis the trend can still be seen but it is no longer significant (p=0.08).

The half-normal detection function model used observations from both observers 1 and 2 and was chosen for conventional distance sampling on the basis of AIC (Fig. S11). The associated Cramer-von Mises goodness-of-fit statistic had a p-value greater than 0.9 and the conventional distance sampling analysis yielded an abundance estimate of 7,546 whales (cv=0.38, 95% CI 3,461-16,450).

The half-normal detection function model with no variables was chosen for the mark-recapture distance sampling on the basis of AIC (Table 3, Fig. 4) and yielded an abundance estimate of 9,072 whales (cv=0.32, 95% CI 4,895; 16,815) with a g(0)=0.94 for both observers. Both the conventional and the mark-recapture distance sampling estimates were corrected for availability bias.

The abundance estimate of 9,072 belugas (cv=32%) in 2012 from the mrds represents a slight decrease in abundance compared to the last abundance estimate of 10,595 (cv=43%) belugas in 2006 (see Fig. 5, Heide-Jørgensen et al., 2010) but is larger than an estimate from 1999 of 7,941 belugas (95% CI 3,650–17,278) (Heide-Jørgensen & Acquarone 2002). None of the estimates are significantly different. Several scenarios may explain the recent (after 2004) fluctuations in abundance of belugas in West Greenland. The continued hunting causes direct mortality which of course affects the abundance and density of belugas in West Greenland. Accessibility of the whales to the hunters is affected by the extent of sea ice. A large extent of sea ice forces the whales closer to the coast and within reach of the hunting communities (Fig. 7, Heide-Jørgensen et al., 2010). The stock of belugas that winters in West Greenland is part of the larger aggregation that is found in the summer in inlets and bays along Somerset Island in northern Canada. Only a portion of the whales from Somerset Island move to West Greenland for the winter whilst the other portion winters in the North Water area in northern Baffin Bay (Heide-Jørgensen et al., 2003). It is unknown if the apportioning of whales between the two wintering areas fluctuates from year to year. It is also possible that undiscovered ice entrapments cause mortality or that excessive disturbance from fishing activities or seismic survey activity reduces the fraction of the Somerset Island belugas that ends up wintering off West Greenland in a given year (cf. Heide-Jørgensen et al., 2012).

4. Anthropogenic removals

Heide-Jørgensen and Rosing-Asvid (2002) calculated a correction factor of 1.29 for killed and lost plus unreported catches. Stewart and Innes (2002) calculated that struck, killed and lost, or not reported whales was 1.41 (1.02-2.42) or 1.4 whales killed for each whale landed and recorded.

Canada

This stock is hunted by communities in Nunavut, Canada and West Greenland. In Nunavut, hunts occur during spring, summer, and fall while belugas are migrating to and from, and residing in, their summer aggregation area around Somerset Island. Nunavut communities that hunt the HA-BB beluga include Arctic Bay, Clyde River, Gjoa Haven, Grise Fiord, Hall Beach, Igloolik, Kugaaruk, Kugluktuk, Pond Inlet, Qikiqtarjuaq, Resolute Bay, and Taloyoak (Table 1).

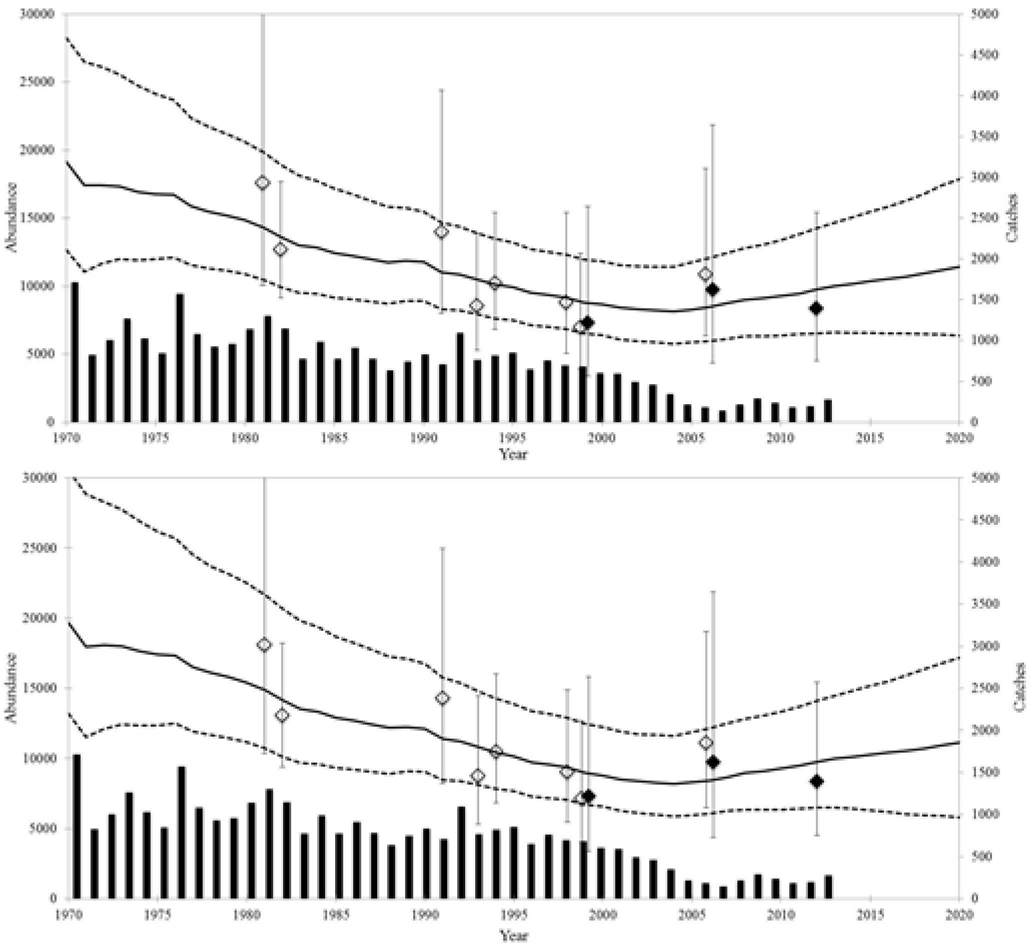

Figure 3. The projected median and 90% credibility interval, together with the absolute abundance estimates (solid diamonds) and the rescaled relative estimates (open diamonds). Top plot, dB model; Bottom plot, dBl model. Catches shown as histograms below.

West Greenland

Correcting for underreporting and killed-but-lost whales (Table 2) increases the catch reports by 42% on average for 1954-1998. If the whales killed in ice entrapments are removed then the corrected catch estimate is on average 28% larger than the reported catches. Catches declined during 1979-2014 to levels below 300 whales per year after 2004.

All catches in West Greenland are presumably taken from the fraction of the Somerset Island summering stock of belugas that winters in West Greenland. The exception is the winter catches in Qaanaaq (approx. 5% of annual catches in Qaanaaq) that likely are taken from the fraction that winter in the North Water. It is unknown which stock is supplying the summer hunt in Qaaanaq (approx. 15% of annual catches in Qaanaaq).

Table 1. Landed catches of beluga whales reported by Nunavut communities that hunt from the High Arctic-Baffin Bay population, 2011-2015.

			Landed Catches by Harvest Year º				
Beluga Population	Community	Quota ¥	2011-2012	2012-2013	2013-2014	2014-2015	2015-2016
Baffin Bay	Arctic Bay	NRQ	0	2	0	0	0
	Clyde River	NRQ	0	0	0	n.r.	1
	Gjoa Haven	NRQ	10	4	5	0	10
	Grise Fiord	NRQ	0	n.r.	0	3	3
	Hall Beach	NRQ	8	n.r.	0	19	7
	Igloolik	NRQ	42	n.r.	0	n.r.	n.r.
	Kugaaruk	NRQ	0	0	0	1	0
	Kugluktuk	NRQ	21	0	0	n.r.	0
	Pond Inlet	NRQ	0	0	0	n.r.	0
	Qikiqtarjuaq	NRQ	0	n.r.	n.r.	n.r.	0
	Resolute Bay	NRQ	4	6	76	8	4
	Taloyoak	NRQ	0	0	n.r.	n.r.	3
	Total						

¥ NRQ = No Regulatory Quota
º n.r. = no record received

Table 2. Catches of belugas from official reports by municipality with corrections for under-reportings for 1954 to 2016. The column 'under-reporting' shows the sum of the corrections for under-reporting or 'ALL' if it is a general correction factor for all regions. 'Disko Bay' includes the municipalities Kangaatsiaq, Aasiaat, Qasigiannguit, Ilulissat and Qeqertarsuaq. The catches before 1975 are extracted from Kapel (1977), between 1975 and 1990 from unpublished statistics from the Ministry of Greenland, Kapel (1983), Kapel and Larsen (1984), Kapel (1985), Born and Kapel (1986), Born (1987) and Heide-Jørgensen (1994), and from 1993 to 2016 from 'Piniarneq'.

Year	Qaanaaq	Upernavik	Uummannaq	Disko Bay	Sisimiut	Maniitsoq	Nuuk	Paamiut qaqortoq	Under-reporting		Total	Mortality Ice Entrap-ment
									All	Regions		
1954		16	61	1774	23						1874	1774
1955		10	3	275	11	1					300	
1956		9	8	373	29	5					424	
1957		6	11	391	95						503	
1958		3	4	182	35	1					225	
1959		12	12	243	42						309	50
1960		13	6	179	17		1				216	
1961	32	15	6	219	47	1	11	14			345	
1962	85	9	7	186	23	8	11				329	
1963	75	18	12	93	8	12	11				229	
1964	125	4	6	166	8	4	18				331	
1965	150	20	53	214	24	18	9				488	
1966		25	88	398	24	13	12	1			561	
1967		34	66	369	76	47	4				596	50
1968		97	65	1013	46	38					1259	234
1969		111	36	661	100	40	30				978	
1970	17	334	6	1133	10	24					1524	1050
1971	2	238	3	328	123	4	41				739	
1972		293	25	362	135	11	14	1			841	
1973		262	33	581	121		70				1067	

1974	21	195	15	512	135	8	25	2			913	
1975	50	150	19	268	130	4	33			47 Q	654	
1976	50	77	12	953	72		48			37 Q	1212	653
1977	50	240	49	379	43	13	65			36 Q	839	
1978	20	104	44	452	77	5	17				719	
1979	25	250	22	379	35	12	18				741	
1980	30	191	100	412	109	45	1				888	
1981	76	343	95	340	62	23	78				1017	
1982	1 27	329	17	313	95	13					894	100
1983	53	233	19	194	99	2	1			10 Q, 165 UP, 100 DB, 50S	601	
1984	21	333	15	352	25	16	1			60 UP, 150 DB, 25 S	763	220
1985	190	188	6	177	25	17	8			135 UP, 75 DB, 25 S	611	
1986		500	4	114		2			75	335 UP	695	
1987		550	13	29		8	6		90		696	
1988		125		125					25		275	125
1989		427	2	30		40				311 UP, 18 DB	499	
1990	2	346	8	684		23				2 Q, 346 UP, 591 DB	1063	500
1991	50	400		100						50 Q, 400 UP, 100 DB	550	
1992		661		26						661 UP, 26 DB	687	
1993	119	328	26	191	79	24	14	1		169 UP	782	
1994	24	188	19	239	105	38	3	2		90 UP	618	
1995	26	252	18	301	117	56	10	4		111 UP	784	
1996	7	86	21	244	131	26	25	1			541	
1997	17	162	29	228	100	7	11	2			556	
1998	71	163	41	304	105	15	4	11			714	
1999	36	189	25	184	38	4	10	6	0		492	
2000	8	303	21	202	57	6	7	8			612	
2001	4	131	26	207	64	19	1	3			455	
2002	5	203	38	149	15	11	1	8			430	
2003	54	119	16	149	48	19	0	7			412	
2004	2	14	8	96	61	4	1	7			193	
2005	3	26	13	102	36	4	0	0			184	

2006	9	31	13	49	28	3	3	1			137	
2007	7	20	2	59	19	9	0	0			116	
2008	45	159	13	58	8	3	1	0			287	
2009	20	114	31	53	17	9	1	0			245	
2010	2	104	15	60	1	5	1	0			188	
2011	7	63	5	67	6	2	0	0			143	
2012	24	120	4	58	2	3	0	0			187	
2013	26	167	19	52	26	14	0	0			304	
2014	31	125	20	71	9	13	0	2			271	
2015	7	27	2	73	6	9	1	0			125	
2016	16	76	3	79	17	12	0	0			203	

Correction for losses

Losses during drive hunts are considered minimal and a catch correction factor of 1.10 is applied to the reported catches to correct for whales lost during drive hunt operations. The drive hunt was the most important way of hunting belugas in Qaanaaq and Upernavik until it was banned in 1995 (effective from the 1996 hunting season).

Shooting whales in open or ice-covered waters (=non-drive hunt) has a much larger proportion of lost whales and a catch correction figure of 1.30 is applied to the statistics from this type of hunting. This hunt type is practiced in all areas south of Upernavik and from 1996 even in Upernavik and Qaanaaq. Therefore a *high option* for the catch statistics after 1954 (applied to the *medium option*) includes a correction of the harvest in northern municipalities (Qaanaaq and Upernavik) of 1.10 and a correction factor in all other areas of 1.30 to adjust for losses during the catch operations.

Catches from multiple stocks in Qaanaaq

The hunt in the municipality/district of Qaanaaq includes both catches of animals migrating towards the wintering ground in West Greenland and therefore part of the same stock that is exploited in West Greenland, catches in spring (May) from the wintering stock of belugas in the North Water and catches during summer months (June through August) of whales of unknown stock identity. The monthly distribution of the catches indicate the relative contribution of the three stocks and it is evident that the southernmost settlement of Savissivik (part of the municipality of Qaanaaq) is solely exploiting the fall migration of belugas that are moving along the coast towards West Greenland (Fig. 3). Catches in Qaanaaq (incl. Siorapaluk) are however dispersed over a longer period of the year but peaks in September where the belugas are moving south along the coast. Summer catches (June-August) in the municipality of Qaanaaq is about 15% of all the catches, whereas spring catches that are assumed to be from the North Water stock, is about 5% of the catches, the rest of the catches (80%) are presumably whales moving towards West Greenland.

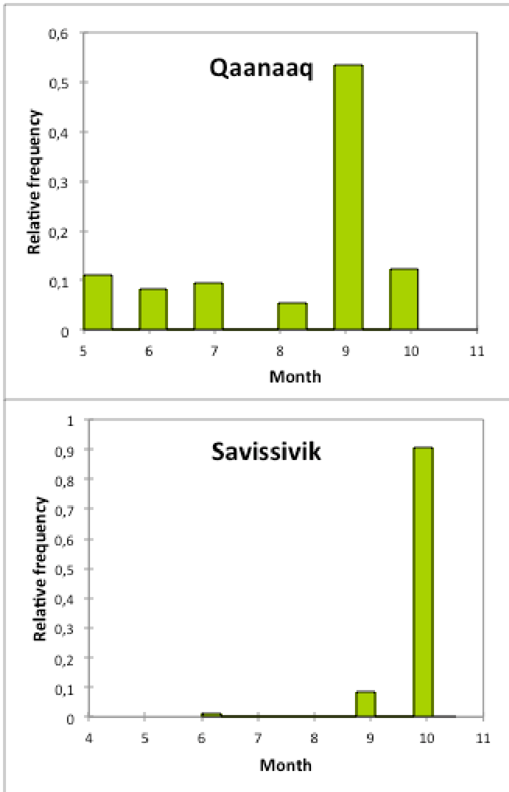

Fig. 3. Distribution of catches per month in Qaanaaq and Savissivik (both part of Qaanaaq municipality). Data from 2008-2014 (n=168).

5. Population trajectory

Innes and Stewart (2002) estimated the pristine/carrying capacity abundacne of the HA-BB population as 39,790 (19,812 - 78,588) for the Baffin Bay wintering stock and 15,966 (5,053 - 30,748) for the North Water portion of the wintering population (total population size = 55,756). Their analysis indicated a decline of about 50% of the population between 1981 and 1994 largely due to Inuit overharvest. However, considerable commercial harvesting of beluga in their estuaries has occured and likely resulted in the initial population decline.

A Bayesian population dynamics model was fitted to relative and absolute indices of winter abundance of belugas off West Greenland (Heide-Jorgensen et al. 2016) using an age- and sex-structured population dynamics model with an even sex ratio and a Pella–Tomlinson form of density regulation on the birth rate (Pella & Tomlinson, 1969). The model suggests a population trajectory that shows a continuous decline from 1970 through 2003. After 2003, with the introduction of catch limits, a slight increase of abundance occured. The high survival (0.98) model estimated a decline from 18 600 (90% CI: 13 400, 26 000) whales in 1970 to 8000 (90% CI: 5830, 11 200) in 2004, and it projects an increase of 11 600 (90% CI: 6760, 17 600) individuals by 2020 (assuming annual removals after 2014 of 294 belugas).

A modelling estimate by Stewart and Innes (2002) came up with a similar population abundance estimate as the aerial survey of 21,093 using a surplus production model within a SIR Bayesian analysis based on index surveys of belugas off the West coast of Greenland and harvest data from Canada and Greenland. Their results estimated the North Water portion of the wintering population (NW) of 14,839 and the West Greenland portion of the wintering population (WG) of 6,254.

6. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

Canada does not have a quota system in place for Inuit harvesting of beluga whales of the High Arctic-Baffin Bay population for Nunavut. Greenland sets their quota for their winter hunt based on science advice from the Joint Commission on Narwhal and Beluga (e.g., JCNB/NAMMCO 2015).

7. Habitat and other concerns

Beluga Whales are occasionally attacked by Polar Bears (*Ursus maritimus*) and Killer Whales (*Orcinus orca*). Their predisposition to return to the same estuaries year after year makes them vulnerable to human hunting and disturbance. For the High Arctic-Baffin Bay population, exploitation by commercial whaling had an adverse effect on the population abundance and over harvest may have occurred to whales while overwinter along the West Greenland coast. The Beluga that winter in the North Water polynya area has not been adversely affected by Inuit harvesting.

8. Status of the stock

Harvest levels from Greenland and Canada have stabilized since 2004 at about 100 in Canada and 300 in Greenland. The population shows signs of recovery although the population is likely less than 50% of its original pristine population level (21,000 in 1996 compared to a carrying capacity of 56,000: Innes and Stewart 2002). Current harvests of about 100 by Canada and 300 by Greenland appear to be sustainable; however, the population will likely take many decades to return to carrying capacity. The COSEWIC status for this stock is Special Concern in 2004.

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Annex 11: Western Hudson Bay Beluga Stock

By: Matthews, C.J.D. and S.H. Ferguson

1. Distribution and stock identity

Canadian belugas are managed as populations and/or stocks based mostly on the disjunct distribution of summer aggregations (Richard 2010). These putative stocks have been characterized using body size and behaviour (Martin et al. 2001), genetics (Brown Gladden et al. 1997, Brown Gladden et al. 1999, de March et al. 2002, de March and Postma 2003, Turgeon et al. 2012, Colbeck et al. 2013), contaminants (de March et al. 2004), biomarkers such as stable isotopes and fatty acids (Rioux et al. 2012), as well satellite telemetry studies that have revealed beluga site fidelity to distinct summer areas (Caron and Smith 1990, Richard et al. 2001).

The Western Hudson Bay (WHB) beluga stock overwinters in Hudson Strait (Figure 1), where it overlaps with belugas from eastern Hudson Bay (Turgeon et al. 2012). The summer distribution of WHB belugas is centred around the Seal, Churchill, and Nelson River estuaries off the coast of Manitoba, although belugas occur further north along the Nunavut coast and south along the coast of northwestern Ontario (Figure 1). During spring and fall migrations, WHB belugas overlap with belugas in eastern Hudson Bay (COSEWIC 2004).

Western Hudson Bay belugas are genetically more diverse than other Canadian beluga stocks, and possess haplotypes common to all other Canadian stocks (de March and Postma 2003). However, haplotypes common to belugas from the neighbouring Eastern Hudson Bay (EHB) stock are rare in WHB belugas (de March and Postma 2003). Not all of the entire range has been sampled, and there is uncertainty about further stock structure along the west and south coasts of Hudson Bay (e.g. belugas off the coast of northwestern Ontario).

2. Abundance

WHB beluga stock abundance has been estimated three times using visual and photographic aerial surveys: in 1987, 2004, and 2015 (Richard et al. 1990, Richard 2005, Matthews et al. 2017). Abundance estimates from these three surveys are compared and discussed below (‘Population trajectory’), while the abundance estimate from the most recent survey, along with survey design, methods, and analysis, are presented here.

The WHB beluga aerial survey in 2015 was originally planned to span the entire western coast of Hudson Bay, encompassing the coasts of Nunavut, Manitoba, and Ontario (Matthews et al. 2017). However, weather delays forced cancellation of surveys off the Nunavut and Ontario coasts, and the survey focused on high-use areas encompassing the Seal, Churchill, and Nelson River estuaries (Figure 2). The survey was divided into five strata with varying degrees of effort, with the Churchill River estuary and the mouth of the Seal River surveyed completely using aerial photographs, two nearshore strata that surrounded the river estuaries surveyed visually along parallel transects, and a final offshore stratum that was surveyed visually with reduced effort (zigzag transects; Matthews et al. 2017; Figure 2).

Surveys were flown in a Twin Otter with bubble windows to facilitate viewing, and a camera hatch at the rear underbelly of the plane for taking photographs. Four observers seated two on each side of the aircraft focused on the area closest to the track line, and used their peripheral vision for sightings farther afield. Observers recorded group size, the perpendicular declination angle to the centre of each group, and, when time permitted, additional details such as the direction of travel, presence of calves, and behaviour. The two primary observers also described ice concentration, sea state, fog, glare, and cloud cover. A digital SLR camera equipped with a 25 mm lens was directed straight down through the camera hatch, capturing an approximate ground area of 875 m x 585 m at the survey altitude of 2,000 ft (610 m; Matthews et al. 2017).

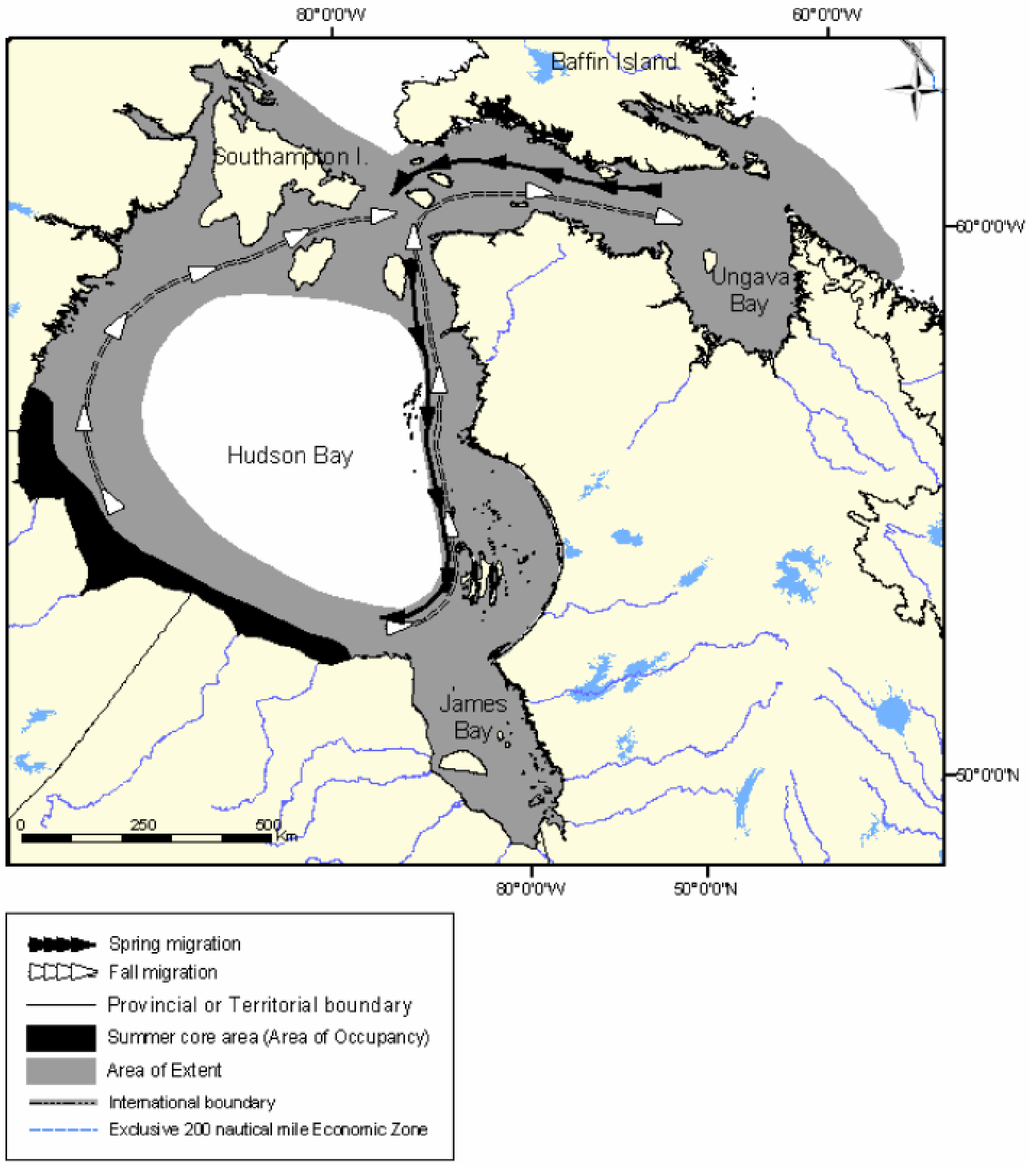

Figure 1. Area of extent of Western Hudson Bay belugas. Summer core-use area shown in black (from COSEWIC 2004). Telemetry results indicate WHB belugas also occur in the middle of Hudson Bay (Smith et al. 2007).

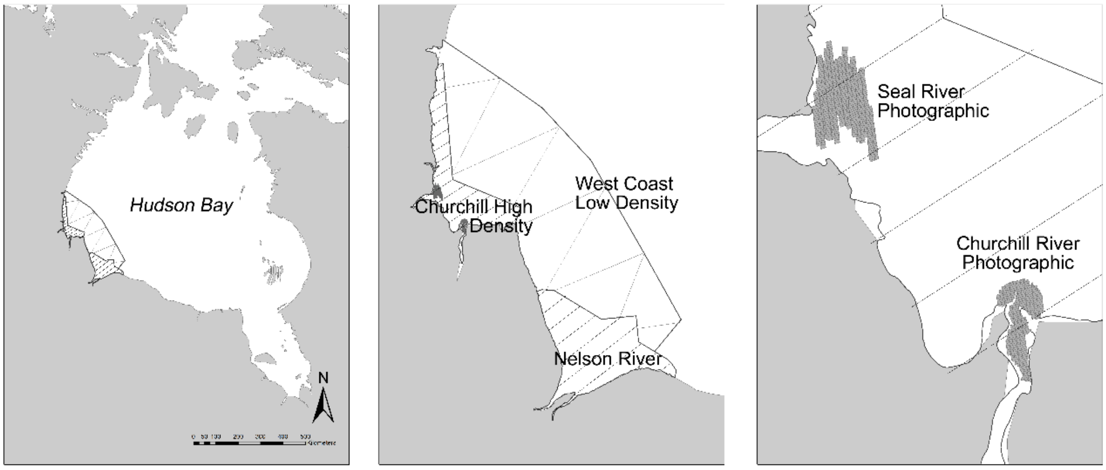

Figure 2. The aerial survey of WHB belugas in 2015 included the major areas of aggregation in western Hudson Bay (left panel), and consisted of strata surveyed visually along transect lines (middle panel), and complete coverage photographic surveys of two strata within the Churchill High Density stratum where beluga densities were too high to count accurately during the visual survey (right panel).

For analysis details, see Matthews et al. (2017). Briefly, visual line-transect survey data were analysed using conventional distance sampling (CDS; Buckland et al. 2001) using Distance 6.2 software (Thomas et al. 2010). Multiple covariates distance sampling (MCDS) was performed to determine whether inclusion of environmental covariates improved model estimates. Poor data quality from less experienced secondary observers prevented mark–recapture distance sampling (MRDS) analysis to estimate the proportion of animals missed due to perception bias (Matthews et al. 2017).

Dive data from satellite tagged belugas in the survey area, however, were used to adjust near surface estimates for availability bias, or the proportion of animals too deep to be observed. Previous studies have shown adult and juvenile beluga whales are visible at depths up to 5 and 2 m in clear water, respectively, and cannot be seen at depths greater than 2 m in murky water. Dive date were therefore used to calculate the proportion of time belugas spent in the 0-1, 0-2, 0-4, and 0-5 m depth bins, and photographs of surveyed areas were assessed qualitatively for turbidity (‘murkiness’). A correction factor based on the 0-2 m depth bin was applied for whales sighted in the Churchill River, which was judged to be murky, and a correction factor based on the 0-5 m depth bin was applied to all other strata, which had clear water. Near surface abundance estimates were corrected for availability bias by multiplying by the correction factor based on the proportion of time belugas spent within the respective depth bin.

Beluga densities in the photographic strata were determined by dividing the total beluga count by the total water area (excluding sun glare, which masked beluga presence) across all photographs. Surface abundance was calculated by multiplying beluga density by the area of the polygon created by merging all photos, with land area subtracted, and adjusted using the availability bias factors as described above (Matthews et al. 2017).

Availability bias adjusted abundance estimates were 7,876 (CV = 0.29) for the Churchill High Density stratum, 23,248 (CV = 0.20) for the Nelson River stratum, 64 (CV = 0.98) for the West Coast Low Density stratum, 20,149 (CV = 0.04) for the Seal River photographic stratum, and 3,173 (CV = 0.30) for the Churchill River photographic stratum. The sum of the availability bias-corrected abundance estimates of the five strata provided a total abundance estimate of 54,473 (cv = 0.098, 95% CI = 44,988–65,957; Matthews et al. 2017). Note that this estimate excludes the coast of Ontario, where ~14,800 belugas were estimated during the 2004 survey (Richard 2005).

3. Anthropogenic removals

Hunting statistics used to calculate the number of WHB beluga harvested annually from 1977 to 2015 are provided below. Table 1 includes only Western Hudson Bay communities whose harvests are made up entirely of WHB belugas. Table 2 includes total harvests in zones along eastern Hudson Bay and Hudson Strait that harvest a mix of belugas from different stocks, and Table 3 presents the estimated number of WHB belugas included in those mixed-stock harvests using the proportion of WHB whales in each zone's harvest (Mosnier et al. 2017). The final table includes estimates of annual WHB harvests, and also includes estimates for struck and lost. The average annual harvest of WHB belugas by communities around Hudson Bay and Hudson Strait (including Sanikiluaq) from 1977 to 2015 was 503 (range 252-784, including struck and lost; Hammill et al. 2017).

Table 1. Reported beluga harvests from communities on the west coast of Hudson Bay, Southhampton Island, and southeastern Baffin Island: Arviat, Baker Lake, Cape Dorset, Chesterfield Inlet (Chest. Inlet), Coral Harbour, Kimmirut, Rankin Inlet, Naujaat (Repulse Bay), Whale Cove, and Iqaluit. 100% of the harvest in these communities is assumed to comprise whales from the WHB beluga stock (nr = not reported; DFO harvest statistics, unpublished data).

Year	Arviat	Baker Lake	Cape Dorset	Chest. Inlet	Coral Harbour	Kimmirut	Rankin Inlet	Naujaat	Whale Cove	Iqaluit
1977	nr	nr	7	18	52	26	12	40	30	0
1978	nr	nr	21	3	24	3	30	0	37	5
1979	nr	nr	7	6	44	35	0	24	0	2
1980	nr	nr	43	11	62	12	14	7	8	18
1981	nr	nr	1	11	8	16	61	56	22	44
1982	nr	nr	3	3	33	4	37	34	6	22
1983	nr	nr	46	5	64	nr	33	18	8	nr
1984	nr	nr	nr	12	116	9	69	30	24	2
1985	nr	nr	21	28	76	9	36	3	19	19
1986	nr	nr	2	23	50	19	30	20	35	20
1987	nr	nr	9	34	29	34	30	30	30	36
1988	45	nr	10	15	38	9	27	47	16	44
1989	70	nr	18	20	67	28	40	20	27	40
1990	70	nr	39	20	67	21	40	20	27	2
1991	25	nr	37	20	125	28	20	13	25	11
1992	0	nr	36	nr	nr	20	nr	9	27	31
1993	23	nr	35	17	20	13	14	12	19	35
1994	32	nr	26	27	30	3	29	28	37	28
1995	3	nr	20	22	50	20	88	35	2	4
1996	100	nr	25	20	31	8	48	20	35	35
1997	100	nr	37	nr	30	4	48	nr	20	23
1998	9	nr	4	15	25	20	35	8	25	17
1999	58	nr	12	nr	50	19	nr	4	nr	70
2000	100	nr	28	1	38	27	45	10	20	22
2001	100	nr	13	25	25	16	35	10	40	45
2002	115	nr	0	18	17	38	130	18	60	35
2003	300	nr	7	20	20	20	25	5	25	28
2004	100	nr	nr	7	3	20	30	0	nr	27
2005	100	nr	21	nr	nr	7	100	3	40	50
2006	45	2	30	3	nr	25	60	50	10	64

2007	50	0	0	12	7	nr	38	21	10	33
2008	100	0	4	3	13	2	50	0	0	0
2009	nr	0	1	0	nr	nr	66	21	nr	66
2010	200	0	3	nr	nr	33	26	8	35	26
2011	100	0	8	25	20	17	62	1	45	18
2012	60	nr	0	29	0	14	26	nr	120	nr
2013	nr	0	15	0	12	0	1	10	50	84
2014	15	2	nr	8	60	17	nr	1	30	53
2015	100	2	0	15	100	22	nr	11	35	8

Table 2. Total number of belugas harvested annually (1977-2015) from zones along eastern Hudson Bay and Hudson Strait (Hammill et al. 2017), where harvests comprise a mix of beluga stocks.

Year	HSUB	SAN	SPRING	FALL	UBSP	UBFA	NEHBSP	NEHBFA
1977	501	14	0	0	0	0	0	0
1978	174	6	0	0	0	0	0	0
1979	224	0	0	0	0	0	0	0
1980	212	0	0	0	0	0	0	0
1981	236	0	0	0	0	0	0	0
1982	271	30	0	0	0	0	0	0
1983	227	7	0	0	0	0	0	0
1984	189	28	0	0	0	0	0	0
1985	166	5	0	0	0	0	0	0
1986	126	25	0	0	0	0	0	0
1987	125	28	0	0	0	0	0	0
1988	117	20	0	0	0	0	0	0
1989	284	19	0	0	0	0	0	0
1990	109	20	0	0	0	0	0	0
1991	178	22	0	0	0	0	0	0
1992	96	20	0	0	0	0	0	0
1993	189	10	0	0	0	0	0	0
1994	207	50	0	0	0	0	0	0
1995	221	30	0	0	0	0	0	0
1996	211	30	0	0	0	0	0	0
1997	239	19	0	0	0	0	0	0
1998	252	54	0	0	0	0	0	0
1999	238	32	0	0	0	0	0	0
2000	208	23	0	0	0	0	0	0
2001	241	27	0	0	66	0	0	0
2002	161	15	0	0	23	0	0	0
2003	168	80	0	0	26	0	0	0
2004	144	94	0	0	4	0	0	0
2005	172	53	0	0	5	0	0	0
2006	147	22	0	0	2	0	0	0

2007	165	35	0	0	6	0	0	0
2008	92	33	0	0	5	0	0	0
2009	0	34	68	70	6	0	0	0
2010	0	47	138	61	8	7	0	0
2011	0	32	115	86	0	17	0	0
2012	0	61	208	56	10	2	0	0
2013	0	76	150	90	8	0	0	0
2014	0	26	208	37	11	0	1	14
2015	0	170	106	94	28	3	0	30

Table 3. Estimated number of WHB belugas harvested annually (1977-2015) from zones along eastern Hudson Bay and Hudson Strait. Estimates were calculated from the proportion of WHB whales contributing to the total reported harvests in each zone as determined by genetics analysis (Mosnier et al. 2017): Hudson Strait-Ungava Bay (HSUB; 0.788, which is an average of Hudson Strait in spring and fall, and Ungava Bay in Spring), Sanikiluaq (SAN; 0.756, which is the value for the extended spring harvest, which represents 86% of the total harvest), Hudson Strait in spring (SPRING; 0.831), Hudson Strait in fall (FALL; 0.711), Ungava Bay in spring (UBSP; 0.823), Ungava Bay in Fall (UBFA; n/a, so spring value of 0.823 used), northeastern Hudson Bay spring (NEHBSP; n/a, so fall value of 0.598 used) and Northeast Hudson Bay fall (NEHBFA; 0.598).

Year	HSUB	SAN	SPRING	FALL	UBSP	UBFA	NEHBSP	NEHBFA
1977	395	11	0	0	0	0	0	0
1978	137	5	0	0	0	0	0	0
1979	177	0	0	0	0	0	0	0
1980	167	0	0	0	0	0	0	0
1981	186	0	0	0	0	0	0	0
1982	214	23	0	0	0	0	0	0
1983	179	5	0	0	0	0	0	0
1984	149	21	0	0	0	0	0	0
1985	131	4	0	0	0	0	0	0
1986	99	19	0	0	0	0	0	0
1987	99	21	0	0	0	0	0	0
1988	92	15	0	0	0	0	0	0
1989	224	14	0	0	0	0	0	0
1990	86	15	0	0	0	0	0	0
1991	140	17	0	0	0	0	0	0
1992	76	15	0	0	0	0	0	0
1993	149	8	0	0	0	0	0	0
1994	163	38	0	0	0	0	0	0
1995	174	23	0	0	0	0	0	0
1996	166	23	0	0	0	0	0	0
1997	188	14	0	0	0	0	0	0
1998	199	41	0	0	0	0	0	0
1999	188	24	0	0	0	0	0	0
2000	164	17	0	0	0	0	0	0

2001	190	20	0	0	54	0	0	0
2002	127	11	0	0	19	0	0	0
2003	132	60	0	0	21	0	0	0
2004	113	71	0	0	3	0	0	0
2005	136	40	0	0	4	0	0	0
2006	116	17	0	0	2	0	0	0
2007	130	26	0	0	5	0	0	0
2008	72	25	0	0	4	0	0	0
2009	0	26	57	50	5	0	0	0
2010	0	36	115	43	7	6	0	0
2011	0	24	96	61	0	14	0	0
2012	0	46	173	40	8	2	0	0
2013	0	57	125	64	7	0	0	0
2014	0	20	173	26	9	0	1	8
2015	0	129	88	67	23	2	0	18

Table 4. Estimated annual WHB harvests (sum of harvests along western Hudson Bay and estimated harvests along eastern Hudson Bay and Hudson Strait) from 1977-2015, incorporating struck and lost. Harvests are calculated including and excluding numbers from Sanikiluaq, reflecting uncertainty in stock identify of those whales (DFO, unpublished data).

Year	Estimated total harvest (including Sanikiluaq)	Estimated total harvest (including Sanikiluaq + S&L (LRC of 1.18; Richard 2008)	Estimated total harvest (excluding Sanikiluaq)	Estimated total harvest (excluding Sanikiluaq + S&L (LRC of 1.18; Richard 2008)
1977	590	697	580	684
1978	265	312	260	307
1979	295	348	295	348
1980	342	404	342	404
1981	405	478	405	478
1982	378	446	356	420
1983	358	423	353	416
1984	432	510	411	485
1985	346	408	342	403
1986	317	374	298	352
1987	352	415	331	390
1988	358	423	343	405
1989	568	670	554	653
1990	407	480	392	462
1991	461	544	444	524
1992	214	252	199	234
1993	344	407	337	398
1994	441	520	403	476
1995	441	520	418	493
1996	511	603	488	576

1997	465	548	450	531
1998	397	469	357	421
1999	425	501	401	473
2000	472	557	455	537
2001	574	677	553	653
2002	588	694	577	681
2003	664	784	604	712
2004	375	442	304	358
2005	501	591	461	544
2006	423	499	406	480
2007	332	392	306	361
2008	274	323	249	293
2009	291	343	265	313
2010	537	634	501	592
2011	491	579	467	551
2012	518	611	472	556
2013	425	501	367	433
2014	423	499	403	476
2015	620	731	491	580

4. Population trajectory

The WHB beluga stock has been surveyed just three times over the past several decades (1987, 2004, and 2015). In addition to the small number of surveys, direct comparison of abundance estimates among surveys to assess population trajectory is complicated by different survey coverage, as well as application of different availability bias correction factors (Richard et al. 1990, Richard 2005, Matthews et al. 2017).

Uncorrected surface counts from the 2004 aerial survey (27,200; Richard 2005) were similar to those from the 1987 survey (25,100; Richard et al. 1990), suggesting that WHB beluga stock abundance had not changed during the interim period (note the area covered by both surveys is not directly comparable). The five strata surveyed in 2015, however, were surveyed in 2004 with similar coverage and effort (Richard 2005), and the near-surface abundance estimates (i.e., not corrected for availability bias) for the same five strata was 43 256 (CV = 0.14) in 2015 and 40 989 (CV = 0.31) in 2004 (Table 4).

Table 4. Comparison of surface abundance estimates (not corrected for availability bias) for the five strata surveyed in 2015 and 2004. 2015 data are from Matthews et al. 2017 (Table 3), and 2004 data can be found in Richard (2005; Table 2).

Stratum	**2015 Surface Abundance (CV)**	**2004 Surface Abundance (CV)**
Churchill High Density	6,352 (30.7)	12,027 (96.0)
Nelson River	18,748 (22.6)	17,544 (28.2)
West Coast Low Density	52 (98.2)	1,753 (79.9)
Churchill Photographic	1,855 (50.0)	2,076 (40.6)
Seal Photographic	16,249 (n/a)	7,589 (17.3)
TOTAL	**43,256 (14.3)**	**40,989 (30.1)**

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR; Wade 1998) is calculated as

$$PBR = N_{min} * 0.5 * R_{max} * F_R$$

where N_{min} is the estimated population size using the 20-percentile of the lognormal distribution (N/[exp(z20*sqrt[ln(1+CV^2)])]), R_{max} is the maximum rate of population increase (unknown for belugas and assumed to be 0.04, the default for cetaceans), and F_R is a recovery factor that varies between 0.1 and 1.

PBR is converted to a total allowable landed catch (TALC) by accounting for the number of animals killed and not recovered (struck and lost) using the following:

$$TALC = PBR/ LRC$$

where LRC is the hunting loss rate correction and is equal to 1.18 ± 0.07 based on reported beluga harvest statistics from three eastern Canadian Arctic communities (Richard 2008).

PBR estimates for the WHB beluga stock using the most recent abundance estimate of 54,473 (CV = 0.098; Matthews et al. 2017) are 1,004, 753, 502 and 251, for recovery factors of 1, 0.75, 0.5 and 0.25 respectively (Hammill et al. 2017). Corresponding TALC values using a LRC of 1.18 (Richard 2008) are 851, 638, 425, and 213. Given the WHB stock is considered healthy, a recovery factor of 1 is appropriate (Wade and Angliss 1997), which corresponds to a PBR and TALC of 1,004 and 851, respectively.

6. Habitat and other concerns

The WHB beluga stock is not well-studied across some parts of its distribution. For example, relatively few samples from belugas off the coast of northern Ontario exist for genetics and other types of studies that could provide information on stock delineation.

7. Status of the stock

The WHB beluga stock is large and similar near-surface counts during surveys conducted in 2004 and 2015 indicate the stock is stable. Annual harvests of belugas from this stock are below estimated PBR (Hammill et al. 2017).

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Annex 12: James Bay Beluga Stock

By: Hammill, M.O., A. Mosnier, and J-F Gosselin

1. Distribution and stock identity

Based on observations of reoccurring aggregations in particular estuaries, Reeves and Mitchell (1987) outlined a management framework where the summering aggregations formed separate management stocks. Since then, photo-identification (Caron and Smith 1990), genetic and contaminant studies (Brennin et al. 1997, Brown Gladden et al. 1999, de March et al. 2004; Turgeon et al. 2011) have provided evidence that individual beluga return every year to the same aggregation areas. Moreover, telemetry studies in Nunavut (Richard et al. 2001, Richard and Stewart 2009) and northern Quebec (Nunavik) (Lewis et al. 2009) have shown that tracked individuals from specific summering aggregations within the summer season did not overlap in distribution. This cumulative evidence provides additional support for the concept of discrete summer stocks (Smith and Hammill 1986) and has led to the current use of summering stocks as management units (Fig. 1)(e.g., Richard 2010).

Large numbers of belugas have been observed in James Bay (Gosselin et al. 2017).

Over three years of tagging (2007-2009), 12 tagged whales showed no movement out of James Bay during the winter (Bailleul et al. 2012). Genetic analyses comparing these whales, along with other samples from James Bay to samples from adjacent locations in western Hudson Bay, eastern Hudson Bay and the Belcher Islands, confirmed that belugas in James Bay form a distinct stock from other management stocks in Hudson Bay (Postma et al. 2012). However, the differentiation between James Bay beluga and the other stocks is weak suggesting the presence of a local breeding population that has recently diverged. The combination of the satellite telemetry and genetic studies indicate that the James Bay beluga population should be considered a separate stock for surveys, population estimates and management (Postma et al. 2012).

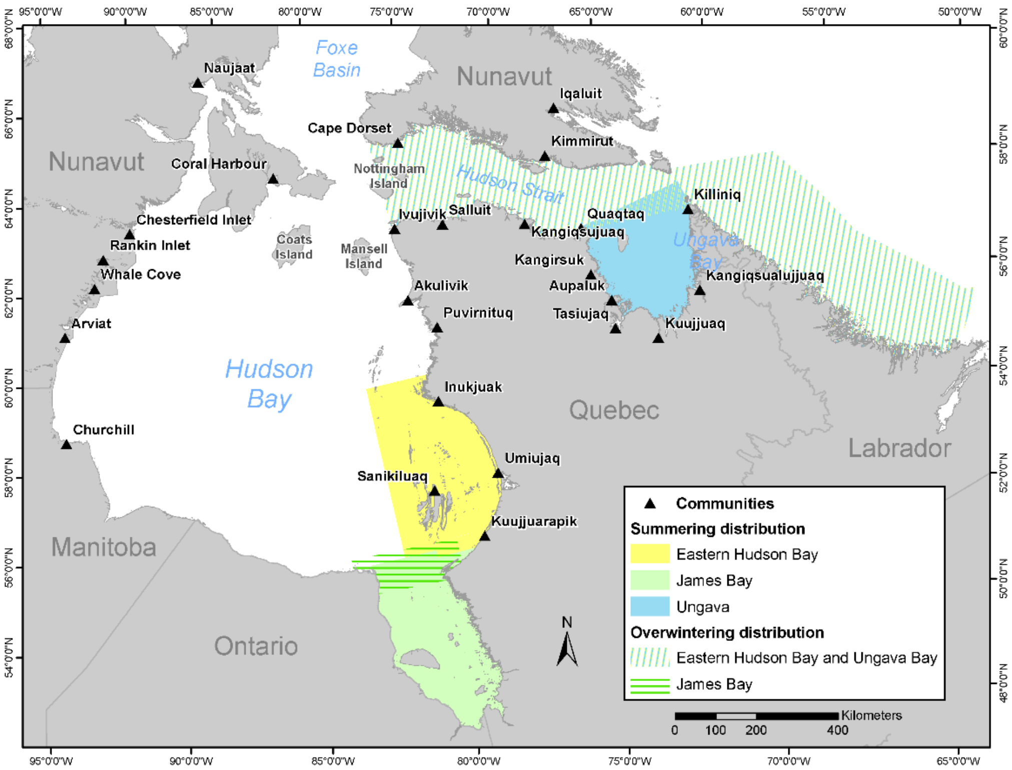

Figure 1. Summering aggregation and overwintering areas of the Eastern Hudson Bay, James Bay and Ungava Bay beluga stocks.

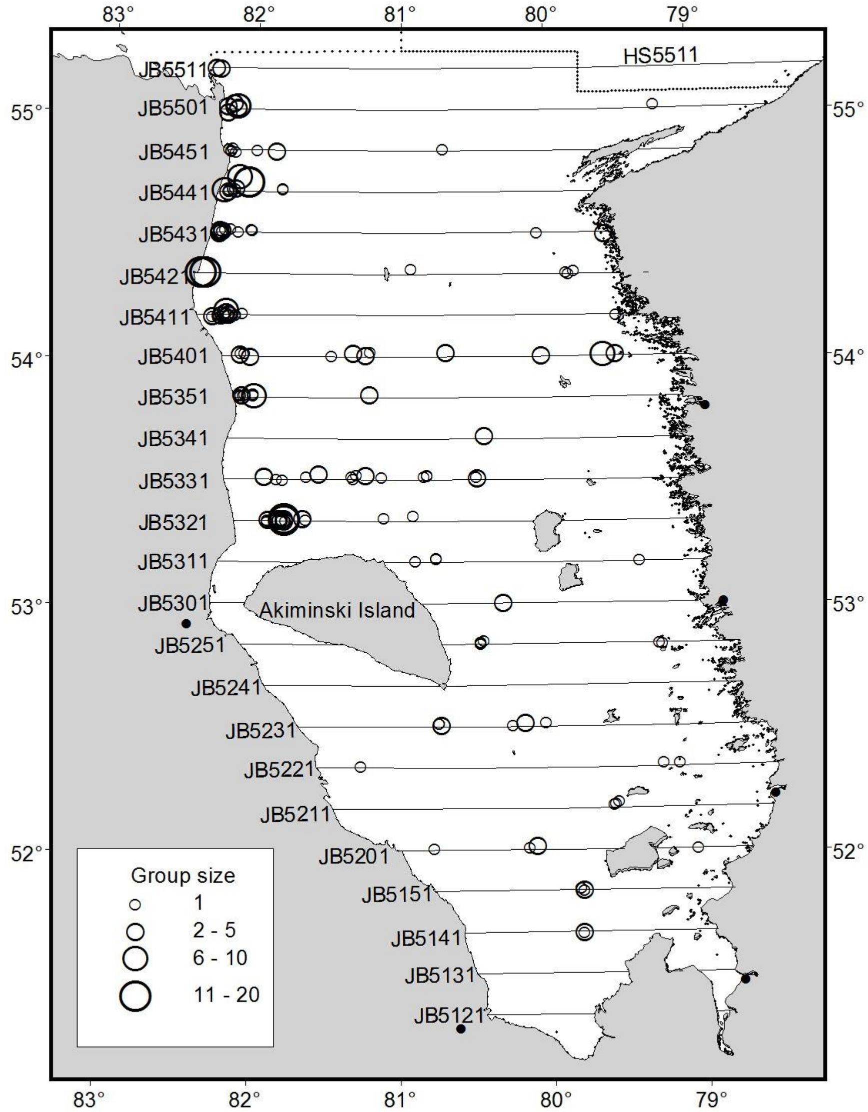

Figure 2. Area of extent of James Bay belugas. Summer core-use area is in James Bay. Figure shows transects flown during summer of 2015 and observations from survey (Gosselin et al. 2017). Animals may overwinter in the northern portion of James Bay or in the shifting pack ice between the Belcher Islands and James Bay.

2. Abundance

Aerial surveys

A total of seven visual systematic aerial surveys have been flown along the same transect lines since 1985. The most recent was flown in 2015 (Fig. 2; Table 1; Gosselin et al. 2017). All surveys have been flown along the same transect lines, but collection of data followed a strip-transect protocol in 1985 while line-transect methods were used for the others (Smith and Hammill 1986; Hammill et al. 2004; Gosselin et al. 2017). A comparative analysis conducted in 2004 allowed adjusting the 1985 survey estimates to make it more comparable to line-transect estimates (Hammill et al 2004). Overall, the number of animals in James Bay appears to have increased, but the rate of increase appears to be too higher than what would be expected from natural growth alone (Hammill unpublished data). In some years, larger numbers of animals are observed in the northwestern region of James Bay and in these years population estimates appear to increase markedly e.g. 2001, and 2008 (Table 1). Beluga are also seen along the Ontario coast of Hudson Bay, but the stock relationships between these animals and other beluga stocks in Hudson bay are not known. In summer there may be some movement of animals between the Ontario coast of Hudson Bay and the northwestern portion of James Bay.

Table 1. Abundance estimates of beluga populations in James Bay obtained from seven systematic aerial surveys. Abundance estimates have been corrected for availability bias and beluga counted in estuaries, but not for perception bias (Kingsley and Gauthier 2002). The 1985 survey data were collected using strip-transect techniques (Smith and Hammill 1986). The other five surveys flew along the same lines as the 1985 surveys, but data were collected using line-transect techniques (Gosselin et al. 2017).

Stratum	Year	Abundance	CV
James Bay	1985	4,720	0.13
	1993	8,205	0.24
	2001	17,285	0.24
	2004	8,364	0.30
	2008	19,439	0.66
	2011	14,967	0.30
	2015	10,615	0.25

3. Anthropogenic removals

In the 19th Century, the Hudson Bay company attempted to develop commercial whaling operations in James Bay, but these were of limited success and efforts were abandoned after only a few years (Reeves and Michell 1989). In James Bay, the Cree have hunted beluga in the past, but this activity was limited (Reeves and Mitchell 1989).

More recent hunting statistics (Table 2) are collected weekly in each village in Nunavik by community wardens, who transmit the data to a coordinator working with the Kativik Regional government in Kuujjuaq. The coordinator collates the data then distributes the weekly harvest information back to the communities as well as to the Department of Fisheries and Oceans (DFO) and other stakeholders. Different management plans have attempted to limit harvesting of the adjacent eastern Hudson Bay stock, and it has been suggested to hunters that beluga could be harvested from James Bay. However, prior to the 2000s harvesting was minimal in the James Bay area because of the distance to travel between James Bay and the nearest Inuit community of Kuujjuarapik (Fig. 1).

4. Population trajectory

Trend of this stock has not been examined in the past because of concerns about the influx of animals from the Ontario coast in certain years.

Fitting a discrete time parameterisation of the Pella and Tomlinson model (1969) to the aerial survey estimates using Bayesian methods and taking into account removals results in a 2015 population estimate of 14,500 (95% credibility intervals= 9500-21,400), a maximum rate of increase of 0.038 (95% C.I. = 0.004-0.059) and carrying capacity (K) of 23,100 (95% C.I.=10,300-39,000; Fig. 4). Although the model suggests an increase of the population size over the last 30 years, the variable movement of animals from the Ontario coast into the James Bay area at the time of aerial surveys flown in August may preclude a correct estimation of the population size Note that the 2001 and 2008 survey estimates are above the 95% Credibility Interval, although the survey 95% confidence limits overlap with the latter.

Table 2. Reported removals from James Bay by hunting (2001-2016). No data are available for animals struck and lost nor for non-reporting. Harvesting occurs near Long Island, which is located at the northern entrance to James Bay (Fig. 1).

Year	Reported harvest	Year	Reported harvest
2001	15	2009	9
2002	8	2010	10
2003	10	2011	6
2004	17	2012	11
2005	13	2013	10
2006	10	2014	5
2007	8	2015	6
2008	14	2016	2

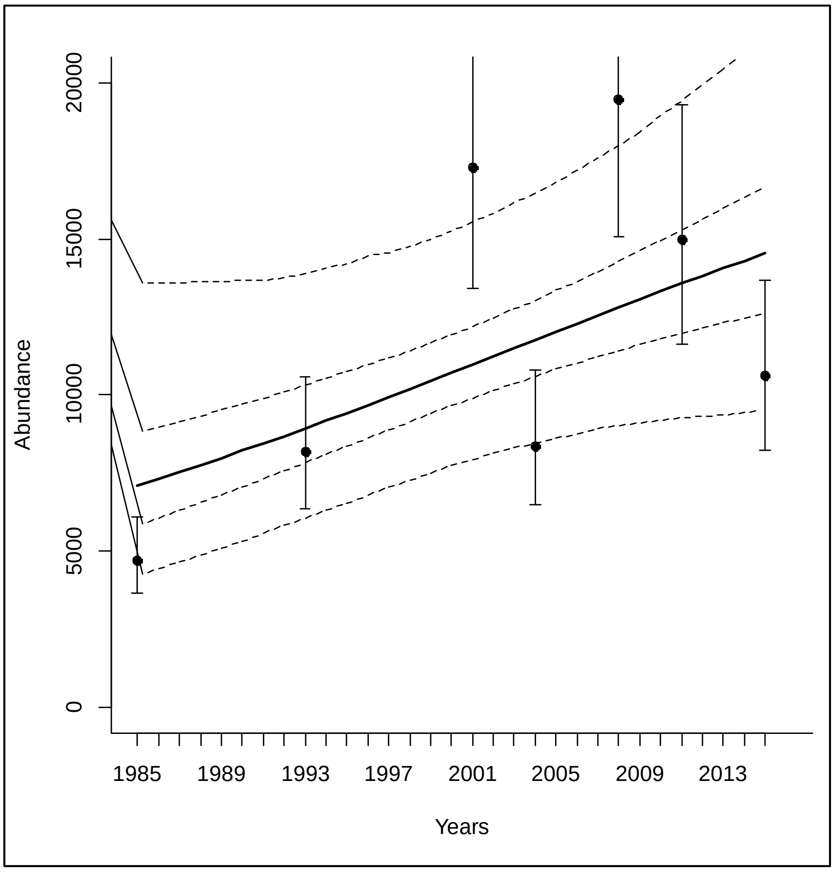

Figure 4. Estimated trajectory (median) of the James Bay beluga stock obtained by fitting a population model to aerial survey estimates of abundance and taking into account harvest removals. Outer dotted lines represent the 95% Credibility Intervals, the inner dotted lines represent the 25th and 75th percentiles. Points represent the aerial survey estimates adjusted for availability bias (±95% confidence intervals).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR; Wade 1998) is calculated as

$$PBR = N_{min} * 0.5 * R_{max} * F_R$$

where N_{min} is the estimated population size using the 20-percentile of the lognormal distribution ($N/[\exp(z_{20}*\mathrm{sqrt}[\ln(1+CV^2)])]$), R_{max} is the maximum rate of population increase (unknown for belugas and assumed to be 0.04, the default for cetaceans), and F_R is a recovery factor that varies between 0.1 and 1.

PBR estimates for the James Bay beluga stock using the most recent aerial survey abundance estimate of 10,615 (CV = 0.25; Gosselin et al. 2017) are 173, 129, 86, and 17, for F_R of 1, 0.75, 0.5 and 0.1 respectively.

6. Habitat and other concerns

The James Bay beluga stock is not well-studied. The limited telemetry work indicates that animals overwinter within the James Bay region. Hunters living on the Belcher Islands (Nunavut) have observed beluga in the pack ice to the south of the islands, suggesting that animals may move northwards in winter to the region between the Belcher Islands and James Bay. Hydroelectric development during the 1970s has altered the hydrological cycle, such that the major period of freshwater discharge has shifted from the fall to the winter, as water is held back in summer then released during winter. The impact on beluga is not known.

7. Status of the stock

The James Bay stock of beluga is large and has been exposed to limited harvesting. Changes may have occurred in habitat over the last 45 years, but these have not been examined in detail. The status of this stock has not been assessed by the Committee on Endangered Species of Wildlife in Canada (COSEWIC)

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Annex 13: Eastern Hudson Bay Beluga Stock

By: Hammill, M.O., A. Mosnier, and J-F Gosselin

1. Distribution and stock identity

Based on observations of reoccurring aggregations in particular estuaries, Reeves and Mitchell (1987) outlined a management framework where the summering aggregations formed separate management stocks. Since then, photo-identification (Caron and Smith 1990), genetic and contaminant studies (Brennin et al. 1997, Brown Gladden et al. 1999, de March et al. 2004) have provided evidence that individual beluga return every year to the same aggregation areas. Moreover, telemetry studies in Nunavut (Richard et al. 2001, Richard and Stewart 2009) and northern Quebec (Nunavik) (Lewis et al. 2009) have shown that tracked individuals from specific summering aggregations within the summer season did not overlap in distribution. This cumulative evidence provides additional support for the concept of discrete summer stocks (Smith and Hammill 1986) and has led to the current use of summering stocks as management units (Fig. 1; e.g., Richard 2010).

In summer, the eastern Hudson Bay (EHB) stock occupies an area bounded in the east by the eastern Hudson Bay arc. In the north, just to the north of the village of Inukjuak (approximately 59° 03' N and in the south by an east-west line running approximately midway between the village of Kuujjuarapik and the top of Long Island, at the entrance to James Bay.(55 °, 11' N). In an east-west direction, the EHB stock includes an area running from the EHB coast westwards to 60 km west of the Belcher Islands (81° W longitude; Fig. 1,2).

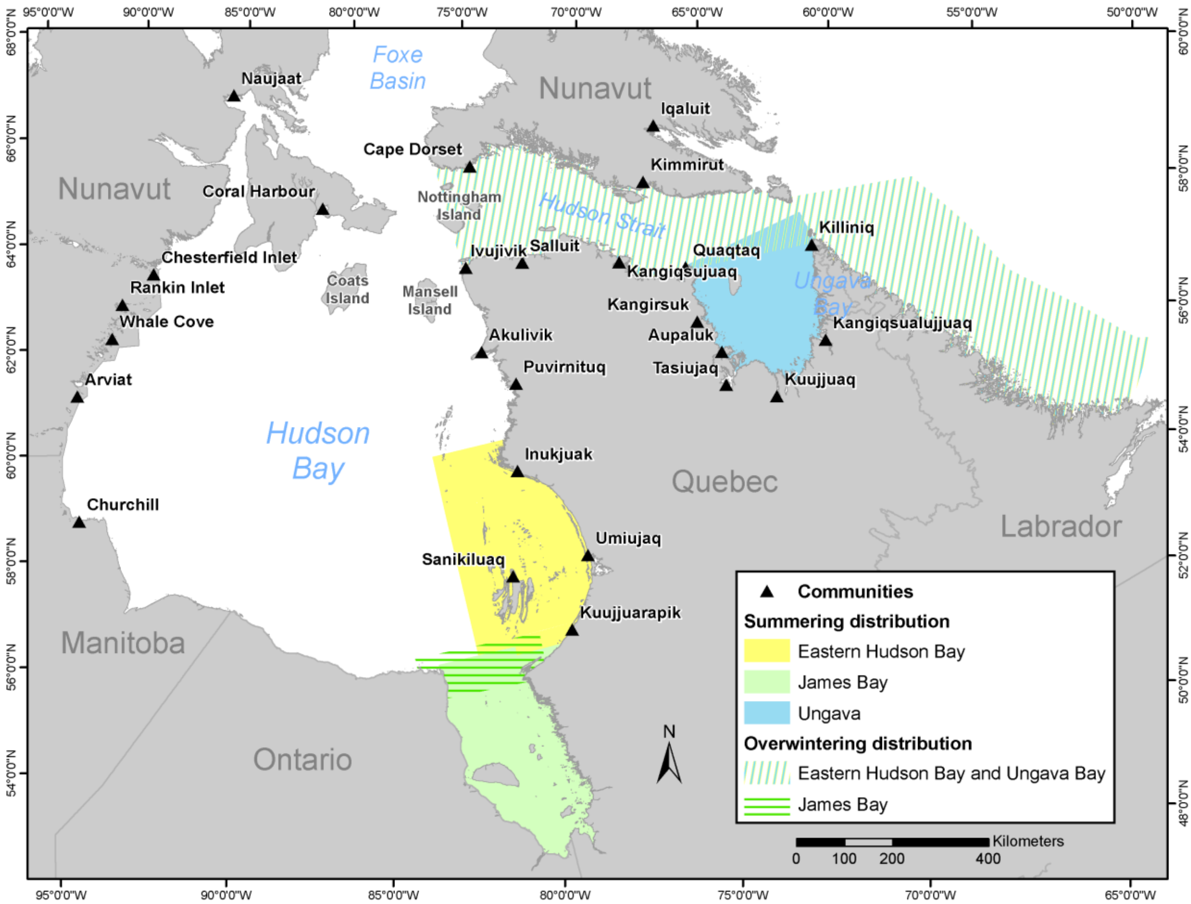


Figure 1. Summering aggregation and overwintering areas of the Eastern Hudson Bay, James Bay and Ungava Bay beluga stocks

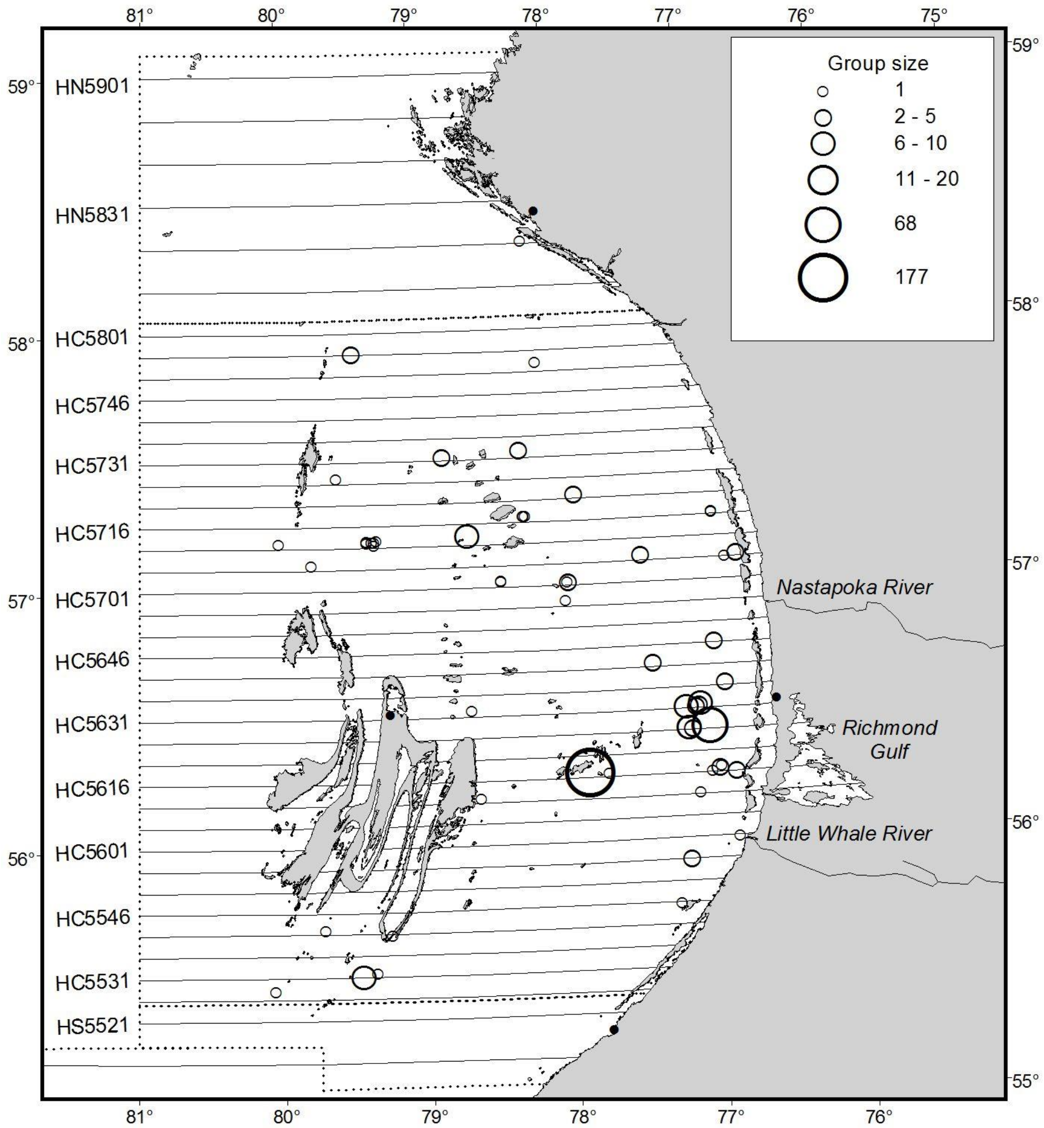


Figure 2. Area of summer extent of Eastern Hudson Bay (EHB) belugas. Figure shows transects flown during summer of 2015 and observations from survey (Gosselin et al. 2017).

Skin samples have been obtained from harvested animals from the Nastapoka and Little Whale River estuaries. Genetic analyses have shown that these animals belong to the same breeding population as the Western Hudson Bay stock (WHB). However, despite interbreeding on wintering grounds (Turgeon et al. 2012), cultural conservatism of maternally-transmitted migration routes seems to prevent substantial exchange between these summering aggregations (Colbeck et al. 2012), providing support for the summer aggregation stock hypothesis, but also making beluga vulnerable to local extirpation (COSEWIC 2004). Information on movements from satellite telemetry provided further support, with summering beluga moving between the coast and areas offshore near the Belcher Islands, but not mixing with WHB animals at this time (Lewis et al. 2009; Bailleul et al. 2012?). In fall, EHB belugas migrate to Hudson Strait and overwinter in the eastern portion of the Strait, near the entrance to Ungava Bay as well as just outside the entrance to eastern Hudson Strait, along the east coast of Labrador (Lewis et al. 2009). This fall migration is characterized by animals remaining along the coast of eastern Hudson Bay and along the southern coast of Hudson Strait (Lewis et al. 2009; Hammill 2013). Little is known about the spring migration. Belugas are seen by harvesters in Hudson Strait during the spring, and animals appear to reach their summering areas by late July. Over the last decade, observations provided by hunters indicate that animals are beginning the spring migration about 7-10 days earlier, and their fall migration 7-10 days later, probably in response to earlier breakup and later freeze-up (Hammill 2013).

2. Abundance

A total of seven visual systematic aerial surveys have been flown to evaluate EHB abundance since 1985. The most recent was flown in 2015 (Fig. 2; Table 1; Gosselin et al. 2017). All surveys have been flown along the same transect lines, but collection of data followed a strip-transect protocol in 1985 while line-transect methods were used for the others (Smith and Hammill 1986; Hammill et al. 2004; Gosselin et al. 2017). A comparative analysis conducted in 2004 allowed adjusting the 1985 survey estimates to make it more comparable to line-transect estimates (Hammill et al 2004). From the surveys, the current abundance estimate is 3,819 animals. Overall, the number of animals in EHB appears to have remained stable.

Table 1. Abundance estimates of beluga populations in Eastern Hudson Bay obtained from seven systematic aerial surveys. Abundance estimates have been corrected for availability bias and beluga counted in estuaries, but not for perception bias (Kingsley and Gauthier 2002). The 1985 survey data were collected using strip-transect techniques (Smith and Hammill 1986) and adjusted to make them comparable to the later surveys (Hammill et al. 2004). The other six surveys flew along the same lines as the 1985 surveys, but data were collected using line-transect techniques (Gosselin et al. 2017).

Stratum	Year	Abundance	CV
Eastern Hudson Bay	1985	4,282	0.13
	1993	2,729	0.40
	2001	2,924	0.48
	2004	4,274	0.37
	2008	2,646	0.47
	2011	3,351	0.49
	2015	3,819	0.43

3. Anthropogenic removals

Commercial harvests in the 19th century initiated the depletion of beluga stocks in eastern Hudson Bay (Reeves and Mitchell 1989). Subsequent subsistence harvests may have limited the opportunity for stocks to recover. In the 1980's, limits were placed on harvesting through a combination of Total Allowable Takes (TAT) and seasonal closures at the Nastapoka and Little Whale rivers. Harvesting in the EHB area was closed from 2001 to 2006, and the Nastapoka and Little Whale River estuaries have remained closed since harvesting resumed in EHB in 2007.

Inuit in northern Quebec (Nunavik) harvest beluga whales belonging to two or more stocks. In summer, hunters from villages in the EHB arc harvest animals belonging to the EHB stock. However, during fall, winter and spring, hunters living in communities in Hudson Strait harvest animals from the small EHB stock (~3,800 belugas) and the much larger WHB stock (~54,500 belugas), as these animals overlap together in the Hudson Strait area at these times of year (DFO unpublished data).

Also, during summer, EHB animals are harvested by hunters living in the community of Sanikilluaq, on the Belcher Islands (Nunavut). Contrary to the regulation in Nunavik, there are no TAT restrictions in Sanikilluaq. However, since 2010, the community has established a community bylaw to close harvesting of belugas between July 1 and September 30, to reduce the probability of harvesting EHB animals. In 2012, the start of the voluntary harvest closure start date was changed to July 15.

Sampling program

A tissue sampling program was initiated in the early 1980s, but has only been operating on a regular basis in Nunavik since the mid-1990s. Hunters from all 14 Nunavik communities provide a tooth, skin samples and information on where animals are harvested. Participation rates vary, but samples are generally obtained from around 30% of the reported catch. There is a slight overrepresentation of male vs female beluga in the harvest, but there have been no significant trends in the sex ratio of the harvest since 1984 (Hammill et al. 2017). The mean age of belugas in the catch (1984-2015) was 18.5 and 23.6 years old for EHB_{type} and Not_EHB_{type} belugas, respectively. No significant time trend was observed in the mean age of the harvest.

Genetic mixture analysis
The hunters in Nunavik, particularly hunters living in villages in Hudson Strait are harvesting animals that belong to the EHB stock and the WHB stock during the spring/fall migration and over winter. Hunters living in Sanikilluaq (Nunavut), also appear to be harvesting animals from mixed stocks (Mosnier et al. 2017). A genetic mixture analysis is used to estimate the proportion of individuals belonging to the different source stocks (i.e. distinct summer stocks) in the composition of the population hunted in different areas and periods, using the tissue samples obtained between 1982 and 2015. Two source stocks were defined by samples taken in July and August, in summering areas of WHB and EHB. The dates and locations for each of the regional harvests were aligned with the definitions of hunting areas and seasons that have been in use since 2014 to manage the Nunavik beluga hunt.

Mixed hunt areas in Nunavik coastal waters that occur along the common seasonal migratory corridor include: northeastern Hudson Bay (NEHB), southern Hudson Strait (HS), and Ungava Bay (UNG; Fig. 3). The Sanikiluaq (SAN) area encompasses hunting zones located around the Belcher Islands (Nunavut). For NEHB, HS and UNG, samples were pooled into two hunting seasons: a “spring” hunt from February 1 to August 31, and a “fall” hunt from September 1 to January 31 (Fig. 3)

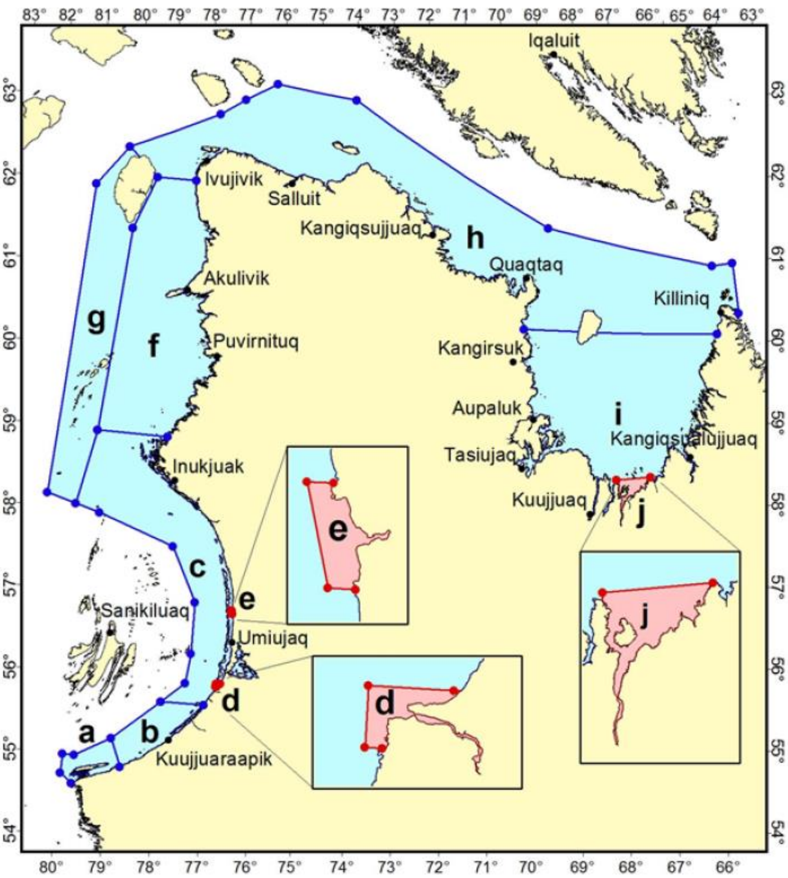


Figure 3. Map of Nunavik communities and management areas for Nunavik beluga (source: Nunavik Marine Region Wildlife Board). In our analysis, the Hudson Strait mixed hunt corresponds to area “h”, Ungava Bay to area “i”,northeast Hudson Bay to area “f”, and the eastern Hudson Bay arc to areas “b”, “c”, “d” and “e”.

Table 2. Estimates of the proportions of beluga (%) from each source stock in the harvest of Nunavik hunt areas (upper part) and Sanikiluaq harvest (lower part) (1982-2015) (Mosnier et al. 2017). N_{sample} is the total number of samples. Animals collected on the same day, may belong to the same group (N_{events}), Eastern Hudson Bay (EHB), Western Hudson Bay (WHB), 95% Confidence interval (95%CI), coefficient of variation (CV samples/events) and percent of the samples not belonging to the two source groups (% Unknown).

	N sample	N events	% WHB	95% CI	(CV samples / events)	% EHB	95% CI	(CV samples / events)	% Unknown
Spring (Feb 1 - Aug 31)									
Hudson Strait	611	278	83.1	78.3 - 87.4	0.02/0.03	10.8	7.1 - 15.2	0.18/0.19	6.1
NE Hudson	2	1	ND			ND	-	-	-
Ungava Bay	75	49	82.3	68.1 – 92.9	0.06/0.08	8.4	0.9 - 23	0.60/0.70	9.3
Fall (Sept 1 - Jan 31)									
Hudson Strait	352	146	71.1	63.4 - 78.1	0.04/0.05	26.1	19.3 - 33.6	0.12/0.14	2.8
NE Hudson	20	8	59.8	31.1 - 85.2	0.21/0.24	30.2	12.1 - 52.3	0.40/0.35	10.0
Ungava Bay	3	3	ND	-	-	ND	-	-	ND

Sanikiluaq

Season	N sample	N events	% WHB	95% CI	(CV samples / events)	% EHB	95% CI	(CV samples / events)	% Unknown
Spring (April 1 - June 30)	297	107	77.3	70.0 - 83.9	0.02/ 0.05	1.5	0.0 - 5.7	1.07/ 1.08	21.2
Extended spring (April 1 - July 14)	320	120	75.6	67.9 - 82.5	0.03/ 0.05	4.4	1.1 - 9.9	0.43/ 0.52	20.0
Summer (July 1 - August 31)	31	18	61.5	33.6 - 85.7	0.16/ 0.22	25.6	5.2 - 55.1	0.37/ 0.51	12.9
Fall (September 1 - November 30)	42	28	97.6	91.3 - 99.9	0.00/ 0.02	0.0	-	-	2.4
Winter (December 1 - March 31)	56	7	31.3	7.4 – 63.0	0.24/ 0.47	36.6	10.5 - 68.2	0.21/ 0.41	32.1

For Sanikiluaq, there are two definitions of the spring hunt: one ("Spring") bounded by the voluntary closure date in place between 2010 and 2012 (i.e., April 1 to June 30) and the other ("Extended Spring") using the post-2012 closure date (i.e., April 1 to July 14).

Some belugas overwinter in the region between the entrance to James Bay and the Belcher Islands (Lewis et al. 2009), but the stock relationships of these animals to the EHB. WHB and James Bay beluga are not clear. Individual belugas are observed from land in late May, but most EHB beluga arrive in eastern Hudson Bay around June-July. Their spring migration route has not been documented but genetic analyses suggest that 8.4% and 10.8% of the whales harvested respectively in the Ungava Bay and the Hudson Strait hunting area during this season belong to the EHB stock. Their proportion in the Ungava Bay harvest declines to 3.1% in summer.

For beluga harvested near Sanikiluaq, EHB beluga represents 1.5% of the harvest in spring (April 1 – June 30), increasing to 4.4% if the spring period was extended to July 14. This proportion increases to 25.6% in summer.

Harvest statistics are available for the years 1974-2016. These statistics represent minimum estimates only, since not all villages provided catch data in all years, and information on the number of animals struck and lost is incomplete. Considering that the proportion of EHB (vs WHB) provided by the genetic mixture analysis is stable over time, we can estimate the number of EHB individual harvested during the whole period (Table 3 and 4). During 1974–2016, an average 119 (SE=14, N=43) EHB whales per year was reported by Nunavik communities. During the recent management plan, the average harvest was 60 (SE=17, N=17) EHB belugas over the last three years (Fig. 4; Tables 3 and4).

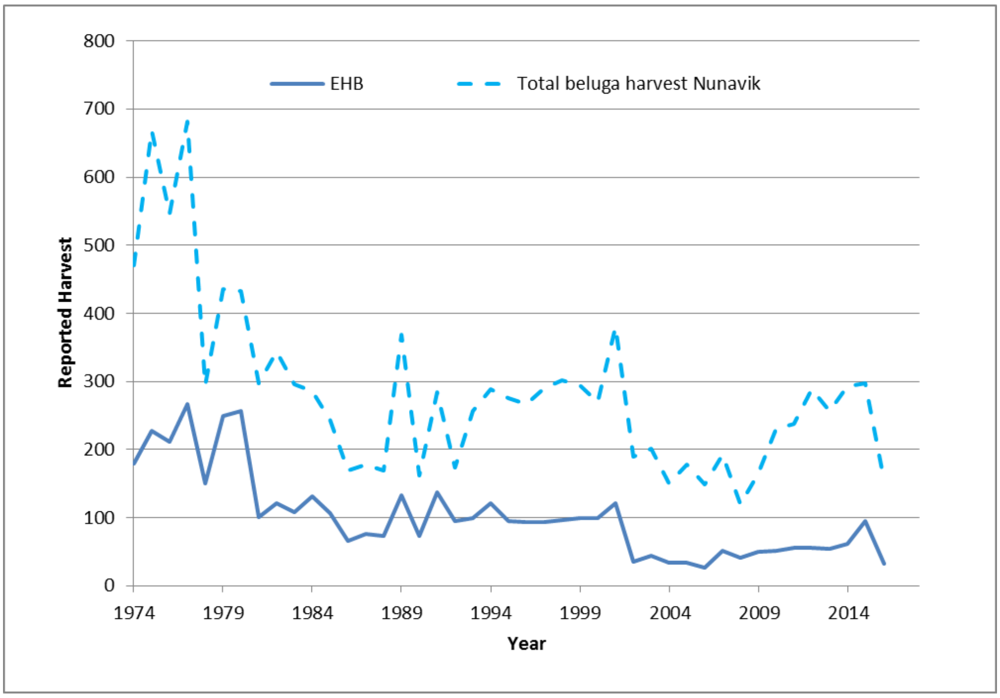


Figure 4. Total reported beluga harvest by hunters in Nunavik, which includes animals from the Eastern Hudson Bay (EHB) and Western Hudson Bay (WHB) stocks; and total reported harvest of EHB belugas only, by hunters from Nunavut and Nunavik.

Table 3. Total number of belugas reported landed by Nunavik communities and the community of Sanikiluaq (Nunavut). Includes animals from all management stocks. The reporting zones are the eastern Hudson Bay arc (ARC), Hudson Strait/Ungava Bay (HSUB), Sanikilluaq (SAN), Hudson Strait spring hunt (SPRING), Hudson Strait fall hunt (FALL), Ungava Bay spring (UBSP), Ungava Bay fall (UBFA), northeastern Hudson Bay spring (NEHBSP), northeastern Hudson Bay fall (NEHBFA)

YEAR	ARC	HSUB	SAN	SPRING	FALL	UBSP	UBFA	NEHBSP	NEHBFA	Nunavik beluga total
1974	119	352	0	0	0	0	0	0	0	471
1975	137	532	0	0	0	0	0	0	0	669
1976	143	403	0	0	0	0	0	0	0	546
1977	181	501	14	0	0	0	0	0	0	682
1978	120	174	6	0	0	0	0	0	0	294
1979	211	224	0	0	0	0	0	0	0	435
1980	220	212	0	0	0	0	0	0	0	432
1981	61	236	6	0	0	0	0	0	0	297
1982	73	271	30	0	0	0	0	0	0	344
1983	69	227	7	0	0	0	0	0	0	296
1984	97	189	28	0	0	0	0	0	0	286
1985	78	166	5	0	0	0	0	0	0	244
1986	43	126	25	0	0	0	0	0	0	169
1987	53	125	28	0	0	0	0	0	0	178
1988	52	117	20	0	0	0	0	0	0	169
1989	84	284	19	0	0	0	0	0	0	368
1990	53	109	20	0	0	0	0	0	0	162
1991	106	178	22	0	0	0	0	0	0	284
1992	78	96	20	0	0	0	0	0	0	174
1993	67	189	10	0	0	0	0	0	0	256
1994	82	207	50	0	0	0	0	0	0	289
1995	55	221	30	0	0	0	0	0	0	276
1996	56	211	30	0	0	0	0	0	0	267

1997	51	239	19	0	0	0	0	0	0	290
1998	50	252	54	0	0	0	0	0	0	302
1999	57	238	32	0	0	0	0	0	0	295
2000	62	208	23	0	0	0	0	0	0	270
2001	73	241	27	0	0	66	0	0	0	380
2002	5	161	15	0	0	23	0	0	0	189
2003	8	168	80	0	0	26	0	0	0	202
2004	3	144	94	0	0	4	0	0	0	151
2005	1	172	53	0	0	5	0	0	0	178
2006	0	147	22	0	0	2	0	0	0	149
2007	21	165	24	0	0	6	0	0	0	192
2008	23	92	33	0	0	5	0	0	0	120
2009	21	0	34	68	70	6	0	0	0	165
2010	16	0	47	138	61	8	7	0	0	230
2011	19	0	32	115	86	0	17	0	0	237
2012	13	0	61	208	56	10	2	0	0	289
2013	8	0	76	150	90	8	0	0	0	256
2014	22	0	26	208	37	11	0	1	14	293
2015	36	0	170	106	94	28	3	0	30	297
2016	16	0	33	116	0	22	0	0	0	154

Table 4. Total number of EHB belugas reported landed by Nunavik communities and the community of Sanikiluaq (Nunavut). The total number of reported belugas landed have been adjusted to account for the proportion of EHB animals in the reported harvest using estimates from the genetic mixed model analysis (Table 2, Mosnier et al. 2017). The reporting zones are the eastern Hudson Bay arc (ARC), Hudson Strait/Ungava Bay (HSUB), Sanikilluaq (SAN), Hudson Strait spring hunt (SPRING), Hudson Strait fall hunt (FALL), Ungava Bay spring (UBSP), Ungava Bay fall (UBFA), northeastern Hudson Bay spring (NEHBSP), northeastern Hudson Bay fall (NEHBFA).

YEAR	ARC	HSUB	SAN	SPRING	FALL	UBSP	UBFA	NEHBSP	NEHBFA	#EHB whales harvested
1974	119	60	0	0	0	0	0	0	0	179
1975	137	90	0	0	0	0	0	0	0	227
1976	143	69	0	0	0	0	0	0	0	212
1977	181	85	1	0	0	0	0	0	0	267
1978	120	30	0	0	0	0	0	0	0	150
1979	211	38	0	0	0	0	0	0	0	249
1980	220	36	0	0	0	0	0	0	0	256
1981	61	41	0	0	0	0	0	0	0	102
1982	73	46	2	0	0	0	0	0	0	121
1983	69	39	0	0	0	0	0	0	0	108
1984	97	32	2	0	0	0	0	0	0	131
1985	78	29	0	0	0	0	0	0	0	107
1986	43	21	2	0	0	0	0	0	0	66
1987	53	21	2	0	0	0	0	0	0	76
1988	52	20	1	0	0	0	0	0	0	73
1989	84	48	1	0	0	0	0	0	0	134
1990	53	19	1	0	0	0	0	0	0	73
1991	106	30	2	0	0	0	0	0	0	138
1992	78	16	2	0	0	0	0	0	0	96
1993	67	32	1	0	0	0	0	0	0	100
1994	82	35	4	0	0	0	0	0	0	121

1995	55	38	2	0	0	0	0	0	0	95
1996	56	36	2	0	0	0	0	0	0	94
1997	51	41	1	0	0	0	0	0	0	93
1998	50	43	4	0	0	0	0	0	0	97
1999	57	40	3	0	0	0	0	0	0	100
2000	62	35	2	0	0	0	0	0	0	99
2001	73	41	2	0	0	5	0	0	0	121
2002	5	27	1	0	0	2	0	0	0	35
2003	8	29	6	0	0	2	0	0	0	45
2004	3	24	7	0	0	0	0	0	0	34
2005	1	29	4	0	0	0	0	0	0	34
2006	0	25	2	0	0	0	0	0	0	27
2007	21	28	2	0	0	0	0	0	0	51
2008	23	16	2	0	0	0	0	0	0	41
2009	21	0	2	8	18	1	0	0	0	50
2010	16	0	3	15,18	16	1	0	0	0	51
2011	19	0	2	13	22	0	0	0	0	56
2012	13	0	4	23	15	1	0	0	0	56
2013	8	0	5	17	23	1	0	0	0	54
2014	22	0	2	23	10	1	0	0	4	62
2015	36	0	12	12	24	2	0	0	9	95
2016	16	0	2	13	0	2	0	0	0	33

4. Population trajectory

Model

A stochastic stock production population model that included information on removals and the stock composition of the catch was fitted to aerial survey estimates of abundance from the Eastern Hudson Bay using Bayesian methods (Hammill et al. 2017). Density-dependent growth was modelled, using a discrete theta-logistic model (Pella and Tomlinson 1969):

$$N_t = N_{t-1} + N_{t-1} \cdot (\lambda_{max} - 1) \cdot \left[1 - (N_{t-1}/K)^{\theta}\right] \cdot \varepsilon_{p_t} - R_t \text{ , with } \varepsilon_{p_t} \sim \mathrm{logN}\left(0, \tau_p\right)$$

where K is environmental carrying capacity and theta (θ) defines the shape of the density-dependent function,

In both models, removals were calculated as

$$R_t = C_t \cdot (1 + SL)$$

Where reported catches, Ct = Landed catch * proportion of EHB animals in the catch * struck and loss (SL), i.e. the proportion of animals that were wounded or killed but not recovered.

The observation process describes the relationship between true population size and observed data. In our model, survey estimates S_t are linked to population size N_t by a multiplicative error term εst:

$$S_t = N_t \cdot \varepsilon_{s_t} \text{ , with } \varepsilon_{s_t} \sim \mathrm{logN}(0, \tau_s)$$

Priors

Existing information, traditional knowledge and expert opinions were used to formulate prior distributions for the random variables included in the model. Beginning with the EHB stock, the initial population size was given a uniform prior between 2000 and 15,000 individuals. The lower bound reflects observations of at least a few hundred beluga in the EHB estuaries, but recognizes that the population had been reduced considerably from pristine sizes (Smith and Hammill 1986; Reeves and Mitchell 1987). Doniol-Valcroze et al. (2012b), estimated a pristine population of around 8,000 (95% CI 7,200-8,700) assuming no losses during the commercial hunt. This estimate does not take into account the subsistence hunt, although compared to the commercial harvest its impact was likely to have been relatively small. For K, a range of 2,000 to 20,000 was used. The upper bound encompassed the possible range of estimates of pristine population size, including if loss rates were as high as 2 and would likely account for subsistence harvests at the time as well (Hammill et al. 2005, Doniol-Valcroze et al. 2012a). The maximum rate of population increase is not known. For the St Lawrence estuary beluga, Beland et al. (1988) using the age distribution of stranded carcasses, estimated a mean rate of increase of 0.049 (95% CL=0.038 to 0.061). Other studies have used maximum rates of increase of 6% (Hobbs et al 2006), 8% (Alvarez-Flores and Heide-Jørgensen 2004; Doniol-Valcroze et al. 2012a) and 10% (Innes and Stewart 2002). The high rates of increase are theoretical estimates that assume survival=1, a three year calving interval, and make additional assumptions concerning reproductive rates. We used a prior with uniform distribution with a range of -0.01 to 0.06. The lower bound allows for the possibility that the rate of growth might be negative in some years. An upper bound of 0.06 assumes an adult mortality rate of 0.97 (Hobbs et al. 2006). In the density-dependent model, the point at which a population attains Maximum Sustainable Yield is also uncertain. Marine mammals are generally considered to attain MSY levels at around 60% of K (Taylor and DeMaster 1993; Butterworth et al. 2002; Hobbs et al. 2006). Therefore, theta (θ) was set to 2.39 which results in maximum productivity at 60% of K (Hobbs et al. 2006).

Reported harvests underestimate the number of beluga killed because of animals wounded or killed but not recovered, as well as under-reporting. The struck and loss (SL) rates in Canadian hunts are not known exactly but are believed to range from around 20% for shallow water hunts up to 60% for deep-water hunting, e.g. along ice edges (Seaman & Burns 1981). Heide-Jørgensen and Rosing-Asvid (2002) calculated a SL factor of 0.29 for Greenland, not including unreported catches. Innes and Stewart (2002) estimated a correction factor that accounted for SL and whales not reported in Baffin Bay at 0.41 whales per whale landed. In Cook Inlet, SL has varied from 33-66% (Hobbs et al. 2006). Richard (2008) estimated SL rates of 18% (CV=13%, range 10-30%). We used a moderately informative prior following a Beta (3, 4) distribution, with a median of 0.42 and quartile points at 0.29 and 0.55, which was used in the previous assessment (Doniol-Valcroze et al. 2012a). These priors result in lower SL estimates than used in earlier assessments where the struck and lost was given a log-normal prior with a median of 0.61 and quartile points at 0.43 and 0.85 (Doniol-Valcroze et al. 2012a,b).

The stochastic process error terms εpt were given a log-normal distribution with a zero location parameter. The precision parameter for this lognormal distribution was assigned a moderately informative prior following a bounded gamma (1.5, 0.001) distribution. These parameters were chosen so that the resulting coefficients of variation (CV) would have quartiles of 5.5% and 8.7%, reflecting our belief that beluga stock dynamics are not highly variable.

Although estimates of uncertainty were available for each survey estimate, they were incorporated into the fitting process only by guiding the formulation of the prior distribution of the survey error. The survey error term εst followed a log-normal distribution with a zero location parameter. Its precision parameter was given a moderately informative prior following a gamma (2.5, 0.4) distribution. These parameters were chosen so that the resulting CV on the survey estimates would have quartiles of 35% and 55%, which are approximately equivalent to the range of actual CV for the survey abundance estimates.

The proportions of EHB beluga harvested in each zone are based on the genetic mixed model analysis (Mosnier et al. 2017). These proportions are incorporated into the model as probabilities. The genetic priors assumed a Beta distribution, with known mean and standard error, but for which the α and β parameters are not available. We solved the system of equations for the mean and variance of a Beta distribution to determine the values of α and β that describe the observed distributions. These Beta distributions were then used as priors for the proportions of EHB animals in the hunt at Sanikiluaq, Hudson Strait (HS) for all season (hunt prior to 2009) and HS for spring and fall (2009–2012), Ungava Bay, and northeastern Hudson Bay spring and fall (Table 4; Mosnier et al. 2017).

Model Results

The model is initiated in 1974, when a relatively continuous period of harvest data are available. Significant updating of priors was observed for the maximum rate of increase (lambda), the initial population in 1974, and estimated environmental carrying capacity (K). Little change was observed between the prior and posterior distribution for Struck and Lost (SL) (Table 4, fig. 4).

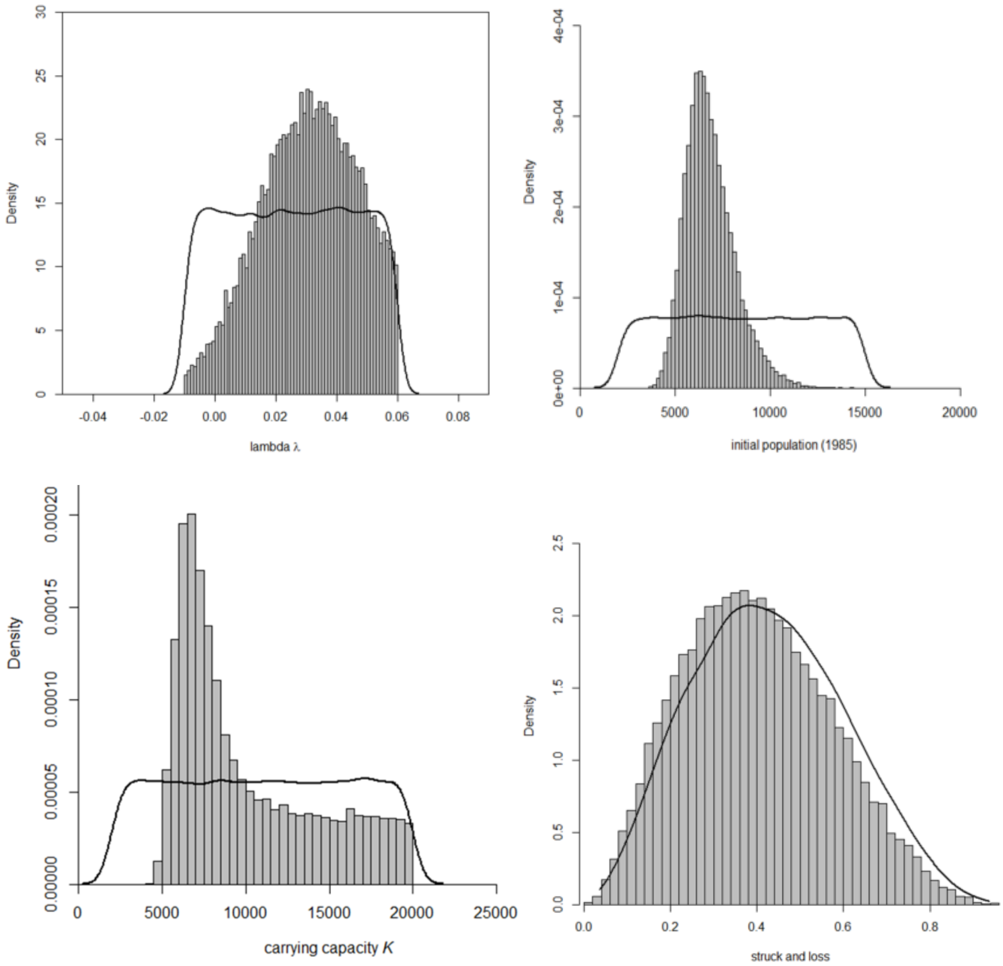
The model estimates K=8,368 (95% CI=5,361-19,250), and a starting population of 6,663 (95% CI=4,791-9,878). The model indicates that the population declined from 1974 reaching a minimum of 3,078 in 2001 and since then has increased to a current population estimate of 3,408 (95% CI=2,091-5,000)(Table 4, fig 5).

The EHB beluga stock is one of three relatively small beluga stocks in Canada. Numbering around 3,400 animals it is approximately three times the size of the other two small stocks, the Cumberland Sound beluga and St Lawrence Estuary beluga stocks which number around 1000 animals each (Marcoux and Hammill 2016; Mosnier et al 2015). The population model trajectory shows that the EHB stock continued to decline even after quotas were introduced in the mid-1980s, because catches of EHB animals remained high throughout this period. Since the early 2000s, there has been an effort to focus harvesting in Hudson Strait, which has reduced the removal of EHB belugas and has resulted in stabilization of the stock (Fig. 5).

Table 4. Parameter estimates from a density dependent model fitted to aerial survey estimates of abundance (1985-2015) and including harvest data (1974-2016). Model priors and posteriors for parameters . The mean, standard deviation (SD), 2.5th , 25th, 50th, 75th and 97.5th quantiles are given for the following model parameters and their priors: carrying capacity (K), maximum rate of increase (lambda), proportion of EHb belugas in harvests from each subzone, struck and lost (SL), and population size in 2016 (N2016). $\hat{R}$ is the Brooks-Gelman-Rubin statistic; values near 1 indicate convergence of chains. N.eff is the number of effective runs after considering autocorrelation.

	Mean	SD	2.5%	25%	50%	75%	97.5 %	Rhat	n.eff
K	10100	4207	5361	6738	8368	13094	19250	1.001	30000
K.prior	11012	5221	2431	6487	11024	15558	19552	1.001	30000
Lambda	0,031	0,016	-0,001	0,02	0,031	0,043	0,058	1.001	24000
Lambda.prior	0,025	0,02	-0,008	0,007	0,025	0,043	0,058	1.001	28000
pFALL	0,261	0,037	0,193	0,236	0,26	0,285	0,336	1.001	15000
pHSUB	0,171	0,023	0,128	0,156	0,171	0,186	0,219	1.001	30000
pNEHBFA	0,301	0,104	0,122	0,224	0,293	0,37	0,52	1.001	30000
pNEHBSP	0,108	0,021	0,071	0,094	0,107	0,122	0,152	1.001	30000
pSAN	0,044	0,023	0,011	0,027	0,04	0,057	0,1	1.001	12000
pSPRING	0,108	0,021	0,071	0,093	0,106	0,121	0,152	1.001	14000
pUBFA	0,261	0,036	0,192	0,236	0,26	0,285	0,334	1.001	30000
pUBSP	0,084	0,058	0,009	0,04	0,071	0,114	0,225	1.001	30000
Startpop	6842	1293	4791	5930	6663	7580	9878	1.001	30000
Startpop.prior	8509	3750	2331	5269	8488	11764	14681	1.001	8900
SL	0,4	0,171	0,106	0,271	0,39	0,52	0,75	1.001	30000
SL.prior	0,428	0,175	0,119	0,297	0,42	0,552	0,777	1.001	28000
N2016	3439	742	2091	2938	3408	3896	5000	1.001	30000

Figure 5. Parameter estimates from a density dependent model fitted to aerial survey estimates of abundance (1985-2015) and including harvest data (1974-2016). Prior (dark lines), and posterior distributions (vertical bars) for the maximum rate of increase (lambda), the initial population size, carrying capacity (K) and struck and loss (SL).



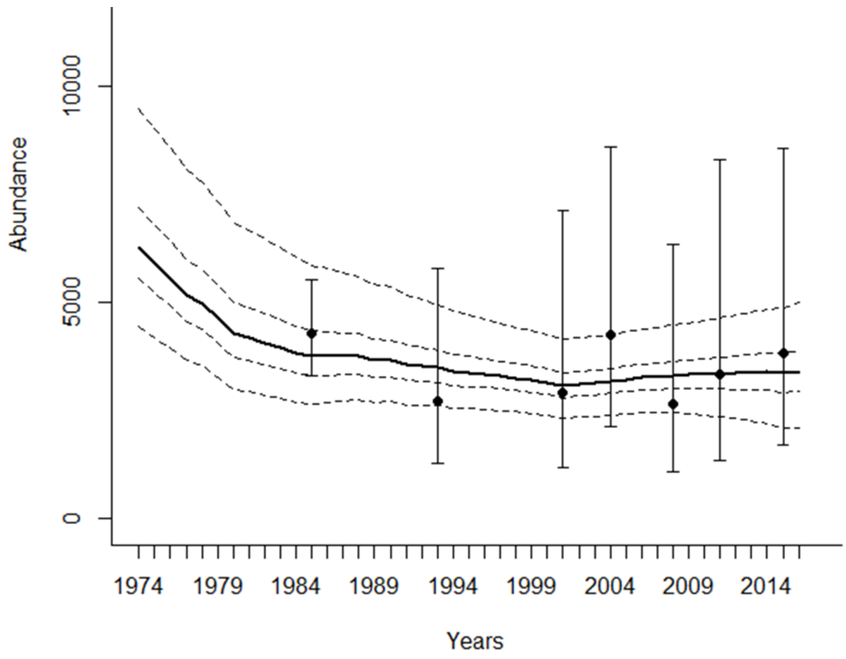


Figure 6. Estimated trajectory of EHB beluga stock obtained by fitting a density dependent model to seven aerial surveys (1985-2015), taking into account harvest data (1974-2016). Surveys (±95%CL), median (solid), 25th,75th quantile (inner dotted lines) and 95% CI (outer dotted lines).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

Harvest levels are set by the Nunavik Marine Region Wildlife Management Board. Under a three year management plan (2015-2017), the management objective was to maintain a constant population. This was identified as the harvest level having a 50% probability of a population decline. A new management plan is being developed for the harvest season set to begin in March 2017. The management objective for this plan has not yet been stated by the Board. There is a 50% probability that a harvest of 67 animals will cause a decline in the population over a 10 year period (Fig 7). This harvest level takes into account SL.

The Potential Biological Removal (PBR; Wade 1998) is calculated as

$$PBR = N_{min} * 0.5 * R_{max} * F_R$$

where N_{min} is the estimated population size using the 20-percentile of the lognormal distribution (N/[exp(z20*sqrt[ln(1+CV^2)])]), R_{max} is the maximum rate of population increase (unknown for belugas and assumed to be 0.04, the default for cetaceans), and F_R is a recovery factor that varies between 0.1 and 1.

A PBR estimate for the EHB beluga stock using the most recent survey abundance estimate of 3,819 (SE=1642; Gosselin et al. 2017) would be 54 animals assuming a F_R of 1. As shown here, beluga surveys can be highly uncertain. However, the PBR estimate uses only the last survey estimate when generating an allowable level of removals, meaning that advice can fluctuate considerably depending on results from the most recent survey. This approach does not make use of the longer time series of abundance information available from the seven surveys. It is also possible to estimate PBR using the model estimates of beluga abundance in 2016 (Table 4). This results in a PBR estimate of 57 EHB whales, assuming a F_R of 1.

PBR represents total removals due to human activity which may include bycatch, ship strikes and SL. The PBR can be converted to a total allowable landed catch (TALC) by accounting for the number of animals killed and not recovered (struck and lost) using the following:

$$TALC = PBR/ LRC$$

where LRC is the hunting loss rate correction. Based on the uncertain estimates of SL for the EHB stock (SL=0.42, Table 4), the TALC for EHB beluga estimated using the PBR approach would be 33 belugas assuming an F_R of 1.

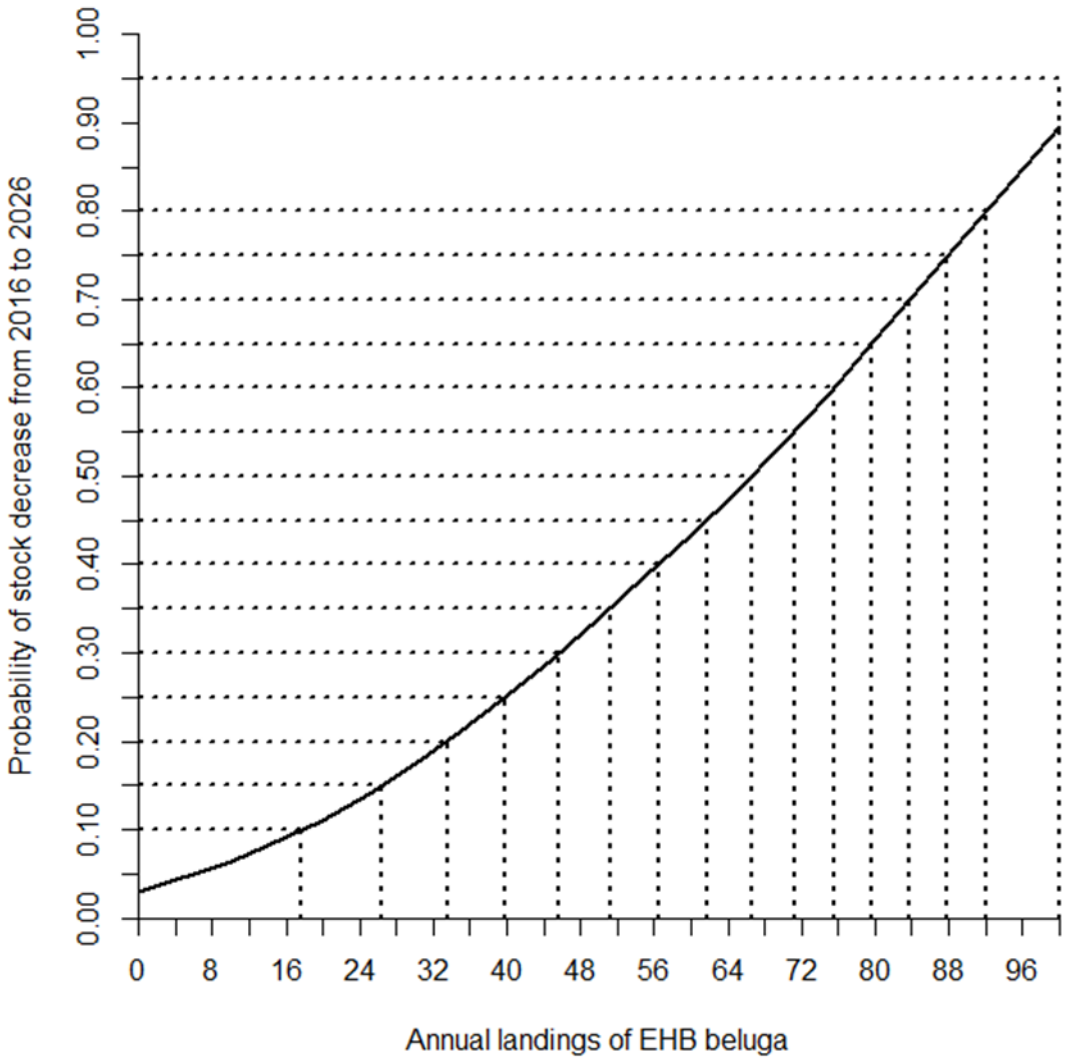


Figure 7. Probability of a population decline from current levels over 10 years at different levels of landings of EHB belugas.

6. Habitat and other concerns

The EHB beluga make extensive use of both the inshore and offshore areas of Hudson Bay in the 'arc area'. The determination of haplotypes considered as representative of the EHB stock is based on samples collected at two river estuaries (Nastapoka and Little Whale River). However, satellite telemetry shows extensive use of the offshore islands in and around the Sanikilluaq area (Belcher Islands; Fig.2) (Lewis et al. 2009; Bailleul et al. 2012). Animals harvested around the Belcher Islands during summer consist of EHB haplotypes as well as other haplotypes suggesting that the genetic composition of the beluga population using the eastern Hudson Bay is more complex than previously thought. However, aerial surveys of the area provide information on overall abundance of all animals without distinguishing between the typical EHB type and other animals.

7. Status of the stock.

The status of this stock has been assessed by the Committee on Endangered Species of Wildlife in Canada (COSEWIC) as 'Endangered", but has not been listed under the Canadian "Species at Risk Act"

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Annex 14: Ungava Bay Beluga Stock

By: Hammill, M.O., A. Mosnier, and J-F Gosselin.

1. Distribution and stock identity

Belugas in Canada were initially designated as separate stocks based on their summering distributions (Finley et al. 1982; Reeves and Mitchell 1987). In most areas, other techniques have also provided some support for the summering stock hypothesis including genetics, satellite telemetry, behavioural observations, trace elements and stable isotopes (Caron and Smith 1990; Brennin et al. 1997, Brown Gladden et al. 1997, de March et al. 2002, de March and Postma 2003; Turgeon et al. 2011; Rioux et al. 2012). Beluga were abundant in the Ungava Bay area, with summer aggregations observed in the Mucalic and adjoining Whale rivers. Animals were also seen in the Kuujjuaq and Leaf Rivers (near Tasiujaq) (fig. 1). Unfortunately, numbers have declined severely since the 1900's. Owing to the small population size, regional closures have limited harvest opportunities, and consequently, no genetic material is available from this 'stock' to evaluate its relatedness to other beluga stocks in the area.

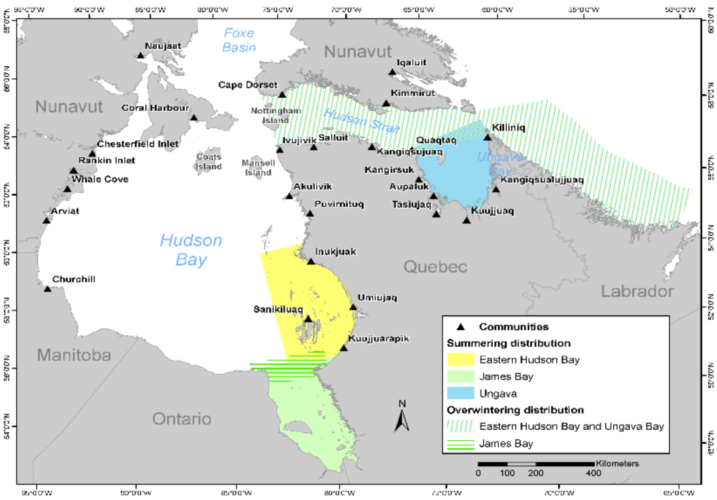

Figure 1. Summering aggregation and overwintering areas of the Eastern Hudson Bay, James Bay and Ungava Bay beluga stocks.

2. Abundance

Although the major summer concentrations of beluga formerly found in southern Ungava Bay are no longer observed, continued sightings and occasional harvesting suggest either that the population persists at some level or that the area is frequented by whales from neighbouring stocks (DFO 2005). The current population size of UB beluga is unknown. Systematic surveys were flown in 1982 by Makivik corporation and in 1985, 1993, 2001, and 2008 by DFO, but no whales were seen within the strip-width of the transects nor during line-transect surveys, flown since 1993 (Smith and Hammill 1986; Hammill et al. 2004, Gosselin et al. 2009). Based on the 1993 survey, using off transect observations, imprecise upper 90% confidence limits of less than 200 individuals in Ungava Bay were proposed (Kingsley 2000).

Since some beluga whales are still seen occasionally in Ungava Bay during summer, it seems more likely that the population still exists but in very small numbers. Doniol-Valcroze and Hammill (2012) developed a Bayesian approach which used of all four surveys with zero-counts. Using the mean group size observed off-transect during these surveys and correction factors for animals underwater, the mean estimate of the current population size was 32 individuals (95% CI 0–94).These estimates are consistent with other off-transect observations of UB beluga made since 1980. Aerial surveys in July 1980 resulted in sightings of 42 animals, including a group of 24 in the Mucalic river (Finley 1982). Surveys made in 1982 by the Makivik Corporation found 11 whales in the southern part of the bay in July and 12 in August. Coastal and offshore surveys in 1985 resulted in the sightings of less than five whales but aerial surveys in 1993 yielded one sighting of 20 whales in July and one sighting of 19 whales in August. During land-based surveys made in 1993, 8 whales were seen in July off Kangirsuk, and 7 sightings totalling 36 whales were made in July and August in the south part of the bay, including a group of 17 animals (Doidge et al. 1994).

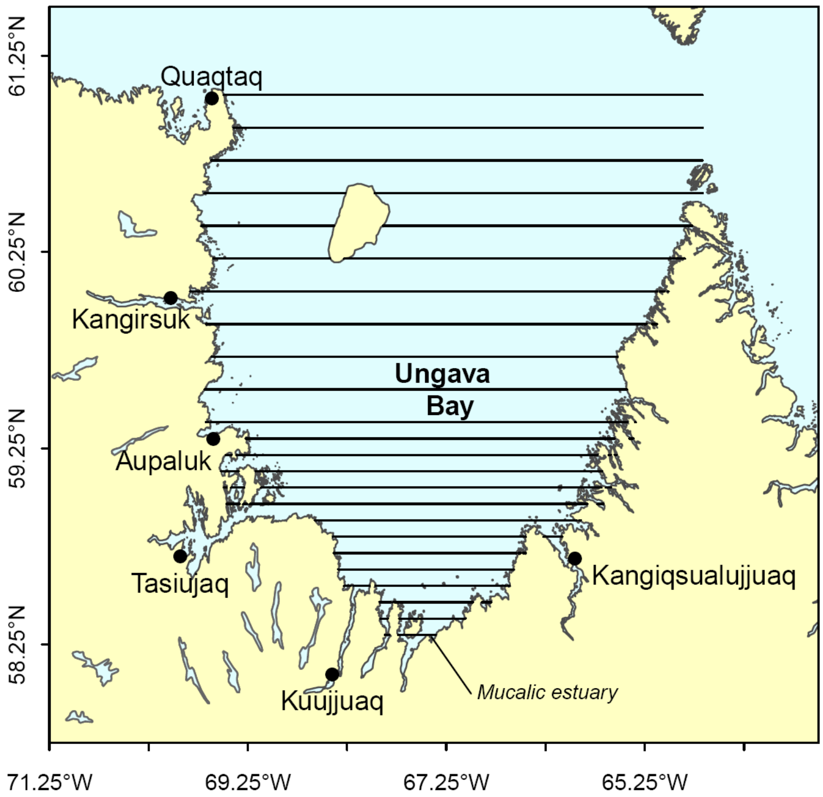

Figure 1. Map showing Ungava Bay and communities around the bay and transect lines flown during past surveys to estimate abundance.

3. Anthropogenic removals

A commercial fishery by the Hudson Bay Company (HBC) took place in Ungava Bay from 1867 to 1911. Using information on HBC catches, it is estimated that the Ungava summer stock numbered at least 1,914 whales in the late 1800s (DFO 2005). The commercial fishery is thought to have severely depleted the number of beluga summering in the bay, but observations and catches made in the 1960's and 70's indicates that a few hundreds were still present in the area. Unregulated subsistence harvesting continued until the early 80's when low numbers observed from aerial and land-based surveys raised concerns that the stock was being overexploited (Boulva 1981, Finley et al. 1982). In 1986, a system of quotas was implemented in Ungava Bay, and the Mucalic estuary was closed to hunting (Lesage et al. 2001). The UB stock was designated "endangered" by the committee on the Status of Endangered Wildlife in Canada in 1988.

4. Population trajectory

Since there is no time series of abundance estimates, it is not possible to provide information on population trend. Current estimates are that the population likely numbers fewer than 100 animals (Doniol-Valcroze and Hammill 2012)

Table 1. Reported harvests in Ungava Bay from 1974-1985 (Smith 1998). These harvests may comprise animals from the Ungava Bay stock as well as animals from multiple stocks that migrate through /overwinter in Ungava Bay. In 1986, catch limits were imposed and the Mucalic river estuary was closed to harvesting. Harvesting continued but may have increasingly included animals from other stocks (see Eastern Hudson Bay).

	1974	75	76	77	78	79	80	81	82	83	84	85
Kangiqsualujjuaq	10	27	20	15	10	37	14	26	12	3	5	3
Kuujjuaq	41	64	102	30	13	34	31	30	29	14	5	2
Tasiujaq	4	9	3	23		3	11	5	6	13	4	9
Aupaluk			6	31	4			4	2	3	2	3
Kangirsuk	37	48	44	79	10	4	4	14	9	12	3	7

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR) method (Wade 1998), is used to calculate total allowable removals from the stock, due to human activities where:

$$PBR = 0.5 * Rmax * Nmin * Fr$$

The mean estimate of the current population size was 32 individuals (95% CI 0–94). We use Nmin = N20% = 12 for the calculation of the PBR, which is equal to 0.16 individuals with a recovery factor of 1. The official guidance for using PBR under the MMPA sets the recovery factor to 0.5 for those that are threatened or depleted, and 0.1 for populations listed as endangered. With those values, the PBR would obviously remain under 1 individual (Doniol-Valcroze and Hammill 2012).

6. Habitat and other concerns

The Mucalic and Whale River estuaries were important aggregation areas for the Ungava Bay beluga stock and sightings are occasionally reported for the area. Unfortunately, there are no recent estimates of abundance, nor observations of frequentation for these areas.

7. Status of the stock.

The Ungava Bay beluga stock was last assessed as Endangered in 2004 by the Committee of Species of endangered Wildlife in Canada (COSEWIC 2004). However, it has not been evaluated under the Species Act Risk Act, so is not afforded any protection.

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Annex 15: Cumberland Sound Beluga Stock

By: Matthews, C.J.D.

1. Distribution and stock identity

Canadian belugas are managed as stocks and/or populations based mostly on the disjunct distribution of summer aggregations (Richard 2010). Putative stocks have been characterized using body size and behaviour (Martin et al. 2001), genetics (Brown Gladden et al. 1997, Brown Gladden et al. 1999, de March et al. 2002, de March and Postma 2003, Turgeon et al. 2012, Colbeck et al. 2013), contaminants (de March et al. 2004), biomarkers such as stable isotopes and fatty acids (Rioux et al. 2012), and satellite telemetry (Caron and Smith 1990, Richard et al. 2001).

Satellite telemetry studies indicate Cumberland Sound (CS) belugas are restricted to Cumberland Sound, with a large aggregation occupying Clearwater Fiord during the summer months (Richard and Stewart 2008; Figure 1). Aerial surveys of the summer range, however, have found up to ~50-60% of the total abundance estimate occurred in the northern portion of Cumberland Sound outside of Clearwater Fiord (Richard 2013, Marcoux et al. 2016).

Genetics and contaminant analyses show CS belugas to be distinct from other Canadian beluga stocks, including belugas sampled from harvests of other southeast Baffin Island communities (Brown-Gladden et al 1997, de March et al 2002, de March et al. 2004, Turgeon et al. 2012). Trace elements and stable isotopes can also be used to differentiate CS belugas from those found in other locations around southeast Baffin Island (Rioux et al. 2012). Inuit traditional knowledge, however, indicates there are more than one type of whale that differ in size, shape, coloration, and taste found within Cumberland Sound (Kilabuk 1998).

2. Abundance

CS beluga abundance was most recently estimated from an aerial visual and photographic survey conducted of the summer range in August 2014 (Marcoux et al. 2016). The survey area, which was based on satellite telemetry studies and Inuit knowledge identifying high-use areas, included Clearwater Fiord and northwestern portions of Cumberland Sound (Figure 2). A complete coverage photographic survey was conducted of Clearwater Fiord, while northwestern Cumberland Sound was divided into the North and West strata, which were surveyed visually using parallel transect lines spaced 10 km apart. The survey was flown using a Twin Otter with a dual-platform design, with two observers seated at bubble windows on each side of the aircraft. Observers recorded species, group size, and declination angles of sightings, as well as weather and environmental conditions (sea state, glare, fog density, and cloud cover). The photographic survey of Clearwater Fiord was conducted using a digital SLR camera mounted at a hatch in the rear underbelly of the plane. When possible, surveys of Clearwater Fiord were flown to coincide with high tide, which provides better water clarity than low tide.

The number of sightings during visual surveys was insufficient for conventional distance analysis, or for estimating perception bias using mark recapture distance analysis (Marcoux et al. 2016). Count data within a 500-m strip on each side of the aircraft were therefore analysed to estimate near-surface abundance using standard methods for strip transects of clustered animals (Marcoux et al. 2016). To account for whales not observed directly beneath the aircraft, the 500-m strip began at 100 m from the track line (Marcoux et al. 2016). Duplicate sightings by both front and rear observers were easily identified (less than 5 s apart and declination angle within 10 degrees) due to the small number of sightings.

Photographs were measured for the proportion of water masked by sun glare and water turbidity (water was ‘murky’ if it was judged to be impossible to detect belugas that were not within 1 meter of the surface). Beluga density was determined by dividing the total beluga count by the summed area of water surveyed (after subtracting land and area covered by glare from each photo). Density was then multiplied by the survey area, which was created by merging the overlapping photographs, to estimate abundance.

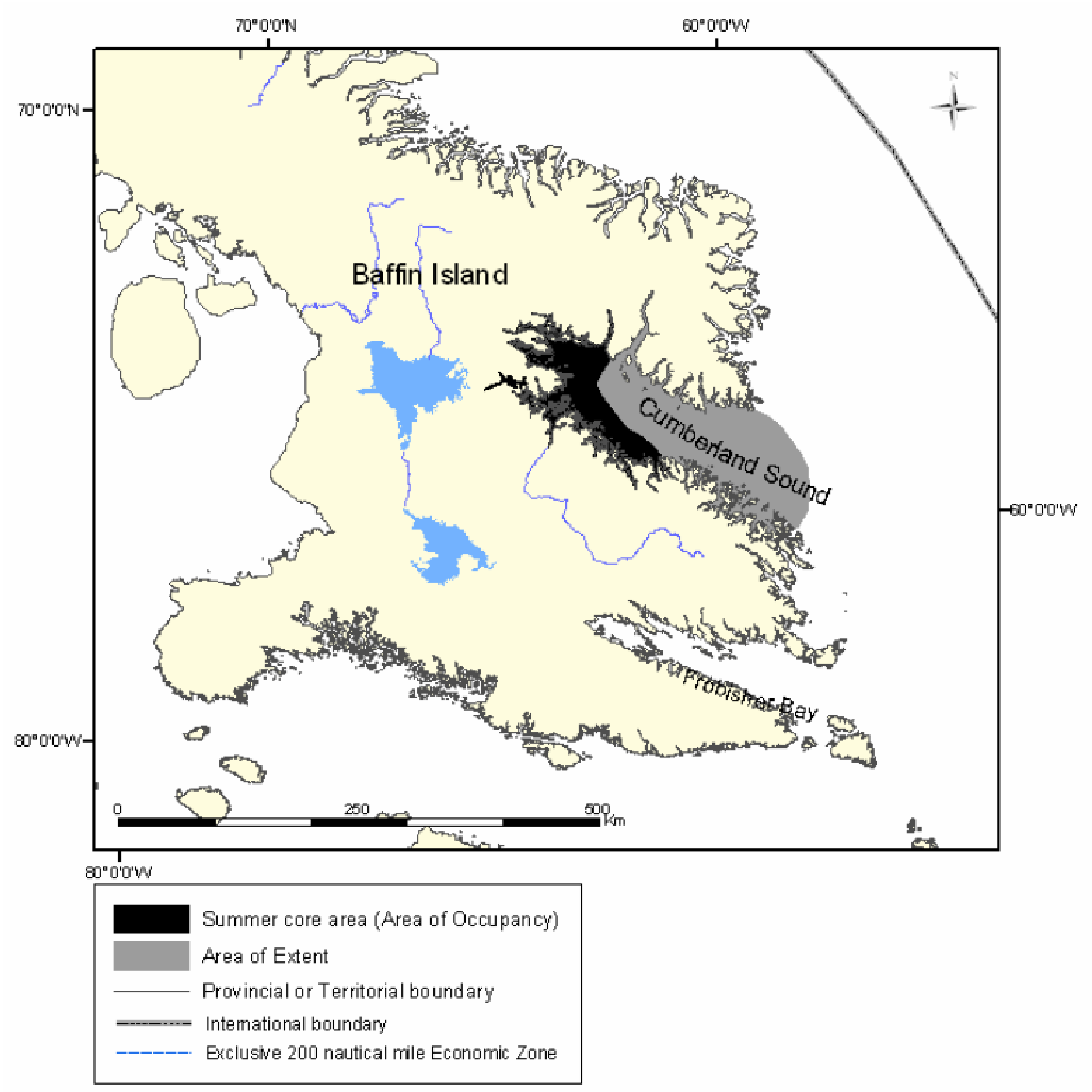

Figure 1. Area of extent of Cumberland Sound belugas. Summer core-use area shown in black (from COSEWIC 2004).

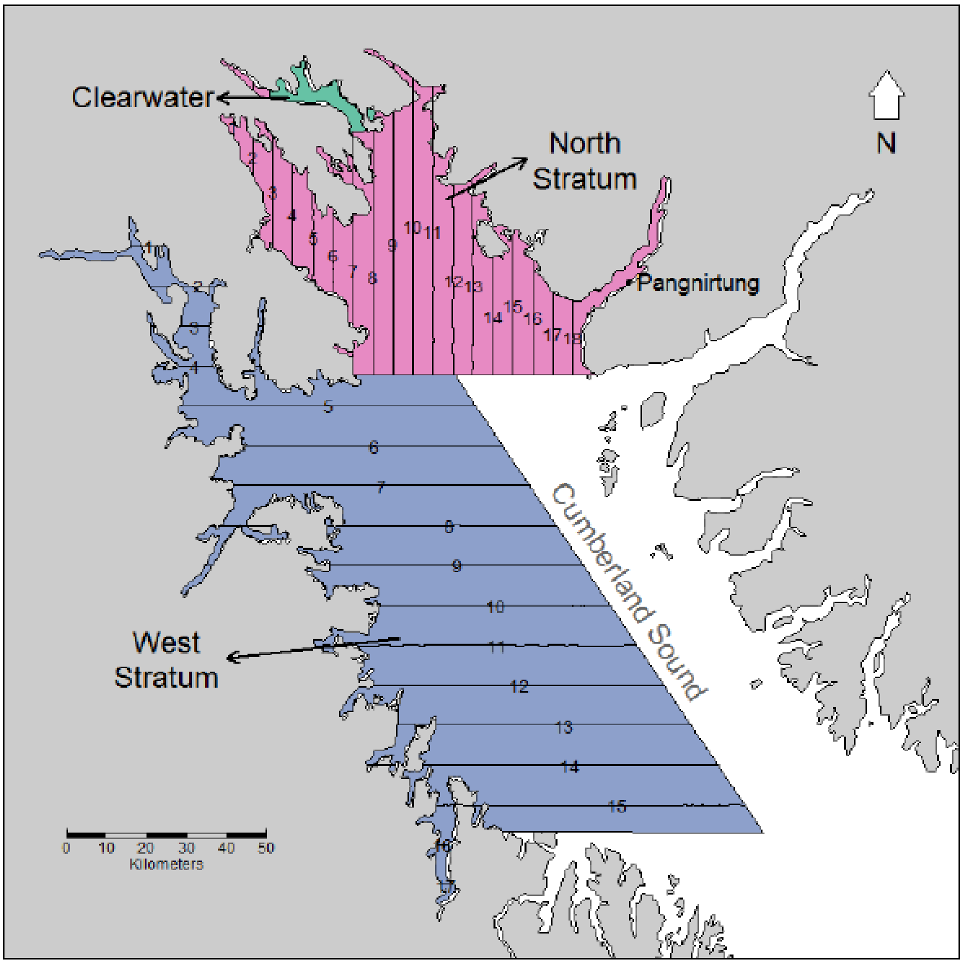

Figure 2. Area surveyed for CS belugas by aircraft in 2014.

Near-surface abundance estimates from both the visual and photographic surveys were corrected for availability bias, the proportion of whales too deep to be observed. Location and dive data from three female CS belugas satellite tagged in 2006-2007 were used to calculate the proportion of time belugas spent in 0–1, 0–2, 0–4, and 0–6 m depth bins in the river estuary and coastal areas. In water judged to be murky, belugas were assumed to be visible only at depths less than 2 m (Richard 2013), while in clear water it was assumed adult belugas could be seen at depths up to 5 m (Richard et al. 1994). Location-specific correction factors for murky water (i.e. Clearwater Fiord) and clear water (i.e. the north and west strata) were determined using location and dive data from the tagged animals (Marcoux et al. 2016).

Two visual surveys of the North stratum and one visual survey of the West stratum were completed, while Clearwater Fiord was surveyed on four different days. The weighted average of the two North stratum survey estimates was 548 (CV = 0.45), while no belugas were observed in the West stratum. The average corrected abundance estimate of the four photographic surveys of Clearwater Fiord was 603 (CV = 0.076). Summing the averaged abundance estimates for each strata resulted in a population estimate (corrected for availability bias) of 1151 (CV = 0.214, 95% CI = 761–1744; Marcoux et al. 2016).

3. Anthropogenic removals

Table 1. Reported landed beluga catches and landed catches plus struck and lost (S&L) for Pangnirtung, NU, from 1977 to 2015 (DFO harvest statistics, unpublished data).

Year	**Landed catches**	**Landed catches + S&L (LRC = 1.18; Richard 2008)**
1977	178	210
1978	85	100
1979	70	83
1980	43	51
1981	45	53
1982	40	47
1983	44	52
1984	40	47
1985	44	52
1986	26	31
1987	40	47
1988	46	54
1989	42	50
1990	36	42
1991	31	37
1992	35	41
1993	15	18
1994	35	41
1995	31	37
1996	41	48
1997	47	55
1998	35	41
1999	50	59
2000	37	44
2001	39	46
2002	41	48
2003	46	54
2004	41	48
2005	41	48
2006	52	61
2007	48	57
2008	41	48
2009	41	48
2010	41	48
2011	42	50
2012	41	48
2013	41	48
2014	41	48
2015	18	21

4. Population trajectory

Nine aerial surveys of the CS beluga population have been conducted between 1980 and 2014 (Richard and Orr 1986, Richard 1991, 2013, Marcoux et al. 2016). Surveys conducted prior to 1999, however, excluded systematic surveys of the three strata surveyed in 1999, 2009, and 2014, and were limited mainly to Clearwater and neighboring Kangilo Fiords (1980, 1981, and 1982), or to Clearwater Fiord and either the North or West Stratum (1985, 1986, and 1990; Richard and Orr 1986, Richard 1991, 2013). Availability bias-adjusted abundance estimates of these earlier surveys, which ranged from 815 to 1775 (Marcoux and Hammill 2016), may be negatively biased since the three most recent surveys have shown 15-64% of the overall population occurred in the North and West strata (Marcoux and Hammill 2016). Adjusted abundance estimates from the three most recent surveys (1999, 2009, and 2014), which are comparable in terms of survey coverage and effort, were 2270 (CV = 0.09), 849 (CV = 0.38), and 1151 (CV = 0.21), respectively (Richard 2013, Marcoux et al. 2016; Figure 3).

DFO (2005) estimated a historical CS beluga population of 8,465 (S.E. = 426) by fitting a population model to aerial survey abundance estimates (1990 and 1999) and reported harvest statistics going back to 1852. A more recent population model fit to survey data from 1990-2014 and reported harvest data (1960-2015) estimated a population of 3,100 animals (rounded to the nearest 100) in 1960 (Figure 3). Determining population trends from the four most recent aerial survey abundance estimates alone is inconclusive, as the sequential estimates are not realistic given our understanding of beluga population growth rates, or harvest removals. The higher population estimate in 1999 relative to 1990 exceeds the purported maximum annual rate of increase of 4% (Marcoux and Hammill 2016). Similarly, the much lower population estimate in 2009 relative to 1999 is only possible if hunting mortality was higher (~180 belugas yr^{-1}) than the currently reported mean of 43 belugas yr^{-1} (Marcoux and Hammill 2016). Marcoux and Hammill (2016) suggest sampling error due variable detection of clumped groups, which has a large impact on abundance estimates, may be a likely explanation (Marcoux and Hammill 2016). The population model by Marcoux and Hammill (2016) indicated the CS beluga population is declining (Figure 3).

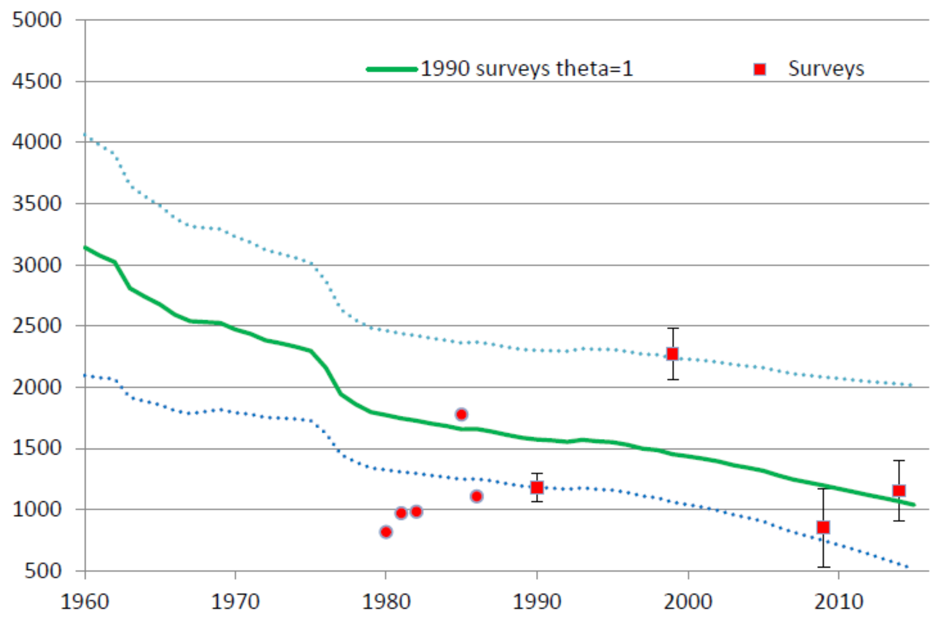

Figure 3. Availability bias-adjusted CS beluga abundance estimates from five surveys conducted over 1980-1986 (red circles) and in 1990, 1999, 2009 and 2014 (red squares). The green line represents model estimates of abundance from fitting to the four most recent surveys and harvest statistics (Marcoux and Hammill 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR; Wade 1998) is calculated as

$PBR = N_{min} * 0.5 * R_{max} * F_R$

where N_{min} is the estimated population size using the 20-percentile of the lognormal distribution ($N/[\exp(z20*\mathrm{sqrt}[\ln(1+CV^2)])]$), R_{max} is the maximum rate of population increase (unknown for belugas and assumed to be 0.04, the default for cetaceans), and F_R is a recovery factor (between 0.1 and 1).

PBR is converted to a total allowable landed catch (TALC) by accounting for the number of animals killed and not recovered (struck and lost) using the following:

$TALC = PBR/ LRC$

where LRC is the hunting loss rate correction and is equal to 1.18 ± 0.07 based on reported beluga harvest statistics from three eastern Canadian Arctic communities (Richard 2008).

Using the most recent CS beluga abundance estimate (1151, CV = 0.214; Marcoux et al. 2016) and assuming a recovery factor of 0.5, which DFO has used as a standard in the past for populations considered as 'threatened' by COSEWIC, PBR = 9.6 and TALC = 8.2. Calculated PBR values based on the modeled 2015 population abundance ranged from 7.0 to 7.9 (Marcoux and Hammill 2016).

6. Habitat and other concerns

The annual subsistence hunt as it is currently set is a demonstrated threat to the CS beluga stock (Marcoux and Hammill 2016). Shifts in the Cumberland Sound ecosystem, notably incursion of capelin over the past several decades, is believed to have resulted in a diet shift of CS belugas from arctic cod to a more capelin-based diet (Marcoux et al. 2012, Watt et al. 2016). While it is unknown if this has had negative impacts on the population, capelin is a major component of the diet of belugas from other regions (Kelley et al. 2010). Stress levels as indicated by cortisol concentrations are higher in CS belugas than other beluga stocks in Canada (DFO, unpublished data), indicating possible negative impacts of potential threats such as anthropogenic noise and disturbance and climate change and associated ecosystem impacts.

7. Status of the stock.

The CS beluga stock is small in abundance and range. The most recent population models indicate the population is in decline and that current harvest rates exceed sustainable levels (Marcoux and Hammill 2016). The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) have designated the Cumberland Sound beluga population as 'Threatened' (COSEWIC 2004). The status of Cumberland Sound belugas under Canada's Species at Risk Act (SARA) is Schedule 1, 'Threatened'.

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Annex 16: St. Lawrence Estuary Beluga Stock

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1. Distribution and stock identity

The current distribution of St. Lawrence Estuary (SLE) beluga (*Delphinapterus leucas*) represents a fraction of that described historically (Figure 1; Vladykov 1944; see Mosnier et al. 2010 for a review). Their core distribution is centered on the Saguenay River, and extends from the Battures-aux-Loups-Marins to Rivière-Portneuf / Rimouski in the Estuary, and to Baie Ste-Marguerite in the Saguenay River. Concentration areas outside of this sector vary seasonally, as they did in the 1930s, but are now constrained within a zone located between Battures-aux-Loups-Marins and Sept-Îles / Cloridorme (vs west of Quebec City to Natashquan in the 1930s), with only rare observations in the Baie des Chaleurs. SLE beluga distribution range is small compared to other beluga populations, and even smaller during summer (Mosnier et al. 2010). Sex- and age-specific spatial segregation is typical of the species during summer (Michaud 1993), and possibly also at other times of year (Colbeck et al. 2013).

Belugas in the SLE represent a relict population, which established themselves in the SLE some 10,000 years ago during the Wisconsin glaciation (Harington 1977; 2008). The SLE population is genetically differentiated from all other Canadian beluga populations, and is the most divergent based on both nuclear and mitochondrial markers (Brown Gladden et al. 1997, 1999; de March and Postma 2003). Molecular genetic studies indicate that their closest relatives are in eastern Hudson Bay, and that their isolation from the other populations persisted over evolutionary timescales (Brennin et al. 1997; Brown Gladden et al. 1997; de March and Postma 2003; O'Corry-Crowe et al. 2010; Postma et al. 2012). SLE beluga show a low nuclear genetic diversity similar to that observed in other isolated, insular populations of mammals (de March and Postma 2003; Patenaude et al. 1994), suggesting that contributions from neighbouring populations are insignificant. Significant ongoing immigration is considered unlikely given that the nearest populations in Ungava Bay, Hudson Bay, and West Greenland are depleted (Smith and Hammill 1986; Reeves and Mitchell 1989; Richard 1991, 1993; Hammill *et al.* 2009).

In addition, there appears to be no overlap in seasonal distribution between SLE beluga and other populations. SLE beluga undertake seasonal movements, but their extent appears to be limited to the northwestern Gulf of St. Lawrence (Mosnier et al. 2010). The winter distribution of eastern Hudson Bay beluga extends into the Labrador Sea, but only to several hundreds of kilometres north of the Gulf of St. Lawrence (Bailleul et al. 2012). Beluga have been reported along the north shore of the St. Lawrence, south coast of Labrador and off Newfoundland (Vladykov 1944; Reeves and Katona 1980; Reeves and Mitchell 1984; Pippard 1985a; Sergeant 1986; Michaud and Chadenet 1990; Curren and Lien 1998; Kingsley and Reeves 1998; Benjamins and Ledwell 2009). However, the origin of these whales was unconfirmed in most cases, although there was confirmation of an Arctic origin for some of the beluga observed on the lower north shore of the Gulf of St. Lawrence, and around Newfoundland (DFO, unpublished data).

2. Abundance

Abundance estimates for this population are obtained on a regular basis since 1988. Survey design has been consistent over time, and consists in systematic strip-transect photographic aerial surveys, with a near-50% coverage of the SLE beluga summer distribution (Kingsley and Hammill 1991; Kingsley 1993; 1996; 1998; 1999; Gosselin et al. 2001; 2007; 2014). One to multiple systematic line-transect visual aerial surveys have also been flown regularly since 2001, and at the same period as photographic surveys, allowing for a second time series of abundance indices to be built (Gosselin et al. 2007; in press; Lawson and Gosselin 2009).

Population size estimates from the photographic surveys were used in an age-structured population model, in combination with information on the proportion of young (< 2 years-old) estimated from 8 photographic surveys flown between 1988 and 2009, and mortalities of newborns and individuals other than newborns documented by the carcass monitoring program over the period 1983-2012 (Mosnier et al. 2015). The model estimated population size to approximately 900 individuals in 2012. Other abundance estimates were obtained since 2012 using visual surveys (e.g., Gosselin et al. in press).

However, these should not be compared to those obtained using photographic surveys until correction factors specific to visual surveys and adequately accounting for availability biases are developed (Gosselin et al. in press).

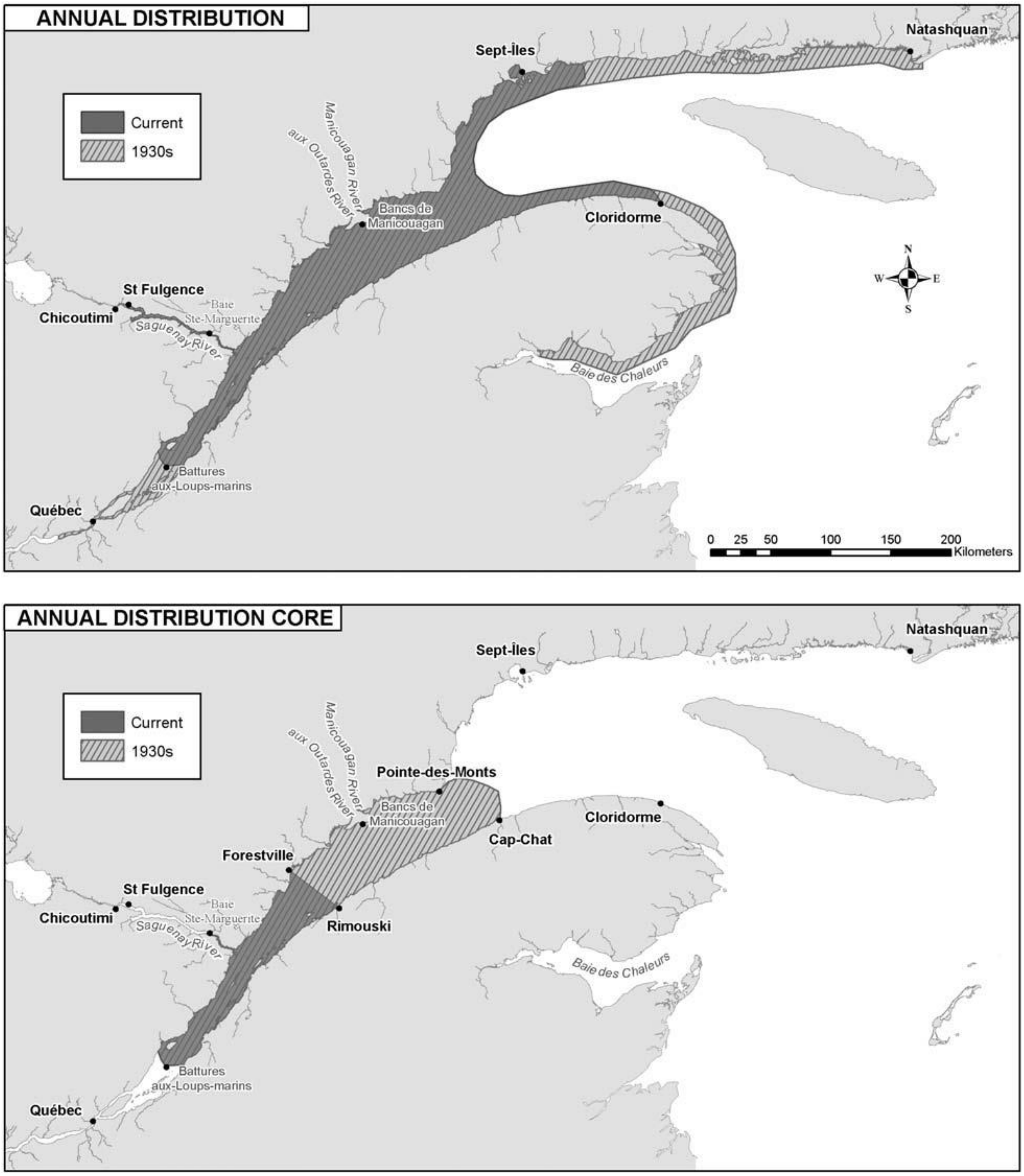

Figure 1. Historical and current annual and core distributions of St. Lawrence Estuary beluga.

3. Anthropogenic removals

Beluga in the SLE have been severely depleted by sustained hunting that took place mainly from the late 1800s to the mid-1900s (Reeves and Mitchel 1984). Beluga hunting in the SLE is prohibited under the Marine Mammal Regulations of the Fisheries Act since 1979.

A program documenting the number and potential causes of SLE beluga deaths has been in place since 1983 (reviewed in Lesage et al. 2014). Out of the 472 beluga carcasses documented over the course of this program, 222 were subjected to complete necropsies. Human activities were responsible for 5% of these documented deaths, and included fishing gear entanglement (1%; n = 2 ind.) and vessel strikes (4%; n = 8 ind.) (Lair et al. 2016).

4. Population trajectory

Population size for SLE beluga was estimated at 5,000–10,000 individuals in the 1800s and less than 1,000 in the late 1970s when there was an official ban on hunting (Reeves and Mitchell 1984; Pippard 1985b; Hammill et al. 2007; Mosnier et al. 2015). A review of the SLE beluga status in 2007 concluded that the population was stable over the period 1988-2006 (Hammill et al. 2007). However, an increase in death reports for young-of-the-year over the period 2008-2012, and in adult female perinatal mortalities, led to a review of SLE beluga status and population trends in 2013 (DFO 2014). This assessment was made using an age-structured population model that included survey estimates, but also other sources of information to describe the dynamics of the population (see section on Abundance; Mosnier et al. 2015). The photographic surveys detected no significant temporal trend in beluga abundance, although the last survey estimate was the lowest of the time series (Gosselin et al. 2014). The photographic surveys also suggested that the proportion of 0-1 year-old calves in the population decreased from 15.1 to 17.8% of the total population in the 1990s to 3.2 to 8.4% in the 2000s (Gosselin et al. 2014). Data from the carcass monitoring program indicated that, over the first 24 years of the program, newborn deaths varied from 0 to 3 per year and followed a 3-4 year cycle (Lesage et al. 2014). In 2008 this cycle changed to biennial peaks, and annual report rates 3 to 5 times higher than the maxima observed previously. Mortality patterns among adults followed no clear temporal trends over the study period (Lesage et al. 2014).

Using the above information, the model estimated that the SLE beluga population was stable or increasing at a slow rate (~ 0.13% per year) between the 1960s and the early 2000s, with around 1000 individuals in 2002. The model then indicated a decline (-1.13% per year) in abundance to an estimated 889 individuals (95% CI 672-1167) in 2012. The model also suggested internal changes in vital rates and age-structure, with two distinct periods. From 1984 to 1998, there was a relative stability in newborn mortality (median values from 14% to 27% with peaks every 3 to 4 years) and pregnancy rates (around 30%, with small peaks every 3 years). During this period, population age structure was stable with approximately 41% of the population being immature beluga, including 7.5% of newborns. From 1999 to 2012, the model suggested demographic instability with major changes in population parameters and age structure. This period was marked by peaks of high newborn mortality interspersed by peaks of high pregnancy rates, themselves separated by periods of lower-than-average fecundity (e.g., ~15% in 2001-2002).

Over the last 6 years of the model, female reproduction appeared to change from a 3-year cycle (with a third of mature females pregnant each year) to a 2-year cycle (with about half of the females pregnant), a phenomenon associated with increased newborn mortality. These changes had strong effects on the population age structure, and proportion of newborns in the population, with a decreasing trend from 6-8% before 1999 to 4-6% after 2007. At the same time, the estimated proportion of immature individuals in the population declined, resulting in a concurrent increase in that of mature beluga even though their absolute numbers remained constant for a ratio of mature : immature of 66 : 34% by 2012. The median of the annual adult mortality was 6% but varied from 4% to 9 depending on years.

Some of the model estimates, particularly in the 2000s, were supported by observations from a long-term program using photo-identification of live SLE beluga conducted over the period 1989-2012 (Michaud 2014). This study revealed a slight increase in the proportion of grey individuals (juveniles and young adults) in the population during the 1990s and early 2000s, with a transition to a decreasing trend in the mid- to late-2000s, a result predicted by the population model. The photo-identification time series also revealed lower-than-average calf production in 1999-2004, followed by high calf production in the mid and late 2000s in the years following those estimated by the model to be characterized by high pregnancy rate.

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

Potential Biological Removal (PBR) is the product of minimum population size (20th percentile), one-half the maximum productivity rate, and a recovery factor (Wade and Angliss 1997). The 20th percentile of the model estimate of 889 individuals in 2012 (Mosnier et al. 2015) is 789 individuals. The recovery factor is 0.1, the default value for endangered populations (Wade and Angliss 1997). The maximum productivity rate is 0.04, the default value for cetaceans (Barlow et al. 1995). A PBR approach indicates that one is the maximum number of whales that could be removed from this population. However, given the declining trend in this population, one could argue that removals should be zero.

6. Habitat and other concerns

Beluga use a variety of habitat, including ice-free and estuarine to coastal and offshore ice-covered environments (Moore et al. 2000; Barber et al. 2001; Suydam et al. 2001; Lydersen et al. 2002). Habitat requirements likely vary according to size, age, sex and reproductive status, as well as energy requirements (Michaud 2005; Loseto et al. 2006). Beluga in the SLE are at the southernmost limit of the species distribution. The continued presence of beluga at these low latitudes since the last glaciation likely results from the substantial freshwater inputs and sub-Arctic conditions (cold, productive waters, and seasonal ice cover) prevailing in the SLE (El-Sabh and Silverberg 1990).

The Critical Habitat of SLE beluga has been defined, and corresponds for the period of June to October to the area of occupancy by females, calves and juveniles (DFO 2012). Adequate food resources and acoustic environment, and processes maintaining cold and productive conditions are habitat features considered essential for beluga vital functions.

Beluga in the SLE are exposed to a number of stressors that can affect the quality of their habitat or that can interfere with their normal activity. The St. Lawrence is a major commercial waterway to Central North America where vessel traffic is chronic, leading to elevated sound levels in some sectors of beluga habitat (McQuinn et al. 2011; Gervaise et al. 2012). In the core of their summer distribution, there is a sustained whale-watching industry operating over 30 vessels and offering several departures a day. This tourism-related activity along with recreational boating peak in July-August, when SLE beluga give birth. Between 2003 and 2012, these activities increased in some sectors of the beluga Critical Habitat as a result of newly established whale-watching companies operating in the Upper SLE and targeting beluga (Ménard et al. 2014). Parturition is tiring for the female as it may take several hours. At this time, females may be more visible and less likely to move away from boaters. Anthropogenic disturbance during parturition or lactation can interfere with calving and/or nursing, resulting in increased calf mortality. Disturbance could also represent an aggravating factor if animals are weakened by other causes (dystocia, health problems due to toxicity or other illnesses).

The St. Lawrence Estuary is also located downstream of highly industrialized areas. As a result, several chemical compounds issued mainly from anthropogenic sources, including some that are known to be carcinogenic such as polycyclic aromatic hydrocarbons (PAH) found their way into beluga and their habitat (DFO 2012). While some of these compounds such as PCBs have shown some decline in beluga following regulations, others such as flame retardants (e.g., polybrominated diphenyl ether), have increased exponentially in beluga during the 1990s, and continue to be at their maxima since then (Lebeuf et al. 2014; Simond et al. in review). The effects on beluga health and their role in the recent elevated frequency of complications at parturition and mortalities of newborns are difficult to demonstrate, but are considered probable; these different classes of chemical substances are known to have various endocrine disrupting effects in mammals with possible impacts on offspring development, and on reproduction, immune system and behaviour (Martineau et al. 2010; Lair et al. 2016).

Recurrent harmful algal blooms have also been reported in the SLE, with the most recent event in 2008 killing multiple specimens of several species, including seven SLE beluga adults and calves in one week (Scarratt et al. 2014). Conditions leading to these events are not uniform and fully understood, but the frequency of harmful algal blooms has increased globally, and could potentially also become more frequent in the SLE, although this has not yet been documented (Anderson et al. 2012; Scarratt et al. 2014).

The ecosystems of the Estuary and Gulf of St. Lawrence have been affected by a number of factors over the past decades, including overfishing which resulted in a major collapse of demersal fishes in 1993, and climate variability (Savenkoff et al. 2007; Galbraith et al. 2012). These changes have likely altered the trophic structure and functioning of these ecosystems, including prey biomasses and distributions, in addition to potentially affecting prey quality. A study incorporating a set of 94 physical and biological parameters, including 28 contributing more directly to the SLE beluga habitat quality, revealed shifts in environmental regimes, characterised by changes in demersal and pelagic fish availability and composition, ocean temperature, and winter sea ice dynamics (Plourde et al. 2014). The shift from a stable to an unstable age structure in SLE beluga, and toward lower than average proportion of calves, and increased number of dead calf reports corresponded approximately with the shift towards negative anomalies in habitat quality indicators, where large demersal fish and spring herring biomass were at their lowest, and ice coverage and temperatures were below normal. This change from a relatively cold period where prey were relatively abundant, to a period of warmer conditions and where prey were less abundant reached extreme values starting in 2010, a period characterized by strong negative anomalies in ice condition (short duration, low volume/coverage), and for 2012, high water temperatures. Such negative anomalies in habitat quality were not observed from 1971 to 1998. In parallel to this study, another analysis examining a time series of isotopic ratios in tissues of SLE beluga over the period 1988-2012 documented a strong and continuing change in beluga isotopic signature since the early 2000s (Lesage 2014). Whether this change is associated with a shift in diet or in other ecosystem characteristics is uncertain (Lesage 2014).

Climate variability may further affect SLE beluga through increases in inter-specific competition as other species expand their range due to temperature change and loss of ice cover. In the short term, efforts can be directed to reducing anthropogenic stressors such as disturbance in sensitive areas and critical periods for females and calves, chemical contamination, nutrient enrichment, habitat loss, and competition for food resources from fisheries (DFO 2014).

In a recent exercise, a population viability analysis incorporating some of the threats identified above was conducted to predict responses of SLE beluga to environmental change and identify management actions most likely to result in population recovery (Williams et al. in review). The main threats considered were: changes in prey abundance, changes in foraging efficiency caused by underwater noise and disturbance, and chemical pollution, namely polychlorinated biphenyls (“PCBs”). Across the range of these stressors, data were only available to link threats to changes in calf mortality. Therefore, the results must be interpreted with caution as there is a need to assess whether stressors could influence pregnancy or adult mortality. This study indicated that the warming conditions and decreased ice may have an important effect on the recovery of this population. The analysis also indicates that improvements to any one threat, within the ranges that seem feasible to change, are not sufficient to achieve positive population growth, but that the population is predicted to do appreciably better and reach a sustained population growth if all three threats could be mitigated (Williams et al. in review).

7. Status of the stock.

Until recently, the SLE beluga population was considered *Threatened* by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), and was listed as such under the Species at Risk Act (SARA) in 2005. Following the 2013 status review by DFO (DFO 2014), COSEWIC proceeded with a re-evaluation of the population status, and concluded that the SLE beluga were now *Endangered* (COSEWIC 2014). This new status was echoed by the SARA in 2016. The SLE beluga population was estimated at 889 individuals in 2012. This population is thus considered small, and in decline at a rate of approximately 1% per year since the early 2000s (Mosnier et al. 2015).

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Annex 17: Svalbard Beluga Stock

Lydersen C and Kovacs KM

1. Distribution and stock identity

Genetic heterogeneity was observed between Svalbard and West-Greenland white whales that reveals limited gene flow over ecological time scales (O'Corry-Crowe et al. 2010). In this study it was also revealed that Svalbard and Beaufort Sea animals diverged 7,600-35,000 years ago, but have experienced recurrent period with gene flow since then, most likely via the Russian Arctic during subsequent warm periods.

Telemetry data show that the Svalbard white whales are extremely coastal in their distribution in the ice free seasons (see figure). They spend most of their time close to glacier fronts, and when they move from one front to another they do so in an apparently directed and rapid manner very close to the shorelines (Lydersen et al. 2001). When sea ice forms in the winter, the whales are "pushed" offshore but still stay in the Svalbard area often occupying areas with more than 90% ice cover (Lydersen et al, 2002).

A survey for various whales in the marginal ice zone north of Svalbard during August 2015 detected no white whales in this area; only bowhead whales and narwhals (Vacquié-Garcia et al. 2017). During the same time period white whales were observed (as is normal) along the coast of Svalbard, further documenting the lack of affiliation with sea ice for this whale species in Svalbard during summer.

Figure showing the tracks of 5 white whales instrumented with satellite relay data loggers in August 2016 revealing the very coastal movement patterns of this species in Svalbard.

2. Abundance

No abundance estimate is available from this area, however a first ever survey planned for July-August 2018.

3. Anthropogenic removals

Totally protected since the 1960s in Svalbard.

4. Population trajectory

No data.

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

Not relevant.

6. Habitat and other concerns

Effects of climate change with impacts on sea ice conditions, prey base composition, competition from more boreal marine mammal species, new parasites and diseases, is a general concern. Levels of various pollutants in white whales from Svalbard are very high and for many compounds higher than what is found in polar bears in the area (Andersen et al. 2001, 2006, Villanger et al. 2011, Wolkers et al. 2004, 2006). These levels are in many cases also higher that what has been shown to impact the physiology and especially the immune system in lab animals.

7. Status of the stock.

Unknown

8. Life History Parameters

Diet and food availability

A diet study based on analyses of fatty acids in the blubber of white whales from Svalbard found that polar cod (*Boreogadus saida*) had the most similar FA composition to the white-whale blubber (Dahl et al. 2000).

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Annex 18: White Sea (Russia) Beluga Stock for the Global Review of Monodontids, 13-16 March 2017, Copenhagen

By: Olga V. Shpak, Vera V. Krasnova, Ilya G. Meshchersky

1. Distribution and stock identity

The data on distribution and migrations (stationary coastal observations, ship-based, aerial surveys, satellite tracking) suggest that belugas in the White Sea form a resident population. Genetic study reveals a heterogeneous population, which consists of discrete reproductive aggregations, probably, spatially associated with the major bays of the White Sea. However, to understand population structure more data is necessary.

Summer distribution

Singletons or small groups are found everywhere, but majority concentrate in major bays, where they form local aggregations (herds) in the coastal waters. Currently, at least eight summer aggregations have been recognised (Figure 1): 2 in Dvina Bay, 2 in Mezen' Bay, 4 in Onega Bay (Chernetskiy et al., 2002; Andrianov et al., 2009, Alekseeva et al., 2012). In summer, belugas do not concentrate in Kandalaksha Bay[5]; however, discrete small groups visit the area but not the bottom of this shallow-watered bay (Glazov et al., 2008; Nikolaeva, 2015; Panova, pers. comm.).

Summer aggregations consist mainly of females with calves of different age. Usually the number of whales in the aggregation doesn't exceed 100 individuals. (Alexeyeva et al., 2012; Krasnova et al., 2012; Andrianov et al., 2009). Photo-identification results showed that the aggregations in Onega Bay are not isolated, and belugas move within the area. (Chernetsky et al., 2011; 2014). However, belugas summering in different bays of the White Sea, are, probably, limited in their movements and do not mix, which is supported with molecular-genetic analysis (see below). Analysis of acoustic repertoires of belugas from Onega and Dvina bays demonstrated statistically significant difference (Panova et al., 2016), which may be caused by a certain isolation.

Such distribution is observed in June-July. Available observations suggest that in August belugas re-distribute in the sea. According to aerial surveys, some belugas shift north-east to the border with the Barents Sea (Matishov and Ognetov, 2006; Glazov et al., 2010; Solovyev et al., 2012). According to aerial survey in the middle of August (Solovyev et al., 2012), belugas distribution was different from previous surveys conducted in July: a lot of whales concentrated at the Funnel, far from shores.

Winter distribution

Matyshov and Ognetov (2006) doubt the existence of a resident White Sea population. They suggest that belugas only «visit» the White Sea for short periods of time in summer; that no (or almost no) belugas stay in the White Sea in January-March, and those who remain are found in the Funnel, not in the Basin. However, according to many experts (multiple sources), belugas, at least a part of population, overwinter in the White Sea. Historical data also support a year-round residency: in March-April, beluga harvest took place along the southern coast of Dvina Bay, when the whales were shot from the shore-fast ice (Alekseeva, 2008). Current knowledge on the White Sea beluga winter distribution is based on aerial survey (2008, 2010) and satellite tracking data (2005, 2010-2011). These studies also showed that at least a considerable part of belugas do not leave the White Sea in winter.

Aerial count was conducted in March 2008 and 2010 using the method of linear transects. The White Sea water surface was uniformly covered with transects, both over the open water and ice (Glazov et al., 2010). In spite of different weather conditions in March 2008 and 2010, distribution patterns were similar. The whales were found not only in the open water, but also in consolidated ice. Most sightings

[5] *The names of bays may be spelled differently in other sources: Kandalaksha=Kandalakshsky Bay, Onega=Onezhsky Bay, Dvina=Dvinskoy Bay, Mezen'=Mezem'sky Bay*

fell on deep-water regions in the central part of the Sea (between the bays I, II and III, Figure 1). A lot of whales were sighted in the areas close to their summer grounds (for example, in the outer Dvina Bay). Both adult and juvenile belugas were seen during the aerial surveys.

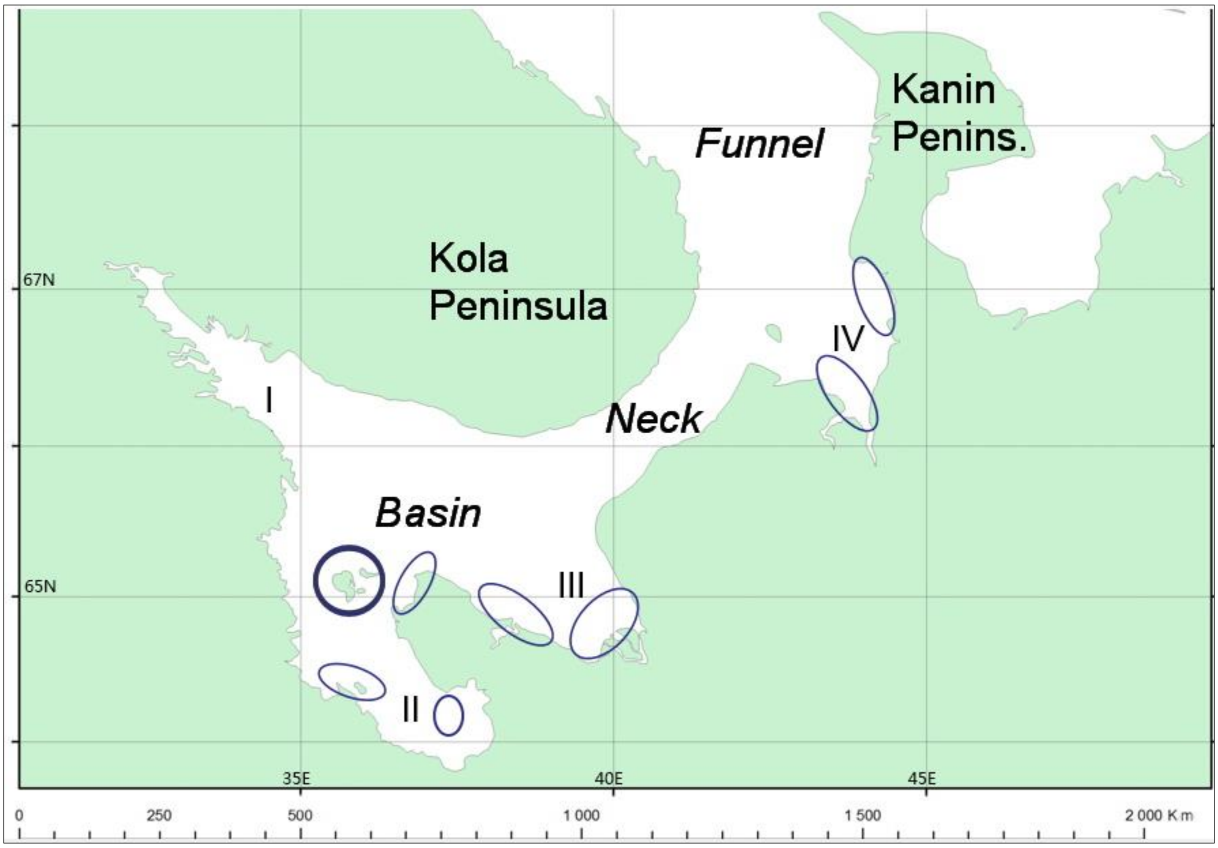

Figure 1. *The White Sea and locations of summer beluga aggregations (different sources, see text). Locations are marked with blue circles; Solovetsky aggregation – in bold. I - Kandalaksha Bay, II – Onega Bay, III – Dvina Bay, IV – Mezen' Bay.*

Several satellite tracking studies on beluga movements in the White Sea have been conducted starting 2003 (summarized in Table 1). The first successful attempt of overwinter tracking (2005) showed that a tagged beluga remained in the White Sea, and her movements in late autumn coincided with the movements of herring (Svetochev et al., 2007; Svetochev and Svetocheva, 2012). Belkovich (2006) admitted that some belugas summering in the White Sea may remain there the entire year, and suggested that this would be the female part of stock, while the male groups spend winters in the Barents Sea and in spring return to the White Sea migrating along the Kola peninsula (See Fig. 1).

To check this hypothesis, eight males were tagged in autumn 2010 and 2011 (Glazov et al., 2012; Kuznetsova et al., 2016). Six tags transmitted for over 180 days: no beluga males left the White Sea waters in winter (Fig. 2). In spring, the males did move into the Basin (the central part of the White Sea) along the Kola peninsula coast. Observing a similar movement may had led Belkovich to hypothesis on male spring migration from the Barents to the White Sea. Residential behaviour of males in November was explained by the peak of autumn spawning migration of Atlantic Salmon (*Salmo salar*) to Varzuga River (Fig 1, red arrow); spring beluga concentration in the SE part of Funnel, in the mouth of Ponoy river, was linked to the seaward migration of Atlantic salmon (Kuznetsova et al., 2016).

Thus, results of aerial surveys and satellite tracking, as well as historic harvest data, suggest that the White Sea belugas do not migrate to the Barents Sea in winter. At the same time, such data are very limited; considerably less belugas are observed in the White Sea in winter compared to summer (see below), and it is unclear whether this lower number is a result of lower detection availability in winter, or part of belugas do leave the White Sea.

Table 1. *Satellite tagging of belugas in the White Sea.*

Location	Sex	Period of transmission	Movements
Dvina Bay, western part.	3 belugas, sex unknown	unsuccessful	N/a
Dvina Bay, western part.	F	26 Jun 2005 - 03 Mar 2006 – 212 days	Mostly Dvina Bay with moves to the Basin in Feb
Kola Peninsula, South, at the mouth of Varzuga River	M	27 Oct 2010 – 29 May 2011 – 215 days	In autumn all remained residential to the area of capture and tagging (Southern coast of Kola penins.). Upon ice formation, moved to the Basin, and to Dvina Bay. In spring, 4 whales moved north, to Ponoy River area (Funnel); while 1 beluga remained in the central part. In June, 1 whale, whose tag still transmitted, returned from the Funnel to the central WS (Fig. 2)
	M	30 Oct 2010 – 29 May 2011 – 212 days	
	M	30 Oct 2010 - 02 May 2011 – 185 days	
	M	30 Oct 2010 – 26 May 2011 – 209 days	
	M	30 Oct 2010 – 24 Jun 2011 – 241 days	
Kola Peninsula, South, at the mouth of Varzuga River	M	01 Oct 2011 – 22 Oct 2011 – 21 days	Wasn’t included in analysis
	M	27 Oct 2011 - ?? May 2011 – ?? days	Autumn: residential to coast of Southern Kola penins., winter and spring – in the Basin and Dvina Bay
	M	29 Oct 2011 – 01 Nov 2011 – 3 days	Wasn’t included in analysis

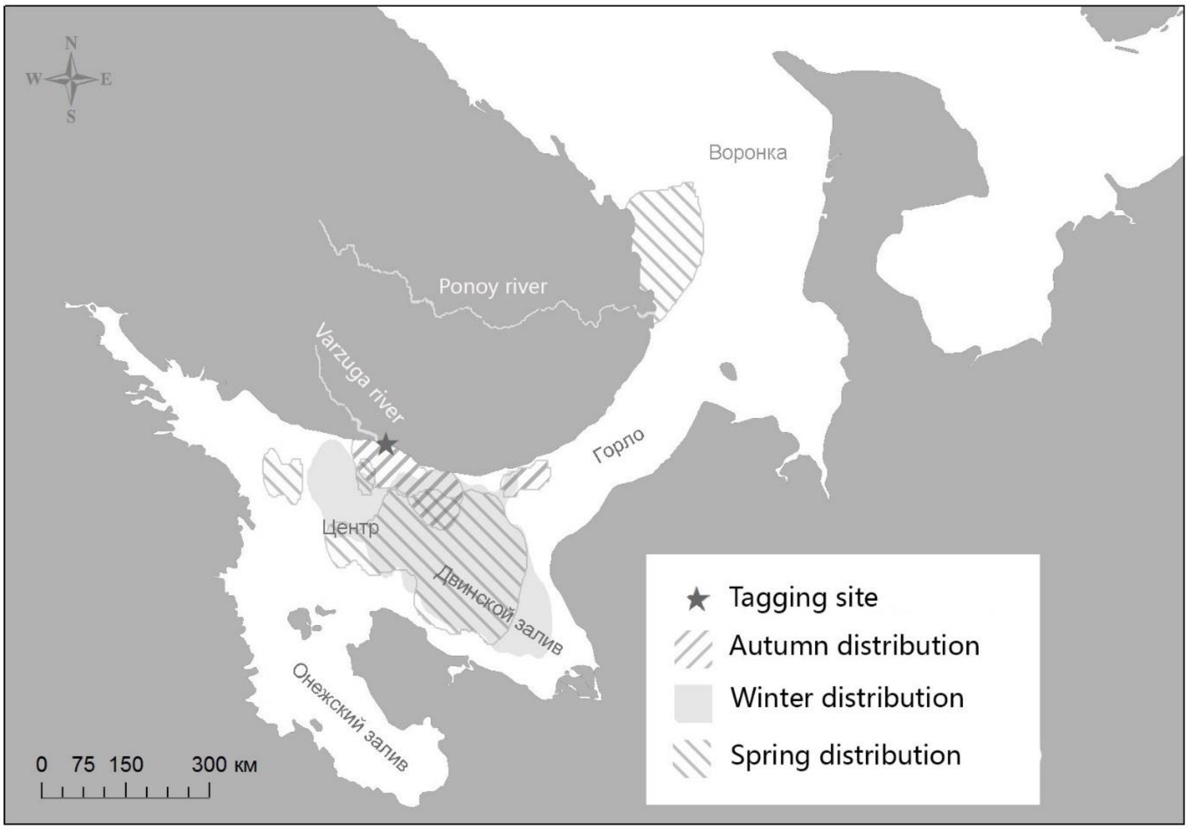

Figure 2. *Seasonal habitats (kernel 95%) of belugas tagged in 2010, 2011. Autumn: tagging date (October) – December; Winter: January –March; Spring – April – end of tagging.*

Genetic studies
Earlier published data based on analysis of 5 individuals (O'Corry-Crowe et al., 2010) did not allow to make a definite conclusion about the White Sea Beluga status due to insufficient sample size. Clustering analysis based on 8 microsatellite loci allele frequencies and carried out for the total sample of belugas from different North Pacific and Arctic regions did not reveal any difference between the White Sea and individuals from the Beaufort Sea, West Greenland and Svalbard (*Ibidem*).

The statements presented here are based on analysis of

-samples from 21(21)[6] individuals biopsied in the Varzuga River mouth (October of 2010-2012, all individuals were males, our data and samples provided by L. Mukhametov);
- samples from 25(22) belugas biopsied or found dead in Onega Bay (July-August of 2010-2016, 4(2) males, 20 females and 1(0) dead individual of unknown sex);
- samples of 4(4) females biopsied and 1(0) male found dead on the coast of Dvina Bay in August 2015.

The samples from Onega and Dvina Bays were provided by P.P. Shirshov Institute of Oceanology of RAS.

It is important that the male-skewed sample from the Varzuga River mouth and the female-skewed sample from Onega and Dvina Bays represent the real sex distribution observed in autumn (Varzuga: only adult males) and summer (the bays: reproductive aggregations with a definite female prevalence among the adult individuals).

[6] *here and below the first number is quantity of individuals analyzed for mtDNA sequence and the second (given in parenthesis) is number of specimens used for microsatellite loci alleles analysis*

As genetic markers we used allelic composition of 17 microsatellite loci (Cb1, Cb2, Cb4, Cb5, Cb8, Cb10, Cb11, Cb13, Cb14, Cb16, Cb17 – Buchanan et al., 1996; Ev37, Ev94 – Valsecchi, Amos, 1996; 415/416, 417/418, 464/465, 468/469 – Schlötterer et al., 1991), 559 bp sequence of mtDNA control region and complete (1140 bp) sequences mtDNA cytochrome *b* gene.

For comparative analysis, in addition to our data for Anadyr Gulf and Chukotka peninsula coast belugas, we used the data of analysis of the samples kindly provided by Mammal Genomic Resources Collection, University of Alaska Museum of the North: 10(8) individuals from the Eastern Chukchi Sea (Point Lay), 3(3) – off Little Diomede Island and 3(3) – from the Beaufort Sea; 5 samples from the Eastern Bering Sea (Norton Sound, for clustering analysis only).

Additionally, we used the published data (O'Corry-Crowe et al., 1997; 2010) on frequency of mtDNA control region (409 bp) haplotypes known for the White Sea (5 individuals), Svalbard (38 individuals), Eastern Chukchi Sea (103 individuals) and the Eastern Beaufort Sea (97 individuals).

The analysis of 17 microsatellite loci allele frequencies (Fst criterion, Arlequin 3.1 Software) showed that the White Sea belugas combined in a single sample (n=47) are significantly reproductively isolated from the sample representing Bering-Chukchi-Beaufort Seas region (n=93, including Anadyr Gulf - n=71, and Chukotka peninsula coast + Little Diomede Island + Point Lay + Beaufort Sea - n=22) with Fst = 0.09221 at p-level = 0.00000.

The same result was found for Onega Bay and Varzuga samples when each of them was compared to Anadyr Gulf and all other B-C-B region as independent samples. Furthermore, the lesser but significant level of isolation was found for Onega Bay and the Varzuga River mouth samples when compared to each other (Fst = 0.03689, p= 0.00010 Genetic diversity level for the two samples was found to be similar: mean values of allele numbers per locus were 4.824 ± 1.590 and 5.125 ± 1.586, and average gene diversity over loci - 0.631078 and 0.593086 respectively.

The Bayesian clustering approach (Structure v. 2.3.4 software) in case of using admixture model demonstrated genetic unity of all the White Sea belugas and their strong isolation from other regions (Fig.3 -C). But in case of assigning the White Sea belugas (as well as belugas of other seas) to a single population and using no admixture model, the genotypes of some individuals from the Varzuga river mouth and Dvina Bay were determined as more probably belonging to the B-C-B group than to the group of other belugas from the White Sea (Fig.3 - B).

We do not assume the possibility of direct gene flow between B-C-B region and the White Sea as a result of individual migrations, and should take into account that the small sample sizes from Dvina Bay and B-C-B region, possibly, led to incorrect results. On the other hand, we can not exclude restricted indirect gene flow resulting from migrations of some belugas between the White and Barents seas and other animals - between the Barents and other Arctic seas. To some extent, it is confirmed by the absence of genetic difference between the White Sea sample and the samples from other Arctic Seas found in G.O’Corry-Crowe with co-authors (2010).

In case the genetic pool of the White Sea belugas is – to some extent - affected by the High Arctic/Siberian Seas population(s), females summering in Onega Bay seem to be less exposed to this gene flow than the males who spend autumn near the Varzuga river mouth are. However, satellite tracking showed that at least some of Varzuga males as permanent White Sea residents. In any case, the two samples represent genetically different groups, and this was additionally confirmed by clustering analysis carried out for the White Sea belugas only (Meschersky et al., in prep.).

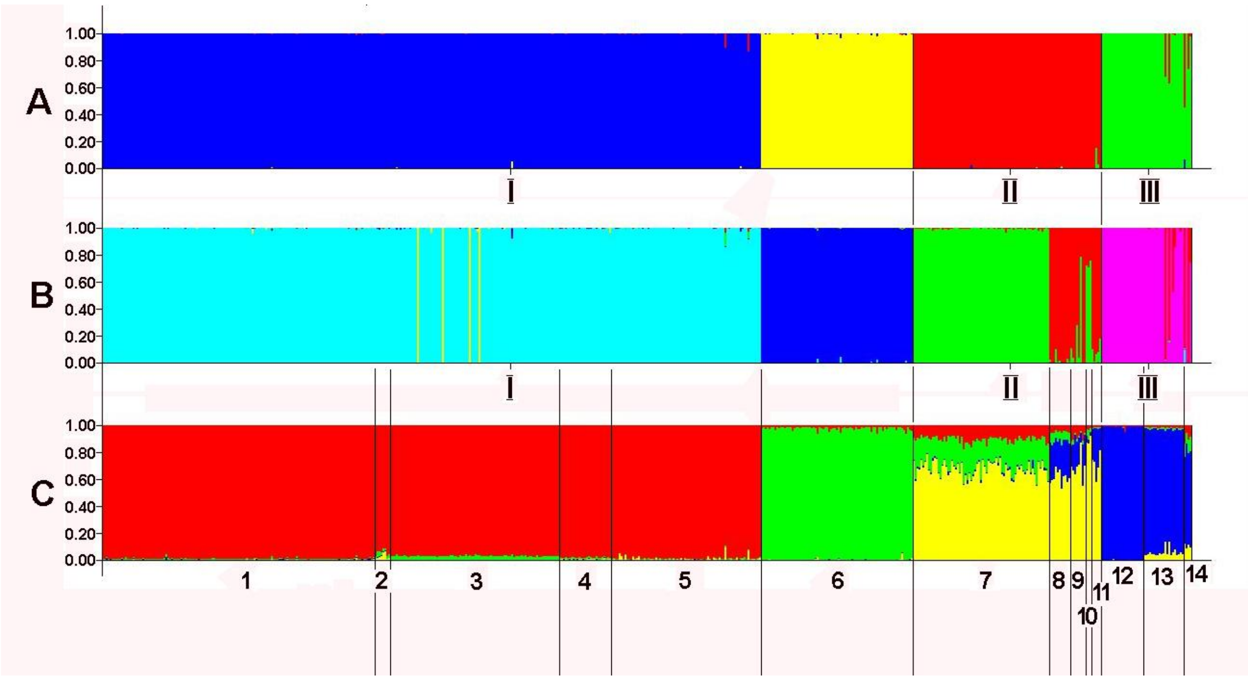

Figure 3. *The results of clustering analysis.*
A - locprior no admixture model for the layout where all individuals from each sea (I - the Okhotsk Sea, II - Bering-Chukchi-Beaufort Seas region, III - the White Sea) were "assigned" to a single population, resulted with K=4 as optimal (Evanno method, Structure Harvester online analysis).
B - the same model as A, resulted with K=6 in accordance with minimal Mean LnP(K) value.
C - locprior admixture model for the layout where the individuals from each location were «assigned» to a separate population, resulted with K=4 in accordance with minimal Mean LnP(K) value. Locations: 1-5 - Western Okhotsk Sea groups, 6 – Shelikhov Bay, 7 – Anadyr Liman, 8 – Chukotka peninsula coast together with 3 samples from Little Diomede Island, 9 - Point Lay, 10 - Beaufort Sea, 11 - Norton Sound, 12-14 - White Sea.

The analysis of mtDNA lineages occurrence and frequencies (Fst criterion - haplotype frequencies only, Arlequin 3.1 Software) based on 409 bp fragment of the control region showed that that both Onega Bay and Varzuga river mouth samples significantly differ from:

- Svalbard belugas (Fst= 0.20056 p= 0.00000 for Onega and Fst= 0.08734 p= 0.01049 for Varzuga), and
- belugas ofrom the Noth Pacific (Anadyr Gulf, Chukotka peninsula coastal waters, Eastern Chukchi Sea and Eastern Beaufort Sea samples: Fst= 0.12255-0.45546, p= 0.00000- 0.00238 for Onega and Fst= 0.05606-0.35474 p= 0.00000-0.03722 for Varzuga).

Nevertheless, a single haplotype - ***hp9*** – prevails in all the White Sea samples. This haplotype is known as widely spread across the Arctic seas: 50% for Svalbard sample, 34% for the West Greenland, 52% for the Eastern Beaufort Sea, - O'Corry-Crowe et al., 1997; 2010; 63% both in Anadyr Gulf and in Chukotka peninsula coast sample, - our data.

For Onega Bay, ***hp9*** was found for 24 of 25 analyzed animals (96%, and the single case of another sequence belonged to an individual found dead on an island located outside the main area of biopsy collection). So, the haplotypic diversity found for Onega Bay belugas was very low: 2 haplotypes, H= 0.080.

For the Varzuga river mouth, ***hp9*** frequency was found to be 76%, and haplotypic diversity value was higher: 3 haplotypes, H= 0.400. The difference in haplotype frequencies between the samples for this marker is statistically significant: Fst= 0.11681 p= 0.04257 (1 shared haplotype).

For the Dvina Bay the ***hp9*** was found for 4 of 5 animals (80%, H= 0.400), and no statistically significant differences were found between Dvina Bay and Svalbard and between Dvina and some of B-C-B samples (keeping in mind extremely small sample size). Haplotype ***hp53*** found in the White Sea by G.O'Corry-Crowe and co-authors (2010) was not present in our samples.

The use of a longer control region fragment (559 bp) did not affect the results in this case due to no nucleotide substitutions or indels were found for the additional part of control region sequence for the White Sea belugas. However, the use of cytochrome *b* sequence changed the result notably.

For the case of the two concatenated markers, 5 mtDNA haplotypes (H= 0.5099, and 3 haplotypes are unique for the sample) were found for belugas of Onega Bay sample and 4 (H= 0.4771, and 3 haplotypes are unique for the sample) for the Varzuga mouth sample.

Nevertheless, a single variant corresponding to ***C425*** (***=hp9*** for 409 bp) control region and ***CB07*** cytochrome b sequence (Meschersky et al., in prep.) predominated in all samples (64% in Onega Bay, 62% in Varzuga mouth and 80% in Dvina Bay samples).

The same sequence is major for Chukotka peninsula coast sample (50%) as well as for Anadyr Gulf (*Meschersky et al.*, in prep.), but was not found at all in B-C-B samples analyzed by us: 10 belugas from Point Lay, 3 from Little Diomede Island and 3 from the Beaufort Sea.

Most of the other haplotypes found for the White Sea samples, namely C425 control region in combination with other cytochrome *b* sequences, as well as other control region sequences *per se*, to our knowledge, were not found anywhere outside the White Sea.

Thus, based on maternal lineages composition and frequencies, the belugas of the White Sea are significantly isolated and should be regarded as a separate unit. Presumably, the White Sea beluga whale population is not uniform, but rather subdivided into subpopulations. However, the pattern of this subdivision as well as the level of gene flow between subpopulations and between each of them (and the total White Sea population) and High Arctic belugas remain poorly understood and require more future studies. For today, we propose to consider belugas from the White Sea as a single defined population.

2. Abundance

Based on a so-called expert estimate, the White Sea beluga population in summer was thought to be 2000-2500 whales (Belkovich, 2004). Other experts suggested that beluga abundance changes inter-annually and inter-seasonally, and varies from 300 to 2000-3000 (Matishov and Ognetov, 2006; Svetochev et al., 2002). All agree that the peak of the beluga presence falls on July.

Modern data on the White Sea beluga abundance in different seasons were obtained from the aerial surveys conducted in 2005 – 2011 (Glazov et al., 2008; 2010 a, b).

Summer abundance estimates are summarised in Table 2 and Fig. 4. Aerial surveys conducted in 6 years showed that the lowest summer abundance estimate of belugas was over 5000. The winter (March) beluga estimate was 3.5-4 times less than in July of the corresponding years (Table 3).

3. Anthropogenic removals

Beluga whale commercial harvest ended in the White Sea in 1980s (Matyshov and Ognetov, 2006). In recent years, beluga whales are live-captured in the Varzuga river mouth for scientific-research and cultural display purposes (not every year, exact numbers are unavailable, but usually not more than 5-6). No information is available on beluga illegal harvest by local people. If it takes place at all, it should not exceed «several» whales (expert opinion).

Table 2. *Summarized results of beluga aerial survey conducted in the White Sea in 2005-2008, 2010-2011 (from Solovyev et al., 2012: the abundance was estimated with program BELUKHA, the estimates in DISTANCE presented for method-comparison purposes)*

Dates of flights	2005 July 9, 10, 15, 16	2006 July 13,17,19, 20,22	2007 July 12, 13, 14, 17	2008 July 19, 20, 21, 22	2010 July 12,13,16, 17,19	2011 August 7,8,9,11,12, 13,14,15
Survey transect length (km)	3047	3161	3069	3790	3261	3304
Survey area (km2)	79112	82115	79917	89395	77534	73411
Total number of detected belugas (including calves)	1639 (115)	1559 (152)	1134 (52)	1197 (34)	987 (41)	897 (52)
Number of detected belugas on line transects (including calves)	765 (68)	585 (57)	367 (19)	543 (18)	638 (24)	707 (42)
Estimate of beluga number, BELUKHA, (CV%)	**7464** (17,0)	**5533** (14,6)	**5009** (14,1)	**6498** (16,4)	**7393** (19,3)	**5593** (13,5)
Estimate of beluga number DISTANCE, (CV%)	7010 (15,1)	4891 (18,7)	4527 (14,1)	6432 (15,7)	7488 (22,5)	5663 (16,5)

Table 3. *Results of aerial surveys in March 2008 and 2010 (from Glazov et al., 2010b).*

	2008	**2010**
Area of survey, km^2	24072	45071
Total number of detected belugas	134	237
Number of belugas on line transects	92	149
Abundance estimate (without availability correction)	**1665±634**	**2183±836**
CV	0,381	0,383
Mean density of detected belugas per 1000 km^2 of surveyed area	69	48

Total allowed takes for the White Sea, issued annually by the Ministry of Agriculture, have been 50 beluga whales for at least the last 5 years.

4. Incidental mortality

No information available.

4. Population trajectory

Reports on earlier surveys do not contain enough information on survey design and analysis methods as well as area coverage to enable comparison of the results for assessing the population trend. The estimates of 6 surveys conducted in 2005-2011 show a slight decline within this period, but the general pattern is rather undulating from year to year (fig. 4).

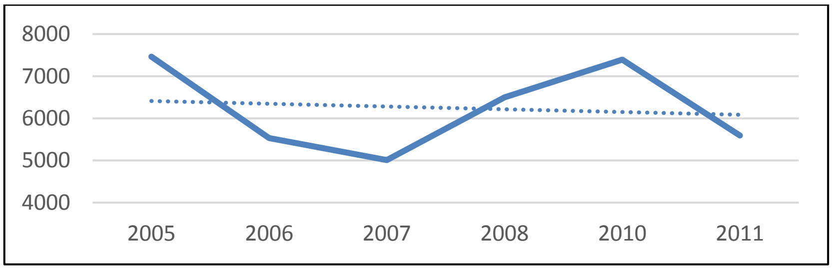

Figure 4. *Summer abundance estimates of belugas in the White Sea.*

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

No management information available, except for TATs issued annually.

6. Habitat and other concerns

- Direct disturbance of local reproductive aggregations by tourists and boat traffic
- Competition with fishermen
- Coastal oil storage bases and oil transport
- Pollution: mostly, via N. Dvina River discharge – wood-processing industry (cellulose-paper plants), companies of "energy sector", discharge from coastal cities and villages. For example, in 2004, river discharge into the White Sea contained 2351 tons of oil-products and 499 tons of phenols (Integrated State Information System on Situation in the World Ocean: http://esimo.oceanography.ru/esp1/index.php?sea_code=12&section=8&menu_code=4256).

7. Status of the stock

It is hard to assess the population trend, since there were no reliable abundance surveys in the past. No official status at the state level is assigned to this population (or any other beluga populations/stocks in Russia. A general expert opinion is that the White Sea stock should be considered **near threatened**, and due to increasing anthropogenic activity and high pollution levels, it should be closely monitored. Certain local reproductive groups, especially, a group concentrating near Bolshoy Solovetsky Island – Solovetskoe local aggregation – do require protection, first, due to an unregulated growing whale-watching industry. Belkovich (2002, 2006) believed that the White Sea is a "maternity home" for all European Arctic belugas, and that reproductive aggregations require protection at regional and federal level. IFAW has been supporting research of this and other reproductive White Sea beluga aggregations for over 10 years. In recent 4 years, Russian Geographic Society dedicated funds as well (at present, discontinued).

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Annex 19: Somerset Island Narwhal Stock

By: Cortney A. Watt

1. Distribution and stock identity

Stock identity is based on consistent summer aggregation reported in TEK, telemetry tracking, and aerial surveys. Summer distribution is indicated in blue and labeled SI on Figure 1. Stock identity is supported by telemetry studies which show narwhals tagged in Somerset Island stay within that region in the summer, and return there after spending the winter in the Baffin Bay region (Heide-Jørgensen et al. 2003). The Somerset Island stock over winters in a region slightly north compared to the other Baffin Bay narwhal stocks (Heide-Jørgensen et al. 2003, Dietz et al. 2008). There is some genetic support for the delineation of this stock as whales from Somerset Island separated out from the other Baffin Bay narwhal stocks in a multivariate analysis (Petersen et al. 2011). Stable isotopes show some discrimination between Somerset Island and Admiralty Inlet, Melville Bay, and East Baffin Island, but there is substantial overlap in the isotope values of whales from Somerset and the Eclipse Sound and Jones Sound stocks (Watt et al. 2012).

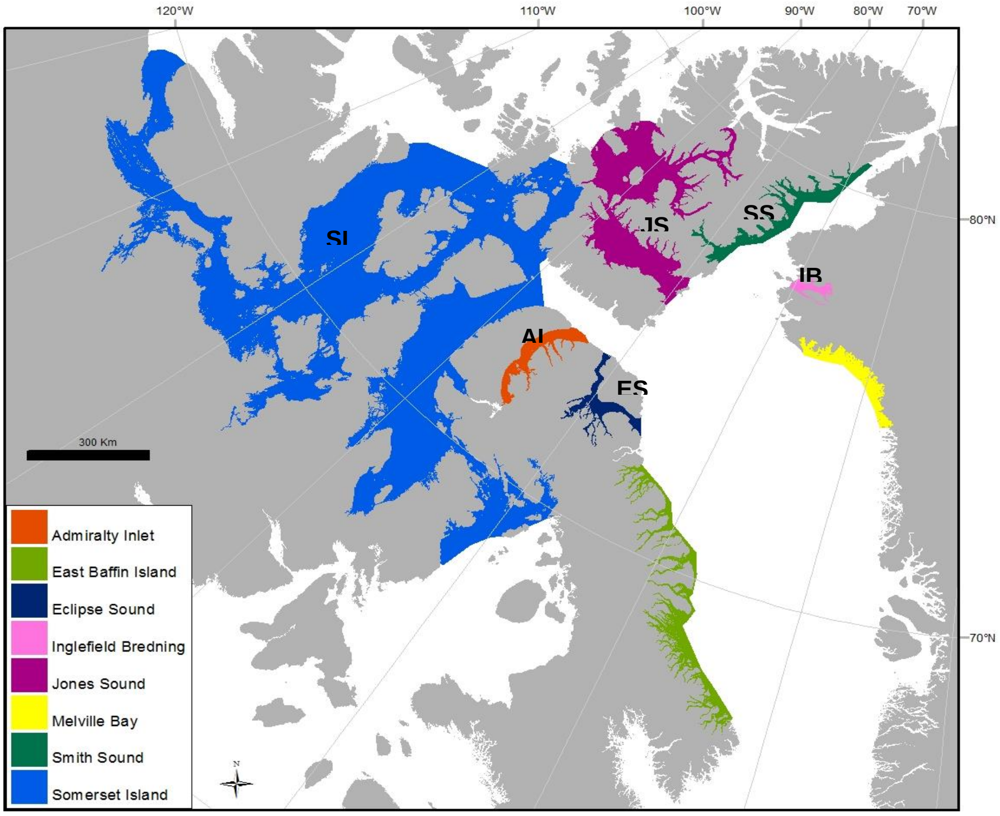

Figure 1. Map indicating the narwhal stocks for the Baffin Bay narwhal population.

2. Abundance

The most recent (2013) abundance estimate for this stock is 49,768 with a CV of 0.20 (Doniol-Valcroze et al. 2015).

This estimate comes from an aerial survey design using a double-platform. Three aircraft were used simultaneously to cover a large area encompassing all of the Canadian narwhal stocks of the Baffin Bay population in August 2013 (Doniol-Valcroze et al. 2015). The extent of the survey areas was based on previous aerial surveys, telemetry tracking studies, TEK, and recent observations by Inuit hunters. Since there has been recent concern about potential movement of narwhals among neighbouring summering regions, the survey with multiple aircraft was designed to survey six of the Baffin Bay stocks during the summer aggregation season - late July through the first three weeks of August prior to the start of fall migration movements. Dates of the survey were chosen to cover areas when sea ice ablation allowed for narwhal to access most of the Arctic Archipelago, and based on the timing of narwhal aggregations in their summering areas as described by TEK and satellite-telemetry data (Doniol-Valcroze et al. 2015). As a result, the last week of July and the first three weeks of August was chosen for the survey, with preference for earlier in August since telemetry data indicated that animals start to move among stocks during the final week of August (Watt et al. 2012).

Transect design was performed in Distance (version 6.1) using coastline shape files. The design was systematic with the first transect line chosen at random. When possible transect lines were oriented in a direction perpendicular to the longest axis of the stratum to maximize the number of lines per stratum (Doniol-Valcroze et al. 2015). For areas where it was assumed narwhal would be in high densities, systematic parallel transects were used. In areas where lower densities were anticipated and landscape patterns permitted, zigzag transects with equally spaced endpoints were used (Doniol-Valcroze et al. 2015).

The survey was flown at an altitude of 1,000 ft, and a target speed of 100 knots using three deHavilland Twin Otter 300 aircraft, each with 4 bubble windows on the sides and run as a double-platform experiment with independent observations platforms at the front and rear of the plane (Doniol-Valcroze et al. 2015). Dual camera systems were mounted under the belly of the plane to allow for continuous digital photography.

Distance sampling methods were used to estimate detection probability away from the track line, while mark-recapture methods were used on sighting data from two observers on either side of the aircraft to correct for perception bias. The distribution of perpendicular distances was different in fiord strata than in the other strata, and thus only non-fiord observations were used to fit the detection function for the non-fiord strata. Examination of the histogram of the perpendicular distances of unique sightings suggested right-truncating the data at 1000 m (i.e., discarding sightings beyond 1000 m), which left 762 unique observations (515 seen by primary observers, 523 by secondary observers, and 276 by both). The shape of the histogram suggested that some narwhals were missed close to the track line despite the bubble windows. Therefore, there was a risk that hazard-rate and half-normal distributions would overestimate the probability of detection and the resulting effective strip width. However, almost a hundred narwhal sightings were made within 100 m of the track line and therefore it seemed inappropriate to lose a large amount of data by left-truncating (i.e., discarding sightings close to the trackline). The shape of the histogram suggested that a gamma distribution would fit better, except that a gamma distribution takes the value zero at zero distance. Therefore, a gamma distribution with an offset term, in addition to half-normal and hazard rate keys, was fitted to the data (Doniol-Valcroze et al. 2015). Model selection was performed on all combinations of covariates (including environmental covariates such as ice cover, cloud cover, sea state, and glare, and a sighting rate covariate which was computed as a rolling average of the number of sightings made by the observer in a 30-second window prior to each sighing). The model with the lowest AIC was one with a truncated gamma key function and the covariates “sighting rate”, “Beaufort” and “glare”. The covariates reduced the detection distance at higher levels (Beaufort >3, Glare=intense, Sighting rate >10 in the last 30 seconds) and resulted in an average probability of detection of 0.48 (CV 2.8%) and an estimated effective strip half width of 481 m (not including perception bias) (Doniol-Valcroze et al. 2015).

For the mark-recapture model to estimate perception bias, models were performed with all combinations of environmental covariates as well as covariates “perpendicular distance”, “observer”, “sighting rate”, “side of aircraft” and “group size”. The best model included “perpendicular distance” and “sighting rate” and the overall probability of detecting a narwhal cluster between the track line and a distance of 1000 m was 0.40 (CV 4.2%) (Doniol-Valcroze et al. 2015).

Fiords were considered their own sampling units and cluster sampling was used to select the fiords to be surveyed (Doniol-Valcroze et al. 2015). In fiords, flights were continuous tracks designed to follow the main axis of the fiord while spreading coverage uniformly based on distance to shore. The resulting data from the fiords was analyzed separately from non-fiord strata. A density surface modelling framework was used to model spatially-referenced count data with the additional information provided by collecting distances to account for imperfect detection (Doniol-Valcroze et al. 2015). First a detection function was fitted to the perpendicular distance data to obtain detection probabilities for clusters of individuals (Doniol-Valcroze et al. 2015). Counts were then summarized for contiguous transect sections and a generalized additive model was constructed with segment counts as the response with areas corrected for detectability (Doniol-Valcroze et al. 2015).

Total surface abundance estimates for stocks were obtained by the additions of the estimated abundances of all the strata that made up that stock's summer range, including results from fiord strata. Variance for the stock-wide abundance estimate was calculated by adding the variances of each stratum; however, identification of duplicates was not straightforward due to the highly aggregated nature of narwhal groups. Because of this, a sensitivity analysis was used to quantify the uncertainty, which allowed the researchers to include an additional variance component to the surface abundance estimate with a CV equal to that of the sensitivity analysis, which ultimately increased the range of uncertainty around the estimate but left the point estimate unchanged (Doniol-Valcroze et al. 2015).

An availability bias correction was also applied to the survey data. For the availability bias correction, the time at depth for 24 narwhals fitted with satellite tags near Arctic Bay and Pond Inlet every August from 2009-2012 (Watt et al. 2015) was used to determine the correction for the number of whales missed as a result of being at depth and unavailable for viewing by the surveyors. The time narwhals spent at 0-2 m depths was used to calculate a correction for areas with clear water, while areas with very murky water, the time spent within 0-1 m of the surface was used. This resulted in a correction factor of 3.18 ± 3.37 for clear water areas and a correction of 4.90 ± 0.187 for murky regions (Watt et al. 2015). This correction is appropriate when sightings are instantaneous, but if they are not (such as in aerial surveys), it can positively bias the estimate and as a result a correction factor incorporating the dive cycle of the animal is needed. Three archival time-depth recorders deployed on whales near Pond Inlet and in Creswell Bay in August 1999 and 2000 were used to evaluate a dive-cycle for narwhals. A weighted availability bias correction factor that took into account both the time at depth and the time in view (dive-cycle) was used (2.94 ± 3.4 for the 0-2 m correction and 4.53 ± 3.8 for the 0-1 m bin).

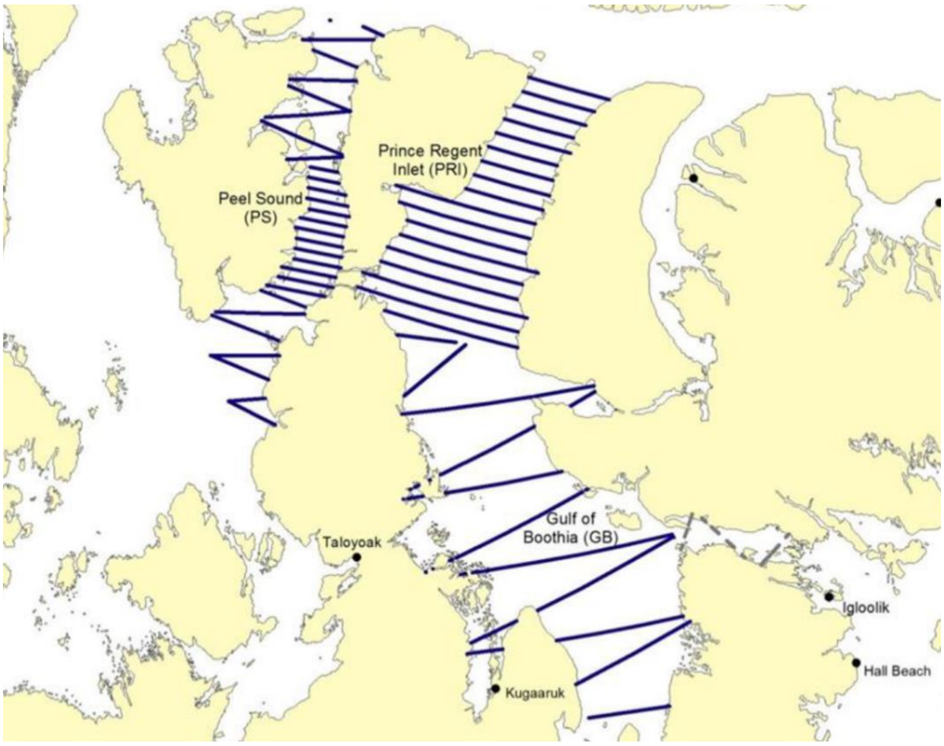

Figure 2. Map of the surveyed strata for the Somerset Island stock. Blue lines indicate surveyed transect and blue areas indicate surveyed fiords (Doniol-Valcroze et al. 2015).

For Somerset Island all strata (Figure 2) three aircraft were flown simultaneously, and both Peel Sound and Prince Regent Inlet were each surveyed in a single day (August 5, 2013 and August 9, 2013, respectively). The Gulf of Boothia was covered a week later on August 15, 2013 and August 16, 2013. Narwhals were aggregated at the southern end of Prince Regent Inlet and in the northern part of the Gulf of Boothia. Despite heavy ice cover, numerous narwhals were observed in the central area of Peel Sound (Doniol-Valcroze et al. 2015). The surface abundance estimate for the Somerset Island stock was 16,921 ± 0.20, and after viewing the photos it was deemed that the water was clear and a correction for the 0-2 m bin should be applied. After a weighted correction of 2.94 ± 0.03 was applied the resulting abundance estimate was 49,768 ± 0.20 (Doniol-Valcroze et al. 2015).

3. Anthropogenic removals

This stock is primarily hunted on the summering grounds in the central Canadian Arctic by the communities of Gjoa Haven, Hall Beach, Igloolik, Kugaaruk, Resolute & Creswell Bay, and Taloyoak (Heide-Jørgensen et al. 2013); however, there is opportunity for hunters from other communities to hunt these whales on their migration to and from the summering grounds and on the wintering grounds (Witting 2016). Catches in Table 1, however, only reflect whales that are hunted within the defined summering region since it is difficult to determine the number of animals from this stock hunted by other communities. In some Canadian communities with a community-based management system, killed-lost and wounded-lost narwhal numbers were documented by hunters between 1999 and 2005 (Table 2). From the narwhal hunts where losses are reported, Richard (2008) calculated a hunting loss rate correction (LRC) (Table 2).

LRC = HM / LC where

HM = the estimated total hunting mortality, or the sum of the landed catch and hunting loss
LC = Landed Catch
The estimated hunting loss was calculated as:
$HM = (HM_{min} + HM_{max})/2$ where
HM_{min} = number of animals landed plus the ones reported sunk and lost
$HM_{max} = HM_{min}$ + the number reported wounded and escaped

This HM estimate used by Richard (2008) assumes that half of the animals wounded and escaped later die from their injuries. This assumption was untested but considered reasonable since both whales with wound scars are later seen alive but dead whales have also washed up after a hunt suggesting some whales survive from their wounds while others perish (Richard 2008). Table 1 indicates the total reported landed catches, and the catches multiplied by a struck and loss factor of 1.28 ± 0.15 (Richard 2008). This data comes from 1999-2005 and is hunter reported for all types of hunt combined for each of the communities. An older study (Roberge and Dunn 1990) investigated struck and lost rates from the community of Arctic Bay in the open water season in 1983 and 1988, on the floe edge in 1988 and 1989, and at the ice crack in 1978, 1988, and 1999 (Table 3). Most of the hunt in Somerset Island occurs in the open water season, which has a struck and loss factor reported by Roberge and Dunn (1990) of 1.40 ± 0.14. In this study researchers monitored the hunt when possible and reported values. Application of this rate rather than the 1.28 reported by Richard (2008), changes catches previous to 1999 by an average of 3 whales, and a maximum of 7 whales (results in brackets in Table 1). Ideally a struck and loss factor would be applied to each catch that occurs through different hunting methods; unfortunately this information is not reported. However based on hunt dates (for which we have some information from 2003-2012 for two of the communities that hunt from the Somerset Island stock), the majority of the hunt occurs in the open water season (98% for Kugaaruk and 88% for Resolute (Doniol-Valcroze 2014)). Currently in Canada the struck and loss rate from Richard (2008) is used, since it is the most up to date.

Table 1. Reported landed catches for the Somerset Island stock from the communities of Gjoa Haven, Hall Beach, Igloolik, Kugaaruk, Resolute & Creswell Bay, and Taloyoak. From 1977 these catches are based on the number of issued tags and recorded by Fisheries and Oceans Canada; prior to 1977 the numbers come from a variety of sources (see reference list) but typically rely on reports by hunters, or RCMP records. Total catch including struck and lost animals is indicated using the newest struck and lost factor (1.28 from Richard (2008)), and using the 1.40 reported for open water hunts by Roberge and Dunn (1990) for years prior to 1999 indicated in brackets.

Year	Gjoa Haven (landed catches)	Hall Beach (landed catches)	Igloolik (landed catches)	Kugaaruk (landed catches)	Resolute & Creswell Bay (landed catches)	Taloyoak (landed catches)	Somerset Island Total (landed catches)	Reference for reported landed catch	Somerset Island Catches + 1.28 S&L factor (1.40 S&L factor)
1970	nr	nr	nr	nr	nr	nr	nr	Mansfield et al. (1975)	nr
1971	nr	nr	nr	nr	nr	nr	nr	Mansfield et al. (1975)	nr
1972	nr	nr	nr	nr	nr	nr	nr	Strong (1989), Mitchell and Reeves (1981)	nr
1973	nr	40	10	nr	4	nr	54	Strong (1989), Mansfield et al. (1975)	69 (76)
1974	nr	nr	nr	nr	nr	nr	nr	Strong (1989), Stewart (2007)	nr
1975	nr	nr	nr	7	nr	nr	7	Strong (1989)	9 (10)
1976	nr	nr	nr	nr	15	nr	15	Strong (1989)	19 (21)
1977	nr	13	0	nr	13	nr	26	Strong (1989)	33 (36)
1978	0	0	0	0	14	0	14	Strong (1989)	18 (20)
1979	0	2	0	0	2	0	4	Strong (1989)	5 (6)
1980	0	11	14	0	nr	0	25	Strong (1989)	32 (35)
1981	0	17	36	0	nr	0	53	Strong (1989)	68 (74)
1982	0	7	25	0	14	0	46	Strong (1989)	59 (64)
1983	22	1	18	0	11	5	57	Strong (1989)	73 (80)
1984	0	0	0	0	0	0	0	Strong (1989)	0 (0)
1985	2	2	4	10	2	0	20	Strong (1989)	26 (28)
1986	0	0	1	0	10	0	11	Strong (1989)	14 (15)
1987	0	0	0	0	12	0	12	Strong (1989)	15 (17)
1988	2	0	0	0	12	0	14	DFO (1991)	18 (20)

1989	0	3	0	1	19	0	23	DFO (1992)	29 (32)
1990	0	0	0	0	22	0	22	DFO (1992)	28 (31)
1991	0	0	0	0	21	7	28	DFO (1993)	36 (39)
1992	0	1	25	0	0	0	26	DFO (1994)	33 (36)
1993	0	0	27	0	8	0	35	DFO (1995)	45 (49)
1994	0	6	25	0	3	0	34	DFO (1996)	44 (48)
1995	0	nr	18	5	4	0	27	DFO (1997)	35 (38)
1996	0	1	5	7	3	0	16	DFO (1999)	20 (22)
1997	0	2	3	15	7	0	27	Stewart (2007)	35 (28)
1998	0	11	25	8	9	0	53	Doniol-Valcroze (2014)	68 (74)
1999	0	0	4	0	14	0	18	Doniol-Valcroze (2014)	23
2000	0	1	5	25	9	0	40	Doniol-Valcroze (2014)	51
2001	1	7	10	37	11	10	76	Doniol-Valcroze (2014), Stewart (2007)	97
2002	0	9	0	17	9	10	45	Doniol-Valcroze (2014)	58
2003	0	2	1	24	2	1	30	Doniol-Valcroze (2014), Stewart (2007)	38
2004	0	11	25	16	4	0	56	Hall et al. (2015)	72
2005	0	3	24	20	16	0	63	Hall et al. (2015)	81
2006	0	1	25	48	28	33	135	Hall et al. (2015)	173
2007	1	0	1	40	9	0	51	Hall et al. (2015)	65
2008	0	0	0	35	10	3	48	Hall et al. (2015)	61
2009	1	0	1	42	16	5	65	Hall et al. (2015)	83
2010	1	2	0	45	9	2	59	Hall et al. (2015)	76
2011	1	1	0	50	4	1	57	Hall et al. (2015)	73
2012	0	1	0	43	16	5	65	Hall et al. (2015)	83
2013	0	3	3	43	0	3	52	Hall et al. (2015)	67
2014	0	2	0	135	0	0	137	Hall et al. (2015)	175
2015	2	0	0	63	4	4	73	Hall et al. (2015)	93

Table 2. Table indicating how the struck and loss factor for this stock is calculated. Table is directly from Richard (2008).

Community	Year	Landed	Wounded/ Escaped	Sunk and Lost	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Community specific average Loss Rate Correction
Pond Inlet	1999	130	14	16	146	160	153	1.18	
	2000	166	21	10	176	197	187	1.12	
	2001	63	5	27	90	95	93	1.47	
	2002	92	1	13	105	106	106	1.15	1.23 ± 0.16
Qikiqtarjuaq	1999	81	30	25	106	136	121	1.49	
	2000	137	79	40	177	256	217	1.58	
	2001	89	8	9	98	106	102	1.15	
	2002	81	40	16	97	137	117	1.44	
	2004	96	12	9	105	117	111	1.16	1.36 ± 0.20
Repulse	1999	156	68	30	186	254	220	1.41	
	2000	49	9	5	54	63	59	1.19	
	2001	100	38	21	121	159	140	1.4	
	2002	57	0	8	65	65	65	1.14	
	2003	30	0	5	35	35	35	1.17	
	2005	72	25	3	75	100	88	1.22	1.26 ± 0.12
Arctic Bay	2001	134	20	4	138	158	148	1.1	
	2003	129	14	22	151	165	158	1.22	
	2004	122	22	33	155	177	166	1.36	1.23 ± 0.13
Kugaaruk	2001	41	18	8	49	67	58	1.41	

	2003	24	4	2	26	30	28	1.17	1.29 ± 0.17
Average across communities									**1.28 ± 0.15**

Table 3. Table indicating how an older struck and loss factor for Arctic Bay was calculated from observations of different hunting types from Roberge and Dunn (1990).

Hunt	Year	Landed	Wounded/ Escaped	Sunk and Lost/mortally wounded	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Hunt specific average Loss Rate Correction
Floe edge	1988	6	6	8	14	20	17	2.83	
	1989	16	0	5	21	21	21	1.31	2.07 ± 1.08
Open water	1983	4	2	1	5	7	6	1.50	
	1988	13	6	1	14	30	17	1.31	**1.40 ± 0.14**
Ice crack	1987	15	13	8	23	36	30	1.97	
	1988	29	8	17	46	54	50	1.72	
	1989	50	7	13	63	70	67	1.33	1.67 ± 0.32
Average across hunt types									**1.71 ± 0.55**

The stock is hunted on the wintering grounds in Greenland where 97% of the hunt in Uummannaq is believed to be from the Somerset Island stock.

Table 4. Catches of narwhals from official reports by municipality with corrections for under-reportings (in parenthesis) for 1954 to 2011. Numbers in square brackets are from *special reports*. The column 'under-reporting' shows the sum of the corrections for under-reporting or 'ALL' if it is a general correction factor for all areas. 'na' means that no data are available. Data from 2007-08 are preliminary. DB=Disko Bay, UUM=Uummannaq, UPV=Upernavik. Data were compiled from Prime Minister's Second Department (1951), Kapel (1977), Kapel (1983), Kapel and Larsen (1984), Kapel (1985), Born and Kapel (1986) and Born (1987).

YEAR	QAANAAQ	UPER-NAVIK	UUMMAN-NAQ	DISKO BAY	SISI-MIUT	MANIIT-SOQ	NUUK	PAAMIUT-QAQORTOQ	TOTAL	ICE ENTRAPMENT
1949	38	16	1	6					61	
1950										
1951										85 DB
1952										450 DB
1954	na	45			1			1	47	
1955	na	179	2	14					195	
1956	na	15	282	21					318	156 UPV, 250 UUM
1957	na	55	11	15					81	
1958	na	24	3	45		1			73	
1959	na	32	8	16				1	57	
1960	na	25	296	7	1	1	1	1	332	
1961	134	25	5	38				1	203	272 UUM
1962	182	17	11	12				1	213	
1963	275	10	3	29					317	
1964	275	17	11	11					314	
1965	na	33	37	33	1	1			105	
1966	na	39	23	43		3	2		110	
1967	na	131			9				140	31 DB
1968	na	454			18				472	161 DB, 50 UPV, 84 UUM
1969	na	174			30				204	Some DB, 50 UPV
1970	na	313			9				322	100 DB
1971	na	146			40				186	
1972	na	84			23				107	
1973	na	191			8				199	
1974	8	136			3				147	

Table 4. Continued

YEAR	QAANAAQ	UPER-NAVIK	UUMMAN-NAQ	DISKO BAY	SISIMIUT	MANIITSOQ	NUUK	PAAMIUT-QAQORTOQ-	TOTAL	ICE ENTRAPMENT
1975	1	54	11	44		6		1	266 (149)	
1976	9	22	27	57					264 (141)	
1977	16	62	113	53	8	1			387 (134)	
1978	110	56	183	262		1			612	
1979	120	22	132	100			3		377	
1980	130	61	146	120		4	1		462	
1981	118	83	140	249			18	1	609	
1982	164	59	162	76					461	45 DB
1983	135 (25)	72 (30)	164	68 (10)					439 (65)	
1984	274	80	245	66 (15)	1				666 (15)	35 UUM
1985	115 (115)	34 (20)	39	67		1			256 (135)	
1986	na	81	97	23		36			237	
1987	na	145	334	25			1		505	
1988	na		206						500 (294)	
1989	na	37	288	2			5		332	
1990	na	100 (73)	1019	11					1057 (100)	
1991	na		27	> 40					na	27 UUM
1992	na	37	288	2			5		342	
1993	144	66	301	75	10	6	4	8	614	
1994	183	59	297	268	6	14	7	11	845	150 DB
1995	107	94	159	108	4	5	8		485	
1996	45	69	405	154	10	4	2	2	691	
1997	66	90	381	156	13	5	9	26	746	
1998	94	105	344	163	21	18	6	24	775	
1999	115	119	253	174	28	24	17	15	745	
2000	109	150	106	155	27	8	0	6	561	
2001	145	155	95	119	1	2	15	3	535	
2002	94	164	180	97	12	11	3	2	563	
2003	113	146	174	114	4	0	2	2	554	
2004	178	53	67	73	2	1	0	0	374	
2005	[70] 137	[74] 71	[137] 161	[47] 39	0	0	0	0	[328] 408	
2006	[94] 99	[58] 62	[55] 72	[4] 53	1	2	0		[211] 289	
2007	[21] 139	[17] 102	[52] 67	[56] 63	0	2	0	1	[146] 374	
2008	129	74	87	47	0	0	0	0	337	

2009	90	110	91	88	0	0	0	1	380	41 in Qaanaaq
2010	108	30	42	45	0	0	0	0	225	53 in Qaanaaq
2011	74	60	77	39	0	0	0	1	251	
2012	144	70	42	179	0	0	0	1	311	125 at Kangersuatsiaq
2013	90	64	78	50	0	0	0	1	283	
2014	114	101	69	50	0	0	0	0	334	
2015	92	54	42	29	0	0	1	0	218	
2016	93	79	120	55	0	0	1	0	348	

4. Population trajectory

Four surveys with the goal of assessing abundance have been conducted over the past 30 years for the Somerset Island stock. Figure 3 indicates the trajectory given the abundance estimates and associated confidence intervals for the different surveys. The 1981 survey had a correction of 2.92 (CV = 0.45) applied in order to make it compatible with later surveys that included corrections for perception and availability bias (Richard et al. 2010). Based on the confidence intervals alone, there is no significant change in the abundance estimates over time. The estimated trajectory for the stock comes from a population dynamic model based on a Bayesian modelling framework that is age and sex structured (for details see Witting 2016). Population trend suggests population is increasing slightly, but population estimates are quite variable across years, and abundance estimates have large confidence intervals (Witting 2016).

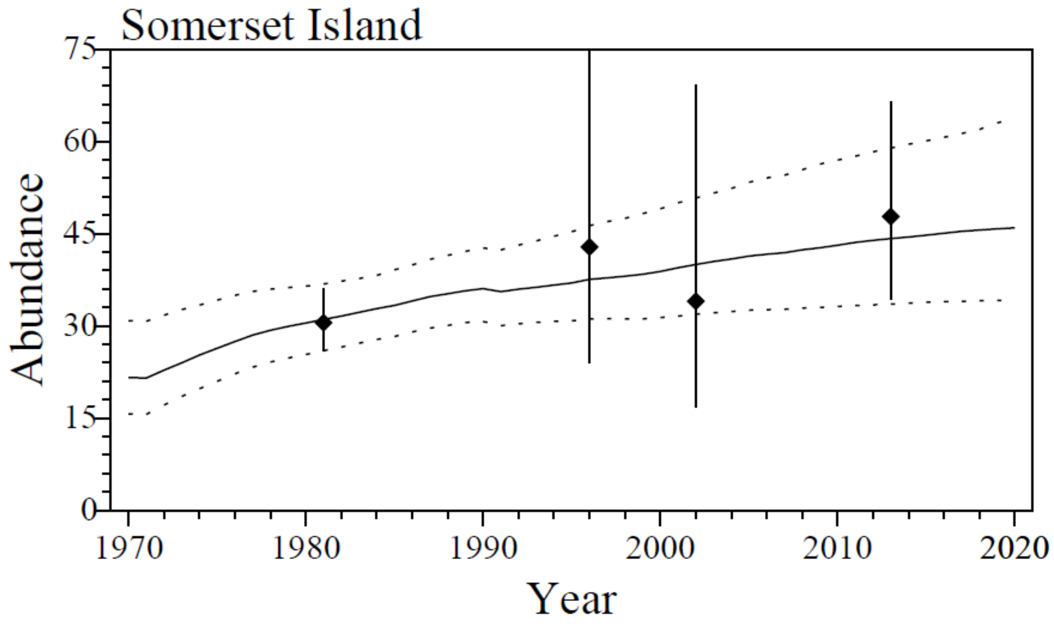

Figure 3. Population trajectory for the Somerset Island stock. Points represent abundance estimates (given in thousands) with 90% confidence intervals. Solid curves indicate the median, and dotted curves the 90 % confidence interval for the estimated models (Witting 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR) method (Wade 1998), corrected for hunting losses (struck and lost), is used to calculate a recommended Total Allowable Landed Catch (TALC):
TALC = PBR/ LRC

Where:

$$PBR = 0.5 * R_{max} * N_{min} * F_r$$

LRC is the hunting loss rate correction and is equal to 1.28 ± 0.15 (Richard 2008). R_{max} is the maximum rate of increase for the stock (unknown so the default for cetaceans of 0.04 is used, N_{min} is the 20th percentile of the log-normal distribution of N (most recent survey estimate), and F_r is the recovery factor (we used a value of 1 which indicates a healthy status for the stock (an assumption)). Therefore, the current TALC is set at 532 for this stock, based on the abundance estimate from the 2002 survey. The new TALC recommendation (which has not yet been implemented) based on the 2013 aerial survey results is 658 (Doniol-Valcroze et al. 2015).

6. Habitat and other concerns

Stock needs to be managed carefully since it is hunted throughout the year by both Canada and Greenland by many different communities. Satellite tagging studies have not been completed since 2001.

7. Status of the stock.

Large stock size that appears to be stable or increasing, but population estimates have large confidence intervals. Current removals are considered to be sustainable (Doniol-Valcroze et al. 2015).

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Annex 20: Jones Sound Narwhal Stock

By: Cortney A. Watt

1. Distribution (provide a map if possible) and stock identity

Stock identity is based on consistent summer aggregation reported in TEK. Summer distribution is indicated in purple on and labelled JS on Figure 1. There have been no tagging studies done on whales from the Jones Sound Stock, although a tagging project was in place there from 2013-2015 no whales were successfully captured or equipped (whales were seen in 2 of the 3 years). Narwhals from the Jones Sound stock separate genetically from other Canadian stocks and from samples from Inglefield Bredning, which are the closest geographically (Petersen et al. 2011) and organochlorine contaminants were notably different for whales hunted in Grise Fiord (de March and Stern 2003). Stable isotopes on skin from narwhals hunted in Grise Fiord from the Jones Sound Stock also show discrimination from all other Baffin Bay narwhal stocks other than the Eclipse Sound and Somerset Island stocks (Watt et al. 2012).

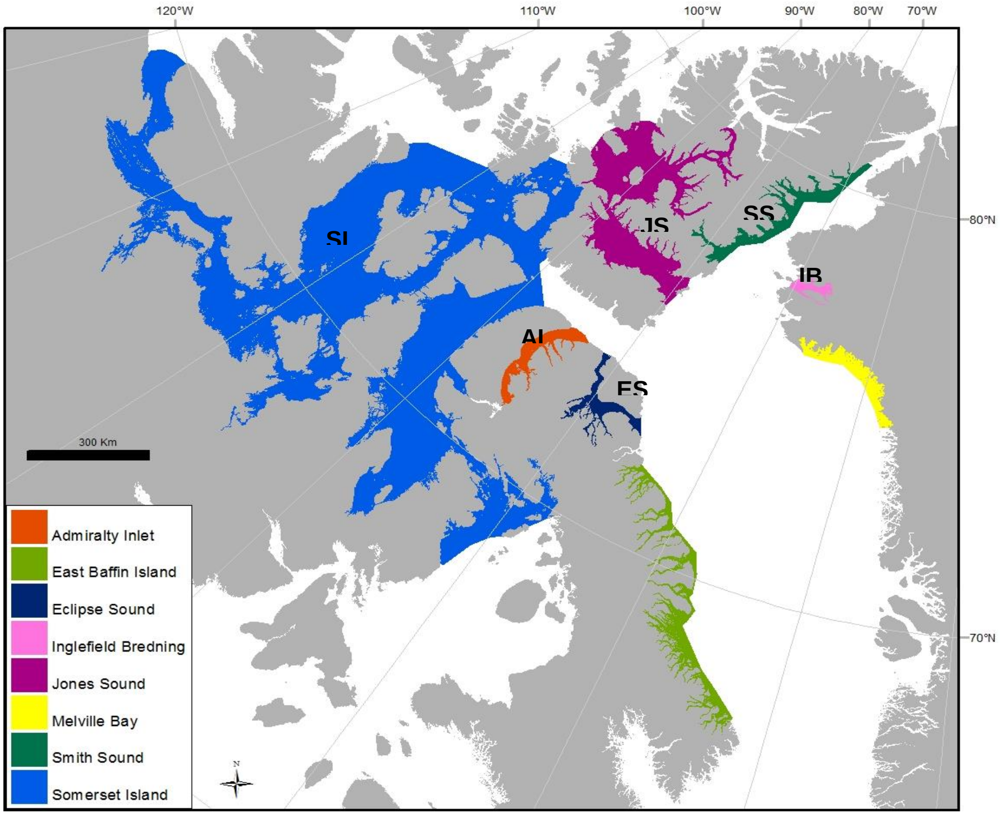

Figure 1. Map indicating the narwhal stocks for the Baffin Bay narwhal population.

2. Abundance

The only abundance estimate for this stock (2013) is 12,694 with a CV of 0.33 (Doniol-Valcroze et al. 2015).

This estimate comes from an aerial survey design using a double-platform. Three aircraft were used simultaneously to cover a large area encompassing all of the Canadian narwhal stocks of the Baffin Bay population in August 2013 (Doniol-Valcroze et al. 2015). The extent of the survey areas was based on previous aerial surveys, telemetry tracking studies, TEK, and recent observations by Inuit hunters. Since there has been recent concern about potential movement of narwhals among neighbouring summering regions, the survey with multiple aircraft was designed to survey six of the Baffin Bay stocks during the summer aggregation season - late July through the first three weeks of August prior to the start of fall migration movements. Dates of the survey were chosen to cover areas when sea ice ablation allowed for narwhal to access most of the Arctic Archipelago, and based on the timing of narwhal aggregations in their summering areas as described by TEK and satellite-telemetry data (Doniol-Valcroze et al. 2015). As a result, the last week of July and the first three weeks of August was chosen for the survey, with preference for earlier in August since telemetry data indicated that animals start to move among stocks during the final week of August (Watt et al. 2012).

Transect design was performed in Distance (version 6.1) using coastline shape files. The design was systematic with the first transect line chosen at random. When possible transect lines were oriented in a direction perpendicular to the longest axis of the stratum to maximize the number of lines per stratum (Doniol-Valcroze et al. 2015). For areas where it was assumed narwhal would be in high densities, systematic parallel transects were used. In areas where lower densities were anticipated and landscape patterns permitted, zigzag transects with equally spaced endpoints were used (Doniol-Valcroze et al. 2015).

The survey was flown at an altitude of 1,000 ft, and a target speed of 100 knots using three deHavilland Twin Otter 300 aircraft, each with 4 bubble windows on the sides and run as a double-platform experiment with independent observations platforms at the front and rear of the plane (Doniol-Valcroze et al. 2015). Dual camera systems were mounted under the belly of the plane to allow for continuous digital photography.

Distance sampling methods were used to estimate detection probability away from the track line, while mark-recapture methods were used on sighting data from two observers on either side of the aircraft to correct for perception bias. The distribution of perpendicular distances was different in fiord strata than in the other strata, and thus only non-fiord observations were used to fit the detection function for the non-fiord strata. Examination of the histogram of the perpendicular distances of unique sightings suggested right-truncating the data at 1000 m (i.e., discarding sightings beyond 1000 m), which left 762 unique observations (515 seen by primary observers, 523 by secondary observers, and 276 by both). The shape of the histogram suggested that some narwhals were missed close to the track line despite the bubble windows. Therefore, there was a risk that hazard-rate and half-normal distributions would overestimate the probability of detection and the resulting effective strip width. However, almost a hundred narwhal sightings were made within 100 m of the track line and therefore it seemed inappropriate to lose a large amount of data by left-truncating (i.e., discarding sightings close to the trackline). The shape of the histogram suggested that a gamma distribution would fit better, except that a gamma distribution takes the value zero at zero distance. Therefore, a gamma distribution with an offset term, in addition to half-normal and hazard rate keys, was fitted to the data (Doniol-Valcroze et al. 2015). Model selection was performed on all combinations of covariates (including environmental covariates such as ice cover, cloud cover, sea state, and glare, and a sighting rate covariate which was computed as a rolling average of the number of sightings made by the observer in a 30-second window prior to each sighing). The model with the lowest AIC was one with a truncated gamma key function and the covariates "sighting rate", "Beaufort" and "glare". The covariates reduced the detection distance at higher levels (Beaufort >3, Glare=intense, Sighting rate >10 in the last 30 seconds) and resulted in an average probability of detection of 0.48 (CV 2.8%) and an estimated effective strip half width of 481 m (not including perception bias) (Doniol-Valcroze et al. 2015).

For the mark-recapture model to estimate perception bias, models were performed with all combinations of environmental covariates as well as covariates "perpendicular distance", "observer", "sighting rate", "side of aircraft" and "group size". The best model included "perpendicular distance" and "sighting rate" and the overall probability of detecting a narwhal cluster between the track line and a distance of 1000 m was 0.40 (CV 4.2%) (Doniol-Valcroze et al. 2015).

Fiords were considered their own sampling units and cluster sampling was used to select the fiords to be surveyed (Doniol-Valcroze et al. 2015). In fiords, flights were continuous tracks designed to follow the main axis of the fiord while spreading coverage uniformly based on distance to shore. The resulting data from the fiords was analyzed separately from non-fiord strata. A density surface modelling framework was used to model spatially-referenced count data with the additional information provided by collecting distances to account for imperfect detection (Doniol-Valcroze et al. 2015). First a detection function was fitted to the perpendicular distance data to obtain detection probabilities for clusters of individuals (Doniol-Valcroze et al. 2015). Counts were then summarized for contiguous transect sections and a generalized additive model was constructed with segment counts as the response with areas corrected for detectability (Doniol-Valcroze et al. 2015).

Total surface abundance estimates for stocks were obtained by the additions of the estimated abundances of all the strata that made up that stock's summer range, including results from fiord strata. Variance for the stock-wide abundance estimate was calculated by adding the variances of each stratum; however, identification of duplicates was not straightforward due to the highly aggregated nature of narwhal groups. Because of this, a sensitivity analysis was used to quantify the uncertainty, which allowed the researchers to include an additional variance component to the surface abundance estimate with a CV equal to that of the sensitivity analysis, which ultimately increased the range of uncertainty around the estimate but left the point estimate unchanged (Doniol-Valcroze et al. 2015).

An availability bias correction was also applied to the survey data. For the availability bias correction, the time at depth for 24 narwhals fitted with satellite tags near Arctic Bay and Pond Inlet every August from 2009-2012 (Watt et al. 2015) was used to determine the correction for the number of whales missed as a result of being at depth and unavailable for viewing by the surveyors. The time narwhals spent at 0-2 m depths was used to calculate a correction for areas with clear water, while areas with very murky water, the time spent within 0-1 m of the surface was used. This resulted in a correction factor of 3.18 ± 3.37 for clear water areas and a correction of 4.90 ± 0.187 for murky regions (Watt et al. 2015). This correction is appropriate when sightings are instantaneous, but if they are not (such as in aerial surveys), it can positively bias the estimate and as a result a correction factor incorporating the dive cycle of the animal is needed. Three archival time-depth recorders deployed on whales near Pond Inlet and in Creswell Bay in August 1999 and 2000 were used to evaluate a dive-cycle for narwhals. A weighted availability bias correction factor that took into account both the time at depth and the time in view (dive-cycle) was used (2.94 ± 3.4 for the 0-2 m correction and 4.53 ± 3.8 for the 0-1 m bin).

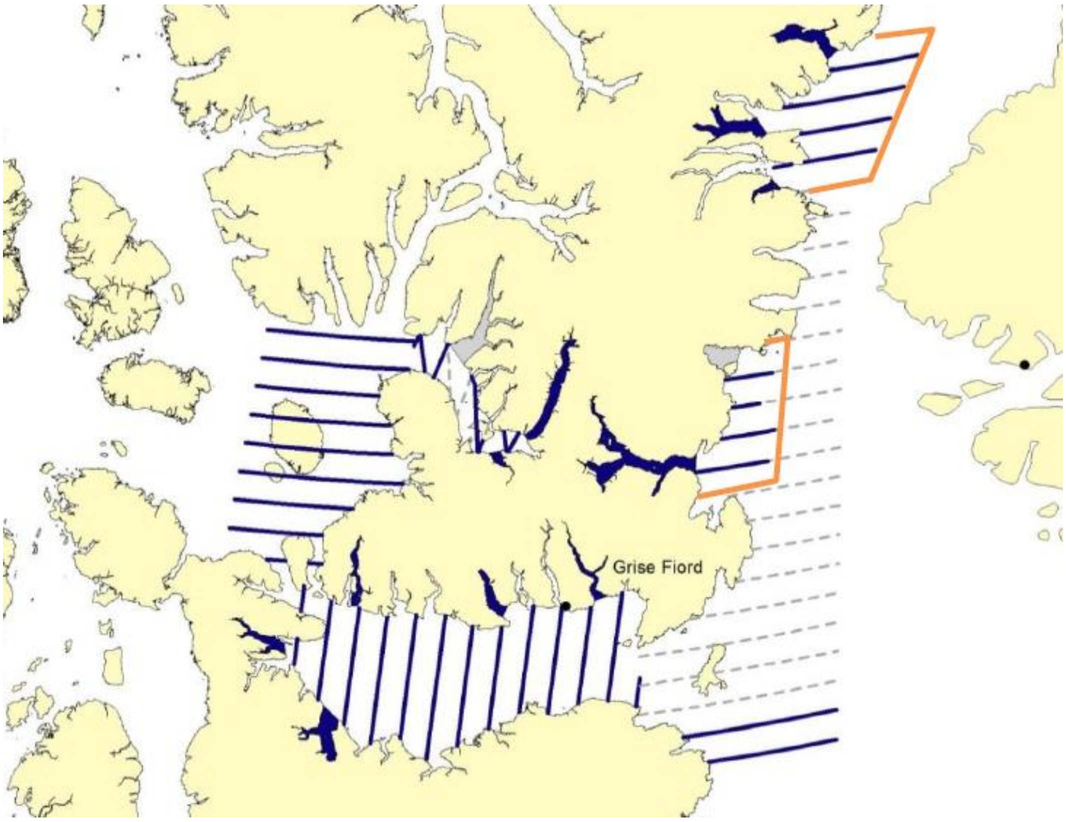

Figure 2. Map of the surveyed strata for the Jones and Smith Sound stocks. Blue lines indicate surveyed transects and blue areas indicate surveyed fiords, while grey dashed

lines and grey areas indicate planned transects and fiords that were unable to be completed as a result of weather (Doniol-Valcroze et al. 2015).

Norwegian Bay was flown in good weather on August 8, 2013, but its northern part and several of its fiords were still frozen. Jones Sound and its fiords (Figure 2) were flown in excellent conditions in a single day (August 10, 2013) although few narwhals were observed. Grise Fiord community members said that narwhals arrived late in 2013. Consequently, efforts were made to fly Jones Sound again at a later time (August 26, 2013), despite deteriorating weather with high winds. From this second survey, only the fiord strata were used because of more sheltered conditions (i.e., some Jones Sound fiords were surveyed a second time two weeks after the first survey and the data were combined) (Doniol-Valcroze et al. 2015). The surface abundance estimate for the Jones Sound stock was 4,316 ± 0.32, and after viewing the photos it was deemed that the water was clear and a correction for the 0-2 m bin should be applied. After a weighted correction of 2.94 ± 0.03 was applied the resulting abundance estimate was 12,694 ± 0.33 (Doniol-Valcroze et al. 2015).

3. Anthropogenic removals

This stock is hunted primarily by the community of Grise Fiord (Heide-Jørgensen et al. 2013); however, there is opportunity for hunters from other communities to hunt these whales on their migration to and from the summering grounds and on the wintering grounds (Witting 2016). Catches in Table 1, however, only reflect whales that are hunted within the defined summering region since it is difficult to determine the number of animals from this stock hunted by other communities. In some Canadian communities with a community-based management system, killed-lost and wounded-lost narwhal numbers were documented by hunters between 1999 and 2005 (Table 2). From the narwhal hunts where losses are reported, Richard (2008) calculated a hunting loss rate correction (LRC) (Table 2).

LRC = HM / LC where
HM = the estimated total hunting mortality, or the sum of the landed catch and hunting loss
LC = Landed Catch
The estimated hunting loss was calculated as:
HM = (HMmin + HMmax)/2 where
HMmin = number of animals landed plus the ones reported sunk and lost
HMmax = HMmin + the number reported wounded and escaped

This HM estimate used by Richard (2008) assumes that half of the animals wounded and escaped later die from their injuries. This assumption was untested but considered reasonable since both whales with wound scars are later seen alive but dead whales have also washed up after a hunt suggesting some whales survive from their wounds while others perish (Richard 2008). Table 1 indicates the total reported landed catches, and the catches multiplied by a struck and loss factor of 1.28 ± 0.15 (Richard 2008). This data comes from 1999-2005 and is hunter reported for all types of hunt combined for each of the communities. An older study (Roberge and Dunn 1990) investigated struck and lost rates from the community of Arctic Bay in the open water season in 1983 and 1988, on the floe edge in 1988 and 1989, and at the ice crack in 1978, 1988, and 1999 (Table 3).

Most of the hunt in Jones Sound occurs in the open water season, which has a struck and loss factor reported by Roberge and Dunn (1990) of 1.40 ± 0.14. In this study researchers monitored the hunt when possible and reported values. Application of this rate rather than the 1.28 reported by Richard (2008), changes catches previous to 1999 by an average of 1 whale, and a maximum of 6 whales (results in brackets in Table 1). Ideally a struck and loss factor would be applied to each catch that occurs through different hunting methods; unfortunately this information is not reported. However based on hunt dates (for which we have some information from 2003-2012), the majority of the hunt occurs in the open water season (80% for Grise Fiord (Doniol-Valcroze 2014)). Currently in Canada the struck and loss rate from Richard (2008) is used, since it is the most up to date.

Table 1. Reported landed catches for Grise Fiord. From 1977 these catches are based on the number of issued tags and recorded by Fisheries and Oceans Canada; prior to 1977 the numbers come from a variety of sources (see reference list) but typically rely on reports by hunters, or RCMP records. Total catch including struck and lost animals is indicated using the newest struck and lost factor (1.28 from Richard (2008)), and using the 1.40 reported for open water hunts by Roberge and Dunn (1990) for years prior to 1999 indicated in brackets.

Year	Grise Fiord (landed catches)	Reference for reported landed catch	Grise Fiord Catches + 1.28 S&L factor (1.40 S&L factor)
1970	49	Mansfield et al. (1975)	63 (69)
1971	25	Mansfield et al. (1975)	32 (35)
1972	nr	Strong (1989), Mitchell and Reeves (1981)	nr
1973	15	Strong (1989), Mansfield et al. (1975)	19 (21)
1974	nr	Strong (1989), Stewart (2007)	nr
1975	nr	Strong (1989)	nr
1976	11	Strong (1989)	14 (15)
1977	0	Strong (1989)	0 (0)
1978	0	Strong (1989)	0 (0)
1979	12	Strong (1989)	15 (17)
1980	0	Strong (1989)	0 (0)
1981	0	Strong (1989)	0 (0)
1982	28	Strong (1989)	36 (39)
1983	3	Strong (1989)	4 (4)
1984	2	Strong (1989)	3 (3)
1985	8	Strong (1989)	10 (11)
1986	2	Strong (1989)	3 (3)
1987	2	Strong (1989)	3 (3)
1988	7	DFO (1991)	9 (10)
1989	5	DFO (1992)	6 (7)
1990	19	DFO (1992)	24 (27)
1991	20	DFO (1993)	26 (28)
1992	1	DFO (1994)	1 (1)
1993	9	DFO (1995)	12 (13)
1994	12	DFO (1996)	15 (17)
1995	9	DFO (1997)	12 (13)
1996	1	DFO (1999)	1 (1)
1997	1	Stewart (2007)	1 (1)
1998	10	Doniol-Valcroze (2014)	13 (13)
1999	16	Doniol-Valcroze (2014)	20
2000	17	Doniol-Valcroze (2014)	22
2001	24	Doniol-Valcroze (2014), Stewart (2007)	31
2002	2	Doniol-Valcroze (2014)	3
2003	8	Doniol-Valcroze (2014), Stewart (2007)	10
2004	9	Hall et al. (2015)	12
2005	1	Hall et al. (2015)	1
2006	21	Hall et al. (2015)	27
2007	20	Hall et al. (2015)	26
2008	23	Hall et al. (2015)	29

2009	5	Hall et al. (2015)	6
2010	21	Hall et al. (2015)	27
2011	21	Hall et al. (2015)	27
2012	16	Hall et al. (2015)	20
2013	7	Hall et al. (2015)	9
2014	8	Hall et al. (2015)	10
2015	7	Hall et al. (2015)	9

Table 2. Table indicating how the struck and loss factor for this stock is calculated. Table is directly from Richard (2008).

Community	Year	Landed	Wounded/ Escaped	Sunk and Lost	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Community specific average Loss Rate Correction
Pond Inlet	1999	130	14	16	146	160	153	1.18	
	2000	166	21	10	176	197	187	1.12	
	2001	63	5	27	90	95	93	1.47	
	2002	92	1	13	105	106	106	1.15	1.23 ± 0.16
Qikiqtarjuaq	1999	81	30	25	106	136	121	1.49	
	2000	137	79	40	177	256	217	1.58	
	2001	89	8	9	98	106	102	1.15	
	2002	81	40	16	97	137	117	1.44	
	2004	96	12	9	105	117	111	1.16	1.36 ± 0.20
Repulse	1999	156	68	30	186	254	220	1.41	
	2000	49	9	5	54	63	59	1.19	
	2001	100	38	21	121	159	140	1.4	
	2002	57	0	8	65	65	65	1.14	
	2003	30	0	5	35	35	35	1.17	
	2005	72	25	3	75	100	88	1.22	1.26 ± 0.12
Arctic Bay	2001	134	20	4	138	158	148	1.1	
	2003	129	14	22	151	165	158	1.22	
	2004	122	22	33	155	177	166	1.36	1.23 ± 0.13
Kugaaruk	2001	41	18	8	49	67	58	1.41	
	2003	24	4	2	26	30	28	1.17	1.29 ± 0.17
Average across communities									**1.28 ± 0.15**

Table 3. Table indicating how an older struck and loss factor for Arctic Bay was calculated from observations of different hunting types from Roberge and Dunn (1990).

Hunt	Year	Landed	Wounded/ Escaped	Sunk and Lost/mortally wounded	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Hunt specific average Loss Rate Correction
Floe edge	1988	6	6	8	14	20	17	2.83	
	1989	16	0	5	21	21	21	1.31	2.07 ± 1.08
Open water	1983	4	2	1	5	7	6	1.50	
	1988	13	6	1	14	30	17	1.31	**1.40 ± 0.14**
Ice crack	1987	15	13	8	23	36	30	1.97	
	1988	29	8	17	46	54	50	1.72	
	1989	50	7	13	63	70	67	1.33	1.67 ± 0.32
Average across hunt types									**1.71 ± 0.55**

4. Population trajectory

Only one survey has been conducted for the Jones Sound stock. Figure 3 indicates the trajectory given the abundance estimate and associated confidence interval for the survey. The estimated trajectory for the stock comes from a population dynamic model based on a Bayesian modelling framework that is age and sex structured (for details see Witting 2016). Unfortunately there are not enough survey estimates to determine a trend for this stock (Witting 2016).

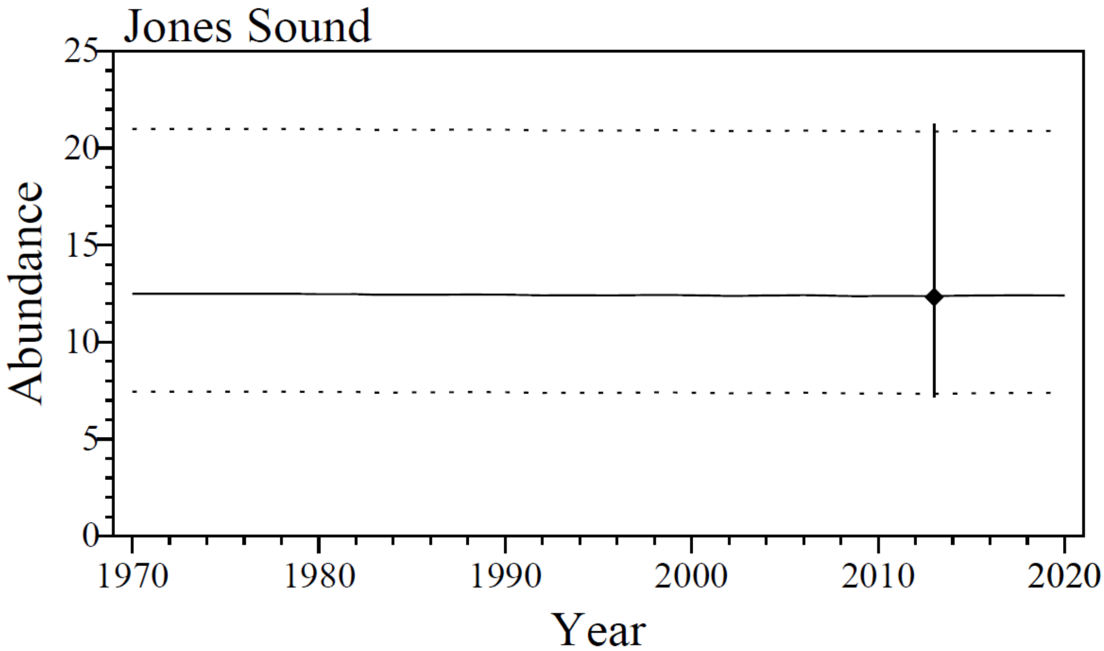

Figure 3. Population trajectory for the Jones Sound stock. The point represents the abundance estimate (given in thousands) with 90% confidence intervals. Solid curves indicate the median, and dotted curves the 90 % confidence interval for the estimated model (Witting 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR) method (Wade 1998), corrected for hunting losses (struck and lost), is used to calculate a recommended Total Allowable Landed Catch (TALC):

$$TALC = PBR/ LRC$$

Where:

$$PBR = 0.5 * R_{max} * N_{min} * F_r$$

LRC is the hunting loss rate correction and is equal to 1.28 ± 0.15 (Richard 2008). R_{max} is the maximum rate of increase for the stock (unknown so the default for cetaceans of 0.04 is used, N_{min} is the 20th percentile of the log-normal distribution of N (most recent survey estimate), and F_r is the recovery factor (we used a value of 1 which indicates a healthy status for the stock (an assumption)). A Total Allowable Harvest of 50 is in place for this stock in combination with the Smith Sound stock. However, now that there is an abundance estimate for this stock a new Total Allowable Landed Catch (TALC) recommendation based on the 2013 aerial survey results is 76.

6. Habitat and other concerns

Little is known since there have been no telemetry studies to show movements/migration or dive behaviour.

7. Status of the stock.

The Jones Sound stock is the second smallest narwhal stock and there is not enough information about the stock to determine its stock status at this time.

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Annex 21: Smith Sound Narwhal Stock
By: Cortney A. Watt

4. Distribution and stock identity

Stock identity is based on consistent summer aggregation reported in TEK. Summer distribution is indicated in dark green and labeled SS on Figure 1. There have been no telemetry studies on whales from the Smith Sound Stock. Skin samples of narwhal from Smith Sound separate genetically from Jones Sound narwhals (Petersen et al. 2011) and from other Greenland stocks (Palsbøll et al. 1997) but were similar to those from Eclipse Sound, Admiralty Inlet, and East Baffin Island (Palsbøll et al. 1997, Petersen et al. 2011).

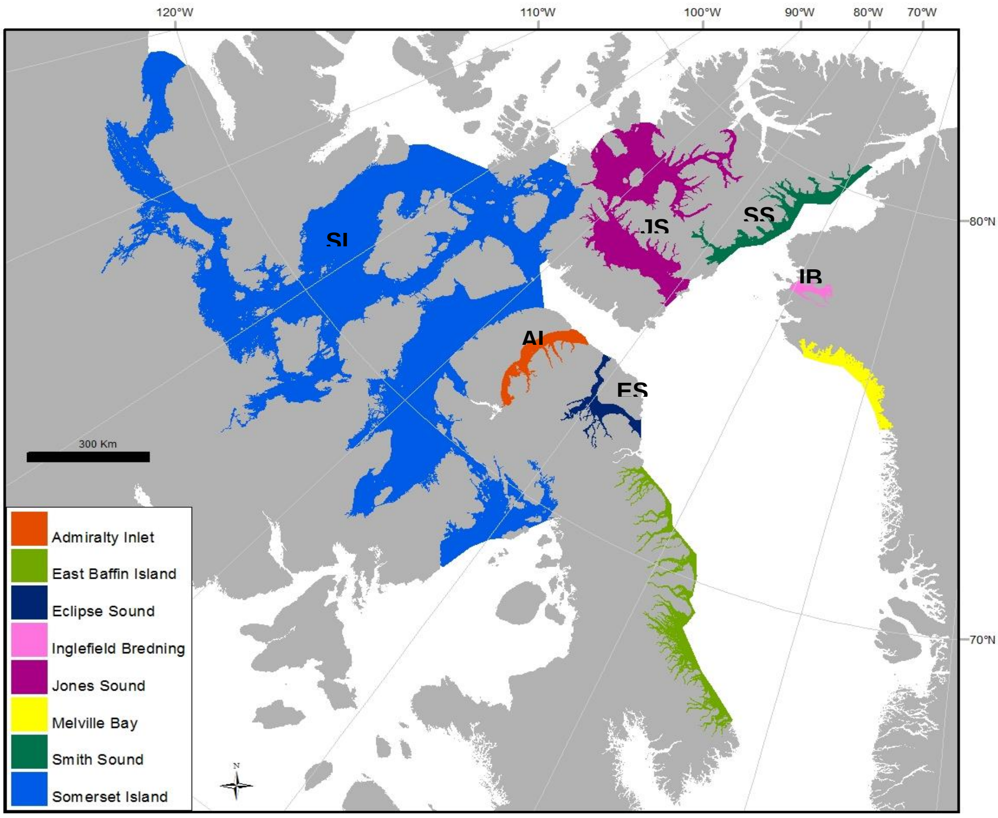

Figure 1. Map indicating the narwhal stocks for the Baffin Bay narwhal population.

5. Abundance

The only dedicated abundance estimate (from 2013) for this stock is 16,360 with a CV of 0.65 (Doniol-Valcroze et al. 2015). An earlier abundance estimate in 2009 of 2,309 with a CV of 1.62 was estimated by subtracting the abundance in Inglefield Bredning from the abundance in the North Water, but no aerial surveys were actually flown over Smith Sound at the time (NAMMCO 2015).

The 2013 estimate comes from an aerial survey design using a double-platform. Three aircraft were used simultaneously to cover a large area encompassing all of the Canadian narwhal stocks of the Baffin Bay population in August 2013 (Doniol-Valcroze et al. 2015). The extent of the survey areas was based on previous aerial surveys, telemetry tracking studies, TEK, and recent observations by Inuit hunters. Since there has been recent concern about potential movement of narwhals among neighbouring summering regions, the survey with multiple aircraft was designed to survey six of the Baffin Bay stocks during the summer aggregation season - late July through the first three weeks of August prior to the start of fall migration movements. Dates of the survey were chosen to cover areas when sea ice ablation allowed for narwhal to access most of the Arctic Archipelago, and based on the timing of narwhal aggregations in their summering areas as described by TEK and satellite-telemetry data (Doniol-Valcroze et al. 2015). As a result, the last week of July and the first three weeks of August was chosen for the survey, with preference for earlier in August since telemetry data indicated that animals start to move among stocks during the final week of August (Watt et al. 2012).

Transect design was performed in Distance (version 6.1) using coastline shape files. The design was systematic with the first transect line chosen at random. When possible transect lines were oriented in a direction perpendicular to the longest axis of the stratum to maximize the number of lines per stratum (Doniol-Valcroze et al. 2015). For areas where it was assumed narwhal would be in high densities, systematic parallel transects were used. In areas where lower densities were anticipated and landscape patterns permitted, zigzag transects with equally spaced endpoints were used (Doniol-Valcroze et al. 2015).

The survey was flown at an altitude of 1,000 ft, and a target speed of 100 knots using three deHavilland Twin Otter 300 aircraft, each with 4 bubble windows on the sides and run as a double-platform experiment with independent observations platforms at the front and rear of the plane (Doniol-Valcroze et al. 2015). Dual camera systems were mounted under the belly of the plane to allow for continuous digital photography.

Distance sampling methods were used to estimate detection probability away from the track line, while mark-recapture methods were used on sighting data from two observers on either side of the aircraft to correct for perception bias. The distribution of perpendicular distances was different in fiord strata than in the other strata, and thus only non-fiord observations were used to fit the detection function for the non-fiord strata. Examination of the histogram of the perpendicular distances of unique sightings suggested right-truncating the data at 1000 m (i.e., discarding sightings beyond 1000 m), which left 762 unique observations (515 seen by primary observers, 523 by secondary observers, and 276 by both). The shape of the histogram suggested that some narwhals were missed close to the track line despite the bubble windows. Therefore, there was a risk that hazard-rate and half-normal distributions would overestimate the probability of detection and the resulting effective strip width. However, almost a hundred narwhal sightings were made within 100 m of the track line and therefore it seemed inappropriate to lose a large amount of data by left-truncating (i.e., discarding sightings close to the trackline). The shape of the histogram suggested that a gamma distribution would fit better, except that a gamma distribution takes the value zero at zero distance. Therefore, a gamma distribution with an offset term, in addition to half-normal and hazard rate keys, was fitted to the data (Doniol-Valcroze et al. 2015). Model selection was performed on all combinations of covariates (including environmental covariates such as ice cover, cloud cover, sea state, and glare, and a sighting rate covariate which was computed as a rolling average of the number of sightings made by the observer in a 30-second window prior to each sighing). The model with the lowest AIC was one with a truncated gamma key function and the covariates “sighting rate”, “Beaufort” and “glare”. The covariates reduced the detection distance at higher levels (Beaufort >3, Glare=intense, Sighting rate >10 in the last 30 seconds) and resulted in an average probability of detection of 0.48 (CV 2.8%) and an estimated effective strip half width of 481 m (not including perception bias) (Doniol-Valcroze et al. 2015).

For the mark-recapture model to estimate perception bias, models were performed with all combinations of environmental covariates as well as covariates “perpendicular distance”, “observer”, “sighting rate”, “side of aircraft” and “group size”. The best model included “perpendicular distance” and “sighting rate” and the overall probability of detecting a narwhal cluster between the track line and a distance of 1000 m was 0.40 (CV 4.2%) (Doniol-Valcroze et al. 2015).

Fiords were considered their own sampling units and cluster sampling was used to select the fiords to be surveyed (Doniol-Valcroze et al. 2015). In fiords, flights were continuous tracks designed to follow the main axis of the fiord while spreading coverage uniformly based on distance to shore. The resulting data from the fiords was analyzed separately from non-fiord strata. A density surface modelling framework was used to model spatially-referenced count data with the additional information provided by collecting distances to account for imperfect detection (Doniol-Valcroze et al. 2015). First a detection function was fitted to the perpendicular distance data to obtain detection probabilities for clusters of individuals (Doniol-Valcroze et al. 2015). Counts were then summarized for contiguous transect sections and a generalized additive model was constructed with segment counts as the response with areas corrected for detectability (Doniol-Valcroze et al. 2015).

Total surface abundance estimates for stocks were obtained by the additions of the estimated abundances of all the strata that made up that stock's summer range, including results from fiord strata. Variance for the stock-wide abundance estimate was calculated by adding the variances of each stratum; however, identification of duplicates was not straightforward due to the highly aggregated nature of narwhal groups. Because of this, a sensitivity analysis was used to quantify the uncertainty, which allowed the researchers to include an additional variance component to the surface abundance estimate with a CV equal to that of the sensitivity analysis, which ultimately increased the range of uncertainty around the estimate but left the point estimate unchanged (Doniol-Valcroze et al. 2015).

An availability bias correction was also applied to the survey data. For the availability bias correction, the time at depth for 24 narwhals fitted with satellite tags near Arctic Bay and Pond Inlet every August from 2009-2012 (Watt et al. 2015) was used to determine the correction for the number of whales missed as a result of being at depth and unavailable for viewing by the surveyors. The time narwhals spent at 0-2 m depths was used to calculate a correction for areas with clear water, while areas with very murky water, the time spent within 0-1 m of the surface was used. This resulted in a correction factor of 3.18 ± 3.37 for clear water areas and a correction of 4.90 ± 0.187 for murky regions (Watt et al. 2015). This correction is appropriate when sightings are instantaneous, but if they are not (such as in aerial surveys), it can positively bias the estimate and as a result a correction factor incorporating the dive cycle of the animal is needed. Three archival time-depth recorders deployed on whales near Pond Inlet and in Creswell Bay in August 1999 and 2000 were used to evaluate a dive-cycle for narwhals. A weighted availability bias correction factor that took into account both the time at depth and the time in view (dive-cycle) was used (2.94 ± 3.4 for the 0-2 m correction and 4.53 ± 3.8 for the 0-1 m bin).

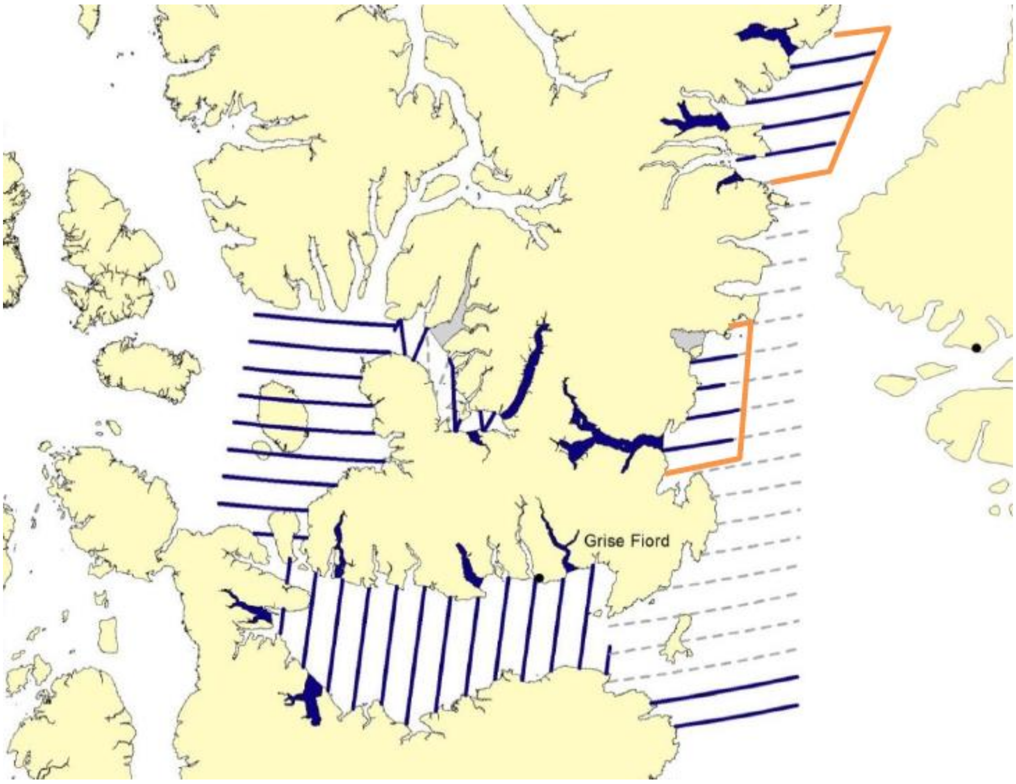

Figure 2. Map of the surveyed strata for the Jones and Smith Sound stocks. Blue lines indicate surveyed transects and blue areas indicate surveyed fiords, while grey dashed lines and grey areas indicate planned transects and fiords that were unable to be completed as a result of weather (Doniol-Valcroze et al. 2015).

For Smith Sound surveys were flown on August 4, 2013 (Figure 2), but fog and strong winds prevented complete coverage of all strata. Several of the eastern Ellesmere fiords could be surveyed, however, and large numbers of narwhals were observed in Mackinson Inlet (Doniol-Valcroze et al. 2015). The surface abundance estimate for the Smith Sound stock was 5,563 ± 0.65, and after a weighted correction of 2.94 ± 0.03 was applied the resulting abundance estimate was 16,360 ± 0.65 (Doniol-Valcroze et al. 2015).

6. Anthropogenic removals

Currently there are no communities in Canada that take from this stock. Historically a quota has been set in conjunction with Jones Sound for the community of Grise Fiord.

In 2015, for the first time, a quota of 5 individuals (2015-2020) were given to the community of Qaanaaq. The quota is set on the basis of the allocation model developed by JCNB SWG.

7. Population trajectory

Only one survey has been conducted for the Smith Sound stock. Figure 3 indicates the trajectory given the abundance estimate and associated confidence interval for the survey. The estimated trajectory for the stock comes from a population dynamic model based on a Bayesian modelling framework that is age and sex structured (for details see Witting 2016). Unfortunately there are not enough survey estimates to determine a trend for this stock (Witting 2016).

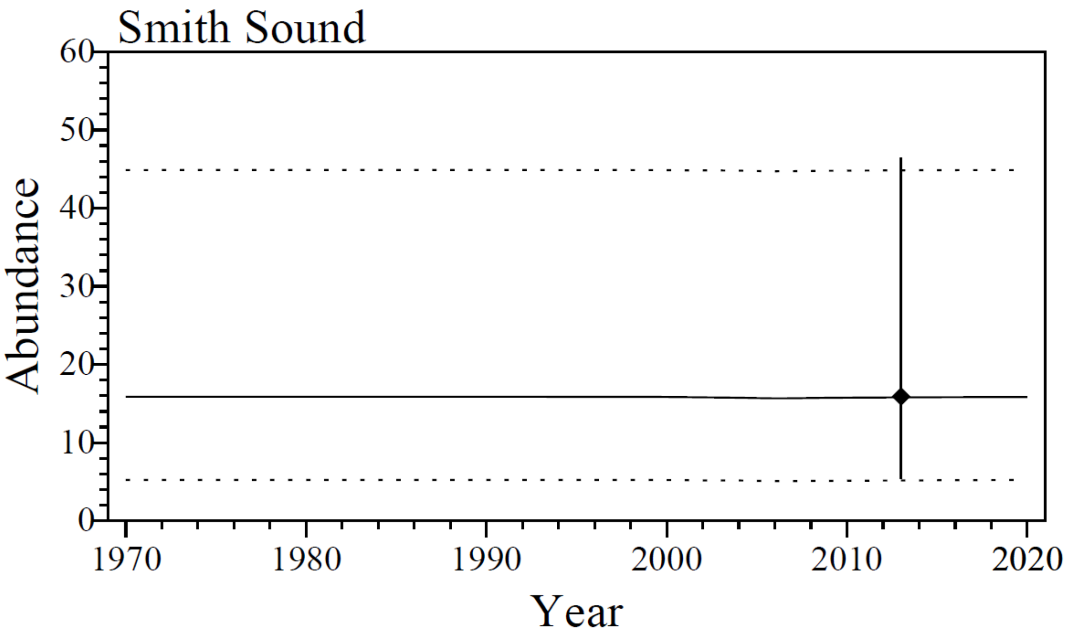

Figure 3. Population trajectory for the Smith Sound stock. The point represents the abundance estimate (given in thousands) with 90% confidence intervals. Solid curves indicate the median, and dotted curves the 90 % confidence interval for the estimated model (Witting 2016).

8. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

In Canada, the Potential Biological Removal (PBR) method (Wade 1998), corrected for hunting losses (struck and lost), is used to calculate a recommended Total Allowable Landed Catch (TALC):

TALC = PBR/ LRC
Where:
$PBR = 0.5 * R_{max} * N_{min} * F_r$

LRC is the hunting loss rate correction and is equal to 1.28 ± 0.15 (Richard 2008). R_{max} is the maximum rate of increase for the stock (unknown so the default for cetaceans of 0.04 is used, N_{min} is the 20th percentile of the log-normal distribution of N (most recent survey estimate), and F_r is the recovery factor (we used a value of 1 which indicates a healthy status for the stock (an assumption)). In Canada, a Total Allowable Harvest of 50 is in place for this stock along with the Jones Sound stock; however, hunters in Grise Fiord have only hunted from the Jones Sound stock historically and there are no other Canadian communities that hunt whales from the region. Now that there are is an abundance estimate for this stock a new Total Allowable Landed Catch (TALC) recommendation based on the 2013 aerial survey results for Smith Sound is 77 for Canada.

9. Habitat and other concerns

One male narwhal (tusk=25 cm) with a standard length of 315 cm was tagged on 14 June from the ice edge at Renselaer Bay on the Greenland side of Smith Sound.

For unknown reasons the tracking only lasted three days but the positions clearly demonstrated the connection across Smith Sound as the whale moved swiftly to Cape D'urville in Canada (Figure 4). No further tagging of narwhals in Smith Sound has been attempted.

Figure 4. Satellite track of a narwhal tagged at Renselaer Bay, Greenland, which moved to the Canadian side of Smith Sound.

10. Status of the stock

The Smith Sound stock is a medium size stock, but there is not enough information about the stock to determine its stock status at this time.

11. Life history

Garde et al. (2015) estimated life history parameters for narwhals (*Monodon monoceros*) from East and West Greenland (n=282) based on age estimates from aspartic acid racemization (AAR) of eye lens nuclei. The species-specific age equation used, $420.32X - 24.02\cdot$year where X is the D/L ratio, was determined from data from Garde et al. (2012) by regressing aspartic acid D/L ratios in eye lens nuclei against growth layer groups in tusks (n=9). Asymptotic body length was estimated to be 399 ± 5.9 cm for females at age 25 years and 456 ± 6.9 cm for males from West Greenland at age 28 years. Age at sexual maturity was assessed based on data from reproductive organs and was estimated to be 8–9 years for females and 12–20 years for males. Length at sexual maturity was ~340 cm for females and 350-400 cm for males. Estimated age at 1st parturition was 9-10 years. Oldest pregnant female was close to 70 years. Pregnancy rates for East and West Greenland were estimated to be 0.38–0.42 and 0.38, respectively. Maximum life span expectancy was found to be approximately 100 years. A population projection matrix was parameterized with the data on age structure and fertility rates. The annual rate of increase of narwhals in East Greenland was estimated to be 3.8% while narwhals in West Greenland had a rate of increase at 2.6%.

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Annex 22: Admiralty Inlet Narwhal Stock
By: Cortney A. Watt and Rikke Guldborg Hansen

1. Distribution and stock identity

Stock identity is based on consistent summer aggregation reported in TEK, telemetry tracking, and aerial surveys. Summer distribution is indicated in orange and labeled "AI" on Figure 1. Stock identity is supported by telemetry studies which show narwhals tagged in Admiralty Inlet stay within that region in the summer, and return there after spending the winter in the Baffin Bay region (Dietz et al. 2008, Watt et al. 2012); however, one whale tagged in Eclipse Sound did enter Admiralty Inlet after winter so there may be some overlap among these stocks (see below for further discussion; Watt et al. 2012). There is not strong genetic support for delineation of this stock as there is a lot of overlap among the stocks during the mating season in late winter-early spring necessary for genetic discrimination (de March et al. 2003, Petersen et al. 2011). However, stable isotope values from skin samples of individuals hunted in Admiralty Inlet differ significantly from those of whales hunted in other regions indicating some separation based on foraging (Watt et al. 2012).

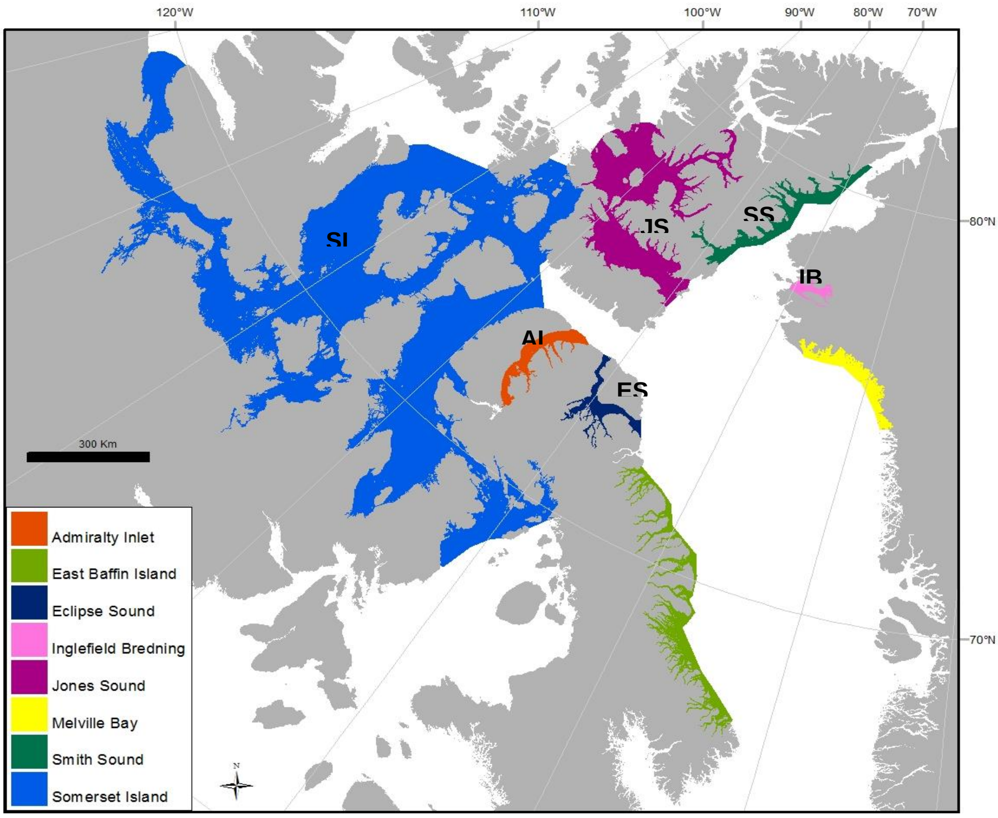

Figure 1. Map indicating the narwhal stocks for the Baffin Bay narwhal population.

2. Abundance

The most recent (2013) abundance estimate for this stock is 35,043 with a CV of 0.42 (Doniol-Valcroze et al. 2015).

This estimate comes from an aerial survey design using a double-platform. Three aircraft were used simultaneously to cover a large area encompassing all of the Canadian narwhal stocks of the Baffin Bay population in August 2013 (Doniol-Valcroze et al. 2015). The extent of the survey areas was based on previous aerial surveys, telemetry tracking studies, TEK, and recent observations by Inuit hunters. Since there has been recent concern about potential movement of narwhals among neighbouring summering regions, the survey with multiple aircraft was designed to survey six of the Baffin Bay stocks during the summer aggregation season - late July through the first three weeks of August prior to the start of fall migration movements. Dates of the survey were chosen to cover areas when sea ice ablation allowed for narwhal to access most of the Arctic Archipelago, and based on the timing of narwhal aggregations in their summering areas as described by TEK and satellite-telemetry data (Doniol-Valcroze et al. 2015). As a result, the last week of July and the first three weeks of August was chosen for the survey, with preference for earlier in August since telemetry data indicated that animals start to move among stocks during the final week of August (Watt et al. 2012).

Transect design was performed in Distance (version 6.1) using coastline shape files. The design was systematic with the first transect line chosen at random. When possible transect lines were oriented in a direction perpendicular to the longest axis of the stratum to maximize the number of lines per stratum (Doniol-Valcroze et al. 2015). For areas where it was assumed narwhal would be in high densities, systematic parallel transects were used. In areas where lower densities were anticipated and landscape patterns permitted, zigzag transects with equally spaced endpoints were used (Doniol-Valcroze et al. 2015).

The survey was flown at an altitude of 1,000 ft, and a target speed of 100 knots using three deHavilland Twin Otter 300 aircraft, each with 4 bubble windows on the sides and run as a double-platform experiment with independent observations platforms at the front and rear of the plane (Doniol-Valcroze et al. 2015). Dual camera systems were mounted under the belly of the plane to allow for continuous digital photography.

Distance sampling methods were used to estimate detection probability away from the track line, while mark-recapture methods were used on sighting data from two observers on either side of the aircraft to correct for perception bias. The distribution of perpendicular distances was different in fiord strata than in the other strata, and thus only non-fiord observations were used to fit the detection function for the non-fiord strata. Examination of the histogram of the perpendicular distances of unique sightings suggested right-truncating the data at 1000 m (i.e., discarding sightings beyond 1000 m), which left 762 unique observations (515 seen by primary observers, 523 by secondary observers, and 276 by both). The shape of the histogram suggested that some narwhals were missed close to the track line despite the bubble windows. Therefore, there was a risk that hazard-rate and half-normal distributions would overestimate the probability of detection and the resulting effective strip width. However, almost a hundred narwhal sightings were made within 100 m of the track line and therefore it seemed inappropriate to lose a large amount of data by left-truncating (i.e., discarding sightings close to the trackline). The shape of the histogram suggested that a gamma distribution would fit better, except that a gamma distribution takes the value zero at zero distance. Therefore, a gamma distribution with an offset term, in addition to half-normal and hazard rate keys, was fitted to the data (Doniol-Valcroze et al. 2015). Model selection was performed on all combinations of covariates (including environmental covariates such as ice cover, cloud cover, sea state, and glare, and a sighting rate covariate which was computed as a rolling average of the number of sightings made by the observer in a 30-second window prior to each sighing). The model with the lowest AIC was one with a truncated gamma key function and the covariates “sighting rate”, “Beaufort” and “glare”. The covariates reduced the detection distance at higher levels (Beaufort >3, Glare=intense, Sighting rate >10 in the last 30 seconds) and resulted in an average probability of detection of 0.48 (CV 2.8%) and an estimated effective strip half width of 481 m (not including perception bias) (Doniol-Valcroze et al. 2015).

For the mark-recapture model to estimate perception bias, models were performed with all combinations of environmental covariates as well as covariates “perpendicular distance”, “observer”, “sighting rate”, “side of aircraft” and “group size”. The best model included “perpendicular distance” and “sighting rate” and the overall probability of detecting a narwhal cluster between the track line and a distance of 1000 m was 0.40 (CV 4.2%) (Doniol-Valcroze et al. 2015).

Fiords were considered their own sampling units and cluster sampling was used to select the fiords to be surveyed (Doniol-Valcroze et al. 2015). In fiords, flights were continuous tracks designed to follow the main axis of the fiord while spreading coverage uniformly based on distance to shore. The resulting data from the fiords was analyzed separately from non-fiord strata. A density surface modelling framework was used to model spatially-referenced count data with the additional information provided by collecting distances to account for imperfect detection (Doniol-Valcroze et al. 2015). First a detection function was fitted to the perpendicular distance data to obtain detection probabilities for clusters of individuals (Doniol-Valcroze et al. 2015). Counts were then summarized for contiguous transect sections and a generalized additive model was constructed with segment counts as the response with areas corrected for detectability (Doniol-Valcroze et al. 2015).

Total surface abundance estimates for stocks were obtained by the additions of the estimated abundances of all the strata that made up that stock's summer range, including results from fiord strata. Variance for the stock-wide abundance estimate was calculated by adding the variances of each stratum; however, identification of duplicates was not straightforward due to the highly aggregated nature of narwhal groups. Because of this, a sensitivity analysis was used to quantify the uncertainty, which allowed the researchers to include an additional variance component to the surface abundance estimate with a CV equal to that of the sensitivity analysis, which ultimately increased the range of uncertainty around the estimate but left the point estimate unchanged (Doniol-Valcroze et al. 2015).

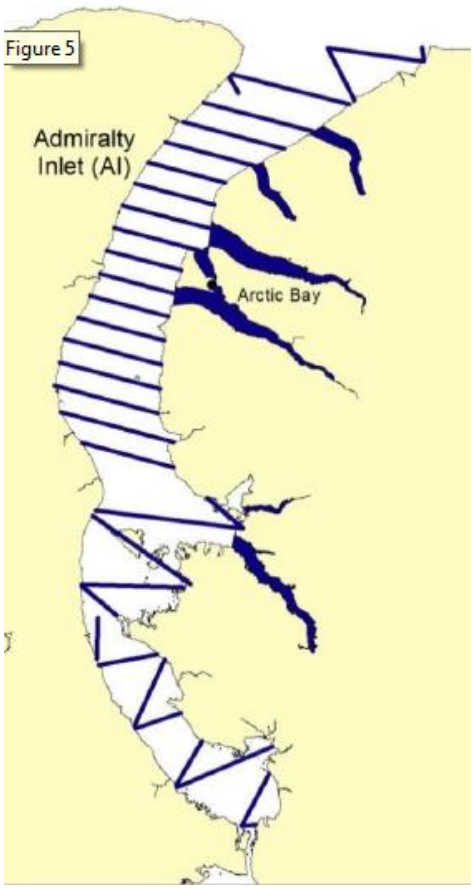

Figure 2. Map of the surveyed strata for the Admiralty Inlet stock. Blue lines indicate surveyed transect and blue areas indicate surveyed fiords (Doniol-Valcroze et al. 2015).

An availability bias correction was also applied to the survey data. For the availability bias correction, the time at depth for 24 narwhals fitted with satellite tags near Arctic Bay and Pond Inlet every August from 2009-2012 (Watt et al. 2015) was used to determine the correction for the number of whales missed as a result of being at depth and unavailable for viewing by the surveyors. The time narwhals spent at 0-2 m depths was used to calculate a correction for areas with clear water, while areas with very murky water, the time spent within 0-1 m of the surface was used. This resulted in a correction factor of 3.18 ± 3.37 for clear water areas and a correction of 4.90 ± 0.187 for murky regions (Watt et al. 2015). This correction is appropriate when sightings are instantaneous, but if they are not (such as in aerial surveys), it can positively bias the estimate and as a result a correction factor incorporating the dive cycle of the animal is needed. Three archival time-depth recorders deployed on whales near Pond Inlet and in Creswell Bay in August 1999 and 2000 were used to evaluate a dive-cycle for narwhals. A weighted availability bias correction factor that took into account both the time at depth and the time in view (dive-cycle) was used (2.94 ± 3.4 for the 0-2 m correction and 4.53 ± 3.8 for the 0-1 m bin).

For Admiralty Inlet all strata (Figure 2) were flown on August 12, 2013 and August 17, 2013. There was a 4-day break between due to bad weather. Few narwhals were observed in the high intensity areas, but instead were aggregated in the southern end of the area, close to shore or within fiords, with a high degree of clumping (Doniol-Valcroze et al. 2015). The surface abundance estimate for the Admiralty Inlet stock was 11,915 ± 0.42, and after viewing the photos it was deemed that the water was clear and a correction for the 0-2 m bin should be applied. After a weighted correction of 2.94 ± 0.03 was applied the resulting abundance estimate was 35,043 ± 0.42 (Doniol-Valcroze et al. 2015).

3. Anthropogenic removals

This stock is hunted primarily by the community of Arctic Bay (Heide-Jørgensen et al. 2013); however, there is opportunity for hunters from other communities to hunt these whales on their migration to and from the summering grounds and on the wintering grounds (Witting 2016). Catches in Table 1, however, only reflect whales that are hunted within the defined summering region since it is difficult to determine the number of animals from this stock hunted by other communities. In some Canadian communities with a community-based management system, killed-lost and wounded-lost narwhal numbers were documented by hunters between 1999 and 2005 (Table 2). From the narwhal hunts where losses are reported, Richard (2008) calculated a hunting loss rate correction (LRC) (Table 2).

LRC = HM / LC where
HM = the estimated total hunting mortality, or the sum of the landed catch and hunting loss
LC = Landed Catch
The estimated hunting loss was calculated as:
$HM = (HM_{min} + HM_{max})/2$ where
HM_{min} = number of animals landed plus the ones reported sunk and lost
$HM_{max} = HM_{min}$ + the number reported wounded and escaped

This HM estimate used by Richard (2008) assumes that half of the animals wounded and escaped later die from their injuries. This assumption was untested but considered reasonable since both whales with wound scars are later seen alive but dead whales have also washed up after a hunt suggesting some whales survive from their wounds while others perish (Richard 2008). Table 1 indicates the total reported landed catches, and the catches multiplied by a struck and loss factor of 1.28 ± 0.15 (Richard 2008). This data comes from 1999-2005 and is hunter reported for all types of hunt combined for each of the communities. An older study (Roberge and Dunn 1990) investigated struck and lost rates from the community of Arctic Bay in the open water season in 1983 and 1988, on the floe edge in 1988 and 1989, and at the ice crack in 1978, 1988, and 1999 (Table 3).

Most of the hunt in Arctic Bay occurs in the open water season, which has a struck and loss factor reported by Roberge and Dunn (1990) of 1.40 ± 0.14. In this study researchers monitored the hunt when possible and reported values. Application of this rate rather than the 1.28 reported by Richard (2008), changes catches previous to 1999 by an average of 11 whales, and a maximum of 20 whales (results in brackets in Table 1). Ideally a struck and loss factor would be applied to each catch that occurs through different hunting methods; unfortunately this information is not reported. However based on hunt dates (for which we have some information from 2003-2012), the majority of the hunt occurs in the open water season (61% for Arctic Bay (Doniol-Valcroze 2014)). Currently in Canada the struck and loss rate from Richard (2008) is used, since it is the most up to date.

The stock is also hunted on the wintering grounds in Greenland where 2% of the hunt in Uummannaq and 32% of the hunt in Disko Bay are believed to be from the Admiralty Inlet stock (see Table 4.).

Table 1. Reported landed catches for Arctic Bay. From 1977 these catches are based on the number of issued tags and recorded by Fisheries and Oceans Canada; prior to 1977 the numbers come from a variety of sources (see reference list) but typically rely on reports by hunters, or RCMP records. Total catch including struck and lost animals is indicated using the newest struck and lost factor (1.28 from Richard (2008)), and using the 1.40 reported for open water hunts by Roberge and Dunn (1990) for years prior to 1999 indicated in brackets.

Year	Arctic Bay (landed catches)	Reference for reported landed catch	Arctic Bay Catches + 1.28 S&L factor (1.40 S&L factor)
1970	nr	Mansfield et al. (1975)	nr
1971	nr	Mansfield et al. (1975)	nr
1972	101	Strong (1989), Mitchell and Reeves (1981)	129 (141)
1973	150	Strong (1989), Mansfield et al. (1975)	192 (210)
1974	52	Strong (1989), Stewart (2007)	67 (73)
1975	167	Strong (1989)	214 (234)
1976	115	Strong (1989)	147 (161)
1977	42	Strong (1989)	54 (59)
1978	65	Strong (1989)	83 (91)
1979	33	Strong (1989)	42 (46)
1980	100	Strong (1989)	128 (140)
1981	100	Strong (1989)	128 (140)
1982	90	Strong (1989)	115 (126)
1983	100	Strong (1989)	128 (140)
1984	93	Strong (1989)	119 (130)
1985	100	Strong (1989)	128 (140)
1986	100	Strong (1989)	128 (140)
1987	25	Strong (1989)	32 (35)
1988	86	DFO (1991)	110 (120)
1989	99	DFO (1992)	127 (139)
1990	67	DFO (1992)	86 (94)
1991	114	DFO (1993)	146 (160)
1992	102	DFO (1994)	131 (143)
1993	85	DFO (1995)	109 (119)
1994	99	DFO (1996)	127 (139)
1995	46	Watt and Hall (2017)	59 (64)
1996	99	DFO (1999)	127 (139)
1997	66	Stewart (2007)	84 (92)
1998	92	Doniol-Valcroze (2014)	118 (129)
1999	89	Doniol-Valcroze (2014)	114
2000	nr	Doniol-Valcroze (2014)	nr
2001	132	Doniol-Valcroze (2014), Stewart (2007)	169
2002	78	Doniol-Valcroze (2014)	100
2003	129	Doniol-Valcroze (2014), Stewart (2007)	165
2004	123	Hall et al. (2015)	157
2005	131	Hall et al. (2015)	168
2006	130	Hall et al. (2015)	166
2007	124	Hall et al. (2015)	159
2008	132	Hall et al. (2015)	169
2009	129	Hall et al. (2015)	165
2010	128	Hall et al. (2015)	164

Year	Arctic Bay (landed catches)	Reference for reported landed catch	Arctic Bay Catches + 1.28 S&L factor (1.40 S&L factor)
2011	130	Hall et al. (2015)	166
2012	125	Hall et al. (2015)	160
2013	159	Hall et al. (2015)	204
2014	141	Hall et al. (2015)	180
2015	216	Hall et al. (2015)	276

Table 2. Table indicating how the struck and loss factor for this stock is calculated. Table is directly from Richard (2008).

Community	Year	Landed	Wounded/ Escaped	Sunk and Lost	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Community specific average Loss Rate Correction
Pond Inlet	1999	130	14	16	146	160	153	1.18	
	2000	166	21	10	176	197	187	1.12	
	2001	63	5	27	90	95	93	1.47	
	2002	92	1	13	105	106	106	1.15	1.23 ± 0.16
Qikiqtarjuaq	1999	81	30	25	106	136	121	1.49	
	2000	137	79	40	177	256	217	1.58	
	2001	89	8	9	98	106	102	1.15	
	2002	81	40	16	97	137	117	1.44	
	2004	96	12	9	105	117	111	1.16	1.36 ± 0.20
Repulse	1999	156	68	30	186	254	220	1.41	
	2000	49	9	5	54	63	59	1.19	
	2001	100	38	21	121	159	140	1.4	
	2002	57	0	8	65	65	65	1.14	
	2003	30	0	5	35	35	35	1.17	
	2005	72	25	3	75	100	88	1.22	1.26 ± 0.12
Arctic Bay	2001	134	20	4	138	158	148	1.1	
	2003	129	14	22	151	165	158	1.22	
	2004	122	22	33	155	177	166	1.36	1.23 ± 0.13
Kugaaruk	2001	41	18	8	49	67	58	1.41	
	2003	24	4	2	26	30	28	1.17	1.29 ± 0.17
Average across communities									**1.28 ± 0.15**

Table 3. Table indicating how an older struck and loss factor for Arctic Bay was calculated from observations of different hunting types from Roberge and Dunn (1990).

Hunt	Year	Landed	Wounded/ Escaped	Sunk and Lost/mortally wounded	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Hunt specific average Loss Rate Correction
Floe edge	1988	6	6	8	14	20	17	2.83	
	1989	16	0	5	21	21	21	1.31	2.07 ± 1.08
Open water	1983	4	2	1	5	7	6	1.50	
	1988	13	6	1	14	30	17	1.31	**1.40 ± 0.14**
Ice crack	1987	15	13	8	23	36	30	1.97	
	1988	29	8	17	46	54	50	1.72	
	1989	50	7	13	63	70	67	1.33	1.67 ± 0.32
Average across hunt types									**1.71 ± 0.55**

Table 4. Catches of narwhals from official reports by municipality with corrections for under-reportings (in parenthesis) for 1954 to 2011. Numbers in square brackets are from *special reports*. The column 'under-reporting' shows the sum of the corrections for under-reporting or 'ALL' if it is a general correction factor for all areas. 'na' means that no data are available. Data from 2007-08 are preliminary. DB=Disko Bay, UUM=Uummannaq, UPV=Upernavik. Data were compiled from Prime Minister's Second Department (1951), Kapel (1977), Kapel (1983), Kapel and Larsen (1984), Kapel (1985), Born and Kapel (1986) and Born (1987).

YEAR	QAANAAQ	UPER-NAVIK	UUMMAN-NAQ	DISKO BAY	SISI-MIUT	MANIIT-SOQ	NUUK	PAAMIUT-QAQORTOQ	TOTAL	ICE ENTRAPMENT
1949	38	16	1	6					61	
1950										
1951										85 DB
1952										450 DB
1954	na		45		1			1	47	
1955	na	179	2	14					195	
1956	na	15	282	21					318	156 UPV, 250 UUM
1957	na	55	11	15					81	
1958	na	24	3	45		1			73	
1959	na	32	8	16				1	57	
1960	na	25	296	7	1	1	1	1	332	
1961	134	25	5	38				1	203	272 UUM
1962	182	17	11	12				1	213	
1963	275	10	3	29					317	
1964	275	17	11	11					314	
1965	na	33	37	33	1	1			105	
1966	na	39	23	43		3	2		110	
1967	na		131			9			140	31 DB
1968	na		454			18			472	161 DB, 50 UPV, 84 UUM
1969	na		174			30			204	Some DB, 50 UPV
1970	na		313			9			322	100 DB
1971	na		146			40			186	
1972	na		84			23			107	
1973	na		191			8			199	
1974	8		136			3			147	

Table 4. Continued

YEAR	QAANAAQ	UPER-NAVIK	UUMMAN-NAQ	DISKO BAY	SISIMIUT	MANIITSOQ	NUUK	PAAMIUT-QAQORTOQ-	TOTAL	ICE ENTRAPMENT
1975	1	54	11	44		6		1	266 (149)	
1976	9	22	27	57					264 (141)	
1977	16	62	113	53	8	1			387 (134)	
1978	110	56	183	262		1			612	
1979	120	22	132	100			3		377	
1980	130	61	146	120		4	1		462	
1981	118	83	140	249			18	1	609	
1982	164	59	162	76					461	45 DB
1983	135 (25)	72 (30)	164	68 (10)					439 (65)	
1984	274	80	245	66 (15)	1				666 (15)	35 UUM
1985	115 (115)	34 (20)	39	67		1			256 (135)	
1986	na	81	97	23		36			237	
1987	na	145	334	25			1		505	
1988	na		206						500 (294)	
1989	na	37	288	2			5		332	
1990	na	100 (73)	1019	11					1057 (100)	
1991	na		27	> 40					na	27 UUM
1992	na	37	288	2			5		342	
1993	144	66	301	75	10	6	4	8	614	
1994	183	59	297	268	6	14	7	11	845	150 DB
1995	107	94	159	108	4	5	8		485	
1996	45	69	405	154	10	4	2	2	691	
1997	66	90	381	156	13	5	9	26	746	
1998	94	105	344	163	21	18	6	24	775	
1999	115	119	253	174	28	24	17	15	745	
2000	109	150	106	155	27	8	0	6	561	
2001	145	155	95	119	1	2	15	3	535	
2002	94	164	180	97	12	11	3	2	563	
2003	113	146	174	114	4	0	2	2	554	
2004	178	53	67	73	2	1	0	0	374	
2005	[70] 137	[74] 71	[137] 161	[47] 39	0	0	0	0	[328] 408	
2006	[94] 99	[58] 62	[55] 72	[4] 53	1	2	0		[211] 289	
2007	[21] 139	[17] 102	[52] 67	[56] 63	0	2	0	1	[146] 374	
2008	129	74	87	47	0	0	0	0	337	
2009	90	110	91	88	0	0	0	1	380	41 in Qaanaaq
2010	108	30	42	45	0	0	0	0	225	53 in Qaanaaq
2011	74	60	77	39	0	0	0	1	251	
2012	144	70	42	179	0	0	0	1	311	125 at Kangersuatsiaq
2013	90	64	78	50	0	0	0	1	283	
2014	114	101	69	50	0	0	0	0	334	
2015	92	54	42	29	0	0	1	0	218	
2016	93	79	120	55	0	0	1	0	348	

4. Population trajectory

Five surveys with the goal of assessing abundance have been conducted over the past 30 years for the Admiralty Inlet stock. Figure 3 indicates the trajectory given the abundance estimates and associated confidence intervals for the different surveys. The 1975 survey had a correction of 2.92 (CV = 0.45) applied in order to make it compatible with later surveys that included corrections for perception and availability bias (Richard et al. 2010), while the 1985 survey was photographic and thus an instantaneous correction of 3.1 was applied (no perception bias in this case) (Asselin and Richard 2011). Based on the confidence intervals alone, there is no significant change in the abundance estimates over time, except for the lowest estimate which came from the 2003 survey. During this particular survey it was noted there were high levels of clumping of narwhal that were not fully captured by the systematic random transect design that was used and the authors thought this may have resulted in a biased estimate (Richard et al. 2010). Despite this, it was a dedicated survey for the Admiralty Inlet stock and is included in the population trajectory. The estimated trajectory for the stock comes from a population dynamic model based on a Bayesian modelling framework that is age and sex structured (for details see Witting 2016). Population trend suggests population is relatively stable, but population estimates are quite variable across years, and abundance estimates have large confidence intervals (Witting 2016).

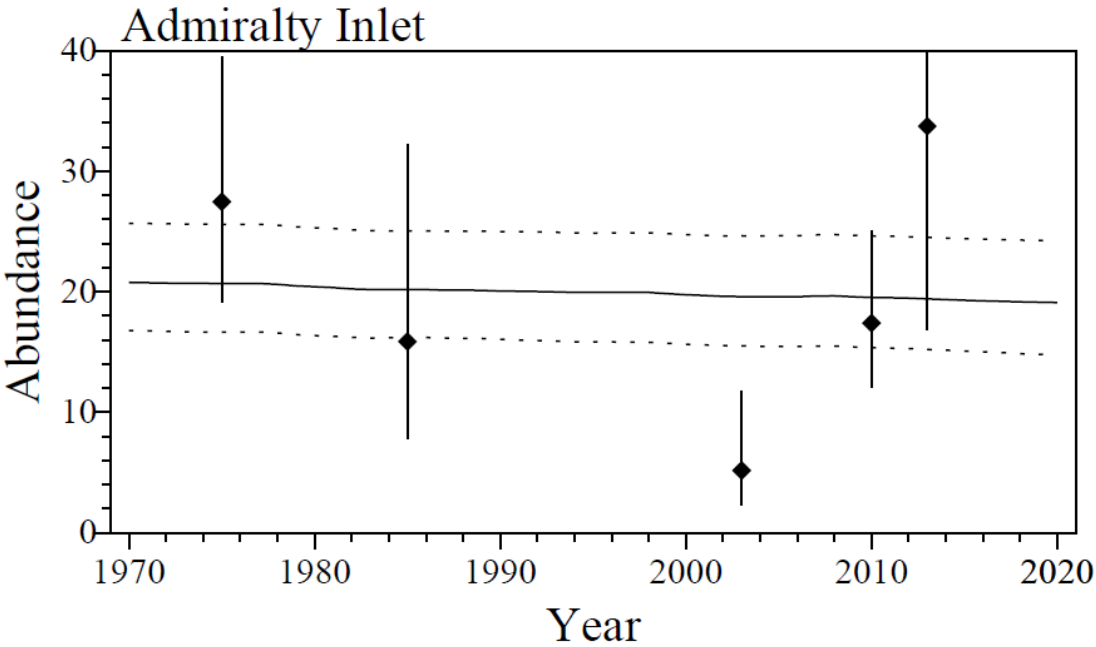

Figure 3. Population trajectory for the Admiralty Inlet stock. Points represent abundance estimates (given in thousands) with 90% confidence intervals. Solid curves indicate the median, and dotted curves the 90 % confidence interval for the estimated models (Witting 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR) method (Wade 1998), corrected for hunting losses (struck and lost), is used to calculate a recommended Total Allowable Landed Catch (TALC):

TALC = PBR/ LRC
Where:
PBR = 0.5 * R_{max} * N_{min} * F_r

LRC is the hunting loss rate correction and is equal to 1.28 ± 0.15 (Richard 2008). R_{max} is the maximum rate of increase for the stock (unknown so the default for cetaceans of 0.04 is used, N_{min} is the 20th percentile of the log-normal distribution of N (most recent survey estimate), and F_r is the recovery factor (we used a value of 1 which indicates a healthy status for the stock (an assumption)). Therefore, the current TALC is set at 233 for this stock, based on the 2010 survey. The new TALC recommendation (which has not yet been implemented) based on the 2013 aerial survey results is 389 (Doniol-Valcroze et al. 2015).

6. Habitat and other concerns

Stock may overlap with Eclipse Sound. The abundance estimate for Admiralty Inlet went up by approximately the same proportion that the Eclipse Sound abundance estimate went down during the 2013 aerial surveys (Doniol-Valcroze et al. 2015), and four narwhals tagged in Eclipse Sound in 2011 travelled into Admiralty Inlet in September – October (Watt et al. 2012). One narwhal's tag applied in Eclipse Sound in 2010 lasted for over a year. This whale made a return migration along the east coast of Baffin Island and to the north shores of Bylot Island, outside of Eclipse Sound from April 17 to June 28, 2011. On July 12, 2011 this whale moved into Navy Board Inlet but, for some reason, turned back and moved into Admiralty Inlet on July 28, 2011, where it remained until the tag finished transmitting on October 10, 2011 (Watt et al. 2012)

7. Status of the stock

Medium size stock that seems stable, but population estimates are quite variable across different survey years. Current removals are considered to be sustainable (Doniol-Valcroze et al. 2015).

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Annex 23: Eclipse Sound Narwhal Stock Eclipse Sound Stock
By: Cortney A. Watt and Rikke Guldborg Hansen

1. Distribution (provide a map if possible) and stock identity

Stock identity is based on consistent summer aggregation reported in TEK, telemetry tracking, and aerial surveys. Summer distribution is indicated in dark blue and labeled ES on Figure 1. Stock identity is supported by telemetry studies which show most narwhals tagged in Eclipse Sound stay within that region in the summer, and typically return there after spending the winter in the Baffin Bay region (Dietz et al. 2001, Heide-Jørgensen et al. 2002, Watt et al. 2012); however, one whale tagged in Eclipse Sound did enter Admiralty Inlet after winter so there may be some overlap among these stocks (see below for further discussion; Watt et al. 2012). There is not strong genetic support as there is a lot of overlap among the stocks for genetic discrimination (de March et al. 2003, Petersen et al. 2011). However, organochlorine contaminants were able to distinguish narwhals hunted in Pond Inlet (de March and Stern 2003), and stable isotopes on narwhal skin showed narwhals hunted in Pond Inlet were significantly different from whales hunted in Admiralty Inlet and East Baffin Island, but overlapped with whales hunted in Jones Sound and Somerset Island (Watt et al. 2012).

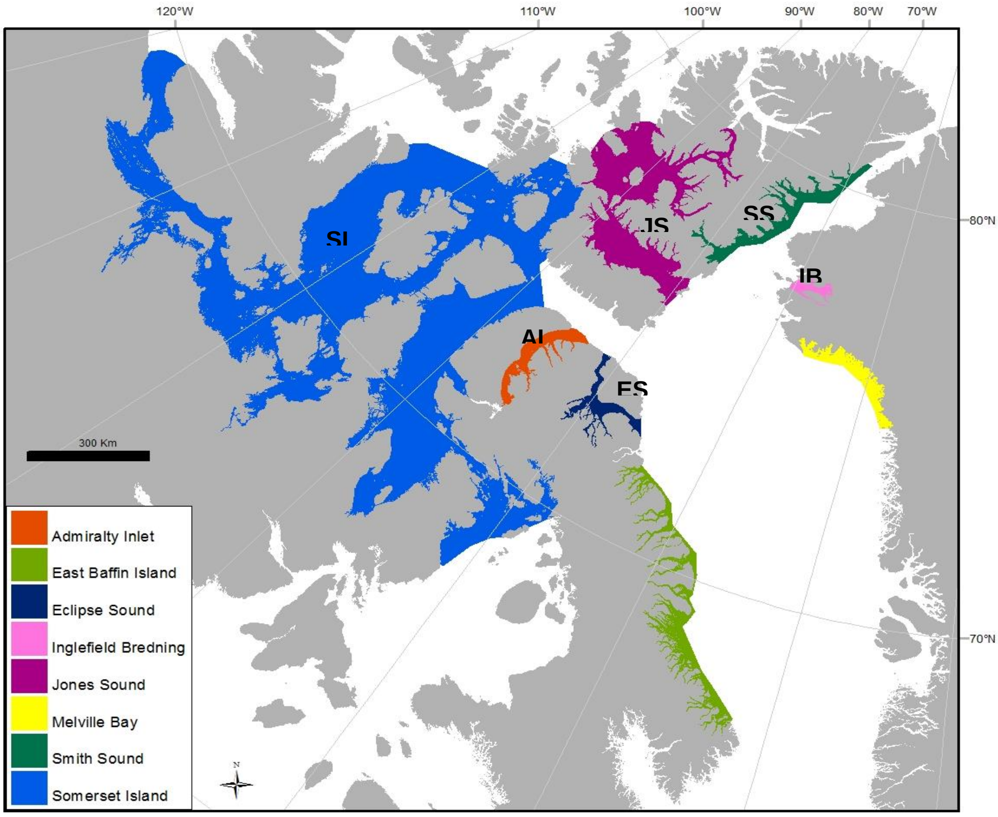

Figure 1. Map indicating the narwhal stocks for the Baffin Bay narwhal population.

2. Abundance

The most recent (2013) abundance estimate for this stock is 10,489 with a CV of 0.24 (Doniol-Valcroze et al. 2015).

This estimate comes from an aerial survey design using a double-platform. Three aircraft were used simultaneously to cover a large area encompassing all of the Canadian narwhal stocks of the Baffin Bay population in August 2013 (Doniol-Valcroze et al. 2015). The extent of the survey areas was based on previous aerial surveys, telemetry tracking studies, TEK, and recent observations by Inuit hunters. Since there has been recent concern about potential movement of narwhals among neighbouring summering regions, the survey with multiple aircraft was designed to survey six of the Baffin Bay stocks during the summer aggregation season - late July through the first three weeks of August prior to the start of fall migration movements. Dates of the survey were chosen to cover areas when sea ice ablation allowed for narwhal to access most of the Arctic Archipelago, and based on the timing of narwhal aggregations in their summering areas as described by TEK and satellite-telemetry data (Doniol-Valcroze et al. 2015). As a result, the last week of July and the first three weeks of August was chosen for the survey, with preference for earlier in August since telemetry data indicated that animals start to move among stocks during the final week of August (Watt et al. 2012).

Transect design was performed in Distance (version 6.1) using coastline shape files. The design was systematic with the first transect line chosen at random. When possible transect lines were oriented in a direction perpendicular to the longest axis of the stratum to maximize the number of lines per stratum (Doniol-Valcroze et al. 2015). For areas where it was assumed narwhal would be in high densities, systematic parallel transects were used. In areas where lower densities were anticipated and landscape patterns permitted, zigzag transects with equally spaced endpoints were used (Doniol-Valcroze et al. 2015).

The survey was flown at an altitude of 1,000 ft, and a target speed of 100 knots using three deHavilland Twin Otter 300 aircraft, each with 4 bubble windows on the sides and run as a double-platform experiment with independent observations platforms at the front and rear of the plane (Doniol-Valcroze et al. 2015). Dual camera systems were mounted under the belly of the plane to allow for continuous digital photography.

Distance sampling methods were used to estimate detection probability away from the track line, while mark-recapture methods were used on sighting data from two observers on either side of the aircraft to correct for perception bias. The distribution of perpendicular distances was different in fiord strata than in the other strata, and thus only non-fiord observations were used to fit the detection function for the non-fiord strata. Examination of the histogram of the perpendicular distances of unique sightings suggested right-truncating the data at 1000 m (i.e., discarding sightings beyond 1000 m), which left 762 unique observations (515 seen by primary observers, 523 by secondary observers, and 276 by both). The shape of the histogram suggested that some narwhals were missed close to the track line despite the bubble windows. Therefore, there was a risk that hazard-rate and half-normal distributions would overestimate the probability of detection and the resulting effective strip width. However, almost a hundred narwhal sightings were made within 100 m of the track line and therefore it seemed inappropriate to lose a large amount of data by left-truncating (i.e., discarding sightings close to the trackline). The shape of the histogram suggested that a gamma distribution would fit better, except that a gamma distribution takes the value zero at zero distance. Therefore, a gamma distribution with an offset term, in addition to half-normal and hazard rate keys, was fitted to the data (Doniol-Valcroze et al. 2015). Model selection was performed on all combinations of covariates (including environmental covariates such as ice cover, cloud cover, sea state, and glare, and a sighting rate covariate which was computed as a rolling average of the number of sightings made by the observer in a 30-second window prior to each sighing). The model with the lowest AIC was one with a truncated gamma key function and the covariates “sighting rate”, “Beaufort” and “glare”. The covariates reduced the detection distance at higher levels (Beaufort >3, Glare=intense, Sighting rate >10 in the last 30 seconds) and resulted in an average probability of detection of 0.48 (CV 2.8%) and an estimated effective strip half width of 481 m (not including perception bias) (Doniol-Valcroze et al. 2015).

For the mark-recapture model to estimate perception bias, models were performed with all combinations of environmental covariates as well as covariates “perpendicular distance”, “observer”, “sighting rate”, “side of aircraft” and “group size”. The best model included “perpendicular distance” and “sighting rate” and the overall probability of detecting a narwhal cluster between the track line and a distance of 1000 m was 0.40 (CV 4.2%) (Doniol-Valcroze et al. 2015).

Fiords were considered their own sampling units and cluster sampling was used to select the fiords to be surveyed (Doniol-Valcroze et al. 2015). In fiords, flights were continuous tracks designed to follow the main axis of the fiord while spreading coverage uniformly based on distance to shore. The resulting data from the fiords was analyzed separately from non-fiord strata. A density surface modelling framework was used to model spatially-referenced count data with the additional information provided by collecting distances to account for imperfect detection (Doniol-Valcroze et al. 2015). First a detection function was fitted to the perpendicular distance data to obtain detection probabilities for clusters of individuals (Doniol-Valcroze et al. 2015). Counts were then summarized for contiguous transect sections and a generalized additive model was constructed with segment counts as the response with areas corrected for detectability (Doniol-Valcroze et al. 2015).

Total surface abundance estimates for stocks were obtained by the additions of the estimated abundances of all the strata that made up that stock's summer range, including results from fiord strata. Variance for the stock-wide abundance estimate was calculated by adding the variances of each stratum; however, identification of duplicates was not straightforward due to the highly aggregated nature of narwhal groups. Because of this, a sensitivity analysis was used to quantify the uncertainty, which allowed the researchers to include an additional variance component to the surface abundance estimate with a CV equal to that of the sensitivity analysis, which ultimately increased the range of uncertainty around the estimate but left the point estimate unchanged (Doniol-Valcroze et al. 2015).

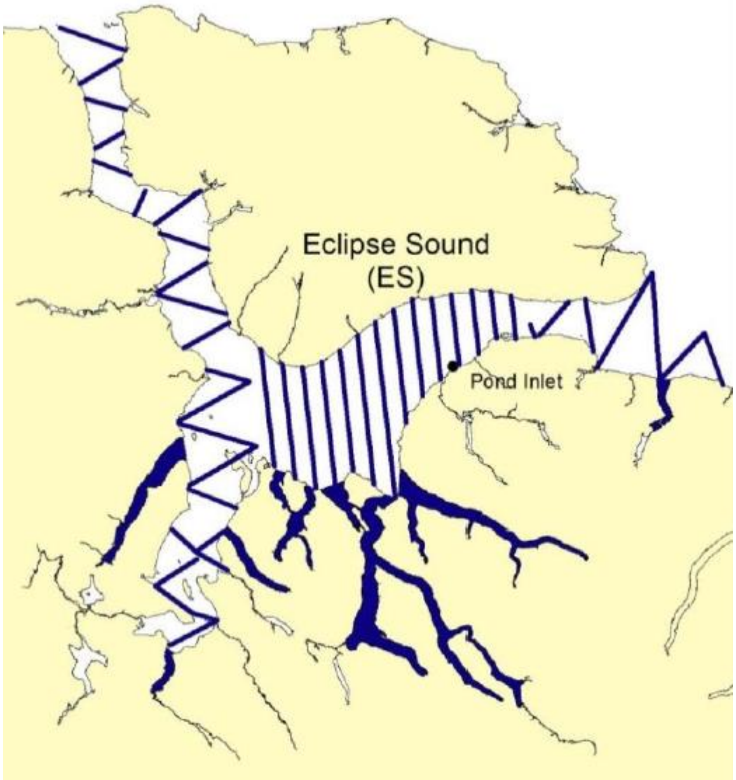

Figure 2. Map of the surveyed strata for the Eclipse Sound stock. Blue lines indicate surveyed transect and blue areas indicate surveyed fiords (Doniol-Valcroze et al. 2015).

An availability bias correction was also applied to the survey data. For the availability bias correction, the time at depth for 24 narwhals fitted with satellite tags near Arctic Bay and Pond Inlet every August from 2009-2012 (Watt et al. 2015) was used to determine the correction for the number of whales missed as a result of being at depth and unavailable for viewing by the surveyors. The time narwhals spent at 0-2 m depths was used to calculate a correction for areas with clear water, while areas with very murky water, the time spent within 0-1 m of the surface was used. This resulted in a correction factor of 3.18 ± 3.37 for clear water areas and a correction of 4.90 ± 0.187 for murky regions (Watt et al. 2015). This correction is appropriate when sightings are instantaneous, but if they are not (such as in aerial surveys), it can positively bias the estimate and as a result a correction factor incorporating the dive cycle of the animal is needed. Three archival time-depth recorders deployed on whales near Pond Inlet and in Creswell Bay in August 1999 and 2000 were used to evaluate a dive-cycle for narwhals. A weighted availability bias correction factor that took into account both the time at depth and the time in view (dive-cycle) was used (2.94 ± 3.4 for the 0-2 m correction and 4.53 ± 3.8 for the 0-1 m bin).

For Eclipse Sound all strata (Figure 2) were flown on August 18, 2013 and August 19, 2013. Few narwhals were observed in the high intensity areas, but instead were aggregated in the southern end of the area, close to shore or within fiords, with a high degree of clumping (Doniol-Valcroze et al. 2015). The surface abundance estimate for the Eclipse Sound stock was 3,566 ± 0.24, and after viewing the photos it was deemed that the water was clear and a correction for the 0-2 m bin should be applied. After a weighted correction of 2.94 ± 0.03 was applied the resulting abundance estimate was 10,489 ± 0.24 (Doniol-Valcroze et al. 2015).

3. Anthropogenic removals

This stock is hunted primarily by the community of Pond Inlet (Heide-Jørgensen et al. 2013); however, there is opportunity for hunters from other communities to hunt these whales on their migration to and from the summering grounds and on the wintering grounds (Witting 2016). Catches in Table 1, however, only reflect whales that are hunted within the defined summering region since it is difficult to determine the number of animals from this stock hunted by other communities. In some Canadian communities with a community-based management system, killed-lost and wounded-lost narwhal numbers were documented by hunters between 1999 and 2005 (Table 2). From the narwhal hunts where losses are reported, Richard (2008) calculated a hunting loss rate correction (LRC) (Table 2).

LRC = HM / LC where
HM = the estimated total hunting mortality, or the sum of the landed catch and hunting loss
LC = Landed Catch

The estimated hunting loss was calculated as:

$HM = (HM_{min} + HM_{max})/2$ where
HM_{min} = number of animals landed plus the ones reported sunk and lost
$HM_{max} = HM_{min}$ + the number reported wounded and escaped

This HM estimate used by Richard (2008) assumes that half of the animals wounded and escaped later die from their injuries. This assumption was untested but considered reasonable since both whales with wound scars are later seen alive but dead whales have also washed up after a hunt suggesting some whales survive from their wounds while others perish (Richard 2008). Table 1 indicates the total reported landed catches, and the catches multiplied by a struck and loss factor of 1.28 ± 0.15 (Richard 2008). This data comes from 1999-2005 and is hunter reported for all types of hunt combined for each of the communities. An older study (Roberge and Dunn 1990) investigated struck and lost rates from the community of Arctic Bay in the open water season in 1983 and 1988, on the floe edge in 1988 and 1989, and at the ice crack in 1978, 1988, and 1999 (Table 3). Most of the hunt in Pond Inlet occurs in the open water season, which has a struck and loss factor reported by Roberge and Dunn (1990) of 1.40 ± 0.14. In this study researchers monitored the hunt when possible and reported values. Application of this rate rather than the 1.28 reported by Richard (2008), changes catches previous to 1999 by an average of 11 whales, and a maximum of 24 whales (results in brackets in Table 1). Ideally a struck and loss factor would be applied to each catch that occurs through different hunting methods; unfortunately this information is not reported. However based on hunt dates (for which we have some information from 2003-2012), the majority of the hunt occurs in the open water season (68% for Pond Inlet (Doniol-Valcroze 2014)). Currently in Canada the struck and loss rate from Richard (2008) is used, since it is the most up to date.

The stock is also hunted on the wintering grounds in Greenland where 1% of the hunt in Uummannaq and 52% of the hunt in Disko Bay are believed to be from the Eclipse Sound stock (see Table 4).

Table 1. Reported landed catches for Pond Inlet. From 1977 these catches are based on the number of issued tags and recorded by Fisheries and Oceans Canada; prior to 1977 the numbers come from a variety of sources (see reference list) but typically rely on reports by hunters, or RCMP records. Total catch including struck and lost animals is indicated using the newest struck and lost factor (1.28 from Richard (2008)), and using the 1.40 reported for open water hunts by Roberge and Dunn (1990) for years prior to 1999 indicated in brackets.

Year	Pond Inlet (landed catches)	Reference for reported landed catch	Pond Inlet Catches + 1.28 S&L factor (1.40 S&L factor)
1970	nr	Mansfield et al. (1975)	nr
1971	nr	Mansfield et al. (1975)	nr
1972	32	Strong (1989), Mitchell and Reeves (1981)	41 (45)
1973	200	Strong (1989), Mansfield et al. (1975)	256 (280)
1974	100	Strong (1989), Stewart (2007)	128 (140)
1975	77	Strong (1989)	99 (108)
1976	125	Strong (1989)	160 (175)
1977	107	Strong (1989)	137 (150)
1978	150	Strong (1989)	192 (210)
1979	94	Strong (1989)	120 (132)
1980	96	Strong (1989)	123 (134)
1981	82	Strong (1989)	105 (115)
1982	100	Strong (1989)	128 (140)
1983	104	Strong (1989)	133 (146)
1984	45	Strong (1989)	58 (63)
1985	99	Watt and Hall 2017	125 (137)
1986	100	Strong (1989)	128 (140)
1987	52	Strong (1989)	67 (73)
1988	53	DFO (1991)	68 (74)
1989	77	DFO (1992)	99 (108)
1990	69	DFO (1992)	88 (97)
1991	100	DFO (1993)	128 (140)
1992	99	DFO (1994)	127 (139)
1993	78	DFO (1995)	100 (109)
1994	91	DFO (1996)	116 (127)
1995	73	DFO (1997)	93 (102)
1996	100	DFO (1999)	128 (140)
1997	75	Stewart (2007)	96 (105)
1998	105	Doniol-Valcroze (2014)	134 (147)
1999	132	Doniol-Valcroze (2014)	169
2000	167	Doniol-Valcroze (2014)	214
2001	65	Doniol-Valcroze (2014), Stewart (2007)	83
2002	63	Doniol-Valcroze (2014)	81
2003	67	Doniol-Valcroze (2014), Stewart (2007)	86
2004	65	Hall et al. (2015)	83
2005	62	Hall et al. (2015)	79
2006	88	Hall et al. (2015)	113
2007	65	Hall et al. (2015)	83
2008	73 [α]	Hall et al. (2015)	93

Year	Pond Inlet (landed catches)	Reference for reported landed catch	Pond Inlet Catches + 1.28 S&L factor (1.40 S&L factor)
2009	44	Hall et al. (2015)	56
2010	62	Hall et al. (2015)	79
2011	112	Hall et al. (2015)	143
2012	97	Hall et al. (2015)	124
2013	147	Hall et al. (2015)	188
2014	135	Watt and Hall (2017)	173
2015	190 [α]	Watt and Hall (2017)	243

[α] In 2008 and 2015 there were ice entrapment events near Pond Inlet where narwhal were harvested (624 and 231 respectively) and these are not included in this table.

Table 2. Table indicating how the struck and loss factor for this stock is calculated. Table is directly from Richard (2008).

Community	Year	Landed	Wounded/ Escaped	Sunk and Lost	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Community specific average Loss Rate Correction
Pond Inlet	1999	130	14	16	146	160	153	1.18	
	2000	166	21	10	176	197	187	1.12	
	2001	63	5	27	90	95	93	1.47	
	2002	92	1	13	105	106	106	1.15	1.23 ± 0.16
Qikiqtarjuaq	1999	81	30	25	106	136	121	1.49	
	2000	137	79	40	177	256	217	1.58	
	2001	89	8	9	98	106	102	1.15	
	2002	81	40	16	97	137	117	1.44	
	2004	96	12	9	105	117	111	1.16	1.36 ± 0.20
Repulse	1999	156	68	30	186	254	220	1.41	
	2000	49	9	5	54	63	59	1.19	
	2001	100	38	21	121	159	140	1.4	
	2002	57	0	8	65	65	65	1.14	
	2003	30	0	5	35	35	35	1.17	
	2005	72	25	3	75	100	88	1.22	1.26 ± 0.12
Arctic Bay	2001	134	20	4	138	158	148	1.1	
	2003	129	14	22	151	165	158	1.22	
	2004	122	22	33	155	177	166	1.36	1.23 ± 0.13
Kugaaruk	2001	41	18	8	49	67	58	1.41	
	2003	24	4	2	26	30	28	1.17	1.29 ± 0.17
Average across communities									**1.28 ± 0.15**

Table 3. Table indicating how an older struck and loss factor for Arctic Bay was calculated from observations of different hunting types from Roberge and Dunn (1990).

Hunt	Year	Landed	Wounded/ Escaped	Sunk and Lost/mortally wounded	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Hunt specific average Loss Rate Correction
Floe edge	1988	6	6	8	14	20	17	2.83	
	1989	16	0	5	21	21	21	1.31	2.07 ± 1.08
Open water	1983	4	2	1	5	7	6	1.50	
	1988	13	6	1	14	30	17	1.31	**1.40 ± 0.14**
Ice crack	1987	15	13	8	23	36	30	1.97	
	1988	29	8	17	46	54	50	1.72	
	1989	50	7	13	63	70	67	1.33	1.67 ± 0.32
Average across hunt types									**1.71 ± 0.55**

Table 4. Catches of narwhals from official reports by municipality with corrections for under-reportings (in parenthesis) for 1954 to 2011. Numbers in square brackets are from *special reports*. The column ‘under-reporting’ shows the sum of the corrections for under-reporting or ‘ALL’ if it is a general correction factor for all areas. ‘na’ means that no data are available. Data from 2007-08 are preliminary. DB=Disko Bay, UUM=Uummannaq, UPV=Upernavik. Data were compiled from Prime Minister’s Second Department (1951), Kapel (1977), Kapel (1983), Kapel and Larsen (1984), Kapel (1985), Born and Kapel (1986) and Born (1987).

YEAR	QAANAAQ	UPER-NAVIK	UUMMAN-NAQ	DISKO BAY	SISI-MIUT	MANIIT-SOQ	NUUK	PAAMIUT-QAQORTOQ	TOTAL	ICE ENTRAPMENT
1949	38	16	1	6					61	
1950										
1951										85 DB
1952										450 DB
1954	na		45		1			1	47	
1955	na	179	2	14					195	
1956	na	15	282	21					318	156 UPV, 250 UUM
1957	na	55	11	15					81	
1958	na	24	3	45		1			73	
1959	na	32	8	16				1	57	
1960	na	25	296	7	1	1	1	1	332	
1961	134	25	5	38				1	203	272 UUM
1962	182	17	11	12				1	213	
1963	275	10	3	29					317	
1964	275	17	11	11					314	
1965	na	33	37	33	1	1			105	
1966	na	39	23	43		3	2		110	
1967	na		131			9			140	31 DB
1968	na		454			18			472	161 DB, 50 UPV, 84 UUM
1969	na		174			30			204	Some DB, 50 UPV
1970	na		313			9			322	100 DB
1971	na		146			40			186	
1972	na		84			23			107	
1973	na		191			8			199	
1974	8		136			3			147	

Table 3. Continued

YEAR	QAANAAQ	UPER-NAVIK	UUMMAN-NAQ	DISKO BAY	SISIMIUT	MANIITSOQ	NUUK	PAAMIUT-QAQORTOQ-	TOTAL	ICE ENTRAPMENT
1975	1	54	11	44		6		1	266 (149)	
1976	9	22	27	57					264 (141)	
1977	16	62	113	53	8	1			387 (134)	
1978	110	56	183	262		1			612	
1979	120	22	132	100			3		377	
1980	130	61	146	120		4	1		462	
1981	118	83	140	249			18	1	609	
1982	164	59	162	76					461	45 DB
1983	135 (25)	72 (30)	164	68 (10)					439 (65)	
1984	274	80	245	66 (15)	1				666 (15)	35 UUM
1985	115 (115)	34 (20)	39	67		1			256 (135)	
1986	na	81	97	23		36			237	
1987	na	145	334	25			1		505	
1988	na		206						500 (294)	
1989	na	37	288	2			5		332	
1990	na	100 (73)	1019	11					1057 (100)	
1991	na		27	> 40					na	27 UUM
1992	na	37	288	2			5		342	
1993	144	66	301	75	10	6	4	8	614	
1994	183	59	297	268	6	14	7	11	845	150 DB
1995	107	94	159	108	4	5	8		485	
1996	45	69	405	154	10	4	2	2	691	
1997	66	90	381	156	13	5	9	26	746	
1998	94	105	344	163	21	18	6	24	775	
1999	115	119	253	174	28	24	17	15	745	
2000	109	150	106	155	27	8	0	6	561	
2001	145	155	95	119	1	2	15	3	535	
2002	94	164	180	97	12	11	3	2	563	
2003	113	146	174	114	4	0	2	2	554	
2004	178	53	67	73	2	1	0	0	374	
2005	[70] 137	[74] 71	[137] 161	[47] 39	0	0	0	0	[328] 408	
2006	[94] 99	[58] 62	[55] 72	[4] 53	1	2	0		[211] 289	
2007	[21] 139	[17] 102	[52] 67	[56] 63	0	2	0	1	[146] 374	
2008	129	74	87	47	0	0	0	0	337	
2009	90	110	91	88	0	0	0	1	380	41 in Qaanaaq
2010	108	30	42	45	0	0	0	0	225	53 in Qaanaaq
2011	74	60	77	39	0	0	0	1	251	
2012	144	70	42	179	0	0	0	1	311	125 at Kangersuatsiaq
2013	90	64	78	50	0	0	0	1	283	
2014	114	101	69	50	0	0	0	0	334	
2015	92	54	42	29	0	0	1	0	218	
2016	93	79	120	55	0	0	1	0	348	

4. Population trajectory

Two surveys with the goal of assessing abundance have been conducted for the Eclipse Sound stock. Figure 3 indicates the trajectory given the abundance estimates and associated confidence intervals for the different surveys. Based on the confidence intervals alone, there is no significant change in the abundance estimates over time. The estimated trajectory for the stock comes from a population dynamic model based on a Bayesian modelling framework that is age and sex structured (for details see Witting 2016). Unfortunately there are not enough survey estimates to determine a trend for this stock (Witting 2016).

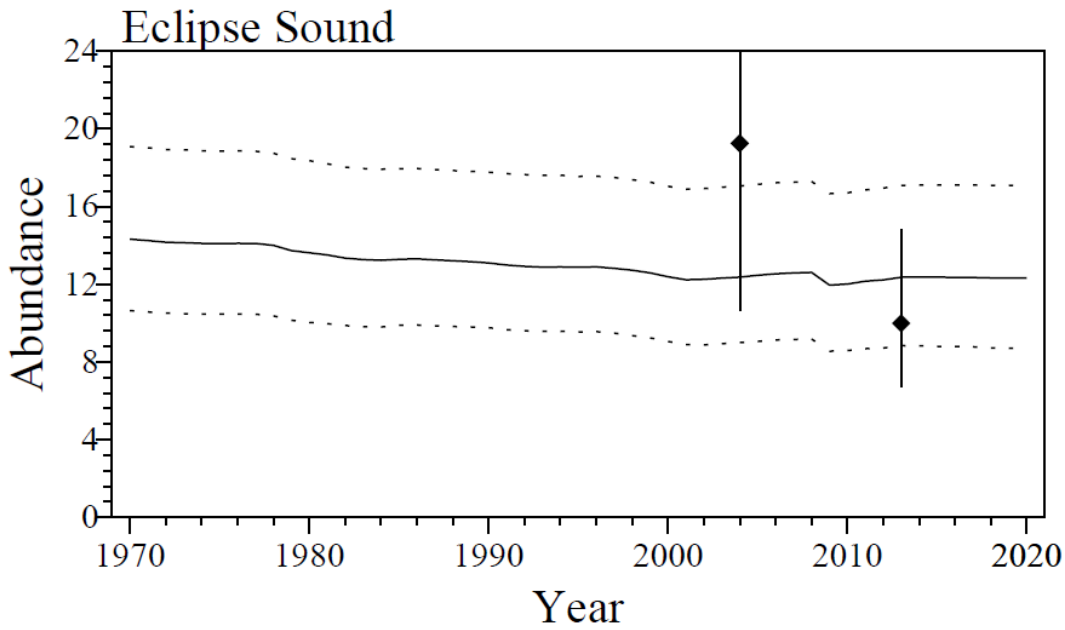

Figure 3. Population trajectory for the Eclipse Sound stock. Points represent abundance estimates (given in thousands) with 90% confidence intervals. Solid curves indicate the median, and dotted curves the 90 % confidence interval for the estimated model (Witting 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR) method (Wade 1998), corrected for hunting losses (struck and lost), is used to calculate a recommended Total Allowable Landed Catch (TALC):
TALC = PBR/ LRC
Where:
PBR = 0.5 * R_{max} * N_{min} * F_r

LRC is the hunting loss rate correction and is equal to 1.28 ± 0.15 (Richard 2008). R_{max} is the maximum rate of increase for the stock (unknown so the default for cetaceans of 0.04 is used, N_{min} is the 20th percentile of the log-normal distribution of N (most recent survey estimate), and F_r is the recovery factor (we used a value of 1 which indicates a healthy status for the stock (an assumption)). Therefore, the current TALC is set at 236 for this stock, based on the 2010 survey. The new TALC recommendation (which has not yet been implemented) based on the 2013 aerial survey results is 134 (Doniol-Valcroze et al. 2015).

6. Habitat and other concerns

Stock may overlap with Admiralty Inlet. The abundance estimate for Eclipse Sound went down by approximately the same proportion that the Admiralty Inlet abundance estimate went up when comparing the surveys done in 2004 (Eclipse Sound (Richard et al. 2010)) and 2010 (Admiralty Inlet (DFO 2012)) to the 2013 aerial survey (Doniol-Valcroze et al. 2015). In addition, four narwhals tagged in Eclipse Sound in 2011 travelled into Admiralty Inlet in September – October (Watt et al. 2012). One narwhal's tag equipped in Eclipse Sound in 2010 lasted for over a year. This whale made a return migration along the east coast of Baffin Island and to the north shores of Bylot Island, outside of Eclipse Sound from April 17 to June 28, 2011. On July 12, 2011 this whale moved into Navy Board Inlet but, for some reason, turned back and moved into Admiralty Inlet on July 28, 2011, where it remained until the tag finished transmitting on October 10, 2011 (Watt et al. 2012)

Eclipse Sound has been identified as an area important for narwhal calving (Mathewson 2016).

7. Status of the stock

Medium size stock that seems stable but population estimates are quite variable across different survey years, and only two surveys have been conducted. Current removals are considered to be sustainable (Doniol-Valcroze et al. 2015).

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Annex 24: Inglefield Bredning Narwhal Stock

By: Rikke Guldborg Hansen, Mads Peter Heide-Jørgensen and Eva Garde

1. Distribution and stock identity

Stock identity is based on consistent summer aggregation, aerial surveys, local knowledge and hunting patterns. Summer distribution of the Inglefield Bredning stock is indicated in light pink on Figure 1. The stock is hunted by the Qaanaaq hunting region (Figure 2).

Migration patterns are unknown.

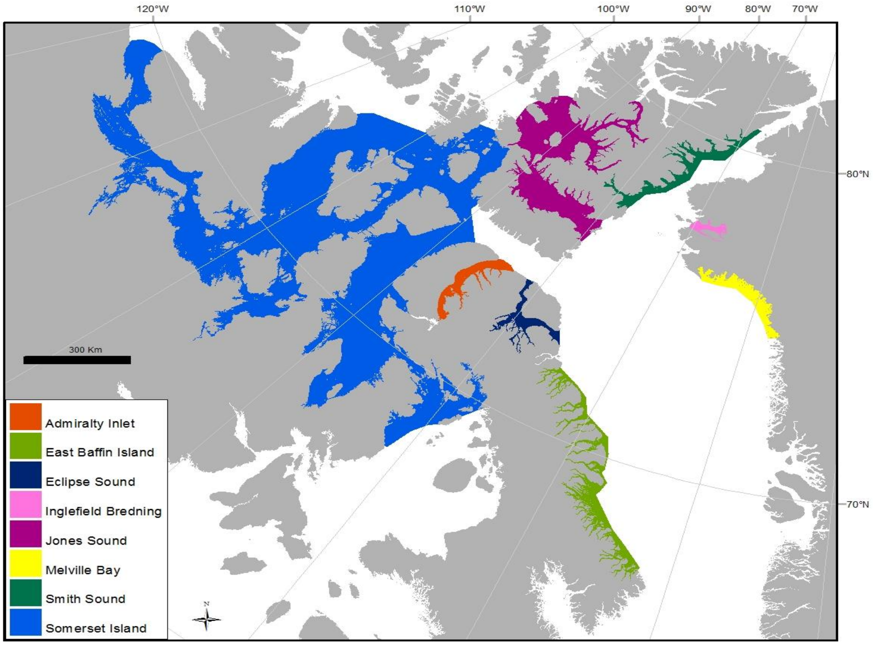

Figure 1. Map indicating the narwhal stocks for the Baffin Bay narwhal population.

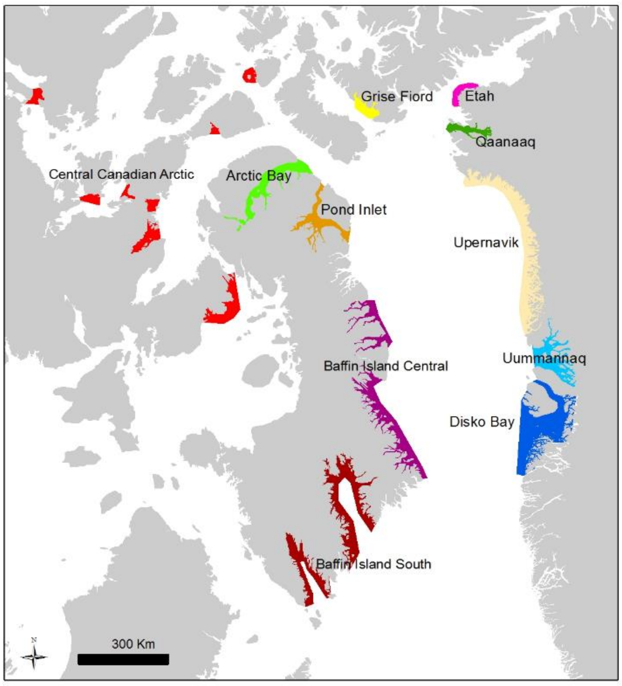

Figure 2. Map indicating the most important hunting regions for the Baffin Bay narwhal population.

Narwhals from the North West Atlantic show low levels of nucleotide and haplotype diversity based on the first 287 base pairs in the mitochondrial control region (Palsbøll et al. 1997). Despite the low degree of variation, frequencies of common haplotypes differed markedly between areas. In East Greenland only one haplotype was found supporting the hypothesis of little or no gene flow between eastern and western Greenland. Heterogeneity was found between Melville Bay narwhals and narwhals from the Avernersuaq district which includes Inglefield Bredning. Hence little gene flow is occurring between the western Greenland summer areas and northern Baffin Bay (eastern Canada and Avanersuaq). Within the northern Baffin Bay samples no significant levels of heterogeneity was found indicating some gene flow between summer grounds within this area. Narwhals show annual fidelity to summer and autumn feeding grounds and pods from these feeding grounds utilizes the same winter grounds.

Watt et al. (2013) conducted stable isotope analysis on narwhal skin samples collected by Inuit hunters during their subsistent narwhal hunt in Canada and Greenland. Stable isotope analysis on carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) revealed the three narwhal populations of Baffin Bay (BB), Northern Hudson Bay (NHB) and East Greenland (EG) to have distinct stable isotope values that were not expected based on geographic differences. Also, males in all populations had significantly higher $\delta^{13}C$.

2. Abundance

The most recent (2007) abundance estimate for this stock is 8,368 (cv=0.25; 95% CI 5,209-13,442. Heide-Jørgensen et al. 2010). The estimate is corrected for both perception and availability bias.

This estimate comes from a visual aerial line transect survey conducted as a double-observer experiment in a fixed-winged, twin-engine aircraft (DeHavilland Twin Otter) with a target altitude and speed of 213m and 168km h-1, respectively. The front observers (observer 1) acted independently of those in the rear (observer 2) and vice versa. Declination angles to sightings, species and group size were recorded when the animals came abeam. Beaufort sea state was recorded at the start of the day and again when it changed. Decisions about duplicate detections (animals seen by both observer 1 and 2) were based on coincidence in timing and location of sightings (Heide-Jørgensen et al. 2010).

Strata delineation was based on previous surveys as well as local knowledge. Four strata were identified and the two strata in Inglefield Bredning were surveyed by transects aligned north-south and the two side fiords were surveyed in a zig-zag manner, covering ~2546 km (Figure 3, Heide-Jørgensen et al. 2010). The survey area was covered once and the sightings were concentrated in the eastern part of Inglefield Bredning.

This survey was conducted at the same time as a survey in Melville Bay. Because of the small number of sightings in Melville Bay, sightings from both regions were combined and a single detection function was estimated. For the DS model both half-normal and hazard-rate functions were fitted. Explanatory variables were included to model any dependency between detections. The available explanatory variables were group size, Beaufort sea state (as a factor with levels 0 to 4), side of plane (left and right), and region (Melville Bay or Inglefield Bredning). The same explanatory variables were included in the MR model in addition to a variable indicating observer (2 levels). Too few sightings in some strata precluded the use of stratum as an explanatory variable. For the DS model, region and side of plane were the most important explanatory variables. For the MR model, group size and observer were the most important explanatory variables. The average probability of detection on the track line was estimated for each observer, and this indicated that observer 2 had a slightly higher probability of detection on the track line than observer 1; 0.83 (cv=0.04) for observer 2 compared with 0.77 (cv=0.05) for observer 1. The probability of detection on the track line for both observers combined was estimated to be 0.96 (cv=0.02). Correction for availability of the at-surface-abundance was based on availability correction factors obtained from two whales from August-September 2007 (*a*=0.21; *cv*=0.09). The fully corrected MRDS abundance estimate was 8,368 narwhals (cv=0.25; 95% CI 5,209-13,442). The time between when a group of whales was first seen and when it passed abeam was 0.85 s (SD=2.0) in Inglefield Bredning, and because of the small interval, probably due to the high density of whales, no corrections were made for the non-instantaneous sighting process.

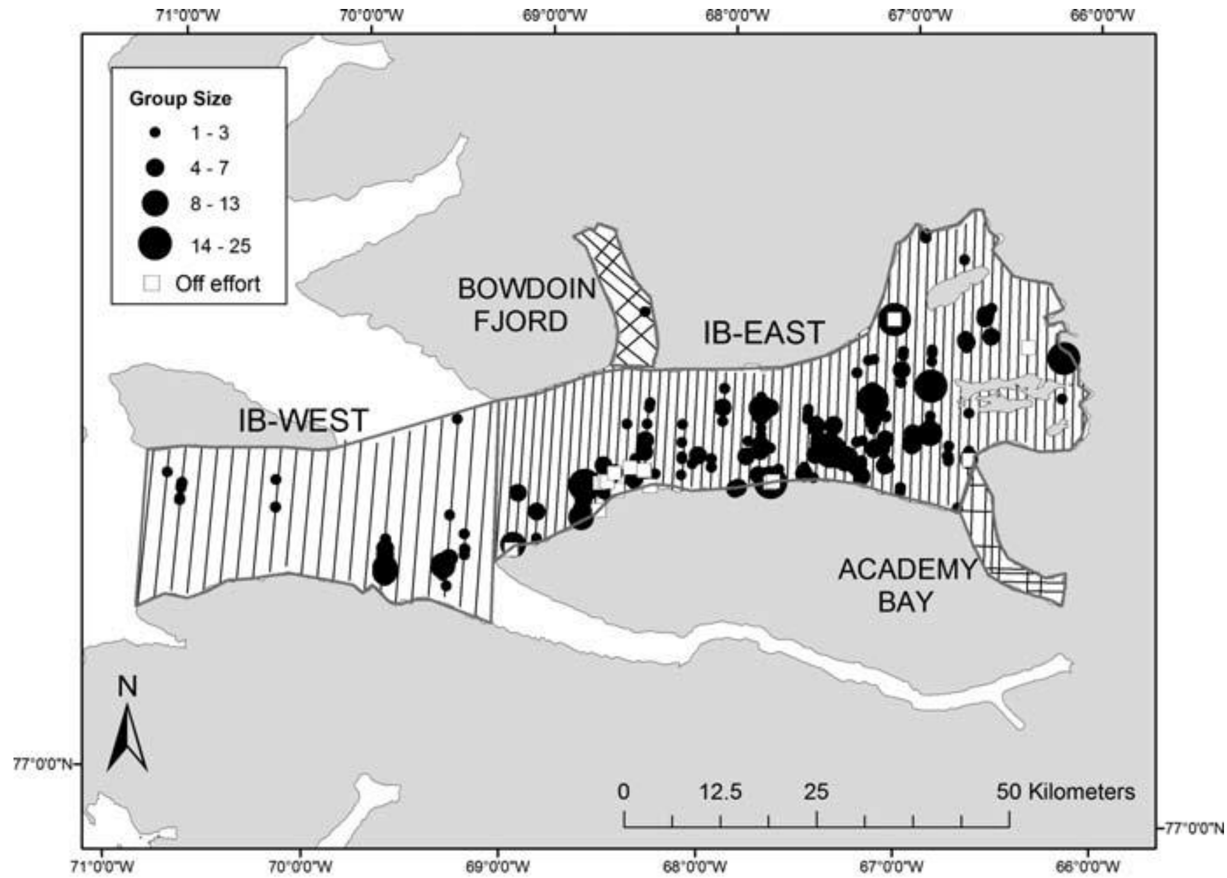

Figure 3. Map of the survey area and transects lines for the Inglefield Bredning stock.

3. Anthropogenic removals

This stock is hunted by the communities in Qaanaaq hunting region during April-September (Figure 2 and 4, Witting 2016). The quota is set on the basis of the allocation model developed by JCNB SWG.

It is generally assumed that the loss rate was low before 1950 where all catches were corrected by 5% to account for some losses. No studies of losses have been conducted in Greenland but inferences can be made from studies in other areas. In the municipality of Qaanaaq local hunting rules requires the attachment of hand-harpoons on the whales before they can be shot. This severely reduces the loss rate and a loss rate of 5% is arbitrarily applied to the catches in Inglefield Bredning to account for both whales that are killed-but-lost and calves that are separated from mothers. Catches in Melville Bay, however, consists of hunting in both the municipality of Qaanaaq and in Upernavik that doesn't require the use of hand-harpoons. Roughly half the whales in Upernavik and Melville Bay are taken under the harpoon requirements (5% loss rate) and the other half is taken in ice edge and open water situations.

For narwhal hunting in open water in Canada Weaver and Walker reported loss rates between 32% and 55%, or catch correction factors of 1.5-2.2. Roberge and Dunn reported catch correction factors for narwhals in Canada to range from 1.11 in open water to 1.41 at the ice crack and 1.56 at the floe edge or ice edge (NAMMCO/SC/22-JCNB/SWG/2015-JWG/06).

For Greenland it is assumed that a catch correction factor of 1.30 covers both the open water hunt and the hunt from ice cracks and the ice edge (for the Melville Bay-Upernavik area a factor of 1.15 is used). The correction factor of 1.30 also covers the open-water hunt in late autumn just before freeze-up, which is a type of hunt where loss rates have not been estimated. If anything the correction factor of 1.30 applied here is downward biased.

Official catch statistics for monodontids from West Greenland include catches that are taken from whale pods entrapped in the ice. It has been suggested that mortality in ice entrapments occasionally is part of the natural mortality (Siegstad and Heide-Jørgensen 1994). To allow for analyses of removals without catches in ice entrapments these are shown separately from the mortality genuinely caused by humans.

4. Population trajectory

Aerial surveys with the goal of assessing abundance have been conducted for the Inglefield Bredning stock. Figure 4 indicates the trajectory given the abundance estimates and associated confidence intervals for the different surveys. Based on the confidence intervals alone, there is no significant change in the abundance estimates over time. The estimated trajectory for the stock comes from a population dynamic model based on a Bayesian modelling framework that is age and sex structured (for details see Witting 2016).

The Inglefield Bredning stock is estimated to be depleted to levels below the MSYL, implying that future harvest levels should be set to ensure an increasing number of narwhals. The stock appears relatively stable.

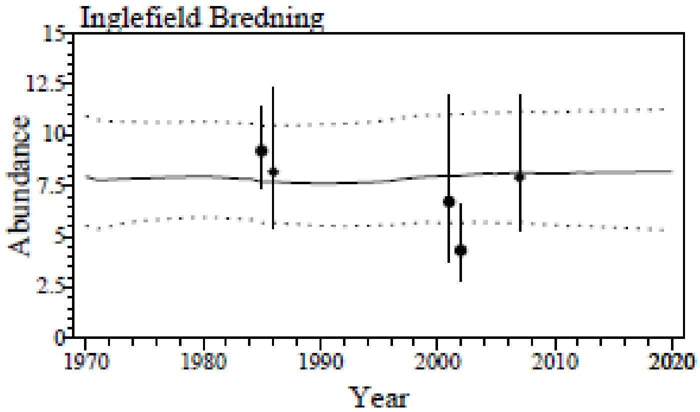

Figure 4. The trajectories of the Inglefield Bredning stock. Points with bars are the abundance estimates (given in thousands) with 90% confidence intervals. Solid curves indicate the median, and dotted curves the 90% CI, of the estimated model (Witting 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

In order to assess the sustainability of catches on this stock, a Bayesian framework was used to estimate the probabilities that an assumed management objective would be fulfilled for potential future catches. While the sustainability of the hunt has to be identified at the population level, recommendations on the sustainability of potential future hunts should preferably be addressed in relation to hunting grounds. To achieve this, for a given set of potential future catches for each hunt, the allocation model developed at JCNB was used to calculate the distributions of future catches for the different populations, with these distributions reflecting the uncertainty in the allocation of catches between the populations (Witting 2016). Then, by having these distributions, for the catches of each percentile of these distributions, the probability that the assumed management objective would be fulfilled for the different populations, could be calculated.

Table 2. Catches of narwhals from official reports by municipality with corrections for underreportings (in parenthesis) for 1954 to 2011. Numbers in square brackets are from *special reports*. The column 'under-reporting' shows the sum of the corrections for under-reporting or 'ALL' if it is a general correction factor for all areas. 'na' means that no data are available. Data from 2007-08 are preliminary. DB=Disko Bay, UUM=Uummannaq, UPV=Upernavik. Data were compiled from Prime Minister's Second Department (1951), Kapel (1977), Kapel (1983), Kapel and Larsen (1984), Kapel (1985), Born and Kapel (1986) and Born (1987; NAMMCO/SC/22-JCNB/SWG/2015-JWG/06)

<table>
<tr><th>YEAR</th><th>QAANAAQ</th><th>UPERNAVIK</th><th>UUMMANNAQ</th><th>DISKO BAY</th><th>SISIMIUT</th><th>MANIITSOQ</th><th>NUUK</th><th>PAAMIUT-QAQORTOQ</th><th>TOTAL</th><th>ICE ENTRAPMENT</th></tr>
<tr><td>1949</td><td>38</td><td>16</td><td>1</td><td>6</td><td colspan="4"></td><td>61</td><td></td></tr>
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<tr><td>1956</td><td>na</td><td>15</td><td>282</td><td>21</td><td></td><td></td><td></td><td></td><td>318</td><td>156 UPV, 250 UUM</td></tr>
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<tr><td>1960</td><td>na</td><td>25</td><td>296</td><td>7</td><td>1</td><td>1</td><td>1</td><td>1</td><td>332</td><td></td></tr>
<tr><td>1961</td><td>134</td><td>25</td><td>5</td><td>38</td><td></td><td></td><td></td><td>1</td><td>203</td><td>272 UUM</td></tr>
<tr><td>1962</td><td>182</td><td>17</td><td>11</td><td>12</td><td></td><td></td><td></td><td>1</td><td>213</td><td></td></tr>
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<tr><td>1965</td><td>na</td><td>33</td><td>37</td><td>33</td><td>1</td><td>1</td><td></td><td></td><td>105</td><td></td></tr>
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<tr><td>1970</td><td>na</td><td colspan="3">313</td><td colspan="4">9</td><td>322</td><td>100 DB</td></tr>
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<tr><td>1974</td><td>8</td><td colspan="3">136</td><td colspan="4">3</td><td>147</td><td></td></tr>
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<tr><td>1981</td><td>118</td><td>83</td><td>140</td><td>249</td><td></td><td></td><td>18</td><td>1</td><td>609</td><td></td></tr>
<tr><td>1982</td><td>164</td><td>59</td><td>162</td><td>76</td><td></td><td></td><td></td><td></td><td>461</td><td>45 DB</td></tr>
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</table>

YEAR	QAANAAQ	UPERNAVIK	UUMMANNAQ	DISKO BAY	SISIMIUT	MANIITSOQ	NUUK	PAAMIUT-QAQORTOQ	TOTAL	ICE ENTRAPMENT
1991	na		27	> 40					na	27 UUM
1992	na	37	288	2			5		342	
1993	144	66	301	75	10	6	4	8	614	
1994	183	59	297	268	6	14	7	11	845	150 DB
1995	107	94	159	108	4	5	8		485	
1996	45	69	405	154	10	4	2	2	691	
1997	66	90	381	156	13	5	9	26	746	
1998	94	105	344	163	21	18	6	24	775	
1999	115	119	253	174	28	24	17	15	745	
2000	109	150	106	155	27	8	0	6	561	
2001	145	155	95	119	1	2	15	3	535	
2002	94	164	180	97	12	11	3	2	563	
2003	113	146	174	114	4	0	2	2	554	
2004	178	53	67	73	2	1	0	0	374	
2005	[70] 137	[74] 71	[137] 161	[47] 39	0	0	0	0	[328] 408	
2006	[94] 99	[58] 62	[55] 72	[4] 53	1	2	0		[211] 289	
2007	[21] 139	[17] 102	[52] 67	[56] 63	0	2	0	1	[146] 374	
2008	129	74	87	47	0	0	0	0	337	
2009	90	110	91	88	0	0	0	1	380	41 in Qaanaaq
2010	108	30	42	45	0	0	0	0	225	53 in Qaanaaq

YEAR	QAANAAQ	UPERNAVIK	UUMMANNAQ	DISKO BAY	SISIMIUT	MANIITSOQ	NUUK	PAAMIUT-QAQORTOQ	TOTAL	ICE ENTRAPMENT
2011	74	60	77	39	0	0	0	1	251	
2012	144	70	42	179	0	0	0	1	311	125 at Kangersuatsiaq
2013	90	64	78	50	0	0	0	1	283	
2014*	114	101	69	62	0	0	0	0	346	

Management defines the total allowable takes for the different hunts (region and season), as these cannot generally be allocated directly to the different summer aggregation. The total allowable takes for the different hunts, with the associated estimates of the probabilities that these takes from 2015 to 2020, will allow the management objective to be fulfilled for the different summer aggregations. These latter probability estimates have 90% confidence limits that reflect the uncertainty of the summer aggregation origin of the animals taken in the different hunts.

The estimated total allowable takes for the different summer aggregations that will meet the management objective with probabilities from 0.5 to 0.95 are presented in Witting et al. 2016. The estimated total allowable take for the Inglefield Bredning stock is 98 individuals per year (2015-2020) with 70% probability for a larger population size in 2020.

6. Habitat and other concerns

Possible concerns include changes in sea ice regime, traffic, seismic exploration and fishing of the halibut resources in central Baffin Bay.

7. Status of the stock

The Inglefield Bredning stock is considered to be a small but stable population.

8. Life history

Garde et al. (2015) estimated life history parameters for narwhals from East and West Greenland (n=282) based on age estimates from aspartic acid racemization (AAR) of eye lens nuclei. The species-specific age equation used, $420.32X - 24.02 \cdot \text{year}$ where X is the D/L ratio, was determined from data from Garde et al. (2012) by regressing aspartic acid D/L ratios in eye lens nuclei against growth layer groups in tusks (n=9). Asymptotic body length was estimated to be 399 ± 5.9 cm for females at age 25 years and 456 ± 6.9 cm for males from West Greenland at age 28 years. Age at sexual maturity was assessed based on data from reproductive organs and was estimated to be 8–9 years for females and 12–20 years for males. Length at sexual maturity was ~340 cm for females and 350-400 cm for males. Estimated age at 1st parturition was 9-10 years. Oldest pregnant female was close to 70 years. Pregnancy rates for West Greenland were estimated to be 0.38. Maximum life span expectancy was found to be approximately 100 years. A population projection matrix was parameterized with the data on age structure and fertility rates. The annual rate of increase of narwhals in West Greenland was estimated to be 2.6%.

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Annex 25: Melville Bay Narwhal Stock

By: Rikke Guldborg Hansen, Mads Peter Heide-Jørgensen and Eva Garde

1. Distribution and stock identity

Stock identity is based on consistent summer aggregations, telemetry tracking, genetics, aerial surveys and local knowledge and hunting patterns. Summer distribution is indicated in yellow on Figure 1. In recent years, the distribution has contracted significantly and whales are now found mainly in the core part of their distribution in areas with glaciers experiencing high freshwater melt (Hansen et al. 2015 and Laidre et al. 2016). Telemetry data show that these narwhals begin their fall migration in October where they travel ~800 km to their wintering grounds in Baffin Bay. Individuals from this stock are susceptible to hunters from three different hunting regions; Upernavik, Uummannaq and Disko Bay that can be seen on Figure 2.

Narwhals from the North West Atlantic show low levels of nucleotide and haplotype diversity based on the first 287 base pairs in the mitochondrial control region (Palsbøll et al. 1997). Despite the low degree of variation, frequencies of common haplotypes differed markedly between areas. In East Greenland only one haplotype was found supporting the hypothesis of little or no gene flow between eastern and western Greenland. Heterogeneity was found between Melville Bay narwhals and narwhals from the Avernersuaq district which includes Inglefield Bredning. Hence little gene flow is occurring between the western Greenland summer areas and northern Baffin Bay (eastern Canada and Avanersuaq). Within the northern Baffin Bay samples no significant levels of heterogeneity was found indicating some gene flow between summer grounds within this area. Narwhals show annual fidelity to summer and autumn feeding grounds and pods from these feeding grounds utilizes the same winter grounds.

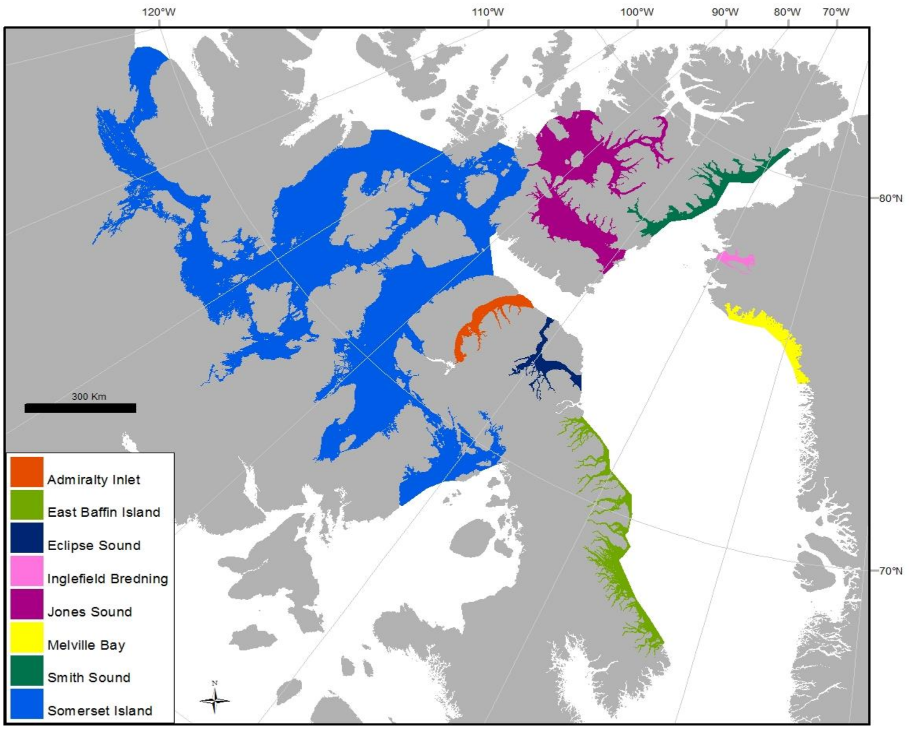

Figure 1. Map indicating the narwhal stocks for the Baffin Bay narwhal population.

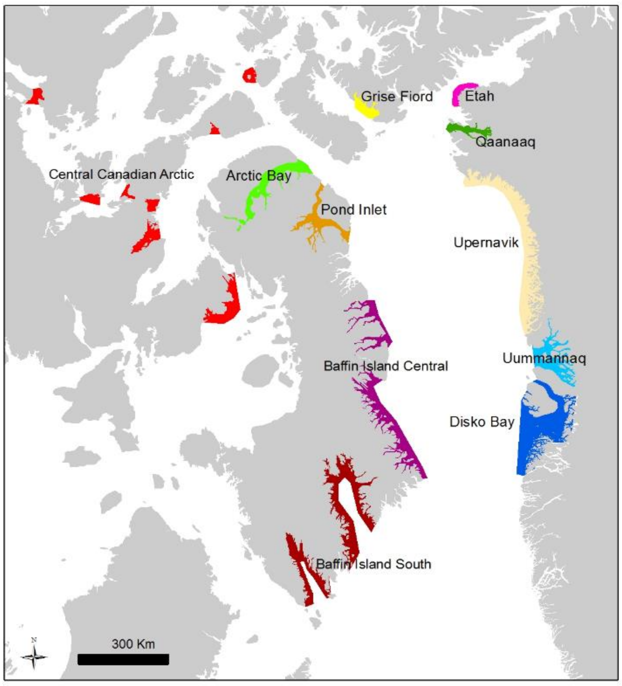

Figure 2. Map indicating the most important hunting regions for the Baffin Bay narwhal population.

Watt et al. (2013) conducted stable isotope analysis on narwhal skin samples collected by Inuit hunters during their subsistent narwhal hunt in Canada and Greenland. Stable isotope analysis on carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) revealed the three narwhal populations of Baffin Bay (BB), Northern Hudson Bay (NHB) and East Greenland (EG) to have distinct stable isotope values that were not expected based on geographic differences. Also, males in all populations had significantly higher $\delta^{13}C$.

2. Abundance

The most recent (2014) abundance estimate for this stock is 3,091 (cv=0.50; 95% CI 1,228-7,783. Hansen et al. 2015). The estimate is corrected for both perception and availability bias.

This estimate comes from a visual aerial line transect survey conducted as a double-observer experiment in a fixed-winged, twin-engine aircraft (DeHavilland Twin Otter) with a target altitude and speed of 213m and 168km h-1, respectively. The front observers (observer 1) acted independently of those in the rear (observer 2) and
vice versa. Declination angles to sightings, species and group size were recorded when the animals came abeam. Beaufort sea state was recorded at the start of the day and then again when it changed. Decisions about duplicate detections (animals seen by both observer 1 and 2) were based on coincidence in timing and location of sightings. The same observers were used for all three surveys except for the 3rd survey where one observer had to be replaced, however, all observers were experienced with both the animals and the data collection schemes from >100 hours participation in other aerial surveys. Instrumentation of the plane and the procedures for data collection were identical to those previously reported by Heide-Jørgensen et al. (2010).

In addition, three aerial surveys were conducted, using the same procedure in 2012 in relation to seismic investigations in Melville Bay. These surveys took place during beginning of August, late August and late September in 2012 and 25-30 August 2014 and covered the area between 74.30°N and 76°N (~14.821 km2, Fig. 1). Strata delineation followed the same design as a previous survey in 2007 based on satellite telemetry as well as local knowledge. Four strata were identified and the two southern strata were surveyed by transects aligned east-west and the two northern were surveyed by north-south transects, systematically placed from the coast to offshore areas crossing bathymetric gradients, covering

~1777km (Figure 3, Heide-Jørgensen et al. 2010). The sightings were concentrated in the central stratum and the two neighbouring strata in all three surveys conducted in 2007, 2012 and 2014.

In the MRDS model a half-normal key functional form and a hazard rate form were tested and the halfnormal was chosen based on its lower AIC with a distance detection range fixed at 0-1200 m. The final DS model in 2012 had distance and group size (as a factor with three levels) as an explanatory variable. The MR model had distance, group size (as a factor with three levels) and 'time to next sighting' as an explanatory factor. The g(0) for observer 1 was 0.76 (cv=0.067) and 0.76 (cv= 0.067) for observer 2 with a combined g(0)= 0.93 (cv= 0.03). The final DS model in 2014 had distance and Beaufort sea state as explanatory variables. The MR model had distance, observer and group size (as a factor with three levels) as explanatory variables (Model 23, Table 5). The g(0) for observer 1 was 0.91 (*cv* =0.047) and 0.77 (*cv*=0.081) for observer 2 with a combined g(0)= 0.98 (*cv*= 0.019). The abundance estimates were stratified by geographic strata. The largest abundance was detected in the central stratum and no sightings were obtained from the northwest stratum. Correction for availability of the at-surface-abundance was based on availability correction factors obtained from five whales from August-September (*a*=0.22; *cv*=0.09). The fully corrected MRDS abundance estimate was 2,983 narwhals (cv=0.39; 95% CI 1,452-6,127) and 3,091 (cv=0.50; 95% CI 1,228-7,783) in 2012 and 2014, respectively.

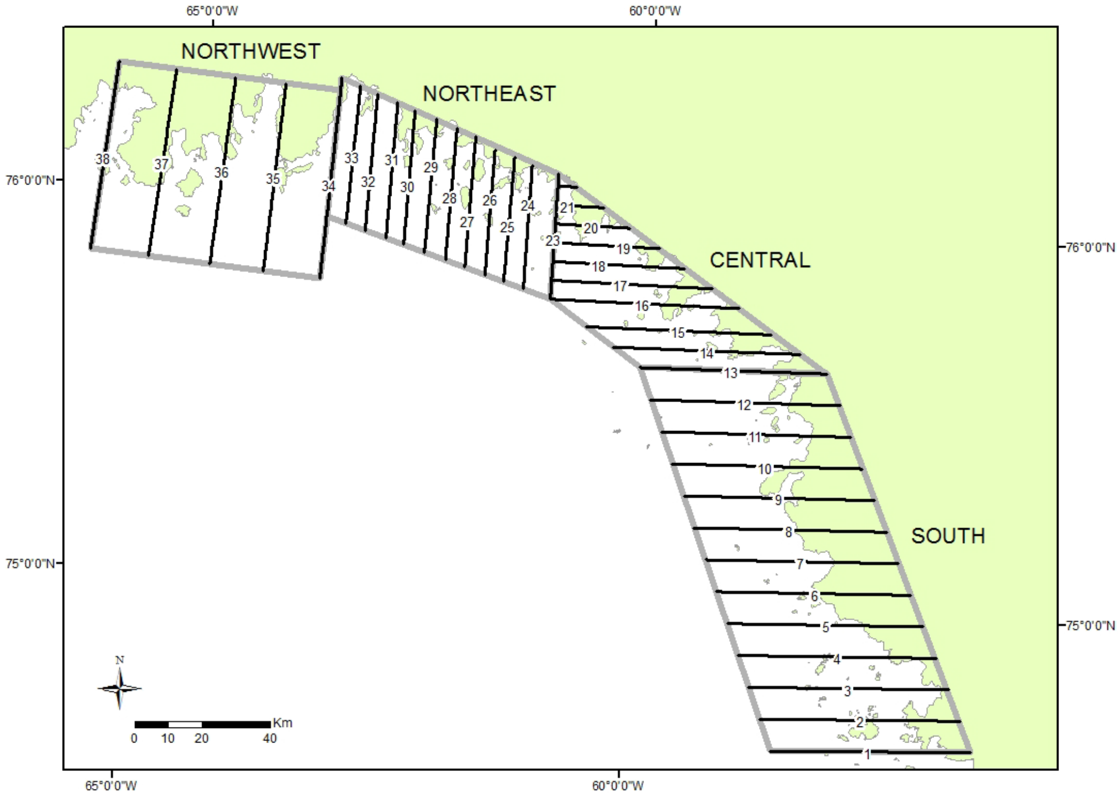

Figure 3. Map of the survey area and transects lines for the Melville Bay stock. All transects were surveyed at least one time.

Data from narwhals instrumented with satellite linked time-depth recorders (Mk10a SLTDRs Wildlife Computers) were used to develop a correction factor for whales that were submerged below the detection depth. Measurements of the time spent above 2m depth were collected in six-hour bins and relayed through the Argos Data Collection and Location System and decoded using Argos Message Decoder (Wildlife Computers). Daily averages were calculated for daylight hours and used for deriving monthly averages that to the extent possible, matched the survey area and dates (Hansen et al. 2015). Detailed dive data with depth recordings every 1 second were obtained from one narwhal equipped with Acousonde in East Greenland in 2013. The Acousonde has a high precision depth recorder and no zero-offset corrections were needed. These time-at-depth data were used for estimating the duration of the dive cycle above and below 2 m depth.

3. Anthropogenic removals

This stock is hunted primarily by the communities in Upernavik hunting region during July-October but are also susceptible for hunt in Uummannaq during November-May and Disko Bay during December-April (Figure 2 and 4, Witting 2016). The quota is set on the basis of the allocation model developed by JCNB SWG.

It is generally assumed that the loss rate was low before 1950 where all catches were corrected by 5% to account for some losses. No studies of losses have been conducted in Greenland but inferences can be made from studies in other areas. In the municipality of Qaanaaq local hunting rules requires the attachment of hand-harpoons on the whales before they can be shot. This severely reduces the loss rate and a loss rate of 5% is arbitrarily applied to the catches in Inglefield Bredning to account for both whales that are killed-but-lost and calves that are separated from mothers. Catches in Melville Bay, however, consists of hunting in both the municipality of Qaanaaq and in Upernavik that doesn’t require the use of hand-harpoons. Roughly half the whales in Upernavik and Melville Bay are taken under the harpoon requirements (5% loss rate) and the other half is taken in ice edge and open water situations.

For narwhal hunting in open water in Canada Weaver and Walker reported loss rates between 32% and 55%, or catch correction factors of 1.5-2.2. Roberge and Dunn reported catch correction factors for narwhals in Canada to range from 1.11 in open water to 1.41 at the ice crack and 1.56 at the floe edge or ice edge (NAMMCO/SC/22-JCNB/SWG/2015-JWG/06).

For Greenland it is assumed that a catch correction factor of 1.30 covers both the open water hunt and the hunt from ice cracks and the ice edge (for the Melville Bay-Upernavik area a factor of 1.15 is used). The correction factor of 1.30 also covers the open-water hunt in late autumn just before freeze-up, which is a type of hunt where loss rates have not been estimated. If anything the correction factor of 1.30 applied here is downward biased.

Official catch statistics for monodontids from West Greenland (Table 2) include catches that are taken from whale pods entrapped in the ice. It has been suggested that mortality in ice entrapments occasionally is part of the natural mortality (Siegstad and Heide-Jørgensen 1994). To allow for analyses of removals without catches in ice entrapments these are shown separately from the mortality genuinely caused by humans.

4. Population trajectory

Three surveys with the goal of assessing abundance have been conducted for the Melville Bay stock. Figure 4 indicates the trajectory given the abundance estimates and associated confidence intervals for the different surveys. Based on the confidence intervals alone, there is no significant change in the abundance estimates over time. The estimated trajectory for the stock comes from a population dynamic model based on a Bayesian modelling framework that is age and sex structured (for details see Witting 2016).

The Melville Bay stock is estimated to be depleted to levels below the MSYL, implying that future harvest levels should be set to ensure an increasing number of narwhals. The stock appears relatively stable.

Table 2. Catches of narwhals from official reports by municipality with corrections for underreportings (in parenthesis) for 1954 to 2011. Numbers in square brackets are from *special reports*. The column ‘under-reporting’ shows the sum of the corrections for under-reporting or ‘ALL’ if it is a general correction factor for all areas. ‘na’ means that no data are available. Data from 2007-08 are preliminary. DB=Disko Bay, UUM=Uummannaq, UPV=Upernavik. Data were compiled from Prime Minister’s Second Department (1951), Kapel (1977), Kapel (1983), Kapel and Larsen (1984), Kapel (1985), Born and Kapel (1986) and Born (1987; NAMMCO/SC/22-JCNB/SWG/2015-JWG/06)

YEAR	QAANAAQ	UPERNAVIK	UUMMAN-NAQ	DISKO BAY	SISIMIUT	MANIITSOQ	NUUK	PAAMIUT-QAQORTOQ	TOTAL	ICE ENTRAPMENT
1949	38	16	1	6					61	
1950										
1951										85 DB
1952										450 DB
1954	na	45			1			1	47	
1955	na	179	2	14					195	
1956	na	15	282	21					318	156 UPV, 250 UUM
1957	na	55	11	15					81	
1958	na	24	3	45		1			73	
1959	na	32	8	16				1	57	
1960	na	25	296	7	1	1	1	1	332	
1961	134	25	5	38				1	203	272 UUM
1962	182	17	11	12				1	213	
1963	275	10	3	29					317	
1964	275	17	11	11					314	
1965	na	33	37	33	1	1			105	
1966	na	39	23	43		3	2		110	
1967	na	131			9				140	31 DB
1968	na	454			18				472	161 DB, 50 UPV, 84 UUM
1969	na	174			30				204	Some DB, 50 UPV

<table>
<tr><th>YEAR</th><th>QAANAAQ</th><th>UPERNAVIK</th><th>UUMMAN-NAQ</th><th>DISKO BAY</th><th>SISIMIUT</th><th>MANIITSOQ</th><th>NUUK</th><th>PAAMIUT-QAQORTOQ</th><th>TOTAL</th><th>ICE ENTRAPMENT</th></tr>
<tr><td>1970</td><td>na</td><td colspan="3">313</td><td colspan="4">9</td><td>322</td><td>100 DB</td></tr>
<tr><td>1971</td><td>na</td><td colspan="3">146</td><td colspan="4">40</td><td>186</td><td></td></tr>
<tr><td>1972</td><td>na</td><td colspan="3">84</td><td colspan="4">23</td><td>107</td><td></td></tr>
<tr><td>1973</td><td>na</td><td colspan="3">191</td><td colspan="4">8</td><td>199</td><td></td></tr>
<tr><td>1974</td><td>8</td><td colspan="3">136</td><td colspan="4">3</td><td>147</td><td></td></tr>
<tr><td>1975</td><td>1</td><td>54</td><td>11</td><td>44</td><td></td><td>6</td><td></td><td>1</td><td>266 (149)</td><td></td></tr>
<tr><td>1976</td><td>9</td><td>22</td><td>27</td><td>57</td><td></td><td></td><td></td><td></td><td>264 (141)</td><td></td></tr>
<tr><td>1977</td><td>16</td><td>62</td><td>113</td><td>53</td><td>8</td><td>1</td><td></td><td></td><td>387 (134)</td><td></td></tr>
<tr><td>1978</td><td>110</td><td>56</td><td>183</td><td>262</td><td></td><td>1</td><td></td><td></td><td>612</td><td></td></tr>
<tr><td>1979</td><td>120</td><td>22</td><td>132</td><td>100</td><td></td><td></td><td>3</td><td></td><td>377</td><td></td></tr>
<tr><td>1980</td><td>130</td><td>61</td><td>146</td><td>120</td><td></td><td>4</td><td>1</td><td></td><td>462</td><td></td></tr>
<tr><td>1981</td><td>118</td><td>83</td><td>140</td><td>249</td><td></td><td></td><td>18</td><td>1</td><td>609</td><td></td></tr>
<tr><td>1982</td><td>164</td><td>59</td><td>162</td><td>76</td><td></td><td></td><td></td><td></td><td>461</td><td>45 DB</td></tr>
<tr><td>1983</td><td>135 (25)</td><td>72 (30)</td><td>164</td><td>68 (10)</td><td></td><td></td><td></td><td></td><td>439 (65)</td><td></td></tr>
<tr><td>1984</td><td>274</td><td>80</td><td>245</td><td>66 (15)</td><td>1</td><td></td><td></td><td></td><td>666 (15)</td><td>35 UUM</td></tr>
<tr><td>1985</td><td>115 (115)</td><td>34 (20)</td><td>39</td><td>67</td><td></td><td>1</td><td></td><td></td><td>256 (135)</td><td></td></tr>
<tr><td>1986</td><td>na</td><td>81</td><td>97</td><td>23</td><td></td><td>36</td><td></td><td></td><td>237</td><td></td></tr>
<tr><td>1987</td><td>na</td><td>145</td><td>334</td><td>25</td><td></td><td></td><td>1</td><td></td><td>505</td><td></td></tr>
<tr><td>1988</td><td>na</td><td></td><td>206</td><td></td><td></td><td></td><td></td><td></td><td>500 (294)</td><td></td></tr>
<tr><td>1989</td><td>na</td><td>37</td><td>288</td><td>2</td><td></td><td></td><td>5</td><td></td><td>332</td><td></td></tr>
<tr><td>1990</td><td>na</td><td>100 (73)</td><td>1019</td><td>11</td><td></td><td></td><td></td><td></td><td>1057 (100)</td><td></td></tr>
<tr><td>1991</td><td>na</td><td></td><td>27</td><td>> 40</td><td></td><td></td><td></td><td></td><td>na</td><td>27 UUM</td></tr>
</table>

YEAR	QAANAAQ	UPERNAVIK	UUMMAN-NAQ	DISKO BAY	SISIMIUT	MANIITSOQ	NUUK	PAAMIUT-QAQORTOQ	TOTAL	ICE ENTRAPMENT
1992	na	37	288	2			5		342	
1993	144	66	301	75	10	6	4	8	614	
1994	183	59	297	268	6	14	7	11	845	150 DB
1995	107	94	159	108	4	5	8		485	
1996	45	69	405	154	10	4	2	2	691	
1997	66	90	381	156	13	5	9	26	746	
1998	94	105	344	163	21	18	6	24	775	
1999	115	119	253	174	28	24	17	15	745	
2000	109	150	106	155	27	8	0	6	561	
2001	145	155	95	119	1	2	15	3	535	
2002	94	164	180	97	12	11	3	2	563	
2003	113	146	174	114	4	0	2	2	554	
2004	178	53	67	73	2	1	0	0	374	
2005	[70] 137	[74] 71	[137] 161	[47] 39	0	0	0	0	[328] 408	
2006	[94] 99	[58] 62	[55] 72	[4] 53	1	2	0		[211] 289	
2007	[21] 139	[17] 102	[52] 67	[56] 63	0	2	0	1	[146] 374	
2008	129	74	87	47	0	0	0	0	337	
2009	90	110	91	88	0	0	0	1	380	41 in Qaanaaq
2010	108	30	42	45	0	0	0	0	225	53 in Qaanaaq
2011	74	60	77	39	0	0	0	1	251	
2012	144	70	42	179	0	0	0	1	311	125 at Kangersuatsiaq
2013	90	64	78	50	0	0	0	1	283	
2014*	114	101	69	62	0	0	0	0	346	

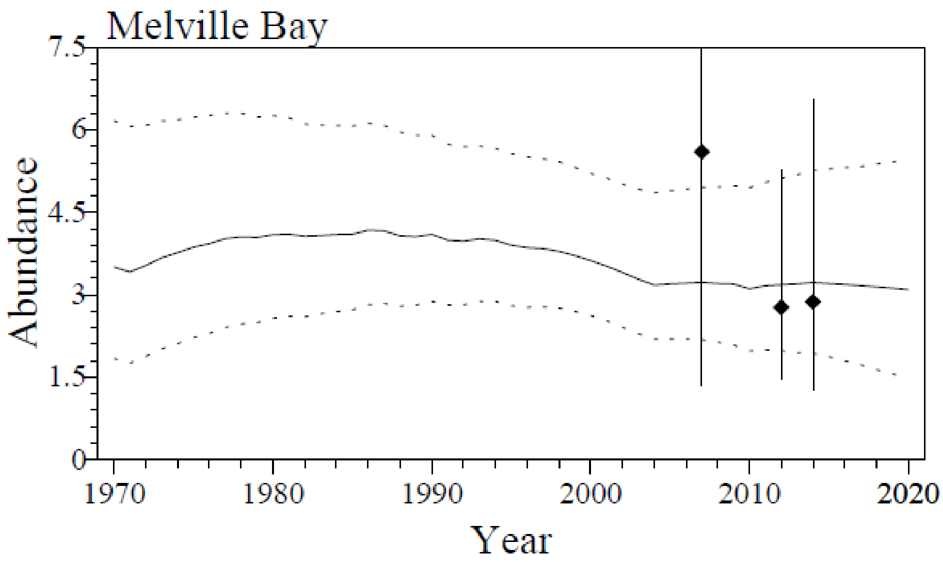

Figure 4. The trajectories of the Melville Bay stock. Points with bars are the abundance estimates (given in thousands) with 90% confidence intervals. Solid curves indicate the median, and dotted curves the 90% CI, of the estimated model (Witting 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

In order to assess the sustainability of catches on this stock, a Bayesian framework was used to estimate the probabilities that an assumed management objective would be fulfilled for potential future catches. While the sustainability of the hunt has to be identified at the population level, recommendations on the sustainability of potential future hunts should preferably be addressed in relation to hunting grounds. To achieve this, for a given set of potential future catches for each hunt, the allocation model developed at JCNB was used to calculate the distributions of future catches for the different populations, with these distributions reflecting the uncertainty in the allocation of catches between the populations (Witting 2016). Then, by having these distributions, for the catches of each percentile of these distributions, the probability that the assumed management objective would be fulfilled for the different populations, could be calculated.

Management defines the total allowable takes for the different hunts (region and season), as these cannot generally be allocated directly to the different summer aggregation. The total allowable takes for the different hunts, with the associated estimates of the probabilities that these takes from 2015 to 2020, will allow the management objective to be fulfilled for the different summer aggregations. These latter probability estimates have 90% confidence limits that reflect the uncertainty of the summer aggregation origin of the animals taken in the different hunts.

The estimated total allowable takes for the different summer aggregations that will meet the management objective with probabilities from 0.5 to 0.95 are presented in Witting et al. 2016. The estimated total allowable take for the Melville Bay stock is 84 individuals per year (2015-2020) with 70% probability for a larger population size in 2020.

6. Habitat and other concerns

Possible concerns include changes in sea ice regime, traffic, seismic exploration and fishing of the halibut resources in central Baffin Bay.

7. Status of the stock.

The Melville Bay is considered to be a small but stable population.

8. Life history

Garde et al. (2015) estimated life history parameters for narwhals from East and West Greenland (n=282) based on age estimates from aspartic acid racemization (AAR) of eye lens nuclei. The species-specific age equation used, $420.32X - 24.02\cdot$year where X is the D/L ratio, was determined from data from Garde et al. (2012) by regressing aspartic acid D/L ratios in eye lens nuclei against growth layer groups in tusks (n=9). Asymptotic body length was estimated to be 399 ± 5.9 cm for females at age 25 years and 456 ± 6.9 cm for males from West Greenland at age 28 years. Age at sexual maturity was assessed based on data from reproductive organs and was estimated to be 8–9 years for females and 12–20 years for males. Length at sexual maturity was ~340 cm for females and 350-400 cm for males. Estimated age at 1st parturition was 9-10 years. Oldest pregnant female was close to 70 years. Pregnancy rates for West Greenland were estimated to be 0.38. Maximum life span expectancy was found to be approximately 100 years. A population projection matrix was parameterized with the data on age structure and fertility rates. The annual rate of increase of narwhals in West Greenland was estimated to be 2.6%.

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Annex 26: Eastern Baffin Island Narwhal Stock
By: Cortney A. Watt

1. Distribution (provide a map if possible) and stock identity

Stock identity is based on consistent summer aggregation reported in TEK. Summer distribution is indicated in green and labeled EBI on the map below. There have been no tagging studies done on whales from the East Baffin Island Stock. There is no genetic support for the delineation of this stock (de March et al. 2003, Petersen et al. 2011), but organochlorine contaminants were significantly different for whales hunted in East Baffin Island compared to other stocks (de March and Stern 2003). Stable isotopes on skin from narwhals hunted in East Baffin Island also show discrimination between East Baffin Island and the other stocks, but this was based on a sample size of only 12 whales (Watt et al. 2012).

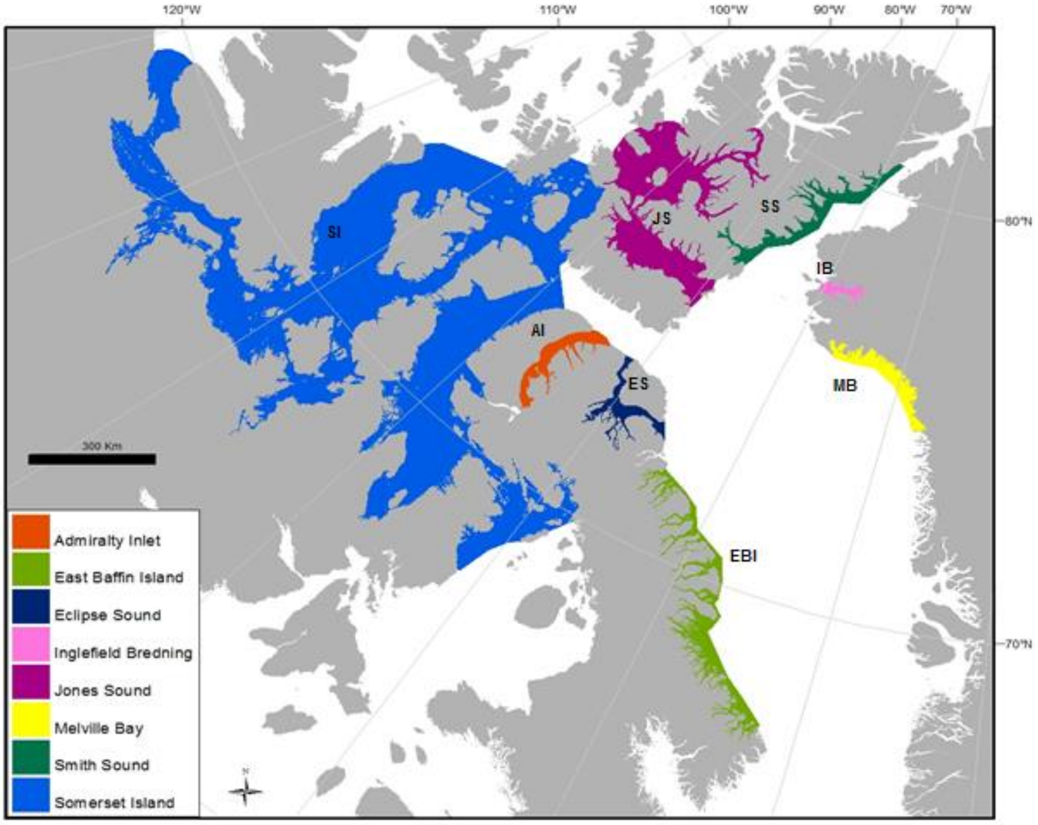

Figure 1. Map indicating the narwhal stocks for the Baffin Bay narwhal population.

2. Abundance

The most recent (2013) abundance estimate for this stock is 17,555 with a CV of 0.35 (Doniol-Valcroze et al. 2015).

This estimate comes from an aerial survey design using a double-platform. Three aircraft were used simultaneously to cover a large area encompassing all of the Canadian narwhal stocks of the Baffin Bay population in August 2013 (Doniol-Valcroze et al. 2015). The extent of the survey areas was based on previous aerial surveys, telemetry tracking studies, TEK, and recent observations by Inuit hunters. Since there has been recent concern about potential movement of narwhals among neighbouring summering regions, the survey with multiple aircraft was designed to survey six of the Baffin Bay stocks during the summer aggregation season - late July through the first three weeks of August prior to the start of fall migration movements. Dates of the survey were chosen to cover areas when sea ice ablation allowed for narwhal to access most of the Arctic Archipelago, and based on the timing of narwhal aggregations in their summering areas as described by TEK and satellite-telemetry data (Doniol-Valcroze et al. 2015). As a result, the last week of July and the first three weeks of August was chosen for the survey, with preference for earlier in August since telemetry data indicated that animals start to move among stocks during the final week of August (Watt et al. 2012).

Transect design was performed in Distance (version 6.1) using coastline shape files. The design was systematic with the first transect line chosen at random. When possible transect lines were oriented in a direction perpendicular to the longest axis of the stratum to maximize the number of lines per stratum (Doniol-Valcroze et al. 2015). For areas where it was assumed narwhal would be in high densities, systematic parallel transects were used. In areas where lower densities were anticipated and landscape patterns permitted, zigzag transects with equally spaced endpoints were used (Doniol-Valcroze et al. 2015).

The survey was flown at an altitude of 1,000 ft, and a target speed of 100 knots using three deHavilland Twin Otter 300 aircraft, each with 4 bubble windows on the sides and run as a double-platform experiment with independent observations platforms at the front and rear of the plane (Doniol-Valcroze et al. 2015). Dual camera systems were mounted under the belly of the plane to allow for continuous digital photography.

Distance sampling methods were used to estimate detection probability away from the track line, while mark-recapture methods were used on sighting data from two observers on either side of the aircraft to correct for perception bias. The distribution of perpendicular distances was different in fiord strata than in the other strata, and thus only non-fiord observations were used to fit the detection function for the non-fiord strata. Examination of the histogram of the perpendicular distances of unique sightings suggested right-truncating the data at 1000 m (i.e., discarding sightings beyond 1000 m), which left 762 unique observations (515 seen by primary observers, 523 by secondary observers, and 276 by both). The shape of the histogram suggested that some narwhals were missed close to the track line despite the bubble windows. Therefore, there was a risk that hazard-rate and half-normal distributions would overestimate the probability of detection and the resulting effective strip width. However, almost a hundred narwhal sightings were made within 100 m of the track line and therefore it seemed inappropriate to lose a large amount of data by left-truncating (i.e., discarding sightings close to the trackline). The shape of the histogram suggested that a gamma distribution would fit better, except that a gamma distribution takes the value zero at zero distance. Therefore, a gamma distribution with an offset term, in addition to half-normal and hazard rate keys, was fitted to the data (Doniol-Valcroze et al. 2015). Model selection was performed on all combinations of covariates (including environmental covariates such as ice cover, cloud cover, sea state, and glare, and a sighting rate covariate which was computed as a rolling average of the number of sightings made by the observer in a 30-second window prior to each sighing). The model with the lowest AIC was one with a truncated gamma key function and the covariates “sighting rate”, “Beaufort” and “glare”. The covariates reduced the detection distance at higher levels (Beaufort >3, Glare=intense, Sighting rate >10 in the last 30 seconds) and resulted in an average probability of detection of 0.48 (CV 2.8%) and an estimated effective strip half width of 481 m (not including perception bias) (Doniol-Valcroze et al. 2015).

For the mark-recapture model to estimate perception bias, models were performed with all combinations of environmental covariates as well as covariates “perpendicular distance”, “observer”, “sighting rate”, “side of aircraft” and “group size”. The best model included “perpendicular distance” and “sighting rate” and the overall probability of detecting a narwhal cluster between the track line and a distance of 1000 m was 0.40 (CV 4.2%) (Doniol-Valcroze et al. 2015).

Fiords were considered their own sampling units and cluster sampling was used to select the fiords to be surveyed (Doniol-Valcroze et al. 2015). In fiords, flights were continuous tracks designed to follow the main axis of the fiord while spreading coverage uniformly based on distance to shore. The resulting data from the fiords was analyzed separately from non-fiord strata. A density surface modelling framework was used to model spatially-referenced count data with the additional information provided by collecting distances to account for imperfect detection (Doniol-Valcroze et al. 2015). First a detection function was fitted to the perpendicular distance data to obtain detection probabilities for clusters of individuals (Doniol-Valcroze et al. 2015). Counts were then summarized for contiguous transect sections and a generalized additive model was constructed with segment counts as the response with areas corrected for detectability (Doniol-Valcroze et al. 2015).

Total surface abundance estimates for stocks were obtained by the additions of the estimated abundances of all the strata that made up that stock's summer range, including results from fiord strata. Variance for the stock-wide abundance estimate was calculated by adding the variances of each stratum; however, identification of duplicates was not straightforward due to the highly aggregated nature of narwhal groups. Because of this, a sensitivity analysis was used to quantify the uncertainty, which allowed the researchers to include an additional variance component to the surface abundance estimate with a CV equal to that of the sensitivity analysis, which ultimately increased the range of uncertainty around the estimate but left the point estimate unchanged (Doniol-Valcroze et al. 2015).

An availability bias correction was also applied to the survey data. For the availability bias correction, the time at depth for 24 narwhals fitted with satellite tags near Arctic Bay and Pond Inlet every August from 2009-2012 (Watt et al. 2015) was used to determine the correction for the number of whales missed as a result of being at depth and unavailable for viewing by the surveyors. The time narwhals spent at 0-2 m depths was used to calculate a correction for areas with clear water, while areas with very murky water, the time spent within 0-1 m of the surface was used. Sightings in all the fiords of the East Baffin Island stratum were reported by observers as having occurred in murky or opaque waters, which was confirmed by examination of the photographs taken underneath the plane. This suggests that observers would not have been able to detect and identify narwhals as deep as 2 m, as is usually assumed. Therefore, for this stratum, a correction factor was calculated based on the assumption that narwhals could only be seen between 0 and 1 m. This resulted in a correction factor of 3.18 ± 3.37 for clear water areas and a correction of 4.90 ± 0.187 for murky regions (the fiords) (Watt et al. 2015). This correction is appropriate when sightings are instantaneous, but if they are not (such as in aerial surveys), it can positively bias the estimate and as a result a correction factor incorporating the dive cycle of the animal is needed. Three archival time-depth recorders deployed on whales near Pond Inlet and in Creswell Bay in August 1999 and 2000 were used to evaluate a dive-cycle for narwhals. A weighted availability bias correction factor that took into account both the time at depth and the time in view (dive-cycle) was used (2.94 ± 3.4 for the 0-2 m correction and 4.53 ± 3.8 for the 0-1 m bin).

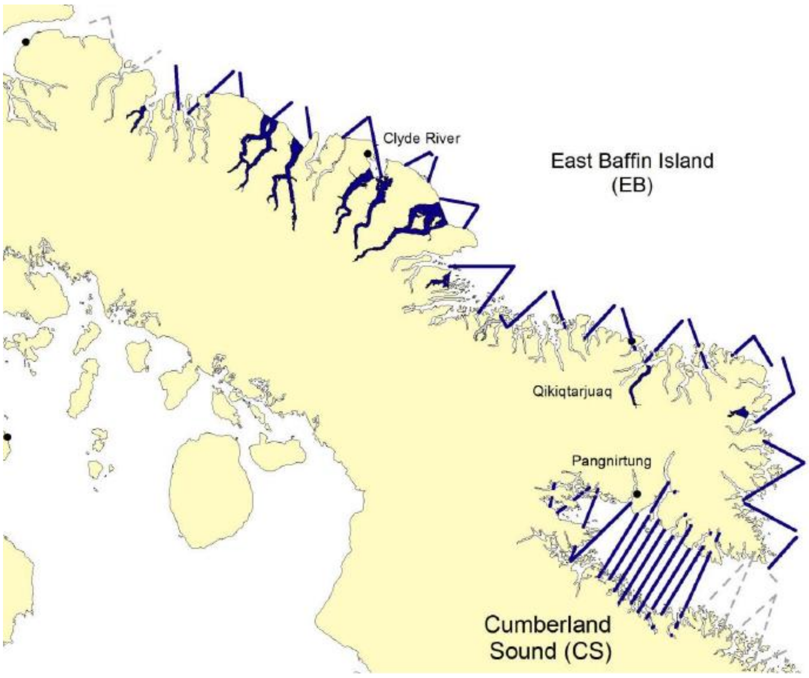

Figure 2. Map of the surveyed strata for the East Baffin Island stock. Blue lines indicate surveyed transects and blue areas indicate surveyed fiords, while grey dashed lines and grey areas indicate planned transects and fiords that were unable to be completed as a result of weather (Doniol-Valcroze et al. 2015).

For East Baffin Island strata (Figure 2) were surveyed using a single airplane over a two-week period (August 11, 2013 – August 25, 2013). Strong winds delayed the offshore strata from being surveyed, but 90% of all planned transects were surveyed and all but one fiord (Doniol-Valcroze et al. 2015). Most narwhal were seen in the fiords of the north-western half of the stratum and a single narwhal was sighted in Cumberland Sound but was not included in the stock estimate due to uncertainty about its stock of origin. The surface abundance estimate for the East Baffin Island stock outside of fiords was 122 ± 0.63, and after viewing the photos it was deemed that the water was clear and a correction for the 0-2 m bin should be applied. After a weighted correction of 2.94 ± 0.03 was applied the resulting abundance estimate was 357 ± 0.63 (Doniol-Valcroze et al. 2015). In the fiords the surface abundance estimate was 3,799 ± 0.35 and after applying a weighted correction of 4.53 ± 0.04 based on the 0-1 m bin (since the water was murky) the corrected abundance was 17,198 ± 0.35, for a total abundance of 17,555 ± 0.35 (Doniol-Valcroze et al. 2015).

3. Anthropogenic removals

This stock is hunted primarily by the communities of Clyde River and Qikitarjuak in the summer (Heide-Jørgensen et al. 2013); however, there is also opportunity for hunters from other communities to hunt these whales on their migration to and from the summering grounds and on the wintering grounds (Witting 2016). Catches in Table 1, however, reflect whales that are hunted within the defined summering region, since it is difficult to determine the number of animals from this stock hunted by other communities. In some Canadian communities with a community-based management system, killed-lost and wounded-lost narwhal numbers were documented by hunters between 1999 and 2005 (Table 2). From the narwhal hunts where losses are reported, Richard (2008) calculated a hunting loss rate correction (LRC) (Table 2).

LRC = HM / LC where
HM = the estimated total hunting mortality, or the sum of the landed catch and hunting loss
LC = Landed Catch
The estimated hunting loss was calculated as:
$HM = (HM_{min} + HM_{max})/2$ where
HM_{min} = number of animals landed plus the ones reported sunk and lost
$HM_{max} = HM_{min}$ + the number reported wounded and escaped

This HM estimate used by Richard (2008) assumes that half of the animals wounded and escaped later die from their injuries. This assumption was untested but considered reasonable since both whales with wound scars are later seen alive but dead whales have also washed up after a hunt suggesting some whales survive from their wounds while others perish (Richard 2008). Table 1 indicates the total reported landed catches, and the catches multiplied by a struck and loss factor of 1.28 ± 0.15 (Richard 2008). This data comes from 1999-2005 and is hunter reported for all types of hunt combined for each of the communities. An older study (Roberge and Dunn 1990) investigated struck and lost rates from the community of Arctic Bay in the open water season in 1983 and 1988, on the floe edge in 1988 and 1989, and at the ice crack in 1978, 1988, and 1999 (Table 3). Most of the hunt in East Baffin Island occurs in the open water season, which has a struck and loss factor reported by Roberge and Dunn (1990) of 1.40 ± 0.14. In this study researchers monitored the hunt when possible and reported values. Application of this rate rather than the 1.28 reported by Richard (2008), changes catches previous to 1999 by an average of 7 whales, and a maximum of 11 whales (results in brackets in Table 1). Ideally a struck and loss factor would be applied to each catch that occurs through different hunting methods; unfortunately this information is not reported. However based on hunt dates (for which we have some information from 2003-2012 for Clyde River and Qikitarjuak), the majority of the hunt occurs in the open water season (59% for Clyde River and 72% for Qikitarjuak (Doniol-Valcroze 2014)). Currently in Canada the struck and loss rate from Richard (2008) is used, since it is the most up to date.

Table 1. Reported landed catches for the communities of Clyde River and Qikiqtarjuaq from the East Baffin Island stock. From 1977 these catches are based on the number of issued tags and recorded by Fisheries and Oceans Canada; prior to 1977 the numbers come from a variety of sources (see reference list) but typically rely on reports by hunters, or RCMP records. Total catch including struck and lost animals is indicated using the newest struck and lost factor (1.28 from Richard (2008)), and using the 1.40 reported for open water hunts by Roberge and Dunn (1990) for years prior to 1999 indicated in brackets.

Year	Clyde River (landed catches)	Qikiqtarjuaq (landed catches)	East Baffin Island (landed catches)	Reference for reported landed catch	East Baffin Island + 1.28 S&L factor (1.40 S&L factor)
1970	9	nr	9	Mansfield et al. (1975)	12 (13)
1971	20	nr	20	Mansfield et al. (1975)	26 (28)
1972	nr	8	8	Strong (1989), Mitchell and Reeves (1981)	10 (11)
1973	8	4	12	Strong (1989), Mansfield et al. (1975)	15 (17)
1974	nr	nr	nr	Strong (1989), Stewart (2007)	nr
1975	15	5	20	Strong (1989)	26 (28)
1976	15	6	21	Strong (1989)	27 (29)
1977	42	35	77	Strong (1989)	99 (108)
1978	4	26	30	Strong (1989)	38 (42)
1979	9	21	30	Strong (1989)	38 (42)
1980	35	49	84	Strong (1989)	108 (118)
1981	37	50	87	Strong (1989)	111 (122)
1982	19	50	69	Strong (1989)	88 (97)
1983	46	20	66	Strong (1989)	84 (92)
1984	49	36	85	Strong (1989)	109 (119)
1985	5	49	54	Strong (1989)	69 (76)
1986	5	7	12	Strong (1989)	15 (17)
1987	19	47	66	Strong (1989)	84 (92)
1988	44	26	70	DFO (1991)	90 (98)
1989	36	46	82	DFO (1992)	105 (115)
1990	26	50	76	DFO (1992)	97 (106)
1991	35	50	85	DFO (1993)	109 (119)
1992	33	40	73	DFO (1994)	93 (102)
1993	34	52	86	DFO (1995)	110 (120)
1994	25	50	75	DFO (1996)	96 (105)
1995	26	50	76	DFO (1997)	97 (106)
1996	10	23	33	DFO (1999)	42 (46)
1997	15	50	65	Stewart (2007)	83 (91)
1998	17	50	67	Doniol-Valcroze (2014)	86 (94)
1999	0	81	81	Doniol-Valcroze (2014)	104
2000	52	131	183	Doniol-Valcroze (2014)	234
2001	41	87	128	Doniol-Valcroze (2014), Stewart (2007)	164
2002	44	82	126	Doniol-Valcroze (2014)	161
2003	50	90	140	Doniol-Valcroze (2014), Stewart (2007)	179
2004	50	96	146	Hall et al. (2015)	187

2005	39	88	127	Hall et al. (2015)	163
2006	43	88	131	Hall et al. (2015)	168
2007	42	89	131	Hall et al. (2015)	168
2008	17	80	97	Hall et al. (2015)	124
2009	13	90	103	Hall et al. (2015)	132
2010	50	89	139	Hall et al. (2015)	178
2011	37	90	127	Hall et al. (2015)	163
2012	21	90	111	Hall et al. (2015)	142
2013	49	83	132	Hall et al. (2015)	169
2014	45	83	128	Hall et al. (2015)	164
2015	62	70	132	Hall et al. (2015)	169

Table 2. Table indicating how the struck and loss factor for this stock is calculated. Table is directly from Richard (2008).

Community	Year	Landed	Wounded/ Escaped	Sunk and Lost	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Community specific average Loss Rate Correction
Pond Inlet	1999	130	14	16	146	160	153	1.18	
	2000	166	21	10	176	197	187	1.12	
	2001	63	5	27	90	95	93	1.47	
	2002	92	1	13	105	106	106	1.15	1.23 ± 0.16
Qikiqtarjuaq	1999	81	30	25	106	136	121	1.49	
	2000	137	79	40	177	256	217	1.58	
	2001	89	8	9	98	106	102	1.15	
	2002	81	40	16	97	137	117	1.44	
	2004	96	12	9	105	117	111	1.16	1.36 ± 0.20
Repulse	1999	156	68	30	186	254	220	1.41	
	2000	49	9	5	54	63	59	1.19	
	2001	100	38	21	121	159	140	1.4	
	2002	57	0	8	65	65	65	1.14	
	2003	30	0	5	35	35	35	1.17	
	2005	72	25	3	75	100	88	1.22	1.26 ± 0.12
Arctic Bay	2001	134	20	4	138	158	148	1.1	
	2003	129	14	22	151	165	158	1.22	
	2004	122	22	33	155	177	166	1.36	1.23 ± 0.13
Kugaaruk	2001	41	18	8	49	67	58	1.41	
	2003	24	4	2	26	30	28	1.17	1.29 ± 0.17
Average across communities									**1.28 ± 0.15**

Table 3. Table indicating how an older struck and loss factor for Arctic Bay was calculated from observations of different hunting types from Roberge and Dunn (1990).

Hunt	Year	Landed	Wounded/ Escaped	Sunk and Lost/mortally wounded	Min mortality	Max mortality	Estimated total kill (average of min and max)	Loss Rate Correction (total/landed)	Hunt specific average Loss Rate Correction
Floe edge	1988	6	6	8	14	20	17	2.83	
	1989	16	0	5	21	21	21	1.31	2.07 ± 1.08
Open water	1983	4	2	1	5	7	6	1.50	
	1988	13	6	1	14	30	17	1.31	**1.40 ± 0.14**
Ice crack	1987	15	13	8	23	36	30	1.97	
	1988	29	8	17	46	54	50	1.72	
	1989	50	7	13	63	70	67	1.33	1.67 ± 0.32
Average across hunt types									**1.71± 0.55**

4. Population trajectory

Two surveys have been conducted for the Eastern Baffin Island stock. Figure 3 indicates the trajectory given the abundance estimates and associated confidence intervals for the surveys. The estimated trajectory for the stock comes from a population dynamic model based on a Bayesian modelling framework that is age and sex structured (for details see Witting 2016). Unfortunately there are not enough survey estimates to determine a trend for this stock (Witting 2016).

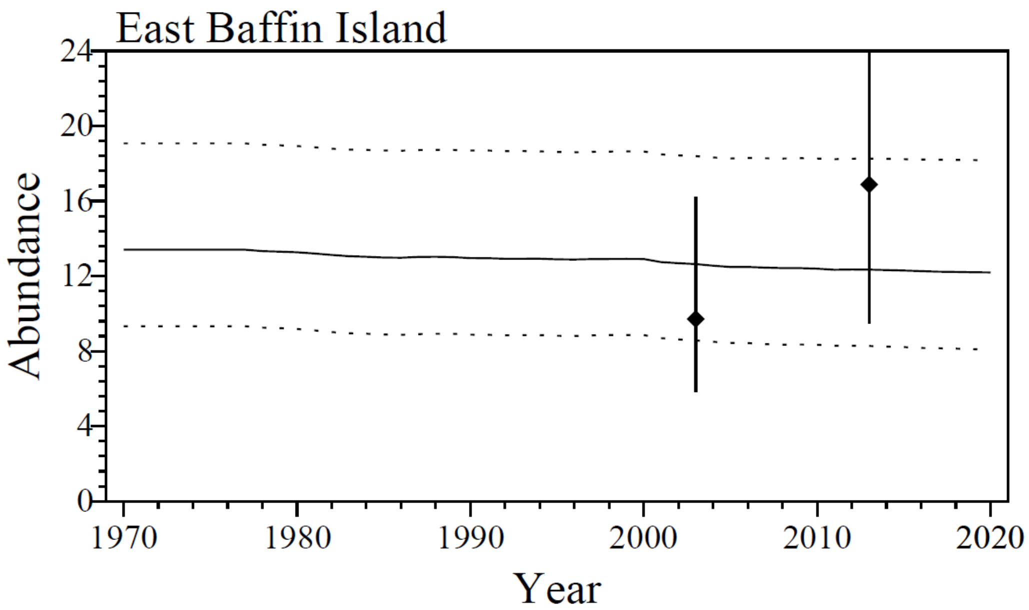

Figure 3. Population trajectory for the East Baffin Island stock. The points represent the abundance estimates (given in thousands) with 90% confidence intervals. Solid curves indicate the median, and dotted curves the 90 % confidence interval for the estimated model (Witting 2016).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The Potential Biological Removal (PBR) method (Wade 1998), corrected for hunting losses (struck and lost), is used to calculate a recommended Total Allowable Landed Catch (TALC):

TALC = PBR/ LRC
Where:
PBR = 0.5 * R_{max} * N_{min} * F_r

LRC is the hunting loss rate correction and is equal to 1.28 ± 0.15 (Richard 2008). R_{max} is the maximum rate of increase for the stock (unknown so the default for cetaceans of 0.04 is used, N_{min} is the 20th percentile of the log-normal distribution of N (most recent survey estimate), and F_r is the recovery factor (we used a value of 1 which indicates a healthy status for the stock (an assumption)). The current Total Allowable Landed Catch (TALC) is set at 122 for this stock. The new TALC recommendation based on the 2013 aerial survey result is 206.

6. Habitat and other concerns

Little is known since there have been no telemetry studies to show movements/migration or dive behaviour.

7. Status of the stock

The stock is quite large, with no conservation concerns at this time, however there is relatively little information about the stock. Current removals are considered to be sustainable (Doniol-Valcroze et al. 2015).

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Annex 27: Northern Hudson Bay Narwhal Stock
Steven Ferguson

1. Stock Definition and Distribution

Narwhals inhabit Arctic waters connected to the North Atlantic north of 60°N (Figure 1) and include the marine waters of Nunavut, west Greenland and the European Arctic but are rare in the East Siberian, Bering, Chukchi and Beaufort seas. This distribution and relative density appears to be relatively unchanged.

Genetic studies of narwhal samples collected from hunts in Canada and Greenland have not found strong differences indicative of population substructure (Palsbøll et al. 1997; de March et al. 2001, 2003; Petersen et al. 2011). However, narwhals from the Northern Hudson Bay population are significantly different from the other two populations (de March et al. 2003; Petersen et al. 2011). Contaminant and biomarker (stable isotope) composition has also been used to help with delineation of narwhal populations (de March et al. 2003; Watt et al. 2012).

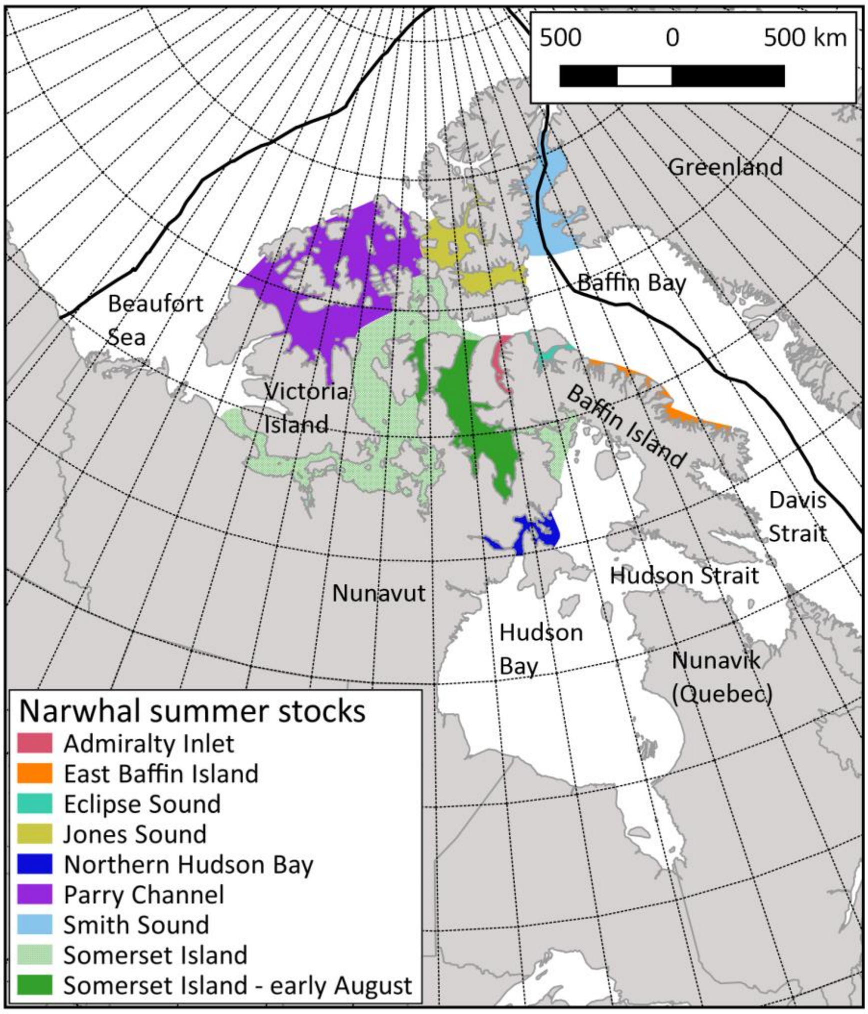

Figure 1. Narwhal whale stocks in the Canadian Arctic. Blue shows core summer aggregation range of Northern Hudson Bay population.

Narwhals that summer in northwest Hudson Bay and winter in eastern Hudson Strait (Westdal et al. 2010; Elliott et al. 2013) are referred to as the northern Hudson Bay population. The summer range of the northern Hudson Bay narwhal population includes the waters surrounding Southampton Island, with the largest aggregations in Repulse Bay, Frozen Strait, and Lyon and Gore Inlets (Richard 1991; Gonzalez 2001; Westdal et al. 2010). Hudson Strait is an important migration route and wintering area for this population in fall, winter, and spring (Westdal et al. 2010; Elliott et al. 2013). The satellite-tagged narwhal made direct movements through Hudson Strait to their wintering area near Resolution Island (Westdal et al. 2010). The population is believed to winter mainly in eastern Hudson Strait (McLaren and Davis 1982; Richard 1991; Koski and Davis 1994; Elliott et al. 2013).

2. Abundance

DFO conducted systematic visual and photographic aerial surveys of narwhals on their main summer range in the Repulse Bay area including Frozen Strait, Gore Bay, Lyon Inlet, the northern part of Roes Welcome Sound, and Duke of York Bay in March 1983 and July of 1982, 1983, and 1984 (Richard 1991). No corrections for availability bias and surveys were limited to open water or areas with 2/10ths ice or less. Therefore, the abundance estimate was negatively biased. Four systematic surveys in total and different stratification in different years resulted in population estimates ranging between 1,038 and 1,517 with varying degree of precision. A strip (600 m/side) and photographic survey was completed in August 2000 that was similar to the earlier surveys with the addition of northern Lyon Inlet and Foxe Channel. Without correcting for availability bias, the Northern Hudson Bay narwhal population was estimated at 1355 (90%CI = 1000-1900 in 1984; Richard 1991) and 1780 (90%CI = 1212-2492 in 2000; Bourassa 2003).

In August 2011 visual (double platform distance sampling) and photographic (directly beneath aircraft) survey was flown of Repulse Bay, Frozen Strait, Wager Bay, Roes Welcome Sound, Lyon Inlet, Gore Bay and parts of Foxe Channel and Foxe Basin (Figures 2 and 3). A fully corrected for availability and perception bias estimate of 12,485 narwhal (CV = 0.26; Asselin et al. 2012).

3. Anthropogenic removals

The northern Hudson Bay narwhals are hunted mostly by Repulse Bay (Naujaat) and to a lesser degree by residents of six other communities: Arviat, Whale Cove, Rankin Inlet, Coral Harbour, Kimmirut, Cape Dorset (Table 1). Most narwhals are harvested in July and August (Hoover et al. 2013).

The number of narwhals actually killed during these hunts is higher than the number reported landed due to animals wounded that subsequently die and are lost to the harvest. However, the number lost is largely unknown because few data have been collected on the hunt. Losses from hunting activity vary seasonally depending upon area, weather, hunter experience, and type of hunt (e.g. floe edge, ice lead, open water) (Weaver and Walker 1988; Roberge and Dunn 1990). Some studies consider only whales that are killed and lost while other studies consider all whales that are wounded and escape resulting in an estimate that underestimates and overestimates the total kill, respectively.

Loss estimates from the community-based management hunts suggest that on average at least 19 (SD 11; killed and lost only) and perhaps as many as 46 (SD 5; killed and lost plus struck and escaped) animals are lost for every 100 landed (COSEWIC 2004) which are comparable to those from earlier studies (e.g. Weaver and Walker 1988; Roberge and Dunn 1990). Due to the economic value of the tusk, there is a preference by hunters to hunt males which may result in an underestimate of females killed since hunters are more likely to retrieve a dead male (Weaver and Walker 1988; Roberge and Dunn 1990).

Landings from the northern Hudson Bay population increased from an average of 21 (SD 8.6) whales per year over the period 1979-1998, to an average of 102 (SD 55) whales per year over the period 1999-2001, and then declined to 83 (SD 30) over the period 2002-2015 (Table 1). A Loss Rate Correction (LRC) of 1.28 has been used for this stock (Asselin et al. 2012). Using the values from COSEWIC (2004) this assumes that approximately one third of the struck and escaped whales will eventually die as a result of their wounds. Using the estimated LRC =1.28 we have an average total removals of 106 per year for the period 2002-2015.

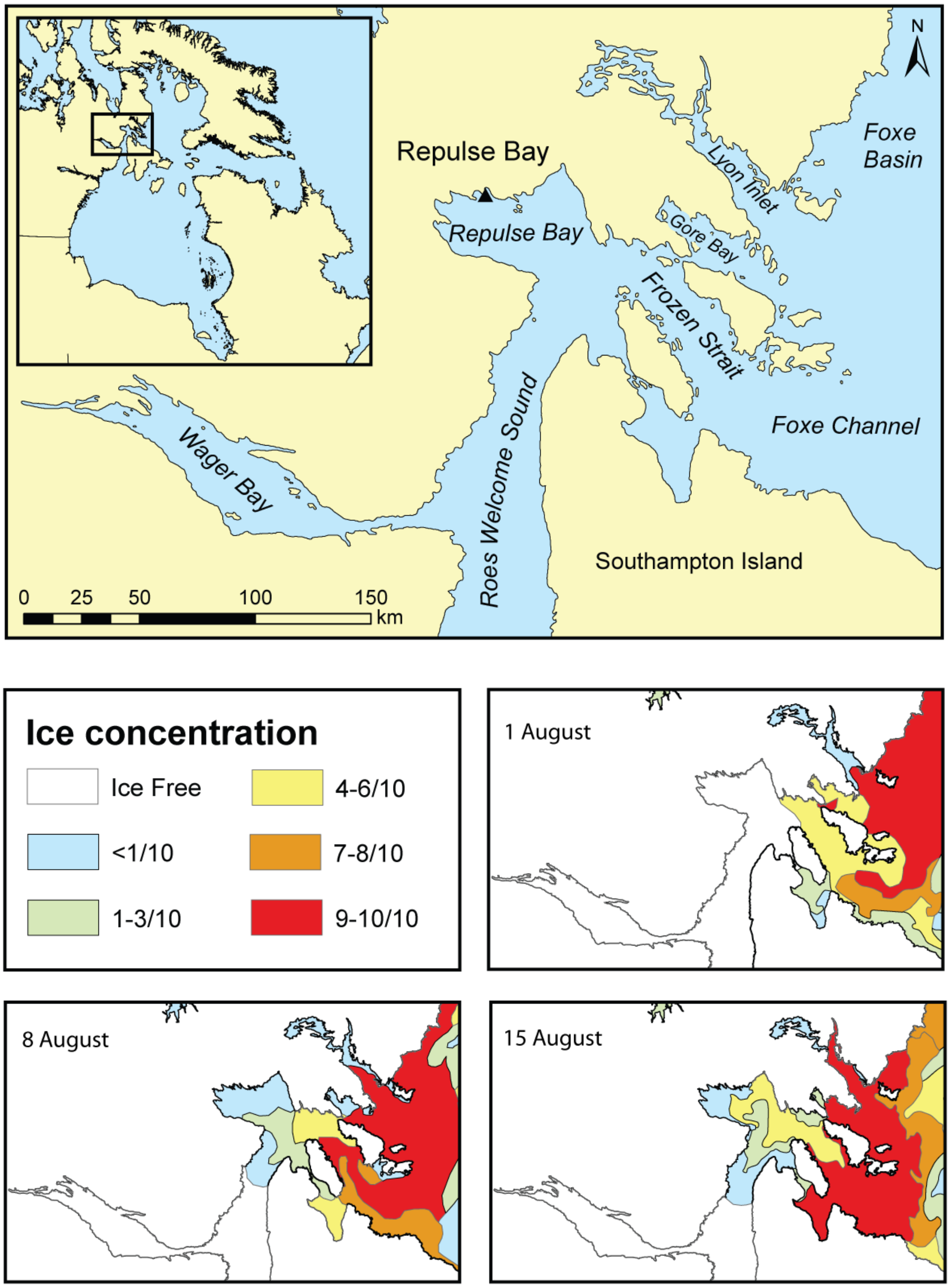

Figure 2. Survey area (above) and ice concentrations from 1 August, 8 August and 15 August 2011 (below). Ice concentrations are from the Canadian Ice Service weekly regional ice charts (available at http://ice-glaces.ec.gc.ca).

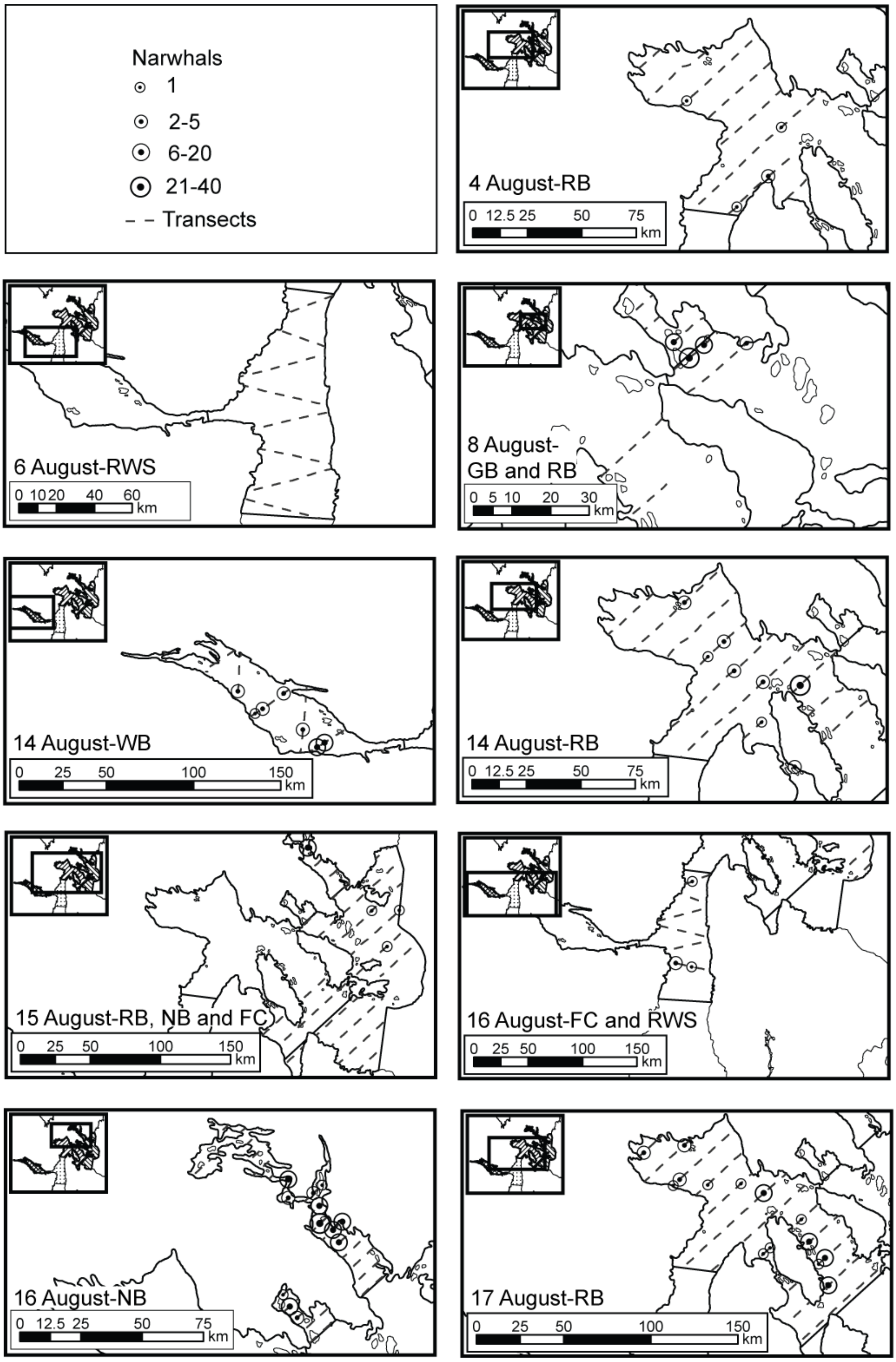

Figure 3. Transects flown and narwhal sightings for the northern Hudson Bay narwhal survey in 2011. Labels indicate date transects were flown and strata: Foxe Channel (FC), Gore Bay (GB), Wager Bay (WB), Northern Bays (NB), Repulse Bay (RB) and Roes Welcome Sound (RWS). Note: narwhal sightings were grouped at 5 km intervals for map clarity.

Table 1. Landed harvests from the northern Hudson Bay narwhal population by community (0 = no harvest, blank cell = no report) from DFO (data on file and Kingsley et al. 2013).

Year[1]	Cape Dorset	Chesterfield Inlet	Coral Harbour	Kimmirut	Rankin Inlet	Repulse Bay[2]	Whale Cove	Total[3]
1977	0		0	0	0		0	**0**
1978	2		0	0	0	4	0	**6**
1979	1		0	0	0	30	0	**31**
1980	1		0	0	0	25	0	**26**
1981	0		0	0	5	29	0	**34**
1982	0		0	0	0	21	1	**22**
1983	0		0		0	11	0	**11**
1984			0	0	2	25	0	**27**
1985	0		0	0	1	15	0	**16**
1986	0		0	0	0	7	0	**7**
1987	0		12	7	0	16	0	**35**
1988	1	0	0	0	0	25	0	**26**
1989	0	0	0	0	0	16	0	**16**
1990	0	0	0	0	0	17	0	**17**
1991	16	0	0	0	0	3	0	**19**
1992	0	0	0	0	0	20	0	**20**
1993	0	0	1	0	0	13	0	**14**
1994	1	0	0	0	0	5	0	**6**
1995	0	0	10	0	6	4	0	**20**
1996	0	0		0	0	16	0	**16**
1997	0	0		0	0	35		**35**
1998	0	4		0		18	0	**22**
1999	0		0	0		154	0	**154**
2000	0	3	0	0		42	0	**45**
2001	1	2	0	0	5	99		**107**
2002	0	4	4	1	2	56	0	**67**
2003	0	1	1	0	3	38		**43**
2004	0	4	3	0	7	106	0	**120**
2005	0	4	6	0	3	72	1	**86**
2006	0	4	3	0	10	75	2	**94**
2007	0	3	1	1	9	74	0	**88**
2008	0	2	1	0	1	25	0	**29**
2009	0	4	8	0	8	97	2	**119**
2010	2	2	6	1	9	82	1	**106**[4]
2011	0	5	7	0	8	70	1	**91**[5]
2012	3	2	0	0	1	48	2	**56**
2013	0	5	6	0	13	83	7[6]	**114**
2014	0	0	0	0	6	89	1[6]	**96**
2015	0	1	1	0	0	43	1	**46**

[1] Starting year of the harvest reporting period. Prior to 1996, the harvest was reported by calendar year. Starting in 1996, the harvest has been reported by fiscal year (April 1 – March 31).

[2] As a participant in the Community Based Management program, Repulse Bay had flexible quota privileges that permitted carry-over of a portion of unused Marine Mammal Tags (MMTs) to the following year.
[3] In some years the community of Hall Beach may have taken narwhals in Lyon Inlet from the Northern Hudson Bay population. 2010 is the only year for which MMT returns indicate that two narwhals were harvested in Lyon Inlet. Landed catches by Hall Beach were not included in the model.
[4] The total includes three narwhals harvested by Arviat. Three MMTs were allocated to Arviat by the other Kivalliq communities in a 2010 Kivalliq Wildlife Board decision (one year only).
[5] The total includes one narwhal harvested by Arviat. As Arviat does not have a regulatory quota, the MMT was borrowed from Whale Cove.
[6] Harvest includes Whale Cove and Baker Lake.

4. Population trajectory

Kingsley et al. (2013) modelled the aerial surveys from the early 1980s, 2000, 2008, and 2011 of northern Hudson Bay narwhal. The stock dynamic model using Bayesian methods and run on the OpenBUGS platform using adjustments for different survey methods (Asselin and Ferguson 2013) estimated a 1.2% growth rate per year and a population that could support a landed catch of no higher than 75 per year. Conclusions were that there is considerable uncertainty in population trend and another survey is needed to corporate the high abundance observed in the 2011 survey. The recommendation was to continue to use the Potential Biological Removal method rather than adopt a risk-based approach until more data is gathered on this population (SAR).

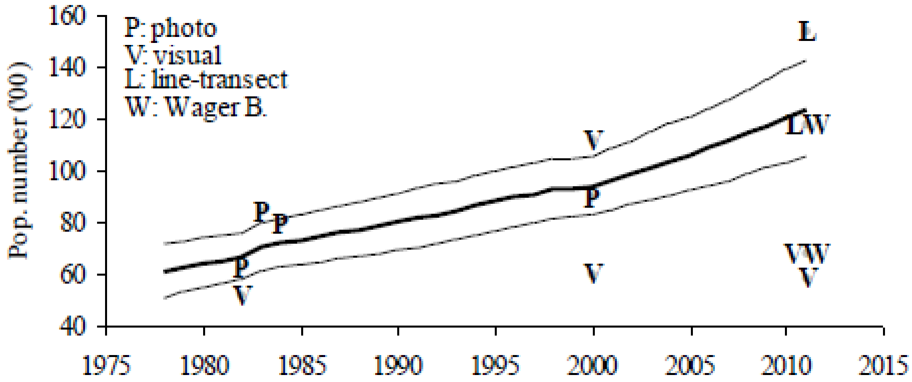

Figure 4. Modelled trend of population abundance (with quartiles) from Kingsley et al. (2013) analysing data from visual and photographic surveys.

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The most recent aerial survey of the population occurred 14-17 August 2011 and covered Repulse Bay, Frozen Strait, Wager Bay, Roes Welcome Sound, Lyon Inlet, Gore Bay and parts of Foxe Channel. The total abundance estimate was 12,485 (CV=0.26) narwhals (Assselin et al. 2012). The Potential Biological Removal (PBR; Wade 1998) is calculated as

PBR = Nmin * 0.5 * Rmax * FR

where Nmin is the estimated population size using the 20-percentile of the lognormal distribution (N/[exp(z20*sqrt[ln(1+CV2)])]), Rmax is the maximum rate of population increase (unknown for narwhals and assumed to be 0.04, the default for cetaceans), and FR is a recovery factor that varies between 0.1 and 1.

Nmin is 10,040 using the most recent survey abundance estimate of 12,485 (CV =0.26) and PBR is 201 animals assuming a FR of 1 (Asselin et al. 2012). With the PBR value and a LRC of 1.28 a Total Allowable Landed Catch (TALC) for the Northern Hudson Bay narwhal stock is 157 narwhals (Asselin et al. 2012). The average annual landed catch of 83 narwhals an average total estimated removals for the period 2002-2015 is therefore considered sustainable.

6. Habitat and other concerns

Narwhal populations in Canada, including the northern Hudson Bay population, may be negatively influenced by hunting, contaminants, industrial activities, and global warming.

7. Status of the stock

No recent population survey since 2011 and the few surveys over time (about 4 since the 1980s) limit the ability to assess population status of the northern Hudson Bay narwhal stock. Although, the most recent survey estimated over twice as many narwhal in the population as had been previously estimated, the 2011 survey covered more area and was overall better designed. The population has probably not increased although it may be reasonable to assume that the population is not depleted from historical times. More active monitoring may be the only requirement to avoiding population depletion.

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Annex 28: East Greenland Narwhal Stock

By: Rikke Guldborg Hansen, Mads Peter Heide-Jørgensen and Eva Garde

1. Distribution and stock identity

Stock identity is based on consistent summer aggregations, telemetry tracking, genetics, aerial surveys and local knowledge and hunting patterns.

Telemetry data show that these narwhals begin their fall migration in October where they travel ~350 km to their wintering grounds in the Greenland Sea. This is the shortest migratory route of all known narwhal populations. Individuals from this stock are susceptible to hunters from two different hunting regions; Ittoqqortotmiit and Tasilaq.

Narwhals from the North West Atlantic show low levels of nucleotide and haplotype diversity based on the first 287 base pairs in the mitochondrial control region (Palsbøll et al. 1997). Despite the low degree of variation, frequencies of common haplotypes differed markedly between areas. In East Greenland only one haplotype was found supporting the hypothesis of little or no gene flow between eastern and western Greenland. Heterogeneity was found between Melville Bay narwhals and narwhals from the Avernersuaq district which includes Inglefield Bredning. Hence little gene flow is occurring between the western Greenland summer areas and northern Baffin Bay (eastern Canada and Avanersuaq). Within the northern Baffin Bay samples no significant levels of heterogeneity was found indicating some gene flow between summer grounds within this area. Narwhals show annual fidelity to summer and autumn feeding grounds and pods from these feeding grounds utilizes the same winter grounds.

Watt et al. (2013) conducted stable isotope analysis on narwhal skin samples collected by Inuit hunters during their subsistent narwhal hunt in Canada and Greenland. Stable isotope analysis on carbon (δ^{13}C) and nitrogen (δ^{15}N) revealed the three narwhal populations of Baffin Bay (BB), Northern Hudson Bay (NHB) and East Greenland (EG) to have distinct stable isotope values that were not expected based on geographic differences. Also, males in all populations had significantly higher δ^{13}C.

2. Abundance

The most recent (2008) abundance estimate for this stock is 6,444 (cv=0.51; 95% CI 2,505-16,575. Heide-Jørgensen et al. 2010). The estimate is corrected for both perception and availability bias.

This estimate comes from a visual aerial line transect survey conducted as a double-observer experiment in a fixed-winged, twin-engine aircraft (DeHavilland Twin Otter) with a target altitude and speed of 213m and 168km h-1, respectively. The front observers (observer 1) acted independently of those in the rear (observer 2) and vice versa. Declination angles to sightings, species and group size were recorded when the animals came abeam. Beaufort sea state was recorded at the start of the day and then again when it changed. Decisions about duplicate detections (animals seen by both observer 1 and 2) were based on coincidence in timing and location of sightings. The same observers were used for all three surveys except for the 3rd survey where one observer had to be replaced, however, all observers were experienced with both the animals and the data collection schemes from >100 hours participation in other aerial surveys. Instrumentation of the plane and the procedures for data collection were identical to those previously reported by Heide-Jørgensen et al. (2010).

Correction for availability of the at-surface-abundance was based on availability correction factors obtained from two whales from August-September in Melville Bay.(a=0.21; cv=0.09).

3. Anthropogenic removals

This stock is hunted by the communities in Ittoqqortotmiit and Tasiilaq during summer.

For narwhal hunting in open water in Canada Weaver and Walker reported loss rates between 32% and 55%, or catch correction factors of 1.5-2.2. Roberge and Dunn reported catch correction factors for narwhals in Canada to range from 1.11 in open water to 1.41 at the ice crack and 1.56 at the floe edge or ice edge (NAMMCO/SC/22-JCNB/SWG/2015-JWG/06).

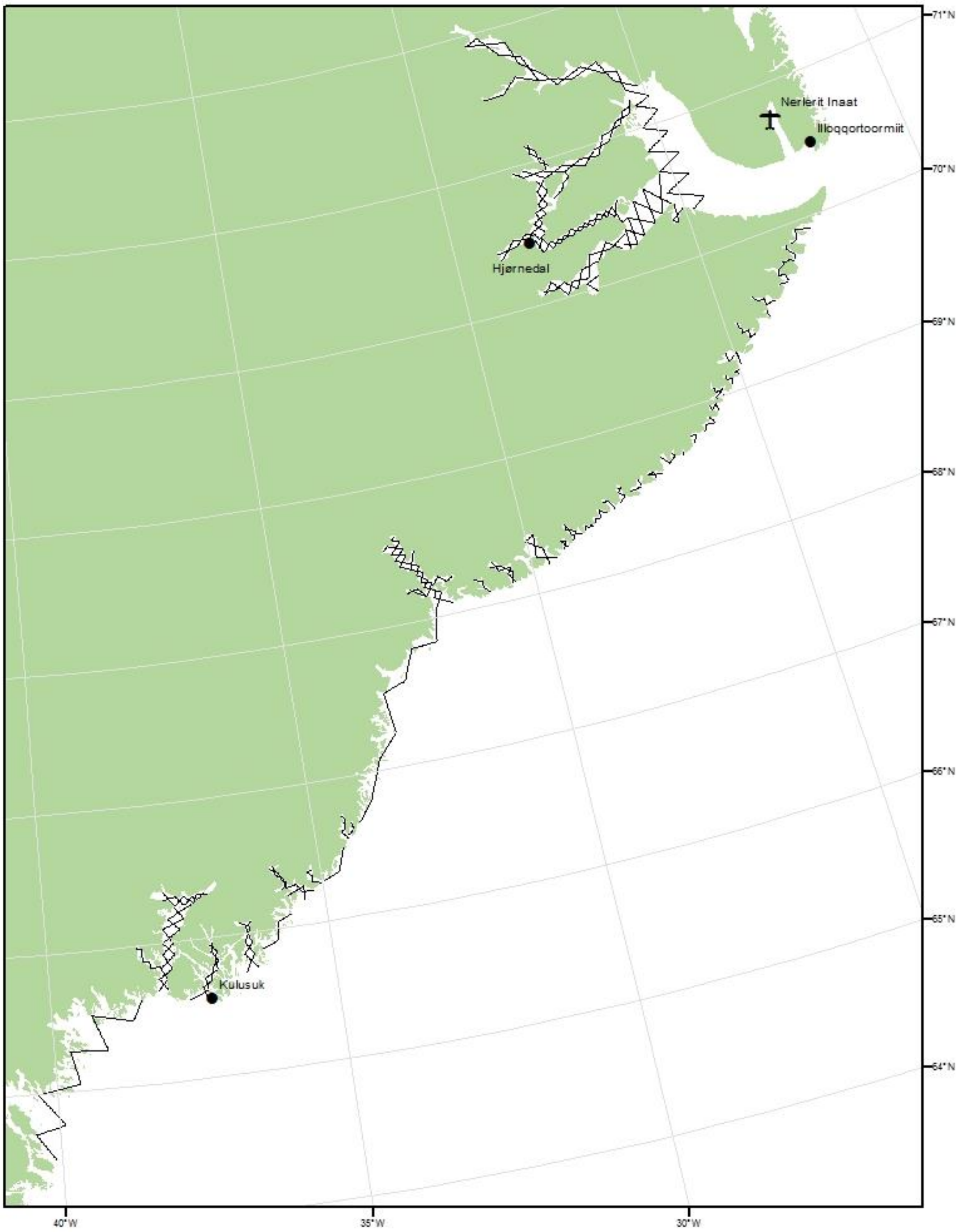

Figure 1. Map of the survey area and transects lines for the East Greenland stock area.

For Greenland it is assumed that a catch correction factor of 1.30 covers both the open water hunt and the hunt from ice cracks and the ice edge (for the Melville Bay-Upernavik area a factor of 1.15 is used). The correction factor of 1.30 also covers the open-water hunt in late autumn just before freeze-up, which is a type of hunt where loss rates have not been estimated. If anything the correction factor of 1.30 applied here is downward biased.

Official catch statistics for monodontids from Greenland include catches that are taken from whale pods entrapped in the ice. It has been suggested that mortality in ice entrapments occasionally is part of the natural mortality (Siegstad and Heide-Jørgensen 1994). To allow for analyses of removals without catches in ice entrapments these are shown separately from the mortality genuinely caused by humans.

Table 1. Catches of narwhals in East Greenland. Data from 1955-1990 from Dietz et al. (1994) and data from 1993-2014 from Piniarneq. Data from 2014 are preliminary. There was one ice entrapment in Tasiilaq in February 2008 that involved about 37 narwhals.

Year	Ittoqqortormiit	Tasiilaq	All
1955	18	6	24
1956	10		10
1957	9	5	14
1958	28	1	29
1959	17	9	26
1960	54	2	56
1961	12	4	16
1962		3	3
1963	8	21	29
1964	8		8
1965			0
1966	2	67	69
1967		20	20
1968		30	30
1969	6	17	23
1970	6	47	53
1971	5	33	38
1972	1	25	26
1973	4	18	22
1974	2	40	42
1975	2	2	4
1976	1	8	9
1977	5	14	19
1978	1	1	2
1979	10	20	30
1980	10	49	59
1981	15	128	143
1982	25	84	109
1983	43	12	55
1984	50		50
1985	28	21	49
1986		63	63
1987		19	19
1988	40	11	51
1989	70	19	89
1990	70	88	158
1991			
1992			

Year	Ittoqqortormiit	Tasiilaq	All
1993	9	16	25
1994	17	20	37
1995	34	35	69
1996	8	39	47
1997	9	42	51
1998	21	26	47
1999	19	99	118
2000	11	28	39
2001	52	70	122
2002	54	55	109
2003	6	87	93
2004	39	96	135
2005	50	68	118
2006	93	29	122
2007	39	40	79
2008	37 *	39	76
2009	12	0	12
2010	20	10	30
2011	30	15	45
2012	31	17	48
2013	47	19	66
2014	63	18	81

4. Population trajectory

Since the survey in 2008 there has been a survey in 2016 but these results have not been reviewed yet.

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

The estimated total allowable take for the East Greenland stock is 66 individuals per year (2015-2020) with 70% probability for a larger population size in 2020.

6. Habitat and other concerns

Possible concerns include changes in sea ice regime, traffic and seismic exploration.

7. Status of the stock.

The East Greenland narwhal stock is considered to be a small, and possibly declining, population.

8. Life history

Garde et al. (2015) estimated life history parameters for narwhals from East and West Greenland (n=282) based on age estimates from aspartic acid racemization (AAR) of eye lens nuclei. The species-specific age equation used, $420.32X - 24.02 \cdot \text{year}$ where X is the D/L ratio, was determined from data from Garde et al. (2012) by regressing aspartic acid D/L ratios in eye lens nuclei against growth layer groups in tusks (n=9). Asymptotic body length was estimated to be 399 ± 5.9 cm for females at age 25 years and 456 ± 6.9 cm for males from West Greenland at age 28 years. Age at sexual maturity was assessed based on data from reproductive organs and was estimated to be 8–9 years for females and 12–20 years for males. Length at sexual maturity was ~340 cm for females and 350-400 cm for males. Estimated age at 1st parturition was 9-10 years. Oldest pregnant female was close to 70 years. Pregnancy rates for West Greenland were estimated to be 0.38. Maximum life span expectancy was found to be approximately 100 years. A population projection matrix was parameterized with the data on age structure and fertility rates. The annual rate of increase of narwhals in West Greenland was estimated to be 2.6%.

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Annex 29: Northeast Greenland Narwhal Stock (North of Scoresby Sound)

By: Rikke Guldborg Hansen, Mads Peter Heide-Jørgensen and Eva Garde

North of Scoresby Sound, narwhals are frequently found in Young Sound (74°N) and along the coast as far north as Nordost Rundingen (82°N) Boertmann and Nielsen (2009 and 2010).

Narwhals north of Scoresby Sound are protected by the Northeast Greenland National Park, no hunting takes place and no attempts have been made to assess the abundance of narwhals in the national park. A survey is planned for 2017.

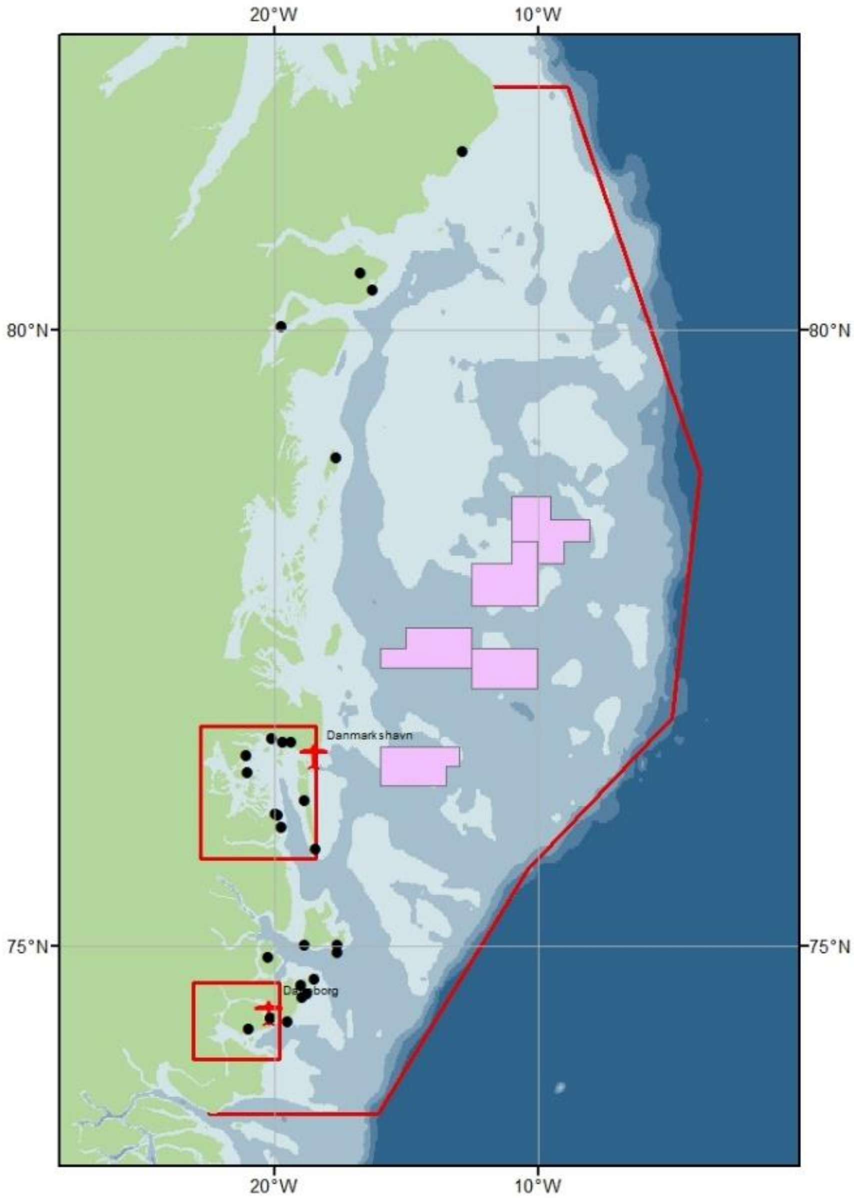

Figure 1. Map of planned survey area in 2017. Off shore area, Dove Bay and Young Sound indicated in red square boxes (black dots are walrus haul out sites). Northeast water area described in references but are within the off shore box.

References
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Boertmann and Nielsen (2010) http://www2.dmu.dk/Pub/FR773.pdf

Annex 30: Svalbard Narwhal Stock

Lydersen C and Kovacs KM

1. Distribution and stock identity

Narwhals are generally rarely observed along the coasts of Svalbard. Three sub-adult narwhals were caught on the east-side of Svalbard summer 1997. Satellite tags were deployed but stayed attached for only short periods (4-46 days) and the tracks are shown in the figure below (Source: Lydersen et al. 2007).

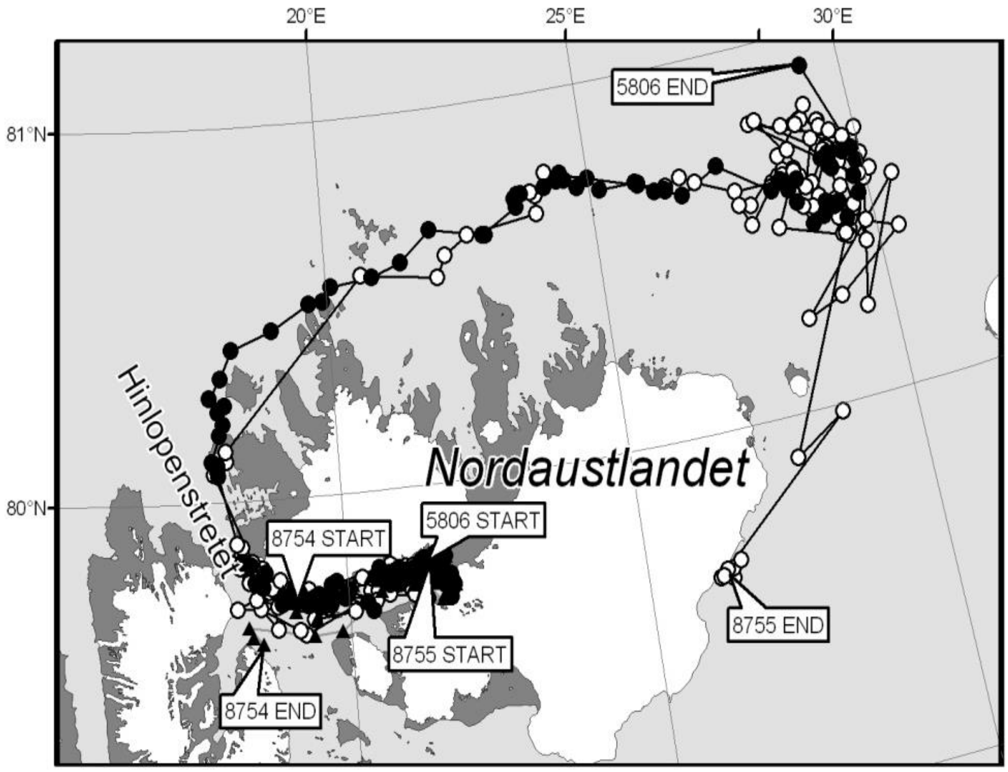

Figure from Lydersen et al. (2007) Tracks of three narwhals instrumented with satellite relay data loggers in Svalbard summer 1997.

2. Abundance

A survey for various whales in the marginal ice zone north of Svalbard August 2015 resulted in an abundance estimate for this species of 837 (CV= 0.501) within the 52919 km^2 study area (Vacquie-Garcia et al. 2017).

The narwhals were all observed from helicopter (not from survey ships) and were located deep into the marginal ice zone. Many of the observations were close to the end of the line-transects, thus indicating that more animals of this species likely would be found even further north.

3. Anthropogenic removals

Narwhals are protected in Svalbard.

4. Population trajectory

No data.

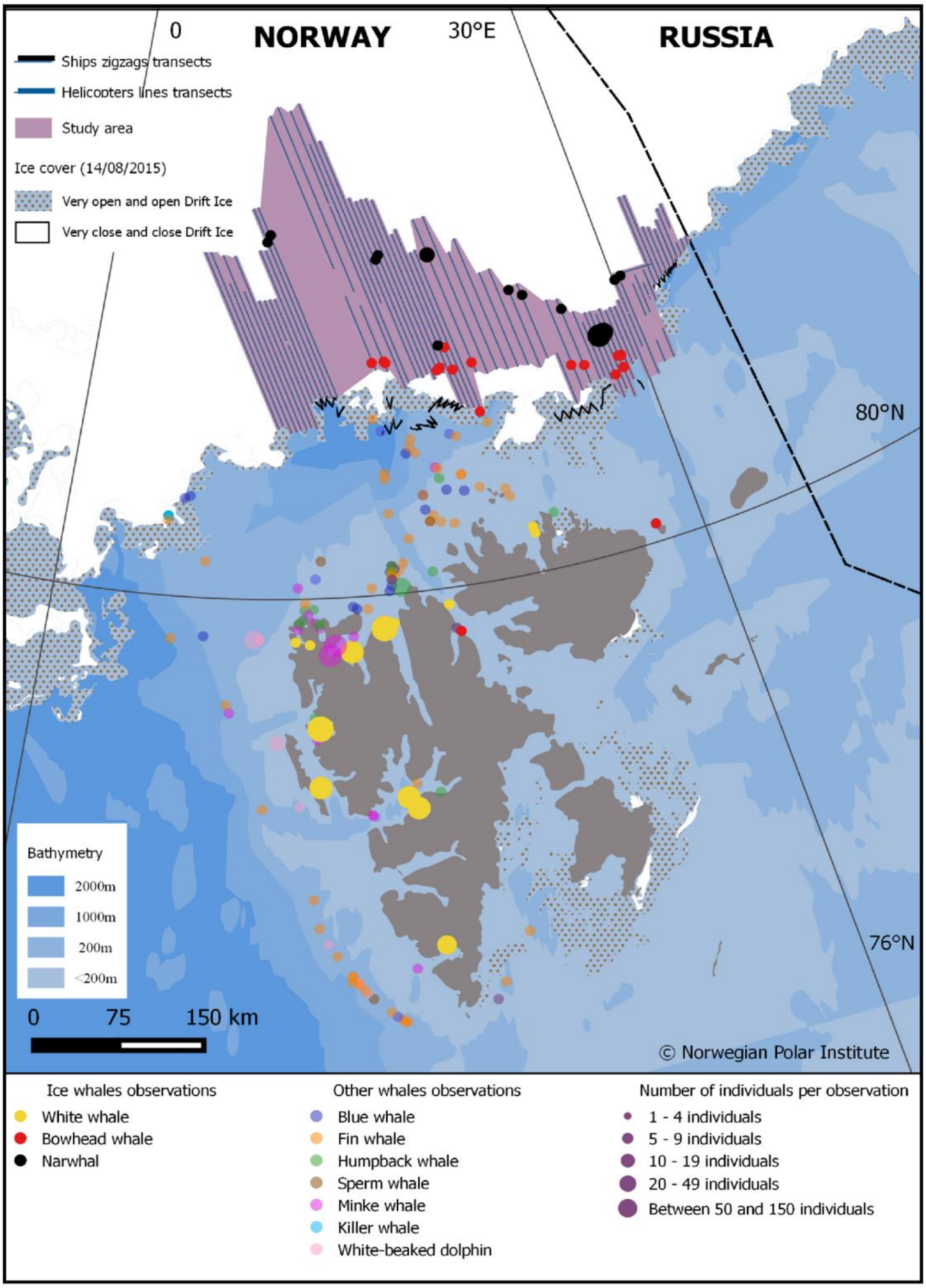

Figure shows narwhals observations as black circles deep inside the marginal ice zone. Data from (Vacquié-Garcia et al. 2017).

5. Potential biological removals or other information on safe (sustainable) limits of anthropogenic removals

Not relevant.

6. Habitat and other concerns

Effects of climate change with impacts on sea ice conditions, prey base composition, competition from more boreal marine mammal species, new parasites and diseases, is a general concern. Levels of various pollutants are even higher than what is found in white whales from this area (Wolkers et al., 2006).

7. Status of the stock

Likely a small stock, but distributed in areas just barely surveyed, so big unknowns here. No data on trends.

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