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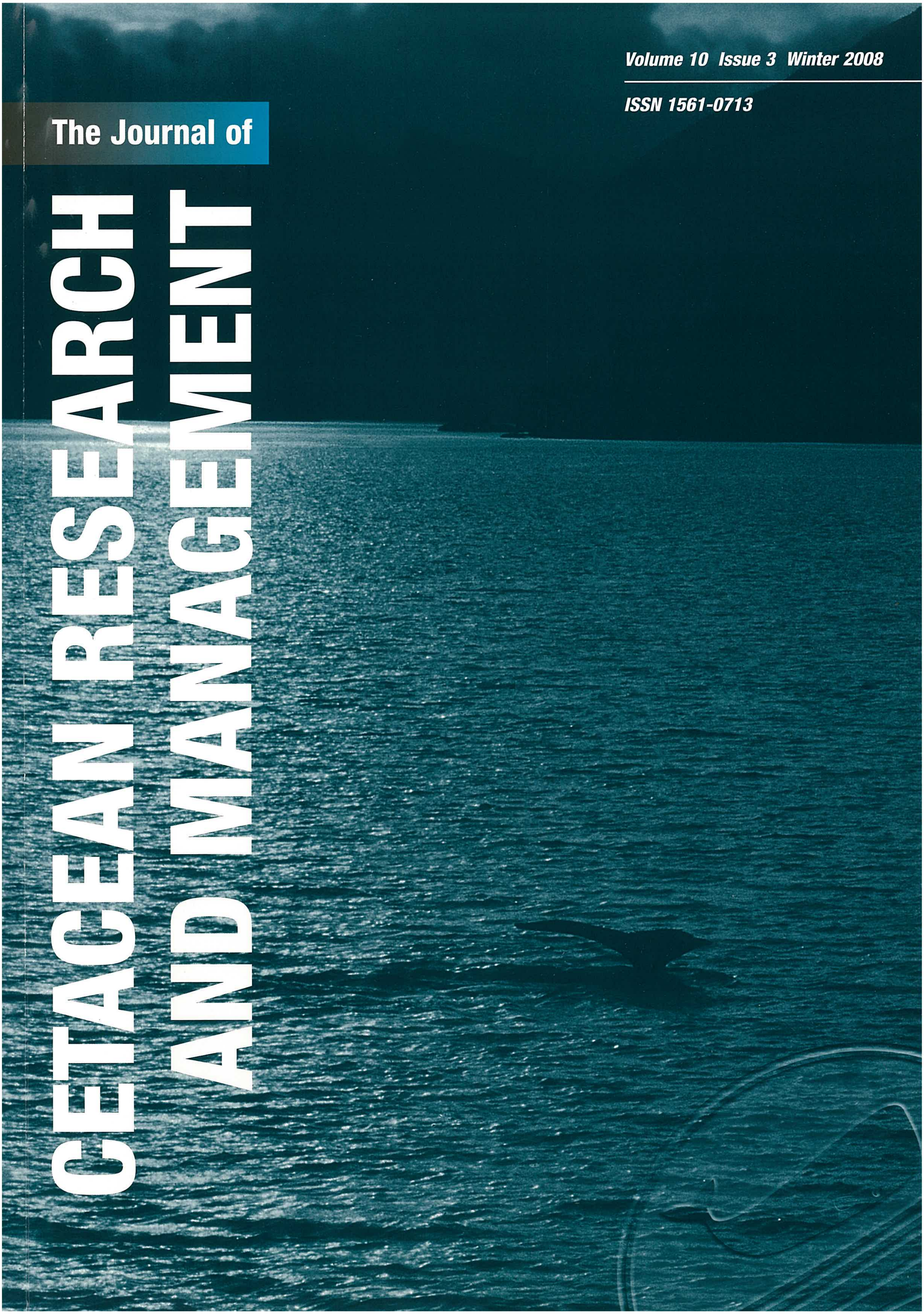

Contents

Editorial

Welcome to this the final issue of Volume 10 of the *Journal of Cetacean Research and Management*.

2008 has been another successful year for the Journal. A total of 30 papers have been published this year covering a wide range of subjects related to the conservation and management of cetaceans. There have been a total of 120 authors from Europe, Asia, Australia, Africa and North and South America. This year's supplement included the full report of the Scientific Committee held in Anchorage, Alaska in May 2007, as well as the results of five important intersessional workshops. The author and subject index can be found at the end of this issue.

The present issue contains three papers of great relevance to the mitigation of one of the major problems facing cetacean populations (especially small cetaceans) – that of their incidental capture in fishing gear. Two of the papers look at the effect of acoustic deterrents. Palka *et al*. look at the perennial question of the use of pingers, and in particular the question of possible habituation. They examined a dataset for the gillnet fishery in the northeastern USA managed under their harbour porpoise take reduction plan for the years 1999-2007 (some 25,000 gillnet hauls). After taking into account a number of variables, they concluded that at least for the fishery and period they examined, there was no evidence for habituation. They also emphasised the importance of mesh size and proper enforcement of the regulations with respect to pinger use. Pingers were developed and first tested for harbour porpoises. Berrow *et al*. undertook at-sea experiments with a variety of acoustic signals, examining their effect on the behaviour of common dolphins in Irish waters. They concluded that at their present stage of development, pingers may not provide a consistently effective deterrent for common dolphins. The final paper by Trippel *et al*. takes a different approach with respect to entanglements of the critically endangered North Atlantic right whale off the eastern coast of Canada. In this case they examine the gear itself and in particular the addition of barium sulphate to rope and twine to produce a neutrally buoyant groundline with a view to maintaining a lower profile in the water and to have a lower (but sufficient) breaking strength. With respect to harbour porpoises in the same area the barium sulphite increased the acoustic detectability of the nets. In both cases the authors consider that barium sulphite modified gear shows potential for reducing entanglement deaths.

The IWC Scientific Committee has pioneered the use of the management procedure approach for the management of large whale fisheries (either commercial or aboriginal subsistence whaling) which *inter alia* involves the use of a simulation modelling framework to incorporate uncertainty. Punt looks at the modelling of parameters related to maximum sustainable yield with stochastic population dynamics. Aldrin *et al*. examine the properties of the *Catch Limit Algorithm* that forms the basis of the Revised Management Procedure for commercial whaling, particularly in the context of 'tuning' and the appropriate simulated management time horizon.

The issue of stock structure and abundance and trends are clearly of major importance in management. Rugh *et al*. examine the use of photo-identification data to examine mixing and stock structure for the Beaufort-Chuckchi-Bering Sea bowhead whales. They conclude that the wide mixing and near-random distribution of resighting dates throughout the spring migration is indicative of single stock of whales. This provides another indication of the value of photo-identification data. Lowry *et al*. examine trends in aerial survey counts of white whales in Bristol Bay Alaska from 1993-2005 to examine increase rates (not absolute abundance). Over the period, they estimated an increase rate of around 4.8% (95% 2.1-7.5%). Afsal *et al*. look at the use of platforms of opportunity to look at cetacean distribution in an area where it is not financially feasible for dedicated surveys to occur (the Indian EEZ and contiguous seas). The final paper in the volume by Reeves *et al*. looks at historical whaling records to determine whether they can shed light on the past occurrence of the critically endangered western gray whale in the Okhotsk Sea for comparison with present day records.

Finally, I would like to thank the 43 scientists that have acted as anonymous reviewers for the papers published in Volume 10 (Aguilar, A.; Bannister, J.L.; Barlow, J.; Best, P.B.; Burt, L.; Butterworth, D.S.; Clark, C.W.; Cooke, J.G.; Desportes, G.; Eigaard, O.R.; Forney, K.A.; Gales, N.; Hammond, P.S.; Heide-Jorgensen, M-P.; Jefferson, T.A.; Kasuya, T.; Larsen, F.; Lawson, J. Leaper, R.; Martin, A.R.; McDonald, M.; Muir, D.; Northridge, S.; Notarbartolo di Sciara, G.; O'Hara, T.; Oien, N.; Palka, D.; Pastene, L.A.; Perrin, W.F.; Punt, A.E.; Raga, J.A.; Read, A.J.; Reeves, R.R.; Rugh, D.J.; Secchi, E.; Skaug, H.J.; Suydam, R.S.; Tougaard, J.; Van Waerebeek, K.; Wade, P.R.; Wells, R.S.; Wilson, B.; Zerzini, A.). Without their diligence and hard work, the papers in the *Journal*, and more importantly the contribution they make to the wise management and conservation of cetaceans, would be considerably poorer. A full list of the reviewers and their affiliations can be found at: http://www.iwcoffice.org/publications/contents_reviewers.htm#review.

G. P. DONOVAN
Editor

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A note on the modelling of MSY-related parameters when population dynamics are stochastic

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ABSTRACT

A method is outlined for calculating the values for the parameters which determine MSYR and MSYL in the types of population dynamics models on which *Implementation Simulation Trials* and *Evaluation Trials* are based in the face of environmental variability in fecundity (birth rate) and survival. The method is illustrated using a minke whale-like biology in which MSYR is defined in terms of harvesting of the mature female component of the population. Results are shown for various levels of environmental variation in survival and fecundity.

KEYWORDS: MANAGEMENT PROCEDURES; MSYR; SIMULATION; MODELLING; BIRTH RATE; COMMON MINKE WHALE

INTRODUCTION

Management advice for whale populations is provided by the Scientific Committee of the International Whaling Commission (IWC SC) on the basis of management procedures. These are pre-specified rules that specify the data on which scientific management recommendations are to be based and how those data are to be analysed to provide recommended levels of removals. Management procedures have been developed to calculate catch limits for commercial whaling of baleen whales on their feeding grounds and to calculate strike limits for aboriginal subsistence whaling. Catch limits for commercial whaling are calculated using the Revised Management Procedure (RMP) (IWC, 1994; 1999) whereas strike limits for aboriginal subsistence whaling are calculated using case-specific *Strike Limit Algorithms* (*SLAs*). To date, *SLAs* have been developed for the Bering-Chukchi-Beaufort (B-C-B) Seas stock of bowhead whales, *Balaena mysticetus* (IWC, 2003a) and for the Eastern North Pacific (ENP) stock of gray whales, *Eschrichtius robustus* (IWC, 2005a).

The ability of candidate management procedures to achieve the management goals is determined using Monte Carlo simulation methods, often referred to as the Management Strategy Evaluation (MSE) approach. For example, the variants of the RMP for specific whale stocks have been selected so that they achieve the highest long-term yield whilst simultaneously achieving pre-agreed standards related to conservation (IWC, 2005b). In contrast, *SLAs* are selected to satisfy the 'need' of aboriginal communities subject to achieving conservation objectives (Punt and Donovan, 2007). Evaluation of anticipated conservation performance and resource use of candidate *SLAs* and RMP variants is based on simulation evaluation using *Implementation Simulation Trials* (RMP variants) and *Evaluation Trials* (*SLAs*) that attempt to capture the primary sources of uncertainty for the stocks concerned (Punt and Donovan, 2007).

A very wide range of uncertainties have been considered by the IWC SC when selecting RMP variants and *SLAs*. For example, trials have explored the impact of bias in survey estimates, levels of stock productivity, changes over time in demographic parameters and stock structuring, amongst very many others. However, all of the RMP *Implementation Simulation Trials* have been based on deterministic population dynamics models, while only a few of the *Evaluation Trials* used during the development of the *SLA* for the B-C-B bowhead whales (IWC, 2003b) included trials that took account of both environmental and demographic stochasticity. This relative lack of consideration of stochasticity in the population dynamics is perhaps surprising given that the extent of variation in recruitment has been found to impact the performance of management procedures for many fish and invertebrate species, as well as the ability to estimate current population size and productivity (Butterworth and Punt, 1999; Punt, 2006). Moreover, Cooke (2007) suggested that not only did the precision of population model-based estimates of the Maximum Sustainable Yield (MSY) rate for whale stocks deteriorate in the face of environmental stochasticity, but also that environmental variation led to biased estimates of MSY rate.

The values for the parameters that determine the MSY rate and MSY level, MSYL (A, the resilience parameter, and z, the degree of compensation), in the operating models in which there was stochastic variation were based on the same approach as is used to calculate the values for these parameters when there is no environmental or demographic stochasticity. This note therefore outlines how the values for A and z can be calculated when there is stochasticity in the population dynamics. The method is general and can be applied to any stochastic population dynamics model. For the purposes of this paper, the method is applied to an extension of the population dynamics model on which *Implementation Simulation Trials* and *Evaluation Trials* have been based which allows for environmental stochasticity in both fecundity (birth rate) and survival. Differences in the values for A and z between the stochastic and deterministic variants of the model are illustrated for a minke whale-like biology.

METHODS

Population dynamics model

The dynamics of the population are governed by the equation[1]:

$$N_{y+1,a} = \begin{cases} B_{y+1} & \text{if } a = 0 \\ N_{y,a-1}S_{y,a-1}(1 - V_{a-1}E_y) & \text{if } 1 \le a \le x-1 \\ N_{y,x-1}S_{y,x-1}(1 - V_{x-1}E_y) & \\ +N_{y,x}S_{y,x}(1 - V_xE_y) & \text{if } a = x \end{cases} \tag{1}$$

where

$N_{y,a}$ is the number of animals of age a at the start of year y,
B_y is the number of births at the start of year y,
V_a is the selectivity of the fishery on animals of age a,
$S_{y,a}$ is the survival rate of animals of age a during year y,
E_y is the exploitation rate during year y[2], and
x is the maximum age (taken to be a plus-group).

Following IWC (2003b), the number of births during year y, B_y, is assumed to be stochastic and related to the expected fecundity, b_y^*, under the assumption that the logit of the birth rate is normally distributed[3],[4]:

$$B_y = N_y^m \frac{1}{1 + e^{\mu_y + \varepsilon_y}} \qquad \varepsilon_y \sim N(0;\sigma_\varepsilon^2) \tag{2}$$

and

$$b_y^* = f_0(1 + A(1 - (N_y^m / K^m)^z)) \tag{3}$$

where

N_y^m is the number of animals that have reached the age-at-first-parturition by the start of year y:

$$N_y^m = \sum_a M_a N_{y,a} \tag{4}$$

M_a is the proportion of females of age a that could have given birth,
K^m is the number of animals that have reached the age-at-first-parturition in the unfished state,
μ_y is a birth-rate parameter, selected so that the expected value of B_y, is $b_y^* N_y^m$, i.e.:

$$b_y^* = \int_{-\infty}^{\infty} (1 + e^{\mu_y + \varepsilon})^{-1} \frac{1}{\sqrt{2\pi}\sigma_\varepsilon} e^{-\frac{\varepsilon^2}{2\sigma_\varepsilon^2}} \, d\varepsilon \tag{5}$$

σ_ε determines the extent of stochasticity in fecundity, and
f_0 is the (expected) fecundity rate at pre-exploitation equilibrium.

The survival rate during year y for animals of age a, $S_{y,a}$, is also assumed to be stochastic (and perfectly correlated among ages[5]). The logit of survival rate is assumed to be normally distributed so annual survival is generated using the equation:

$$S_{y,a} = (1 + e^{\phi + \eta_y})^{-1} \qquad \eta_y \sim N(0;\sigma_\eta^2) \tag{6}$$

where

ϕ is a survival rate parameter, selected so that the expected value of $S_{y,a}$ is $\tilde{S}$, i.e.:

$$\tilde{S} = \int_{-\infty}^{\infty} (1 + e^{\phi + \eta})^{-1} \frac{1}{\sqrt{2\pi}\sigma_\eta} e^{-\frac{\eta^2}{2\sigma_\eta^2}} \, d\eta \tag{7}$$

$\tilde{S}$ is the (pre-specified) expected survival rate, and
σ_η determines the extent of stochasticity in survival.

The catch during year y, C_y, is calculated assuming that the fishery occurs before natural mortality, i.e.:

$$C_y = E_y \sum_a V_a N_{y,a} \tag{8}$$

The catches for a given exploitation rate (i.e. $E_y = E$) are stochastic because the population dynamics (and hence the numbers-at-age) are stochastic.

Solving for *A* and *z*

The values for A and z are selected so that if the exploitation rate is set to MSYR, the derivative of the mean yield function with respect to exploitation rate is zero and so that the mean population size, when expressed relative to the corresponding pre-exploitation equilibrium size, equals MSYL. The mean yield and population size are computed by projecting the population ahead for many (1000) years many times (1000) under an exploitation rate equal to MSYR (i.e. E_y in Equation 1 is set equal to MSYR). The number of years and replicates were selected so that the distribution of population size (and catch) as a function of exploitation rate reached steady-state. The age-structure at the start of the projection period is set equal to that corresponding to the deterministic equilibrium under MSYR (note: this age-structure depends on both A and z).

Application to common minke whales

Table 1 lists the values for the pre-specified parameters of the population dynamics model for the example application. MSYR is defined in terms of harvesting of the mature component of the population (i.e. MSYR_{mat}) for consistency with how *Implementation Simulation Trials* have been parameterised for Bryde's (*Balaenoptera edeni*) and common minke whales (*B. acutorostrata*) (IWC, 2004; 2007), and MSYL is also defined in terms of this population component. Selectivity is set equal to having reached first parturition, and both selectivity and maturity are assumed to be logistic functions of age, parameterised in terms of the ages at 50%- and 95%-maturity (Table 1). Consistent with the *Implementation Simulation Trials* for the North Atlantic and western North Pacific common minke whales, animals of age two and younger are assumed to be immature (and not available for capture). A range of values for the parameters which determine the extent of environmental

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[1] The dependence of numbers-at-age on sex has been omitted for ease of presentation.

[2] Exploitation rate is assumed to be constant over time for the analyses of this paper.

[3] This is equivalent to adding environmental stochasticity to calf survival.

[4] The assumption that the logit of birth-rate is stochastic is made to ensure that the birth-rate falls between 0 and 1. Equation 2 can be modified straightforwardly to allow for gestation periods that are longer than a year.

[5] Assuming perfect correlation of the factors impacting survival will maximise the impact of environmental stochasticity on the dynamics of the population and hence make any simulation trials based on this model 'more difficult'.

variation in fecundity and survival are considered. Note that Equations 2 and 6 ensure that fecundity and survival are never less than zero or greater than one even when σ_ε and σ_η are large. The catch used when finding MSY is set to the average over the last 500 years of the projection period.

Table 1

The parameters of the population dynamics model.

Parameter	Value(s)
$MSYR_{mat}$	0.01, 0.04
$MSYL_{mat}$	0.6
$V_{50\%}$, $V_{95\%}$	7yr; 10.53yr*
$M_{50\%}$, $M_{95\%}$	7yr; 10.53yr
$\tilde{S}$	0.07 yr^{-1}
σ_ε	0, 0.2, 0.4
σ_η	0. 0.2, 0.4

*Set equal to the parameters of the maturity ogive (IWC, 1992).

RESULTS AND DISCUSSION

The primary outcome from the analysis is the steady-state relationship (equivalent to the equilibrium relationship for deterministic dynamics) between catch and exploitation rate and catch and population size (Figs 1 and 2). The upper left panels of Figs 1 and 2 compare the deterministic and stochastic ($\sigma_\varepsilon = \sigma_\eta = 0.2$) evaluations of *A* and *z* for $MSYR_{mat}$=0.01 and 0.04 respectively. As expected, the mean yield curve based on stochastic dynamics is similar to the deterministic relationship, even though the estimates of *A* and *z* differ slightly between the deterministic and stochastic cases (Table 2). The remaining panels of Figs 1 and 2 show the distributions for the number of 'mature' animals (i.e. animals that have reached the age at first parturition) relative to the pre-exploitation number of such animals as a function of exploitation rate, and the distributions of the average catch (over years 500-1000 of the projection period) and catch in year 1000 as a function of exploitation rate. As expected, the distribution for the catch in year 1000 is broader than that of the average catch. However, the extent to which this is the case is lower than might be expected because population sizes (and hence catches) are strongly temporally auto-correlated (Fig. 3).

There is considerable variability in individual trajectories of population size, with the extent of variation higher for $MSYR_{mat}$=0.01 than for $MSYR_{mat}$=0.04 (Fig. 3), and this is reflected in the distributions of catch and population size as a function of exploitation rate. There are some transient effects in the first 200 years of the projection period (particularly for $MSYR_{mat}$=0.01), which presumably reflects the impact of all of the analyses starting from the same age-structure, and in the absence of stochasticity.

Table 2 lists the values for *A* and *z* for each combination of σ_ε and σ_η while Figs 4 and 5 show the relationships between the catch in year 1000 and exploitation rate for the

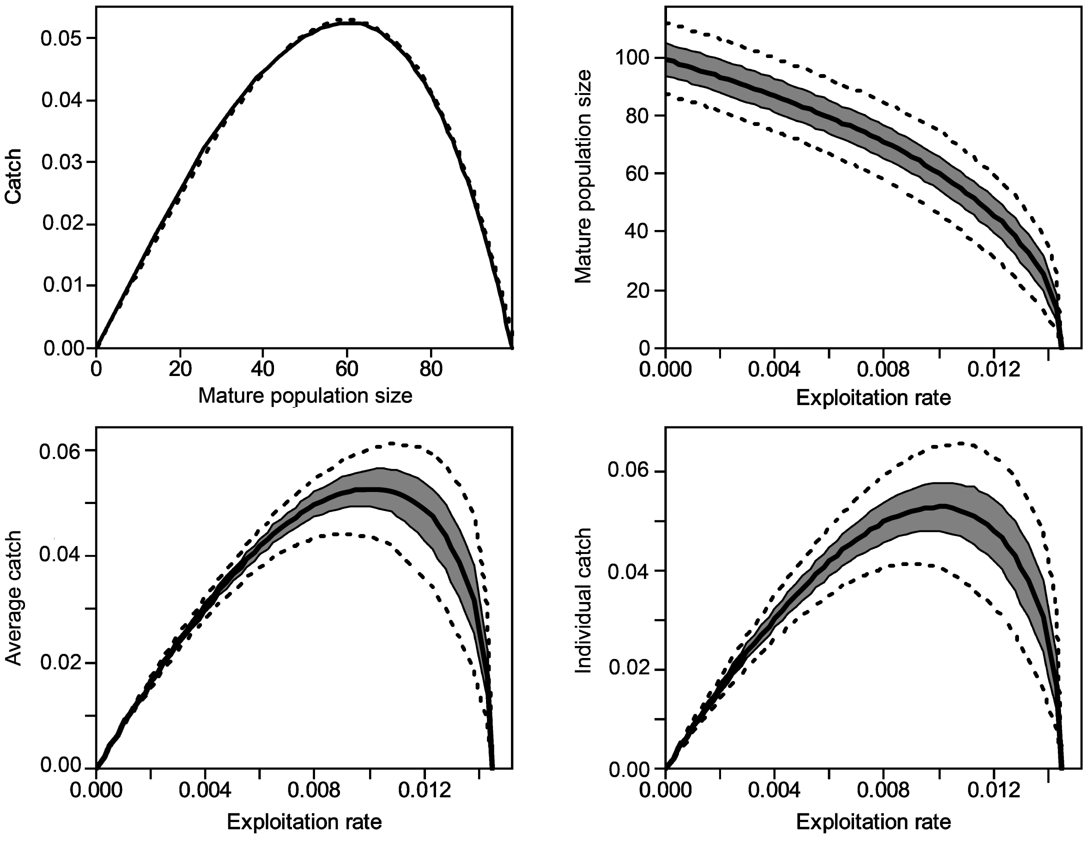

Fig. 1. Relationship between the number of mature animals (expressed as a percentage of the pre-exploitation level) and catch (in units of fractions of carrying capacity) based on deterministic analyses (solid line) and the mean of stochastic realisations (dotted line) (upper left panel), that between the depletion of the mature female component of the population (upper right panel) and exploitation rate (where exploitation rate is the proportion of the selected animals which are removed on an annual basis), and that between average catch and exploitation rate (lower panels). Results are shown in the lower left panel for the average catch over the last 500 years of a 1000-year projection period and in the lower right panel for the catch in the 1000th year. The analyses on which this figure are based assume that $MSYR_{mat}$ =0.01, $MSYL_{mat}$ = 0.6, $\sigma_\varepsilon = 0.2$ and $\sigma_\eta = 0.2$. In the distribution plots, the solid line indicates the median, the shaded region the interquartile range, and the dotted lines the 90% intervals.

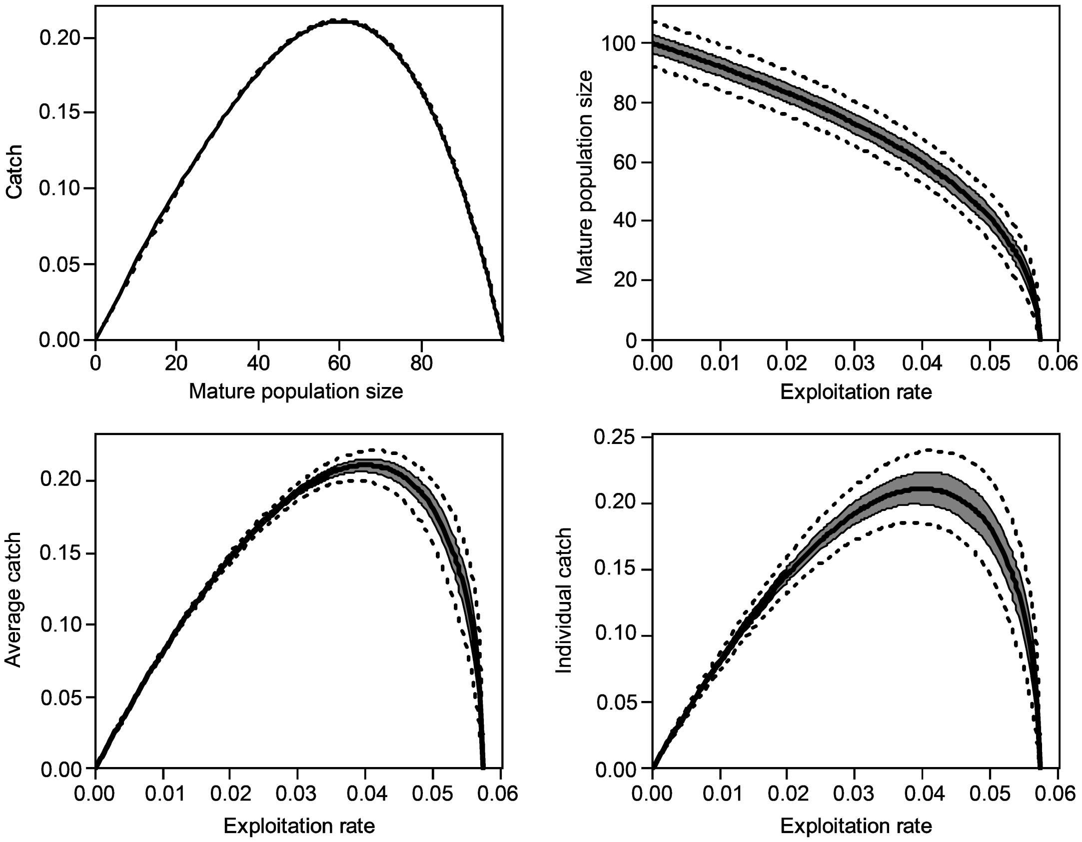

Fig. 2. As for Fig. 1, except that the analyses are based on $MSYR_{mat} = 0.04$.

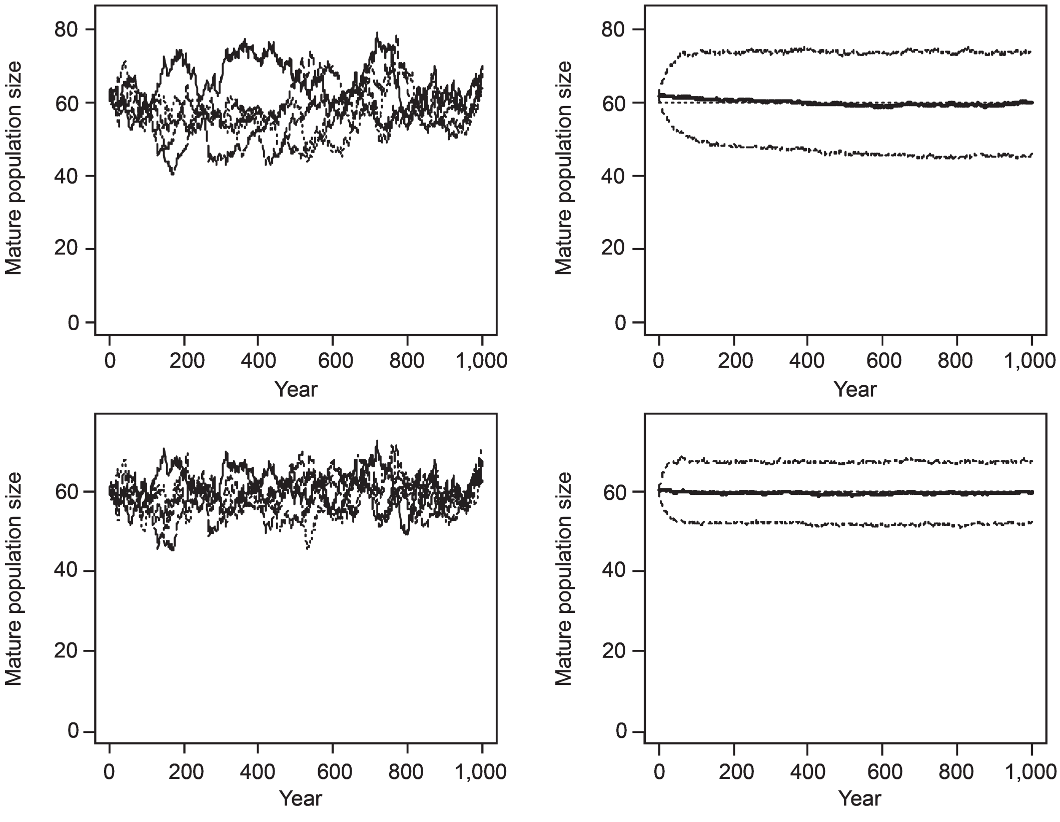

Fig. 3. Time-trajectories for the number of mature females (expressed as a percentage of the pre-exploitation number of mature females). The left panels show the results of five individual replicates and the right panels show the median and 90%iles for these time-trajectories. The results in this figure pertain to $MSYL_{mat} = 0.6$, $\sigma_\varepsilon = 0.2$ and $\sigma_\eta = 0.2$, with the upper panels based on $MSYR_{mat} = 0.01$ and the lower panels on $MSYR_{mat} = 0.04$.

Table 2

Values for the resilience and degree of compensation parameters for various choices for the extent of environmental variation in fecundity and survival.

Scenario	$MSYR_{mat} = 0.01$	$MSYR_{mat} = 0.04$
$\sigma_\varepsilon = 0; \sigma_\eta = 0$	0.1938, 2.393	0.7714, 2.402
$\sigma_\varepsilon = 0; \sigma_\eta = 0.2$	0.1949, 2.491	0.7744, 2.415
$\sigma_\varepsilon = 0.2; \sigma_\eta = 0$	0.1939, 2.434	0.7719, 2.413
$\sigma_\varepsilon = 0.2; \sigma_\eta = 0.2$	0.1966, 2.481	0.7801, 2.381
$\sigma_\varepsilon = 0.2; \sigma_\eta = 0.4$	0.1978, 3.007	0.7895, 2.437
$\sigma_\varepsilon = 0.4; \sigma_\eta = 0.2$	0.1982, 2.550	0.7841, 2.385
$\sigma_\varepsilon = 0.4; \sigma_\eta = 0.4$	0.1993, 3.127	0.7933, 2.443

combinations of σ_ε and σ_η in Table 2. *A* and *z* are not impacted noticeably by the values specified for σ_ε and σ_η (Table 2), except when σ_η is set to 0.4 (see Fig. 6 for examples of individual time-trajectories of population size for this case).

The results in Figs 4-5 highlight that environmental variation in survival has a larger impact on the population dynamics than environmental variation in fecundity for the same amount of environmental variation (contrast the widths of the distributions of catch for cases a and b and those for cases d and e). This is not unexpected because environmental variation in fecundity only impacts a single age-class whereas environmental variation in survival impacts all age-classes simultaneously (given the assumption that deviations in survival are perfectly correlated among ages). It is perhaps noteworthy therefore that the 'stochastic' *Evaluation Trials* for the B-C-B bowhead whales were based only on environmental variation in fecundity (although some *Robustness Trials* examined the impact of catastrophic events – a form of environmental variation in survival).

Although the values for *A* and *z* differ from the deterministic values (Table 2), the effect is small, which suggests that setting the values for *A* and *z* based on deterministic analyses should not lead to results of evaluations of management procedures which differ markedly from those using values for *A* and *z* based on the method of this paper. However, this needs to be confirmed in specific cases.

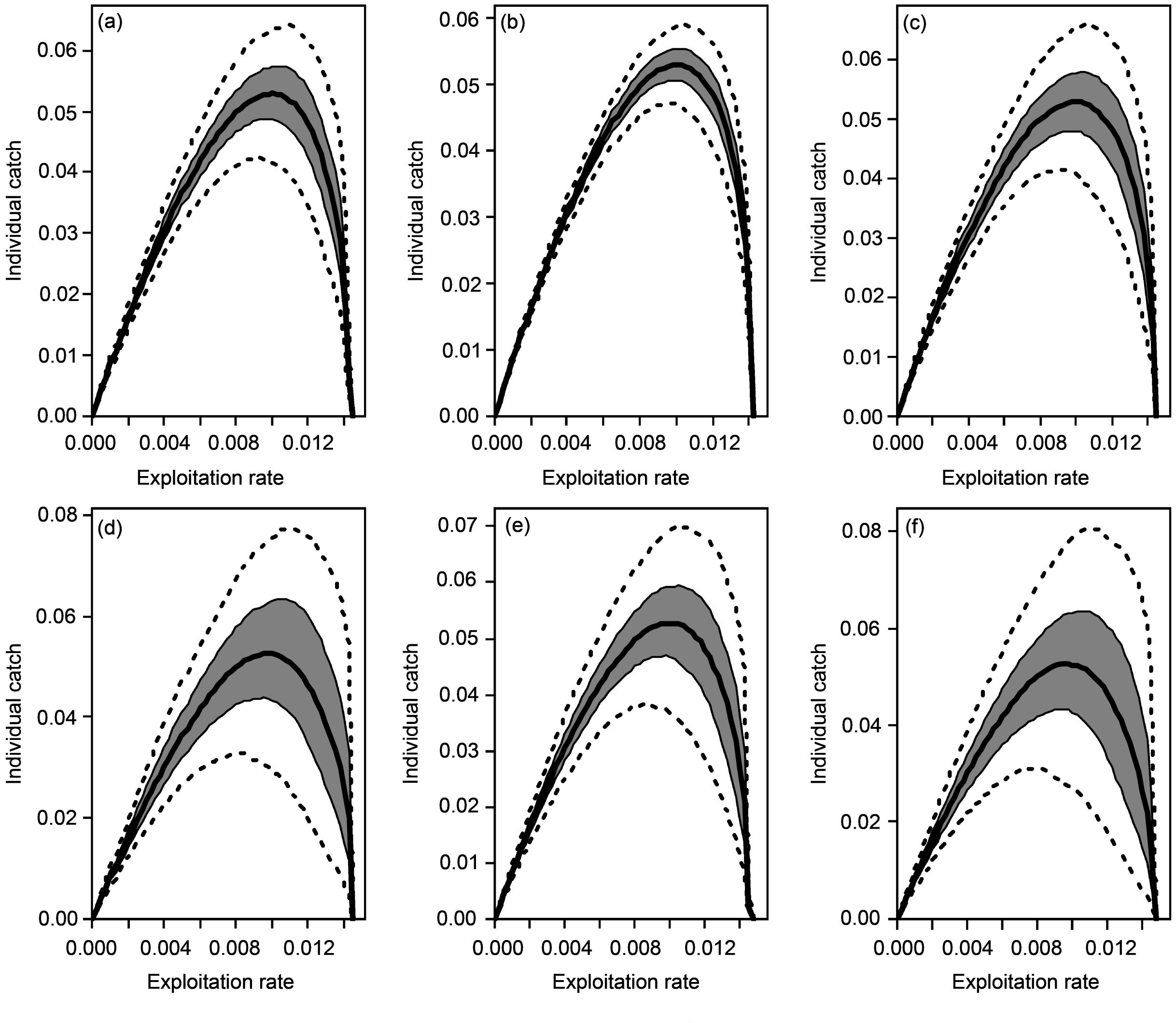

Fig. 4. Relationships between exploitation rate and the catch in year 1000 for $MSYR_{mat}$=0.01. Results are shown in (a) for σ_ε = 0; σ_η = 0.2, (b) for σ_ε = 0.2; σ_η = 0, (c) for σ_ε = 0.2; σ_η = 0.2, (d) for σ_ε = 0.2; σ_η = 0.4, (e) for σ_ε = 0.4; σ_η = 0.2, and (f) for σ_ε = 0.4; σ_η = 0.4. In the distribution plots, the solid line indicates the median, the shaded region the interquartile range, and the dotted lines the 90% intervals.

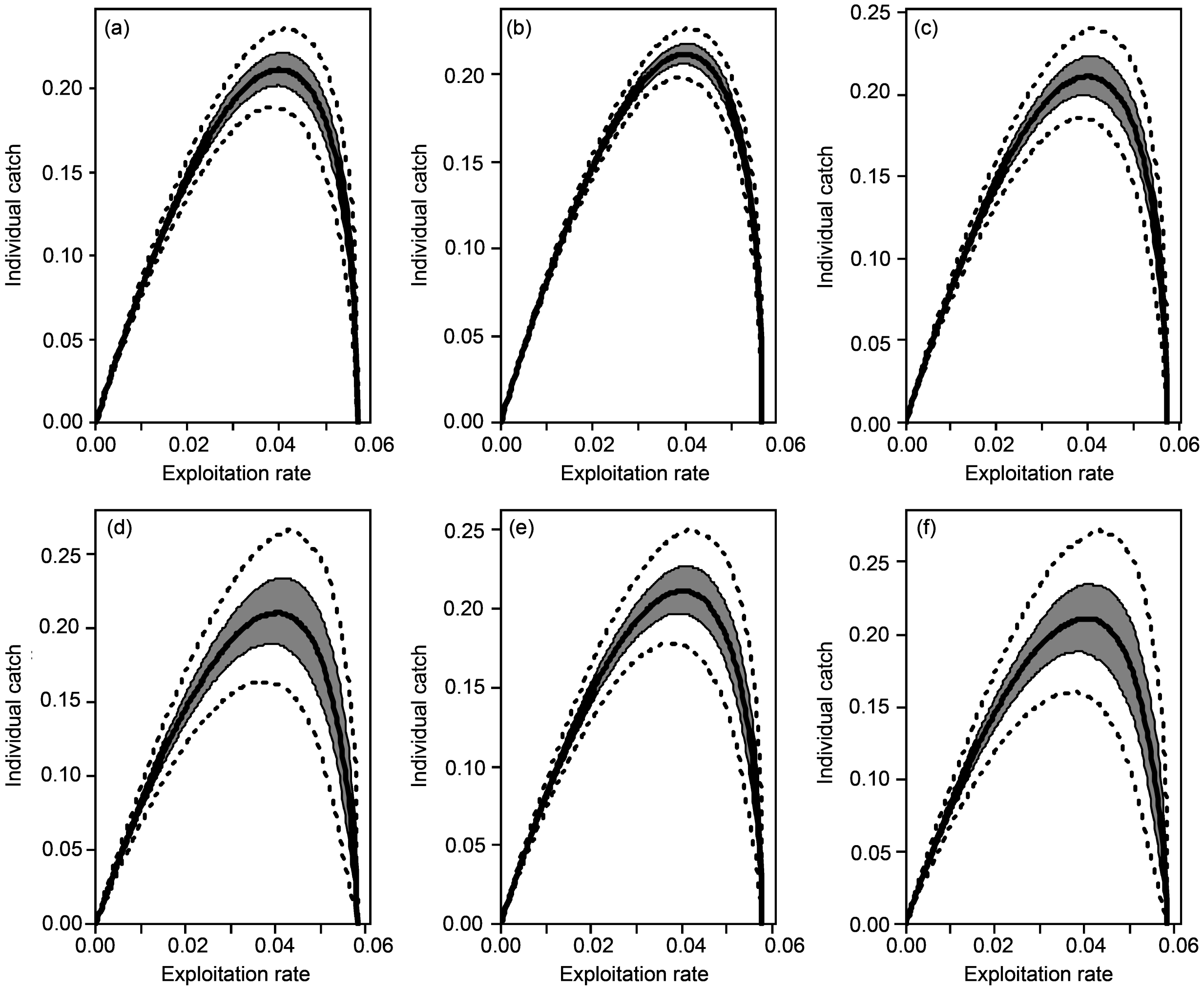

Fig. 5. As for Fig. 4, except that the results pertain to $\text{MSYR}_{\text{mat}} = 0.04$.

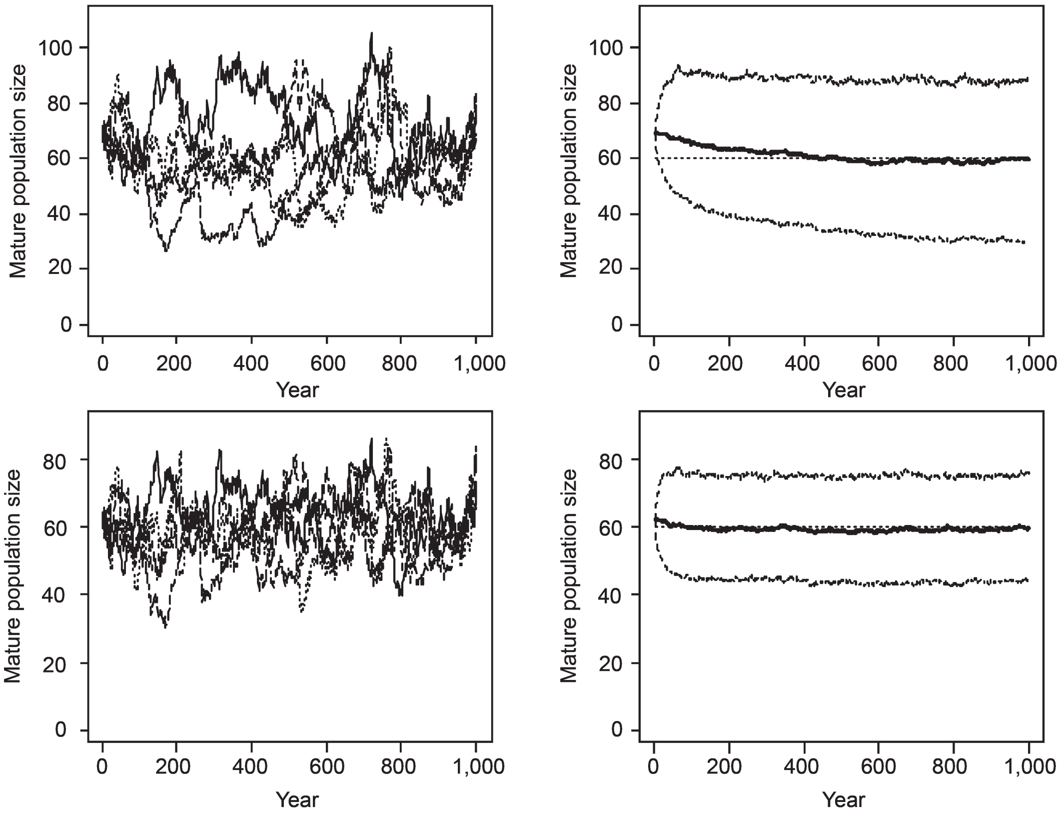

Fig. 6. As for Fig. 3, except that the results pertain to the case $\sigma_\varepsilon = 0.4$; $\sigma_\eta = 0.4$.

The choices for σ_ε and σ_η considered in this paper are arbitrary, there being no way at present to quantify the extent of inter-annual variation in fecundity or survival for minke whales. Fig. 7 shows how the standard deviation of fecundity changes as a function of mean fecundity and the value assumed for σ_ε.

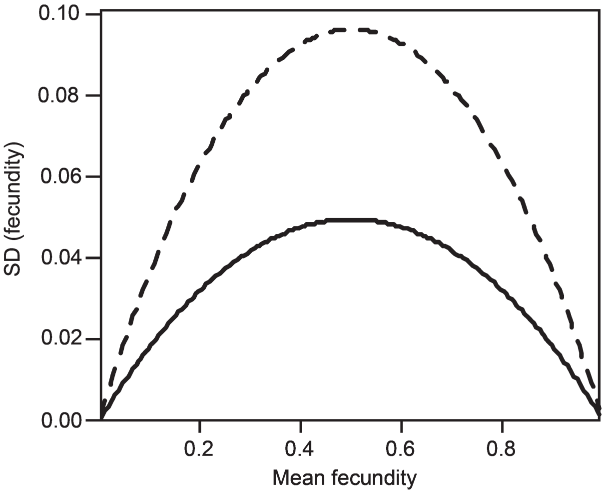

Fig. 7. Relationship between the mean and standard deviation of fecundity for two choices for σ_ε (0.2 – solid line; 0.4 – dashed line).

The approach taken in this paper has been to select values for *A* and *z* using a 'fixed exploitation rate' strategy, i.e. by selecting the values for these parameters so that the expected catch is maximised when a constant proportion of the selected animals is removed each year. This is not the only way to translate the notion of MSY from a deterministic concept into one that accounts for stochastic dynamics; a variety of definitions for stochastic MSY have been defined for fisheries management purposes in New Zealand, for instance (e.g. Francis and Mace, 2005). For example, MSY can be defined using a constant catch strategy. Adoption of this definition for MSY would mean that *A* and *z* would be selected so that the ratio of the constant catch at MSYL is MSYR. A fixed exploitation rate strategy was adopted for this paper because the management strategies used by the IWC are closer to fixed exploitation rate strategies than to constant catch strategies.

The example in this paper focused on environmental rather than demographic stochasticity because demographic stochasticity only has a noteworthy impact on the population dynamics at levels of abundance at which harvests under IWC management procedures would not be permitted anyway. However, for other cases, demographic stochasticity can be important. The method outlined here could be extended to deal with cases in which the population size is small, but it would necessitate the development of an individual-based model as well as accounting for the possibility of extinction even when the exploitation rate equals that corresponding to MSY.

Finally, the analyses of this paper ignore temporal auto-correlation in survival and fecundity caused by environmental variation. This can be incorporated straightforwardly into Equations 2 and 7 (see, for example, IWC, 2003b), although it is likely that it will be necessary for there to be much longer projection periods when there is (high) auto-correlation in fecundity and survival if reliable values for *A* and *z* are needed. Similarly, the deviations in fecundity could be correlated with those in survival to reflect the hypothesis that poor environmental conditions are reflected simultaneously in both survival and fecundity (or *vice versa*). For example, preliminary modelling work (J. Brandon, UW, pers. comm.) based on the time-series of calf counts (Perryman *et al*., 2002) and strandings data for the ENP gray whales suggests that there is a correlation (albeit not linear) between the inter-annual deviations about mean fecundity and survival. Finally, allowance could be made for occasional major reductions in abundance, as has also been postulated for the ENP gray whales.

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REFERENCES

Butterworth, D.S. and Punt, A.E. 1999. Experiences in the evaluation and implementation of management procedures. *ICES J. Mar. Sci.* 56(6): 985-98.

Cooke, J.G. 2007. The influence of environmental variability on baleen whale sustainable yield curves. Paper SC/59/RMP10 presented to the IWC Scientific Committee, May 2007, Anchorage, USA (unpublished). 15pp. [Paper available from the Office of this Journal].

Francis, R.I.C.C. and Mace, P.M. 2005. An evaluation of some alternative harvest strategies. NZ Fisheries Assessment Report 2005/60. 30pp.

International Whaling Commission. 1992. Report of the Scientific Committee, Annex K. Report of the working group on North Atlantic minke whales, Appendix 1. Specifications of the North Atlantic minke trials. *Rep. int. Whal. Commn* 42:246-51.

International Whaling Commission. 1994. Report of the Scientific Committee. Annex H. The Revised Management Procedure (RMP) for Baleen Whales. *Rep. int. Whal. Commn* 44:145-52.

International Whaling Commission. 1999. Report of the Scientific Committee. Annex N. The Revised Management Procedure (RMP) for Baleen Whales. *J. Cetacean Res. Manage. (Suppl.)* 1:251-58.

International Whaling Commission. 2003a. Report of the Scientific Committee. *J. Cetacean Res. Manage. (Suppl.)* 5:1-92.

International Whaling Commission. 2003b. Report of the Scientific Committee. Annex E. Report of the Standing Working Group on the Development of an Aboriginal Subsistence Whaling Management Procedure (AWMP). *J. Cetacean Res. Manage. (Suppl.)* 5:154-255.

International Whaling Commission. 2004. Report of the Scientific Committee. Annex D. Report of the Sub-Committee on the Revised Management Procedure. Appendix 10. North Pacific minke whale *Implementation Simulation Trial* specifications. *J. Cetacean Res. Manage. (Suppl.)* 6:118-29.

International Whaling Commission. 2005a. Report of the Scientific Committee. *J. Cetacean Res. Manage. (Suppl.)* 7:1-62.

International Whaling Commission. 2005b. Report of the Scientific Committee. Annex D. Report of the Sub-Committee on the Revised Management Procedure. Appendix 2. Requirements and Guidelines for *Implementation*. *J. Cetacean Res. Manage. (Suppl.)* 7:84-92.

International Whaling Commission. 2007. Report of the Scientific Committee. Annex D. Report of the Sub-Committee on the Revised Management Procedure. Appendix 6. The specifications for the *Implementation Simulation Trials* for western North Pacific Bryde's whales. *J. Cetacean Res. Manage. (Suppl.)* 9:114-24.

Perryman, W.L., Donahue, M.A., Perkins, P.C. and Reilly, S.B. 2002. Gray whale calf production 1994-2000: Are observed fluctuations related to changes in seasonal ice cover? *Mar. Mammal Sci.* 18(1): 121-44.

Punt, A.E. 2006. The FAO Precautionary Approach after almost 10 years: Have we progressed towards implementing simulation-tested feeback-control management systems for fisheries management. *Nat. Resour. Modeling* 19: 441-64.

Punt, A.E. and Donovan, G.P. 2007. Developing management procedures that are robust to uncertainty: lessons from the International Whaling Commission. *ICES J. Mar. Sci.* 64: 603–12.

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A note on tuning the *Catch Limit Algorithm* for commercial baleen whaling

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ABSTRACT

The *Catch Limit Algorithm* for commercial baleen whaling developed by the International Whaling Commission converges slowly to a steady depletion (proportion of carrying capacity), and consequently 300 years of management is proposed as horizon for tuning and computer simulation. Long-term depletion is rather insensitive to the parameter currently used for tuning, and an alternative control parameter is suggested for this purpose.

KEYWORDS: WHALING-REVISED CATCHES; MANAGEMENT PROCEDURE

INTRODUCTION

In 1982, the International Whaling Commission (IWC) decided to impose a moratorium or pause in commercial whaling. As part of the 'Comprehensive Assessment' that was part of that decision, the IWC's Scientific Committee began the process of developing a procedure for setting safe catch limits for commercial whaling for baleen whales (Donovan, 1989). The agreed objectives for what was to become the 'Revised Management Procedure (RMP)' were: (i) stability of catch limits which would be desirable for the orderly development of the whaling industry; (ii) acceptable risk that a stock not to be depleted (at a certain level of probability) below some chosen level (e.g. fraction of carrying capacity), so that the risk of extinction is not seriously increased by exploitation; (iii) making possible the highest possible continuing yield from the stock (IWC, 1992a). In this context, the term 'continuing yield' refers to the mean (maximum) yearly harvest in the long term, i.e. when the exploited stock has reached a stationary state. This note considers whether the current simulated management horizon (100 years) is sufficiently long to allow exploited stocks of baleen whales to reach a stationary state. Other simulation studies have shown this not to be the case for a number of scenarios (Aldrin and Huseby, 2007; Butterworth and Punt, 1994). Here, we suggest that 300 years is a more appropriate management horizon and provide a re-examination of the best approach to 'tune' the *Catch Limit Algorithm (CLA)*. The current method seems unfit to tune the median depletion level after 300 years of exploitation to targets less than 70% and here we suggest an alternative tuning method.

To expand upon this, the *CLA* (IWC, 1992b; 1999) is a core component of the RMP that is used, as its name suggests, to calculate levels of anthropogenic removals that meet the stated objectives. The conservation- and yield properties of the *CLA* were and are tuned and investigated under a range of plausible conditions by way of stochastic simulation over a management period of 100 years. In this note, it is argued that under the range of productivity of baleen whale populations that has been considered in past simulation testing of the CLA (with a maximum sustainable yield rate relative to the sexually mature population, $MSYR^{mature}$, being as low as 1%), this horizon is not long enough. With such a low value of $MSYR^{mature}$, the dynamics under many scenarios are too slow for the population of a managed whale stock to even come close to a stationary state after 100 years (Fig. 1). We suggest that the management horizon should rather be 300 years, which is now computationally feasible (it was not when the *CLA* was being developed in the late 1980s and early 1990s) and which allows for a reasonable evaluation of continuing yield and long-term depletion (the ratio of current abundance to the pre-exploitation level, the latter being termed 'carrying capacity' or *K*).

The *CLA* is currently tuned to desired target levels for depletion by the internal posterior percentage point for the catch limit given by a catch control law (see equation 3 below). Since we have found it difficult to tune the *CLA* to target depletion levels much below 0.72 (population size relative to carrying capacity) after 300 years by this method, we will propose that the *CLA* rather should be tuned by varying the steepness of the line determining the internal catch level for estimated depletion above the internal protection level in the catch control law.

This short note first reviews the nature of the *CLA* and then examines issues related to the management horizon and tuning method. It does not contribute directly to the important investigation of conservation properties of the *CLA* and the current management regime, nor does it address the question of how productive baleen whales really are (which is currently under review by the IWC Scientific Committee – see IWC, 2009).

THE CATCH LIMIT ALGORITHM

The *CLA* (IWC, 1999) consists of: a simple internal model for the population dynamics of the whale population in question; a prior distribution for its parameters; a procedure for calculating the posterior distribution from historic catches and available abundance estimates, but with nominal standard errors and correlations on the log scale being quadrupled; and a catch control rule to set catch limits. The input data to the *CLA* consists of a time series of historic annual catches and of absolute abundance estimates

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along with their standard errors and correlations on the logarithmic scale.

The internal population model is the age-aggregated Pella-Tomlinson model

$$P_{t+1} = P_t - C_t + 1.4184\mu P_t\left(1 - \left(\frac{P_t}{P_0}\right)^2\right) \quad 0 \le t < T \qquad (1)$$

where P_t and C_t are respectively the number of individuals of age 1+ and the number of individuals removed by whaling (assumed known without error), both at the beginning of year t, and μ is an internal productivity parameter. The catch limit is to be calculated for year T. The population is assumed to be at (internal) carrying capacity P_0 in year 0, which is the first year of recorded catch.

The abundance estimates are assumed to be log-normally distributed with an estimated covariance matrix Σ on the log scale, which is assumed known. Treating the depletion $D_T = P_T/P_0$ as a parameter, and deleting constant factors, the likelihood based on the abundance data at time T is taken to be

$$L(\mu, D_T, \beta) = \exp\left(-\frac{1}{2\tau^2}\left(\underline{a} - \underline{p} - \beta\underline{1}\right)^t \Sigma^{-1}\left(\underline{a} - \underline{p} - \beta\underline{1}\right)\right) \qquad (2)$$

where $\tau = 4$ as discussed below. The underlined terms are vectors of log estimated abundance (a), log abundance (p) and ones, all for years with available abundance estimates. The parameter β is the log of the multiplicative bias in the abundance estimates.

The parameters μ, D_T, $b = exp(\beta)$ are assigned independent uniform prior distributions over the respective intervals (0, 0.05), (0, 1.0) and (0, 5/3). This joint prior distribution is then combined with the likelihood of equation (2) to give a posterior distribution for the three internal parameters. This 'posterior' is in turn used to set the catch limit – the term 'posterior' is used although 'pseudo-posterior' would be more correct since the likelihood in equation (2) is really a pseudo-likelihood.

A distinctive feature of the *CLA* is that abundance data are strongly down-weighted to obtain desired catch stability and conservation properties. In the internal model, all variances and covariances of logarithmic abundance estimates are actually multiplied by $\tau^2 = 16$. The posterior density function of the parameters (μ, D_T, b) over the allowed range is therefore proportional to the function shown in equation (2).

The catch control law provides the internal catch limit L_T as the following function of the internal parameters and the current internal abundance P_T.

$$L_T = 0 \; if \; D_T \le IPL, \quad L_T = \gamma\mu\left(D_T - IPL\right)P_T \; if \; D_T > IPL \qquad (3)$$

The internal protection level *IPL* is a further control parameter.

The internal catch limit inherits its posterior distribution from that of (μ, D_T, b). For a chosen probability level α, the catch limit is calculated as the α-quantile of the derived posterior distribution for L. The parameter α has traditionally been used to tune the *CLA* to a desired target population, while the slope parameter γ has been fixed at $\gamma=3$.

In scenarios with deterministic and stationary population dynamics, the posterior distribution for L will asymptote to a degenerate distribution concentrated at one point, provided the information content expands sufficiently. In the so-called 'base case trial' used for tuning the *CLA*, the population dynamics parameters are fixed and the abundance estimates are uncorrelated and have constant coefficients of variation. If n is the number of absolute abundance estimates, there is simply the familiar $n^{-1/2}$ convergence in the joint posterior for (μ, D_T, b), and consequently also in that for L. In the long term, the deterministic population process will converge in probability, and the catch limit will also converge to the replacement yield at the limiting population level.

The size of the catches early in the process will impact the population trajectory, and hence also the later catches and consequently the population level as well. This feedback mechanism has the effect of making the long-term population level depend on the probability parameter α, but only through the size of the early catches. In the limit, the posterior is degenerate, and the value of α has no impact. The limiting depletion is thus only slightly dependent on the traditional tuning parameter. This is further discussed below.

Note that catches are stochastic through the stochasticity of the abundance estimates. This will impact the early part of the process in particular. The stochasticity of the early catches will actually propagate through the feedback mechanism to an element of stochasticity in the limit. This means that the catches and depletion will stabilize over time to a limit, but this limit will vary slightly between simulation runs, even given a specific scenario.

MANAGEMENT HORIZON

The *CLA* has traditionally been tuned to target a specific median depletion after 100 years of managed whaling of an initially unexploited stock (initially this was to allow for the comparison of various candidate *CLAs* e.g. see Kirkwood, 1992), and its performance properties have been investigated by simulating the process over 100 years for a variety of scenarios. It has however been known that 100 years is too little to reach stability when the stock has low productivity; Butterworth and Punt (1994) found for example in the scenario used for tuning that median population level increases from years 100 to 300 of the management period, but then levels off.

Aldrin and Huseby (2007) also found that median population level keep varying, and often increased appreciably, from 100 to 300 years of whaling for a number of scenarios, see for instance the upper left panel of Fig. 1 for the results for the trial used to tune the *CLA* and the upper right panel for another trial. The reader is referred to Aldrin and Huseby (2007) for plots summarising performance in a collection of scenarios.

The simulation period was set somewhat arbitrarily to 100 years in the early phase of developing the *CLA*. Computing capacity was relatively limited before 1990, and a longer simulation period was found too expensive in computer time. Simulation time does actually grow way faster than linear with period length. With current computing power, we consider that a longer simulation period is feasible and suggest 300 years to be a reasonable compromise between the desire to investigate the long-term performance properties of the *CLA* and the need to keep computing costs within reasonable limits.

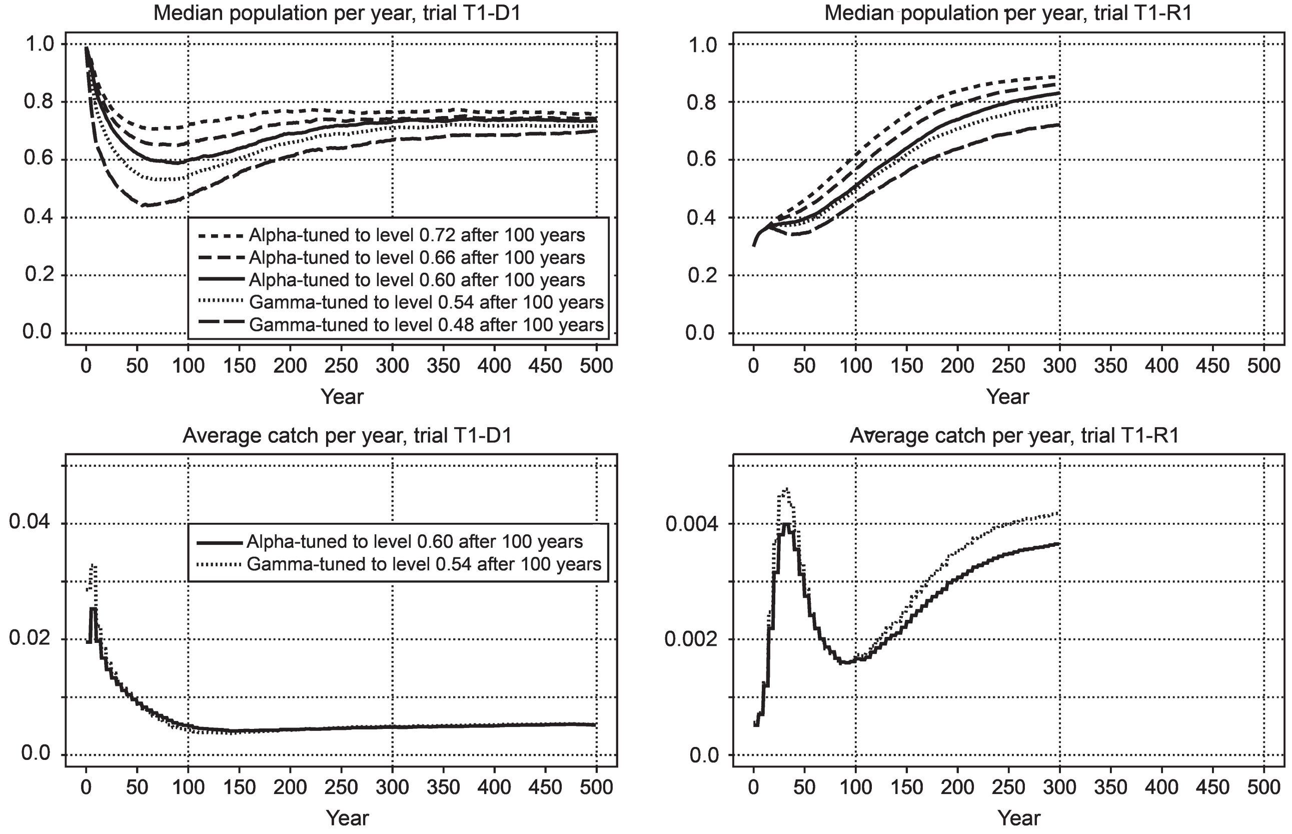

Fig. 1. Median depletion for five different tuning levels and average catch for two tuning levels. Left panels: Trial T1-D1 used for tuning, which commences with an abundance of 99% of the carrying capacity and $MSYR^{mature} = 1\%$. Right panels: Trial T1-R1, which commences with an abundance of 30% of the carrying capacity and $MSYR^{mature} = 1\%$. The results are based on 400 replicate simulations in each case.

Table 1

Median depletion by tuning method and by years of management. The tuning is by α in the three upper rows, and then by γ. 400 replicate simulations were run in each case.

α	γ	Year 100	Year 300	Year 500
0.4015	3	0.72	0.76	0.76
0.4629	3	0.66	0.74	0.74
0.5222	3	0.60	0.73	0.73
0.5	4.7157	0.54	0.71	0.72
0.5	9.3443	0.48	0.67	0.70

TUNING METHOD

From Fig. 1 and Table 1, it is clear that tuning to target depletion in the range of interest (0.66-0.72) after 300 years is difficult when this is attempted by varying α. The long-term population level is actually rather insensitive to the value of α between 0.4 and 0.5. Values of this tuning parameter above 0.5 are inappropriate, since then the catch limit increases with the spread of the posterior distribution for L, i.e. less certainty results in greater catches contrary to the precautionary approach.

There are two other parameters that could be used for tuning, namely the internal protection level *IPL* and the scaling parameter γ, see equation (3). *IPL* is not considered as a viable option as a tuning parameter since it was determined by the Commission that $IPL = 0.54$ (IWC, 1992a).

This leaves γ as the candidate for tuning parameter. This slope parameter simply scales the non-zero catch limits, and should therefore enable tuning to any desired limiting depletion level.

The lower panels of Fig. 1 show the average catch per year for two tuning levels, one tuned by α and one by γ, for the two trials. Table 2 shows other summary statistics for the same trials and tuning levels.

DISCUSSION

As one management objective, the Commission (IWC, 1992a) agreed that whaling ought to be managed on a long term basis. In this note we consider that 100 years of management is too short for the process to come close to a stationary state at the lower end of what is presently assumed to be the range of productivity in baleen whales, and that a management horizon of 300 years is more appropriate. We further suggest an alternative method of tuning the catch limit algorithm to obtain (long term) continuing yield of a size leading to the stock approximating around 70% of carrying capacity in the tuning scenario.

The plausible range of productivity in baleen whales is currently under investigation (IWC, 2009). This note does not contribute to that investigation. Aldrin and Huseby (2007), found that the risk of extinction of a single stock is not seriously increased by exploitation when managed by the *CLA* with α=0.5, γ=4.7157. In the trials considered (with 400 replicate runs in each), they found no cases where extinction occurred over 300 years of management. They found the low 5% quantile for population size after 100 years of management to be at least 24% of K for all trials considered where at least one whale was harvested, reported catches were at least 50% of true catch and survey bias was at most +50%. This happened in a trial where the initial population was 30% of K. In trials with episodic 'events' (i.e. a major depletion of a population in a single year), the

Table 2

Median and 5% percentile depletion at year 100 and 300, and 5% percentile of lowest population size and average catch in years 1-100 and 101-300, for two tuning levels and two trials. Their unit is K (carrying capacity). 400 replicate simulations were run in each case.

			Median depletion		5% percentile depletion		5% percentile of lowest population		Average catch per 100 years	
Trial	α	γ	Year 100	Year 300	Year 100	Year 300	Years 1-100	Years 101-300	Years 1-100	Years 101-300
T1D1	0.5222	3	0.60	0.73	0.46	0.54	0.45	0.41	1.10	0.45
	0.5	4.7157	0.54	0.71	0.39	0.53	0.37	0.35	1.19	0.44
T1R1	0.5222	3	0.51	0.83	0.42	0.73	0.30	0.42	0.23	0.29
	0.5	4.7157	0.49	0.78	0.39	0.67	0.30	0.39	0.24	0.33

population was however driven to near extinction in a few replicate simulations regardless of tuning levels. This happened also in a trial where *K* being doubled over the first 100 years for some levels of tuning. They did in fact run all the trials required at that time (IWC, 2007) to demonstrate whether a candidate procedure is an improvement compared to the current version of the RMP. Many of these trials assumed low productivity, with $MSYR^{mature} = 1\%$. It has been agreed by the IWC Scientific Committee that discussion of Aldrin and Huseby (2007) would await completion on the review of values for MSYR to be considered in the context of the RMP (IWC, 2008a; 2008b). We suggest that the version of the *CLA*, α=0.5, γ=4.7157, should be a candidate for consideration when the plausible range for productivity in baleen whales has been settled.

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REFERENCES

Aldrin, M. and Huseby, R.B. 2007. Simulation trials 2007 for a re-tuned *Catch Limit Algorithm*. 143pp. Paper SC/59/RMP4 presented to the IWC Scientific Committee, May 2007, Anchorage, USA. 143pp. [Paper available from the Office of this Journal].

Butterworth, D.S. and Punt, A.E. 1994. An investigation of the merits or otherwise of the proposal for an Antarctic-wide whale sanctuary by means of adaptations of the simulation trials used to test the Revised Management Procedure. *Rep. int. Whal. Commn* 44: 289-302.

Donovan, G.P. 1989. *Report of the International Whaling Commission (Special Issue 11). The Comprehensive Assessment of Whale Stocks: the early years.* International Whaling Commission, Cambridge, UK. [vi]+210pp.

International Whaling Commission. 1992a. Chairman’s Report of the Forty-Third Meeting. *Rep. int. Whal. Commn* 42:11-50.

International Whaling Commission. 1992b. Report of the Scientific Committee, Annex D. Report of the Sub-Committee on Management Procedures. *Rep. int. Whal. Commn* 42:87-136.

International Whaling Commission. 1999. Report of the Scientific Committee. Annex N. The Revised Management Procedure (RMP) for Baleen Whales. *J. Cetacean Res. Manage. (Suppl.)* 1:251-58.

International Whaling Commission. 2007. Report of the Scientific Committee. Annex D. Report of the Sub-Committee on the Revised Management Procedure. Appendix 5. Report of *Catch Limit Algorithm* (*CLA*) trials group. *J. Cetacean Res. Manage. (Suppl.)* 9:110-13.

International Whaling Commission. 2008a. Report of the Scientific Committee. *J. Cetacean Res. Manage. (Suppl.)* 10:1-74.

International Whaling Commission. 2008b. Report of the Scientific Committee. Annex D. Report of the sub-committee on the revised management procedure. *J. Cetacean Res. Manage. (Suppl.)* 10:90-120.

International Whaling Commission. 2009. Report of the MSYR Workshop, 16-19 November 2007, National Marine Mammal Laboratory, Alaska Fisheries Science Center, Seattle, WA, USA. *J. Cetacean Res. Manage. (Suppl.)* 11:467-80.

Kirkwood, G.P. 1992. Report of the Scientific Committee. Annex I. Background to the development of revised management procedures. *Rep. int. Whal. Commn* 42: 236-43.

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Interyear re-identifications of bowhead whales during their spring migration past Point Barrow, Alaska, 1984-1994

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ABSTRACT

As a part of a review of bowhead whale (*Balaena mysticetus*) stocks, a study was conducted to evaluate how much mixing occurs in the whales' spring migration, a period which immediately follows the mating season. This study has used aerial photography of bowhead whales during their spring migration near Point Barrow, which has resulted in 5,800 images, primarily from 1984 through 1994. These photographs included 40 different whales seen in at least two years, and of these, two were seen in three different years, making for a pair-wise sample size of 42 matches between years. Differences between dates of initial sightings and subsequent sightings (i.e. resightings) ranged from -31 to +23 days comparing month and day only, irrespective of year. These resightings were well dispersed across most of the bowhead spring migration; 98% of the photographs were taken across 45 days from 19 April through 2 June. Models for predicting resighting date from initial sighting date, whale length, presence of a calf, year of initial sighting and year of subsequent sightings were considered, and the best model was chosen using Akaike's Information Criterion (AIC). The best model included most predictors but did not include initial sighting date. Thus, all of the available evidence indicates that individual mature bowheads do not have a consistent migration timing past Barrow; instead, in subsequent years they may appear on almost any date within the normal migratory period. This wide mixing and near-random distribution of resighting dates throughout the spring migration is indicative of a single stock of whales that have a somewhat plastic schedule.

KEYWORDS: BOWHEAD WHALE; ARCTIC; NORTH AMERICA; DISTRIBUTION; MIGRATION; PHOTO-ID

INTRODUCTION

In preparation for the intensive review of bowhead whale (*Balaena mysticetus*) stock structure conducted by the International Whaling Commission Scientific Committee (IWC SC) in 2007, a large research programme was developed that coordinated a variety of studies covering many aspects of bowhead biology. This included: (1) research planning and hypothesis testing; (2) genetics sampling and analysis; (3) animal mixing and abundance; (4) spatial distribution and abundance; and (5) migration patterns (George *et al*., 2007). The focus was on bowhead whales in Alaskan waters, referred to as the Bering-Chukchi-Beaufort (BCB) stock. Some concern had been raised when evidence suggesting multiple stocks (Jorde *et al*., 2007) was found in microsatellite DNA data from BCB bowheads sampled via subsistence hunts during the spring and autumn migration. As a part of the evaluation of stock discreteness, data from aerial photographs of bowhead whales were reviewed. Individual bowhead whales have unique markings, some of which are genetically acquired, and some of which are acquired through trauma such as contact with sea ice or the seafloor. In many cases, markings on dorsal surfaces are distinct enough to be recognised in aerial photographs (Koski *et al*., 1992; Rugh, 1990; Rugh *et al*., 1992a). Data from individually identified bowhead whales have been used in population abundance estimates (da-Silva *et al*., 2000; Rugh, 1990; Schweder, 2003), survival analysis (da-Silva *et al*., 2007; Zeh *et al*., 2002), determination of calving intervals (Miller *et al*., 1992; Rugh *et al*., 1992b) and photogrammetric analyses of whale lengths and growth (Angliss *et al*., 1995; Koski *et al*., 1992; 1993; Koski *et al*., 2006).

This paper examines dates of reidentifications for bowhead whales photographed in different years during their spring migration past Point Barrow, the northernmost tip of Alaska. The spring migration near Barrow has been more thoroughly and systematically surveyed on more years than other seasons or places, thus it serves as a measure of whale migratory timing[1]. In particular, differences in passage dates of the same whales in different years provide a measure of variation in behaviour of individual whales, perhaps as a function of the presence of a calf, sea ice conditions, interactions with predators (including humans) or availability of prey. Variation in migratory dates of individual whales can provide an indication of how much mixing there might be within the stock of bowheads photographed near Barrow. That is, if there is little variation in migratory dates, there is a lowered probability that whales will mix between years, but if each whale migrates on a wide variety of dates, there is an increased probability that there is genetic mixing during the spring migration because March to May is when mating occurs (Koski *et al*., 1993; Nerini *et al*., 1984), dates which overlap or occur only shortly before the spring migration (April to early June) (Moore and Reeves, 1993).

During much of the migration period, bowheads are thought to be moving through the survey area in a continuous manner so that residence time in a given area is

[1] Although many aerial photographic surveys have been conducted prior to and during the fall migration of bowhead whales, these surveys have been across much of the Beaufort Sea and lack the geographic focus that is available near Barrow in the spring migration. Therefore, the migratory timing of individual whales is harder to determine in the fall when sighting dates over a wide range of locations must be considered.

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usually only one day; however, later in the spring, residence time and availability to photograph may be as long as several days, perhaps as a function of feeding opportunities. Consequently, time differences of more than a day through most of the migration period indicate variation in behaviour from year to year. This paper considers the observed pattern of variation and the effect of measured whale length (as a proxy for age) and the presence of a calf on timing and variation in migration. Finally, it examines whether the observed variation is consistent with an assumption that individuals (1) may appear at random within the migrating population or (2) tend to vary more narrowly around each individual's mean migration date.

METHODS

Aerial photographs of bowhead whales have been collected systematically during the spring migration near Point Barrow in many years during the past two decades, particularly from 1984 to 1994. Procedures for collecting these aerial photographs have been described in Rugh (1990), Rugh *et al.* (1992a) and Koski *et al.* (1993). Techniques for categorising images and reidentifying individual whales have been summarised by Rugh (1990) and Rugh *et al.* (1992a; 1998). Following each field season, systematic searches were conducted among the images to find whales photographed more than once; comparisons were then done between years. No equivocal matches are included in the data set (13 potential matches were not included because they were not definitely of the same whale). Each match was confirmed by three different researchers (DJR, WRK and Gary Miller of LGL Ltd). Data used in this study were limited to the area near Point Barrow (between 160°W and 153°W longitude; from the coast north to 72°N; see Fig. 1) during the spring migration (April-June).

Bowhead images obtained near Point Barrow during spring migration were binned relative to the respective 'week' (<23 April, 23-29 April, 30 April-6 May, 7-13 May, 14-20 May, 21-27 May and >27 May) of the migration as given in table 6 of Koski *et al.* (2006). The first and last 'weeks' are more than 7 days long. The dates defining the weeks are based on the temporal distribution of sighting data from the ice-based census of bowhead whales near Point Barrow in the spring (George *et al.*, 2004). Hypotheses related to these weeks as well as hypotheses concerning the differences between dates of initial sightings and subsequent sighting (i.e. resightings) were tested.

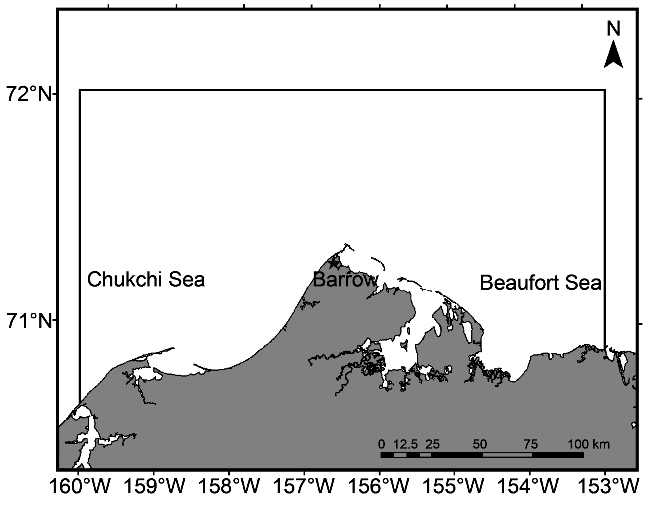

Fig. 1. Map of the sample area where aerial photographs of bowhead whales were taken during the spring migration past Barrow, north of Alaska.

It is well known that the bowhead migration is length structured (Angliss *et al.*, 1995; Koski *et al.*, 2006). Most notably, small whales (except for calves) pass Point Barrow primarily during the first half of the migration period and cows with calves during the latter half. There are no calves among the resighted whales because they lack sufficient marks to be identified in aerial photographs, but some resightings are of adults accompanied by calves in at least one year. It is also known that the timing of the whole migration might be shifted somewhat from one year to the next (e.g. Koski *et al.*, 2006). It is thus natural to examine correlation between initial and subsequent sighting dates by constructing a model for predicting resighting date based on the initial sighting date, years of the sightings, whale length and presence of a calf.

Koski *et al.* (2006) determined that the 1985 migration was delayed by nine days; accordingly, 9 days were subtracted from 1985 dates. Shifts were also estimated for the other years with >6 sightings among the resighted whales (1986, 1989, 1990, 1991 and 1992). To do this, dummy variables were created; for example, Y_{86}=1 if the resighting year was 1986, 0 otherwise and $Y1_{86}$ =1 if the year of the initial sighting was 1986, 0 otherwise. The date variables represent month and day of the sighting date (*Date1* for initial sighting and *Date2* for a subsequent sighting) in days after 31 March. The variable *Mom* is 1 for a cow accompanied by a calf, 0 otherwise. The variable *Length* is the length of the whale in the year resighted, except for the four cases in which this length is missing; in those cases, *Length* is length in the year of the initial sighting.

The resulting full nonlinear model is $Date2 = Constant + Clength \times Length + Cmom \times Mom + \Sigma_y\ Shift_y \times Y_y + Cdate1 \times (Date1 - \Sigma_y\ Shift_y \times Y1_y)$ where the summations are over the applicable years *y*, with *Y* and *Y1* the dummy variables for years of subsequent and initial sightings, respectively. The best model is defined as the one including the subset of $\{Constant, Clength, Cmom, Cdate1, Shift_{86}, Shift_{89}, Shift_{90}, Shift_{91}, Shift_{92}\}$ that minimises Akaike's Information Criterion (AIC) (Venables and Ripley, 1999). All possible subsets of coefficients were considered.

RESULTS AND DISCUSSION

Aerial photography of bowhead whales has resulted in >12,000 images collected between 1976 and 2000. Over 1,330 individual whales have sufficient marks to be considered unique and identifiable using these techniques, and 157 interyear reidentifications have been made of 118 different whales seen in two different years, 19 seen in three years and 2 seen in four years. Around 5,800 of the photographs were taken from 1984 to 1994 near Point Barrow, Alaska, during the spring migration, all from 15 April to 7 June (median=12 May). Among the 5,800 photographs, there were images of 40 whales seen more than once between years, and two of these whales were seen in three different years, making for a total pair-wise sample size of 42 matches between the first year seen and a subsequent year (Table 1). Fig. 2 traces the matches between paired sightings of the respective whales.

Table 1

Bowhead whales resighted during 1984-1994 spring migrations past Point Barrow, Alaska. Whale numbers are as defined in the database. Differences in dates (ΔT) are subsequent sighting dates minus initial sighting dates, irrespective of year.

Whale no.	Sighting 1	Length 1	Sighting 2	Length 2	ΔT
1890	05/08/1984	N/A	04/23/1992	13.37	-15
1921	05/02/1985	10.15	05/05/1990	10.55	3
1937	05/11/1985	13.23	04/27/1992	13.36	-14
2024	05/14/1985	10.37	05/12/1986	N/A	-2
2037	05/17/1985	14.97	05/29/1986	15.17	12
1058	05/18/1985	13.46	05/13/1986	12.88	-5
2200	05/22/1985	16.31	05/26/1991	16.16	4
2217	05/23/1985	14.63	05/10/1991	14.49	-13
2246	05/26/1985	13.39	05/06/1989	14.05	-20
2247	05/26/1985	13.38	05/17/1989	N/A	-9
2291	05/27/1985	13.50	05/18/1989	N/A	-9
2312	05/29/1985	14.59	05/19/1990	13.71	-10
2347	05/31/1985	14.56	05/11/1986	14.67	-20
2371	06/01/1985	15.05	05/26/1992	15.46	-6
2374	06/01/1985	13.88	*05/29/1986	14.29	-3
2384	06/02/1985	12.97	05/15/1989	14.01	-18
2392	06/02/1985	14.45	05/22/1986	13.97	-11
2392	06/02/1985	14.45	#05/18/1989	14.66	-15
2403	06/02/1985	14.34	05/19/1986	13.98	-14
2428	06/06/1985	16.70	05/27/1989	16.01	-10
7946	05/06/1986	12.99	05/03/1989	13.60	-3
3963	05/11/1986	9.80	05/14/1992	11.26	3
4020	05/11/1986	13.33	05/06/1989	13.79	-5
8002	05/11/1986	13.44	05/10/1991	14.17	-1
8015	05/11/1986	13.51	*06/02/1990	13.80	22
8026	05/11/1986	N/A	05/16/1992	13.71	5
8033	05/11/1986	14.60	05/19/1990	14.72	8
8090	05/14/1986	N/A	04/19/1989	12.84	-25
8622	05/19/1986	13.55	*05/26/1989	13.94	7
8135	05/22/1986	13.65	04/21/1989	13.05	-31
8142	05/22/1986	13.78	05/19/1990	13.34	-3
8250	05/04/1987	N/A	05/11/1990	14.56	7
1184	05/07/1987	N/A	04/23/1992	15.17	-14
8288	05/08/1987	N/A	*05/25/1991	16.03	17
8312	05/18/1987	N/A	05/11/1990	13.76	-7
8744	04/20/1989	12.61	05/13/1992	13.56	23
8824	04/25/1989	12.75	05/14/1992	14.57	19
9304	05/31/1989	N/A	05/29/1990	14.18	-2
9304	05/31/1989	N/A	#*05/10/1991	14.95	-21
1880	05/08/1991	13.59	05/13/1992	N/A	5
10573	05/11/1991	14.68	05/26/1992	15.26	15
5149	05/09/1992	13.57	05/25/1994	14.45	16

*Accompanied by a calf. #Third sighting.

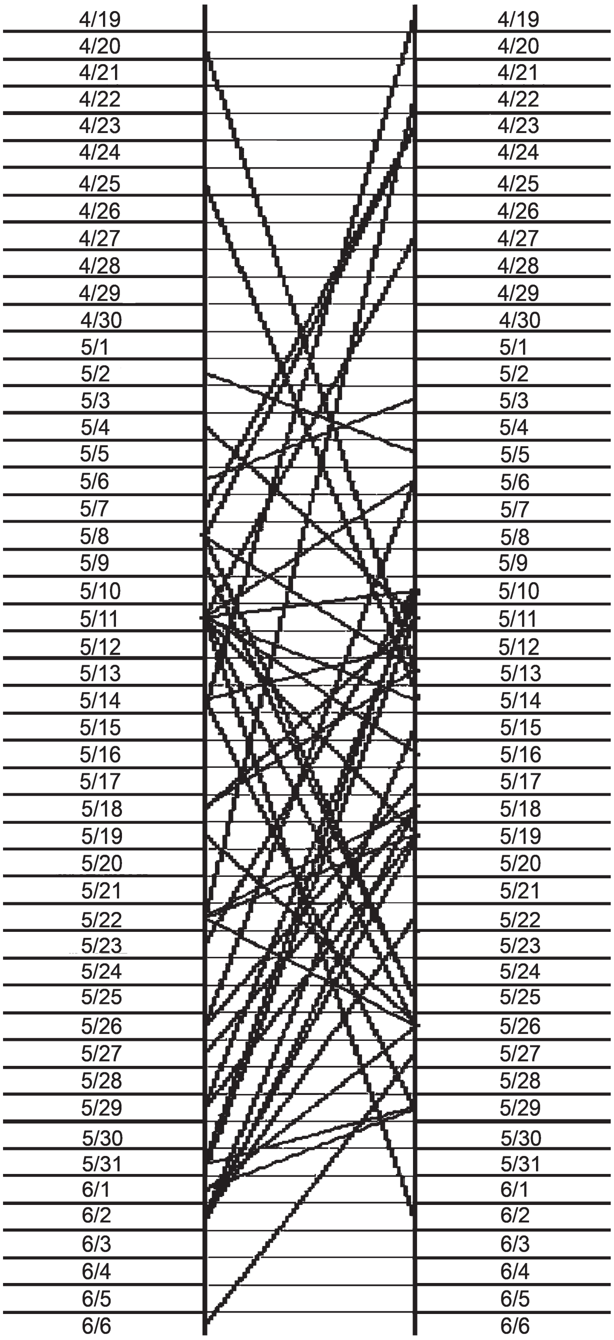

Fig. 2. Differences in dates when individual bowhead whales were photographed migrating past Point Barrow, Alaska, through the spring migration. The lines connect the pair of dates for resightings of each whale. The left column represents the initial sighting date, and the right column represents the resighting date.

Differences between dates of initial sightings and subsequent sightings ranged from -31 to +23 days, comparing dates irrespective of year. The range of the differences did not depend on whether the 1985 dates were shifted by nine days, but the mean was closer to zero (0.7 compared to -3.3), and the standard deviation was somewhat smaller (12.3 compared to 13.1) with the shift. Only three whales were resighted within two days of their original sighting date, but many (52%) were resighted within ten days (Fig. 3). This is not surprising given that more than half of the photographic images (53%) were obtained within a two-week period from 7 to 20 May in a typical year (Table 2).

When lengths were compared relative to absolute differences (|ΔT|) in sighting dates (Fig. 4), it appears that smaller, immature whales (<12m) may be less variable in the date that they pass Point Barrow (|ΔT| = 2-3 days) than larger whales (>12m), which have a wide range in dates (|ΔT| = 1-31 days). Although the sample size of resighted immature whales is very small (*n*=3), if it is representative, it supports the consistent observation that immature whales tend to pass Barrow early in the migration (Angliss *et al*., 1995; Koski *et al*., 2006; Nerini *et al*., 1984; Rugh, 1990; Zeh *et al*., 1993). However, these data suggest that the migration is less structured than previously thought. Since bowhead whales acquire marks over time, young whales have a low probability of being identifiable in aerial photographs. It is likely that there are unrecognised resightings of immature whales among our photographs.

As shown in Table 1, five resightings were of an adult with a calf. These resightings had differences in migration dates that ranged from -21 to +22 days, which is similar in range (-31 to 23 days) for the other 37 resightings of whales without calves. Whether or not an adult was accompanied by a calf did not seem to affect inter-year differences in

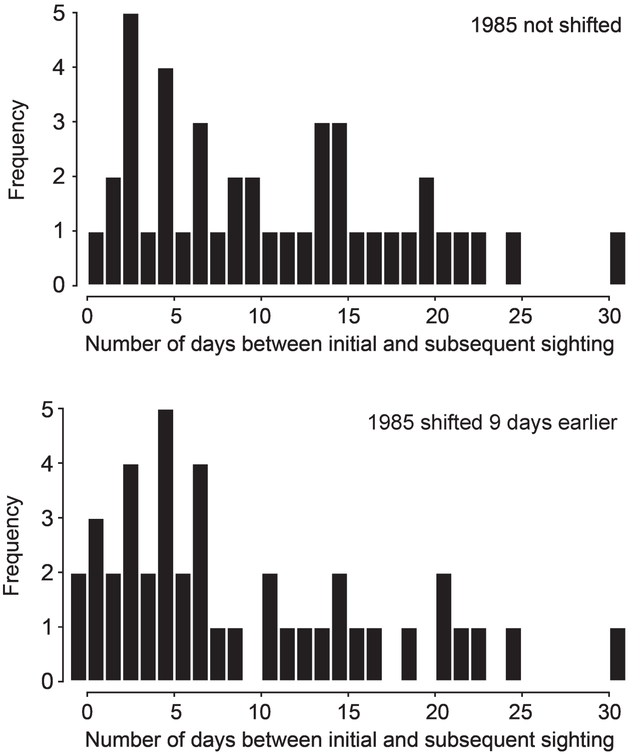

Fig. 3. Histograms showing ΔT in days between the initial sighting of a bowhead whale in the spring migration near Barrow and its resighting in a subsequent year. The top panel shows ΔT with uncorrected dates, and the lower panel shows ΔT based on a −9 day shift of dates in 1985 because that year the migration was late.

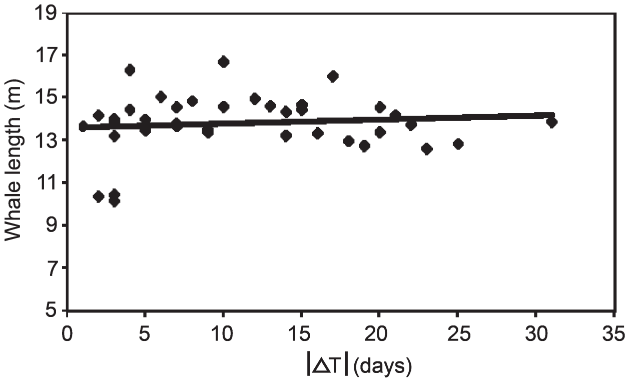

Fig. 4. Bowhead whale length relative to absolute differences (ΔT) in migratory dates determined from aerial photographs taken near Point Barrow. Smaller whales (<11m) appear to be less variable in their migration date (2-3 days) than mature whales (>12m), which range 1-31 days.

migratory timing (*t* test, P=0.29, based on differences corrected for the shift in 1985), in part because all of the bowhead whales had a wide range of date differences.

To test hypotheses regarding the resighting dates in Table 1, week bins were considered (Table 2), showing the number of resighting dates expected in each of these weeks under two different null hypotheses. The first is that resightings are equally likely to occur in any week. The second is that the expected resightings in a week are proportional to the number of photographic images obtained in that week. Not surprisingly, chi-squared tests reject the first of these hypotheses (P=0.043) but not the second (P=0.099) at the 5% level.

Table 2

Proportions of photographic images of bowhead whales by 'week' during the 1984-1994 spring migrations near Point Barrow and expected numbers of resightings under two different null hypotheses. Hypothesis H_{01} is that all weeks are expected to have the same number of resightings. Hypothesis H_{02} is that the expected number of resightings is proportional to the number of photographic images.

		Expected number of resightings	
Weeks	Proportion of images*	H_{01}	H_{02}
15-22 Apr.	0.034	6	1.428
23-29 Apr.	0.129	6	5.418
30 Apr.-6 May	0.119	6	4.998
7-13 May	0.374	6	15.708
14-20 May	0.156	6	6.552
21-27 May	0.150	6	6.300
27 May-7 Jun.	0.038	6	1.596
Total	1.000	42	42.000

*From Table 6 of Koski *et al.* (2006), in which 1985 images are shifted 9d earlier. The H_{02} expected numbers are obtained by multiplying the proportion of images by 42.

The second test is a rather crude test of random resighting dates. A more appropriate test uses a theoretical distribution taking into account that a resighting can only occur in a year subsequent to the year of the initial sighting and only on a day in that year with photographs comparable to those of the resighted whales in terms of quality and identifiability. Fig. 5 shows the empirical distribution function (EDF) of resighting dates and the theoretical distribution under the null hypothesis of random resighting dates. A Kolmogorov-Smirnov test at the 5% level (Birnbaum, 1962) rejects the null hypothesis of random resighting dates. The date with the largest discrepancy between the EDF and the theoretical distribution was 9 May, when young unmarked or marginally marked whales predominate (Angliss *et al.*, 1995). The 9 May discrepancy determined the significance of the test. Since two of the resighted whales were scored as nearly but not completely unmarked in two adjacent body parts in their best photo, such whales were considered to be marked in constructing the theoretical distribution function. Such marginally marked whales are less likely to be reidentified than whales scored as at least moderately marked, which may explain the significant test result.

All of the sightings and resightings reported here occurred during the spring migrations from 1984 to 1994. The timing of most of these migrations was about the same, generally starting around the middle of April and continuing into early June with the peak week 7-13 May (Koski *et al.*, 2006). However, the migration in 1985 was relatively late (Fig. 6), apparently delayed by nine days (as described earlier), even though the aerial survey provided thorough coverage from 23 April to 6 June.

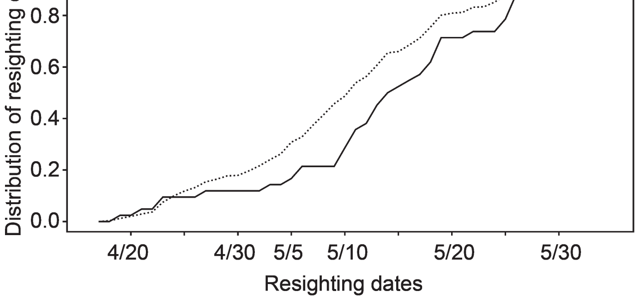

Fig. 5. The empirical distribution function (solid line) and the theoretical distribution function (dotted line) for dates resighted bowhead whales passed Point Barrow during the spring migration.

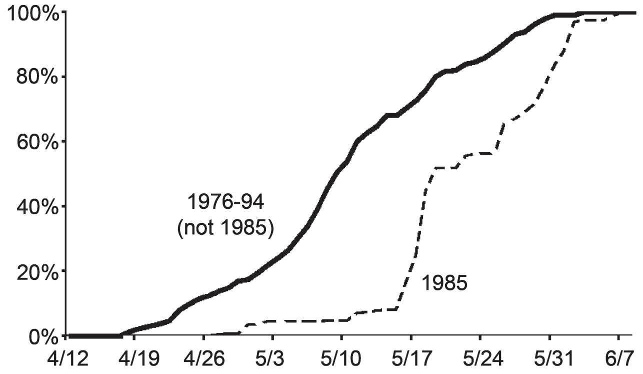

Fig. 6. Cumulative percentile frequencies of bowhead photographs by date showing that the sampled migration in 1985 was significantly later (9 days) than the average of other years.

Comparing dates among the 6 years (1985-86, 1989-92) with sufficient sample sizes (*n*>6) of whales that have been seen in different years, there were significant differences in passage dates (*P*=0.001; ANOVA). However, when 1985 dates were either removed or shifted by nine days, differences were no longer significant (*P*=0.38 for both tests). The mean date for photos of resighted whales in 1985 ($\bar{x}$=25 May; SE=2.1 days) was 11 days later than in 1986-92 ($\bar{x}$=14 May; SE=1.4d). This difference in mean dates is consistent with the shift estimated by Koski *et al.* (2006) from their more comprehensive data set.

A test on the absolute differences $|\Delta T|$ between initial sighting and subsequent sighting dates also provided no evidence for temporal fidelity. In the presence of temporal fidelity, small values of $|\Delta T|$ are expected to be more probable than large ones. The null hypothesis that $\Pr(|\Delta T| \leq 5) \leq 0.5$ was tested against the alternative that this probability is >0.5. Using $|\Delta T|$ uncorrected for the late 1985 migration, the observed proportion of $|\Delta T| \leq 5$ is 0.31 (*P*=0.996, exact binomial test). Using the corrected $|\Delta T|$, the observed proportion is 0.43 (*P*=0.86). Thus, there is no evidence that values of $|\Delta T| \leq 5$ are more probable than larger values.

A problem with a test like the one just described is that the choice of ± 5 days for defining temporal fidelity is arbitrary, and the test does not account for such factors as whale length, presence of a calf and possible less dramatic shifts in migration timing than that observed in 1985. Therefore, a model predicting resighting date from initial sighting date and other relevant factors is a better approach for examining the correlation between initial and subsequent sighting dates. To examine this, the first step was to subtract nine days from all initial sighting dates in 1985; there were no resightings in 1985, subsequently all possible subsets of the potential predictor variables described in our Methods section were considered.

The best single predictor was *Length*, and the best pair of predictors was *Length* and *Mom*; *Length* was present in the best model of each size. The *Constant* term appeared in only the full (9-parameter) model and the best 7-parameter model; it was omitted in the best 8-parameter model and all models with 6 or fewer parameters. Shift parameters for 1986 and 1990 appeared in the best 4-parameter and 3-parameter models, with and without *Mom*, respectively, but they did not appear in the best 7-parameter model. When they appeared, both were positive, suggesting that these years had somewhat delayed migrations, although not as delayed as 1985, relative to the remaining years considered (1989, 1991 and 1992).

The best model was the 5-parameter model $Date2 = Clength \times Length + Cmom \times Mom + \Sigma\, Shift \times Y$, where the summation is over the years 1989, 1991 and 1992. Thus, the best model, like all the models with fewer parameters, does not include *Date1* as a predictor. Although all the models with more parameters included *Date1*, the coefficient *Cdate1* was never statistically significant. The coefficients of the best model are given in Table 3. They suggest that larger whales arrive later than smaller ones, cow-calf pairs arrive late in the migration and the migrations in 1989, 1991 and 1992 were early compared to 1985, 1986 and 1990. The large negative shift coefficient for 1991 was no doubt influenced by the large whale seen with a calf in 1991 on 10 May, earlier than cow-calf pairs are usually seen. Fig. 7 shows $Date2 - \Sigma_y\, Shift_y \times Y_y$ from the best model plotted vs *Length*; cow-calf pairs are shown in black. The residuals from this model had a mean of -0.05 and a standard deviation of 8.55, considerably less than the standard deviation of the differences between resighting and initial sighting dates with 1985 shifted by nine days. Accordingly, this model, which does not incorporate initial sighting date, is considerably more precise than assuming that initial and resighting dates should be similar.

Table 3

Coefficients in the best model for predicting resighting dates, chosen using Akaike's Information Criterion.

	coef	std.err	t.stat	P.value
Clength	3.5213	0.1690	20.8305	0.0000
Cmom	9.4073	4.5933	2.0480	0.0477
Shift89	-9.2327	3.5414	-2.6071	0.0131
Shift91	-10.9451	4.8683	-2.2482	0.0306
Shift92	-8.5546	3.6931	-2.3164	0.0262

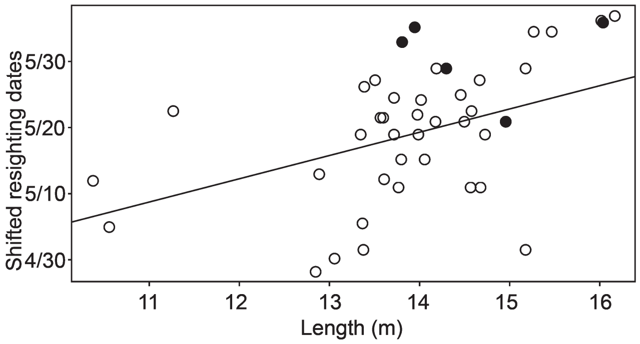

Fig. 7. Resighting dates with resightings in 1989, 1991 and 1992 shifted to reflect the Shift coefficients shown in Table 3. These shifted resighting dates are plotted against whale length, the strongest predictor of resighting date. The line is determined by the Clength coefficient in Table 3. Adults accompanied by a calf are the solid points.

Aerial photography of bowhead whales in the Point Barrow area has occurred as early as 15 April and as late as 7 June, covering much of the spring migration (Table 2). These dates spread across a 54 day period. However, 98% of the photographs have been taken between 19 April and 2 June, a range of 45 days. Some bowhead whales have been photographed as much as 31 days apart in different years. This wide mixing in dates is demonstrated in Fig. 2 (treated here as our null hypothesis with no significant difference from a random distribution). The alternate (failed) hypothesis is that bowhead whales do not significantly change travel dates between migrations, which would mean interyear resightings would be only a few days apart. Instead, the wide mixing and near-random distribution of resighting dates of larger whales throughout the spring migration is indicative of a single stock of whales that have a somewhat plastic schedule[2].

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[2] Smaller whales (<12m) might migrate past Barrow in a tighter timeframe than larger (>12m) whales, but we are limited by a small sample size (n=3).

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The National Marine Mammal Laboratory (NMML) and LGL, Ltd. (LGL) are the organisations that have led in the bowhead aerial photography projects. Many researchers have been involved at many different levels. We appreciate the dedication and help from co-principal investigators for field projects, including Mary Nerini (NMML), David Withrow (NMML), W. John Richardson (LGL) and Rolph Davis (LGL). Photographic assistance in different years was provided by Robyn Angliss, Laurie Briggs, Dan Cheng, Rich Ferrero, Camille Goebel, Merrill Gosho, Bryan Herczeg, Wayne Perryman, Kim Shelden, Kathleen Strickland and Larry Tsunoda of NMML and Gary Miller and Martie Crone of LGL. Gary Miller also conducted most of the interyear matching of images. Discussions with Dinara Sadykova and Tore Schweder helped guide some of the approaches used here. Document reviews were provided by Robyn Angliss and Rod Hobbs of NMML and by W. John Richardson of LGL, Robert Suydam of Alaska's North Slope Borough and Peter Best of the Mammal Research Institute, Pretoria, South Africa. NMML operated under Permit Number 580 during the survey period reported here, and LGL operated under Permit Numbers 517, 518, 670 and 481-1464, all issued under the provisions of the US Marine Mammal Protection Act and Endangered Species Act.

REFERENCES

Angliss, R.P., Rugh, D.J., Withrow, D.E. and Hobbs, R.C. 1995. Evaluations of aerial photogrammetric length measurements of the Bering-Chukchi-Beaufort Seas stock of bowhead whales (*Balaena mysticetus*). *Rep. int. Whal. Commn* 45: 313-24.

Birnbaum, Z.W. 1962. *Introduction to Probability and Mathematical Statistics*. Harper and Brothers Publishers, New York. 325pp.

da-Silva, C.Q., Gomes, P.V.S. and Straditoto, M.A. 2007. Bayesian estimation of survival and capture probabilities using logit link and photoidentification data. *Computational Stat. and Data Analysis* 51: 6521-34.

da-Silva, C.Q., Zeh, J., Madigan, D., Laake, J., Rugh, D., Baraff, L., Koski, W. and Miller, G. 2000. Capture-recapture estimation of bowhead whale population size using photo-identification data. *J. Cetacean Res. Manage.* 2(1): 45-61.

George, J.C., Moore, S.E. and Suydam, R.S. 2007. Summary of stock structure research on the Bering-Chukchi-Beaufort Seas stock of bowhead whales: 2003-2007. Paper SC/59/BRG3 presented to the IWC Scientific Committee, May 2007, Anchorage, USA (unpublished). 15pp. [Paper available at the Office of this Journal].

George, J.C., Zeh, J., Suydam, R. and Clark, C. 2004. Abundance and population trend (1978-2001) of western Arctic bowhead whales surveyed near Barrow, Alaska. *Mar. Mammal Sci*. 20(4): 755-73.

Jorde, P.E., Schweder, T., Bickham, J.W., Givens, G.H., Suydam, R. and Stenseth, N.C. 2007. Detecting genetic structure in migrating bowhead whales off the coast of Barrow, Alaska. *Mol. Ecol*. 16(10): 1993-2004.

Koski, W.R., Davis, R.A., Miller, G.W. and Withrow, D.E. 1992. Growth rates of bowhead whales as determined from low-level aerial photogrammetry. *Rep. int. Whal. Commn* 42: 491-99.

Koski, W.R., Davis, R.A., Miller, G.W. and Withrow, D.E. 1993. Reproduction. pp.239-74. *In*: Burns, J.J., Montague, J.J. and Cowles, C.J. (eds). *The Bowhead Whale*. Special Publication No.2. The Society for Marine Mammalogy, Lawrence, Kansas. 787pp.

Koski, W.R., Rugh, D.J., Punt, A.E. and Zeh, J. 2006. An approach to minimise bias in estimation of the length-frequency distribution of bowhead whales (*Balaena mysticetus*) from aerial photogrammetric data. *J. Cetacean Res. Manage.* 8(1): 45-54.

Miller, G.W., Davis, R.A., Koski, W.R., Crone, M.J., Rugh, D.J., Withrow, D.E. and Fraker, M.A. 1992. Calving intervals of bowhead whales – an analysis of photographic data. *Rep. int. Whal. Commn* 42: 501-06.

Moore, S.E. and Reeves, R.R. 1993. Distribution and movement. pp.313-86. *In*: Burns, J.J., Montague, J.J. and Cowles, C.J. (eds). *The Bowhead Whale*. Special Publication No. 2. Society for Marine Mammalogy, Lawrence, KS. 787pp.

Nerini, M.K., Braham, H.W., Marquette, W.M. and Rugh, D.J. 1984. Life history of the bowhead whale, *Balaena mysticetus,* (Mammalia, Cetacea). *J. Zool., London*. 204: 443-68.

Rugh, D.J. 1990. Bowhead whales reidentified through aerial photography near Point Barrow, Alaska. *Rep. int. Whal. Commn (special issue)* 12: 289-94.

Rugh, D.J., Braham, H.W. and Miller, G.W. 1992a. Methods for photographic identification of bowhead whales, *Balaena mysticetus*. *Can. J. Zool*. 70: 617-24.

Rugh, D.J., Miller, G.W., Withrow, D.E. and Koski, W.R. 1992b. Calving intervals of bowhead whales established through photographic identifications. *J. Mammal*. 73(3): 487-90.

Rugh, D.J., Zeh, J.E., Koski, W.R., Baraff, L.S., Miller, G.W. and Shelden, K.E.W. 1998. An improved system for scoring photo quality and whale identifiability in aerial photographs of bowhead whales. *Rep. int. Whal. Commn* 48: 501-12.

Schweder, T. 2003. Abundance estimation from multiple photo surveys: confidence distributions and reduced likelihoods for bowhead whales off Alaska. *Biometrics* 59: 974-83.

Venables, W.N. and Ripley, B.D. 1999. *Modern Applied Statistics with S-Plus*. 3rd ed. Springer-Verlag Inc., New York. xi+501pp.

Zeh, J., Poole, D., Miller, G., Koski, W., Baraff, L. and Rugh, D. 2002. Survival of bowhead whales, *Balaena mysticetus*, estimated from 1981-98 photo-identification data. *Biometrics* 58: 832-40.

Zeh, J.E., Clark, C.W., George, J.C., Withrow, D., Carroll, G.M. and Koski, W.R. 1993. Current population size and dynamics. pp.409-89. *In*: Burns, J.J., Montague, J.J. and Cowles, C.J. (eds). *The Bowhead Whale*. Special Publication No.2. The Society for Marine Mammalogy, Lawrence, Kansas. 787pp.

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Trend in aerial counts of beluga[1] or white whales (*Delphinapterus leucas*) in Bristol Bay, Alaska, 1993-2005

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ABSTRACT

Thirty-eight aerial surveys of beluga or white whales (*Delphinapterus leucas*) were conducted in Bristol Bay, Alaska, during six different years between 1993 and 2005. Belugas were sighted mainly close to shore in the upper parts of Nushagak and Kvichak bays, as well as along the coast between these bays and in the lower parts of major rivers. Data from 28 complete counts made in good or excellent survey conditions were analysed for trend. Counts ranged from 264 to 1,067. The estimated rate of increase over the 12-year period was 4.8%/year (95% CI = 2.1%-7.5%). Such a rate of increase suggests that either the population was below the environmental carrying capacity in the early 1990s or, alternatively, that factors that had been limiting population increase were alleviated after that time. A review of possible changes in human-caused mortality, predation and prey availability did not reveal a single likely cause of the increase. Among the factors that could have played a role are recovery from research kills in the 1960s, a modest decline in subsistence removals and a delayed response to increases in Pacific salmon (*Oncorhynchus* spp.) abundance in the 1980s. The positive growth rate for this population shows that in recent years there has been no substantial negative impact of human or natural factors, acting either alone or in combination, and there is no need for changes to the current management regime.

KEYWORDS: BELUGA WHALE; WHITE WHALE; INDEX OF ABUNDANCE; MONITORING; SURVEY-AERIAL; TRENDS

INTRODUCTION

Beluga whales (*Delphinapterus leucas*) are common in coastal and offshore waters of western Alaska. During summer months they congregate predictably at certain coastal locations; that distribution pattern was used to identify provisional management stocks (Frost and Lowry, 1990). Subsequent studies of mitochondrial DNA confirmed that three beluga stocks occur in waters off western Alaska during summer, the Bristol Bay stock, the eastern Bering Sea stock and the eastern Chukchi Sea stock (O'Corry-Crowe *et al*., 1997). Those stocks are considered to be separate management units (Angliss and Outlaw, 2007).

The Bristol Bay region (Fig. 1) supports an abundance of fish and wildlife, and beluga whales are present in this area throughout the year. They are seen most commonly in Kvichak and Nushagak bays, especially in the months from April to August (Chythlook and Coiley, 1994; Frost and Lowry, 1990; Frost *et al*., 1984). Kvichak and Nushagak bays also support large runs of anadromous fishes, especially red salmon (*Oncorhynchus nerka*), which are a major prey item of the belugas in this region (Brooks, 1955; Frost *et al*., 1984). Scientific studies conducted in the region in the 1950s and in 1982-1983 provided information on beluga distribution and abundance, movements and diving patterns, food habits and entanglements in fishing nets (Brooks, 1955; Frost *et al*., 1984; Frost *et al*., 1985; Lensink, 1961).

The Alaska Beluga Whale Committee (ABWC) was formed in 1988 to conserve beluga whales and manage beluga subsistence hunting in western Alaska in cooperation with the National Oceanic and Atmospheric Administration (Adams *et al*., 1993; Fernandez-Gimenez *et al*., 2006). Aerial surveys to estimate the abundance and trends of western Alaska beluga stocks have been a part of the ABWC research program since the early 1990s. This paper presents the results of ABWC surveys flown in Bristol Bay during 1993-2005, and estimates the trend in counts of belugas over that period.

METHODS

Aerial surveys

Aerial surveys were flown using a high-wing, twin-engine *AeroCommander* with oversized bubble windows. The survey crew included the pilot, a data recorder and two observers (from a group of three individuals) seated behind the pilot on the left and right sides of the aircraft. The survey was designed to cover all of Kvichak and Nushagak bays (Fig. 1), which includes the region where essentially all reported June-July sightings of belugas in Bristol Bay have occurred (see fig. 3 in Frost and Lowry, 1990). The standard survey track followed the entire coast of both bays 0.9km off shore, including the lower parts of major rivers. In the wider portions of the bays we also flew east-west transects at 1.8km intervals to cover the entire area. On those lines, observers counted whales in a strip 0.9km wide on each side. Strip widths were measured by inclinometers and angles were marked on the aircraft windows with grease pencils. Survey altitude was 305m except that when surveying rivers it was 91m. Airspeed was maintained at approximately 222km hr^{-1} during all surveys. Years in which surveys were conducted were determined by availability of funding and needs to conduct other activities of the ABWC. For each year the objective was to complete five replicate surveys of the entire area known to be used by belugas.

[1] The common name agreed for *Delphinapterus leucas* by the Scientific Committee of the International Whaling Commission is the white whale. However, 'beluga' is commonly used in several parts of the world, including Alaska, and is used in this paper.

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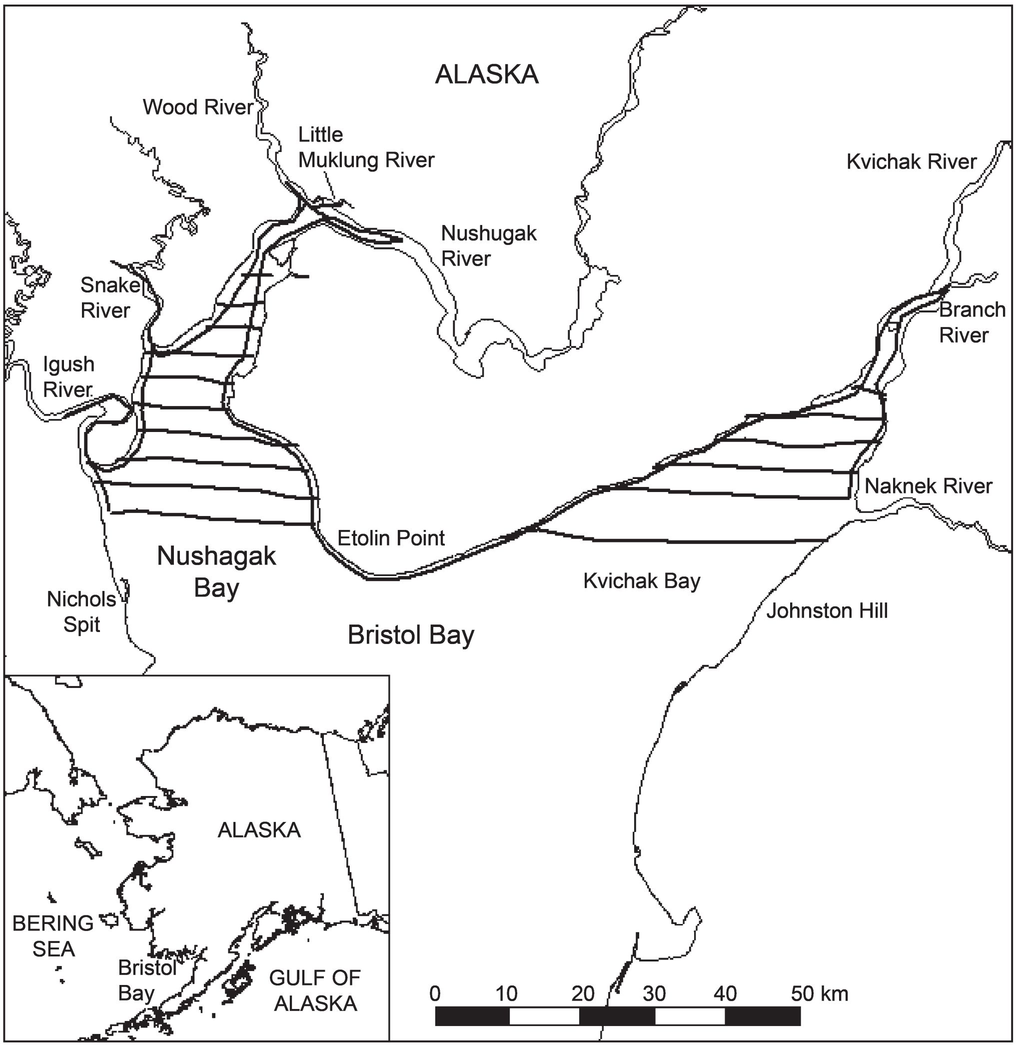

Fig. 1. Map of the Bristol Bay study area showing representative survey flight lines (heavy dark lines).

All belugas visible at the surface along the survey track were counted and counts were recorded either on datasheets by the observers or on a computer by the recorder. When large groups were encountered two or more counts were made. In those situations the aircraft circled after passing by the group and flew past again on a line oriented to provide one observer the clearest view of the entire group. Multiple counts, usually by both observers, were recorded individually, and observers identified which count was best (e.g. minimum glare and no whales in the blind area directly under the plane) and that single count was used in analysis of the data.

A computerised data logging system recorded the time and position determined by the global positioning system at the beginning and end of each transect, at 60 second intervals along the transect, and at every beluga sighting. Beluga counts, weather, sea state, glare, overall sighting conditions and other potentially relevant observations were also entered into the database.

Data analysis

As soon as possible after the survey, computer records were checked for accuracy and edited as necessary. Later all data were entered into a geographic information system (GIS; ArcInfo and ArcView). All sightings and survey tracklines were plotted in the GIS, and the results were examined to identify any possible duplicate sightings which, when found, were removed from the database. Duplication occurred only when cross-bay lines intersected coastal transects, and duplicate sightings were identified based on location, timing and group size.

Sea state (DeMaster *et al*., 2000) and glare (pers. obs.) can have a great effect on counts of belugas. Those and other environmental factors (e.g. low clouds or fog) were integrated in the parameter ‘sighting conditions’, which was recorded as ‘excellent’, ‘good’, ‘fair’ or ‘poor.’ The GIS was used to examine the relationship between sighting conditions on individual surveys and beluga counts. This examination revealed that counts were much lower when conditions on the shoreward-looking side of coastal transects were recorded as fair or poor – within years, the means of counts made under fair or poor conditions were 48%-79% lower than those made under good or excellent conditions. Therefore, it was decided to include in the analysis only those surveys for which the entire study area was surveyed with good or excellent sighting conditions on the shoreward side of alongshore transects (which is where nearly all sightings occurred, see below).

The rate of increase for the period 1993-2005 was estimated by fitting an exponential model to the individual counts assuming a normal error distribution. In this model, the instantaneous rate of increase (r) in the population (N) is constant over time (t): $N_t = N_0 * e^{rt}$. The rate of increase can be estimated in a linear regression framework ($\ln[Nt] = \ln[N_0]+rt$, where r is the slope of the regression). Confidence intervals of the estimated rate of increase were calculated as: 95% $\text{CI}=r \pm t_{0.025,\ df} * \text{SE}(r)$. The instantaneous rate (r) was converted into an annual rate of increase as e^r-1.

RESULTS

Thirty-eight aerial surveys were conducted during 1993-2005. Belugas were sighted mainly close to shore in the upper parts of Nushagak and Kvichak bays, as well as along the coast between these bays and in the lower parts of major rivers (Fig. 2).

Three of the 38 surveys were incomplete due to weather that precluded counting in parts of the study area, and 7 failed to meet the criterion of good or excellent sighting conditions. The dataset used to analyse for trend therefore included 28 counts, with 3 to 6 counts in each year (Table 1). The number of belugas counted per survey ranged from 264 to 1,067. The annual counts showed an increase over time, with the 2005 average count being 76% greater than the average count in 1993.

The estimated rate of increase from the linear regression model for the period 1993-2005 was 4.8%/year (95% CI=2.1%-7.5%). The fit of the model through the count data is illustrated in Fig. 3.

Table 1

Aerial survey counts of beluga whales in Bristol Bay, Alaska, 1993-2005.

Survey date	Flight	Total count	Mean count	CV
29/06/93	1	311		
30/06/93	1	415		
03/07/93	1	269		
17/07/93	1	452		
18/07/93	1	362		
1993 combined			**362**	**0.21**
06/07/94	1	473		
07/07/94	1	391		
07/07/94	2	264		
1994 combined			**376**	**0.28**
14/07/99	1	287		
15/07/99	1	441		
15/07/99	2	454		
16/07/99	1	349		
17/07/99	1	690		
1999 combined			**444**	**0.35**
10/07/00	1	284		
11/07/00	1	531		
11/07/00	2	432		
12/07/00	1	496		
12/07/00	2	361		
2000 combined			**421**	**0.24**
06/07/04	1	362		
08/07/04	1	674		
08/07/04	2	674		
09/07/04	1	556		
09/07/04	2	779		
10/07/04	1	794		
2004 combined			**609**	**0.16**
13/07/05	1	393		
13/07/05	2	480		
14/07/05	1	1,067		
14/07/05	2	607		
2005 combined			**637**	**0.47**

DISCUSSION

Distribution of belugas within Bristol Bay

Mitochondrial DNA analyses indicate that the belugas inhabiting Bristol Bay are demographically distinct from other belugas in western Alaska (O'Corry-Crowe *et al*., 1997), and they are considered by the National Marine Fisheries Service (NMFS) to be a management stock separate from other western Alaska beluga stocks (Angliss and Outlaw, 2007). The total range occupied by this population throughout the year is not well described, but compilations of sightings (Frost and Lowry, 1990), recent surveys (this study) and satellite-linked telemetry (L. Quakenbush, pers. comm.) indicate that essentially all the animals are in the Kvichak Bay-Nushagak Bay region during the months of June and July.

Within Kvichak and Nushagak bays several concentration areas have been used consistently during the summer since at least the mid-1950s (Brooks, 1955; Frost and Lowry, 1990; Frost *et al*., 1984) and this study. In fact, the specific sites (e.g. certain portions of rivers) occupied by concentrations of belugas in June and July were essentially identical over the 24-year period in which we (LFL and KJF) have surveyed the area. Belugas occurred mostly very near shore in the upper portions of Kvichak and Nushagak bays and along the intervening coastline. The vast majority of sightings made during this study were within 0.9km of the shoreline. Although the sightings shown in Fig. 2 suggest a fairly broad onshore-offshore distribution, this actually reflects the large tidal range and gently sloping topography which, in combination, cause the location of the shoreline in some places to vary by three kilometres or more during a tidal cycle. Their predictable distribution pattern, which is apparently stable over time, makes beluga whales in Bristol Bay relatively easy to locate and count.

Trend in counts

With an annual rate of increase of 4.8% per year, we estimate that the abundance of Bristol Bay belugas increased by 65% over the 12-year period. This result is quite similar to the 76% increase in the mean of counts over this same period. Our results are consistent with the observations of long-time residents and Alaska Native beluga hunters who report that more belugas are present in Kvichak and Nushagak bays now than there were 10-20 years ago.

Prior to this study there had been no rigorous effort to estimate the number of belugas in Bristol Bay. Brooks (1955) studied belugas in Kvichak and Nushagak bays, and estimated their abundance as 1,000 in 1954 and 525 in 1955 '…based on surface observations, aerial observations, and fishermen and pilot reports'. Lensink (1961) continued the work of Brooks in the late 1950s and concluded that '…accurate counts are impossible in the turbid waters of this area, but the population probably numbers between 1,000 and 1,500 animals'. It is not possible to compare these earlier estimates to one another or to our recent counts, therefore we cannot say how the population size may have changed since the 1950s.

Frost *et al*. (1984) studied belugas in Kvichak and Nushagak bays in 1982-1983, and one aerial survey of the entire study area in good sighting conditions on 29 June 1983 resulted in a count of 334 belugas. The 1983 count was only slightly lower than the mean counts for 1993 and 1994, suggesting that there was little population growth over that 11-12 year period. It appears therefore that the period of rapid growth of this population probably began in the early 1990s.

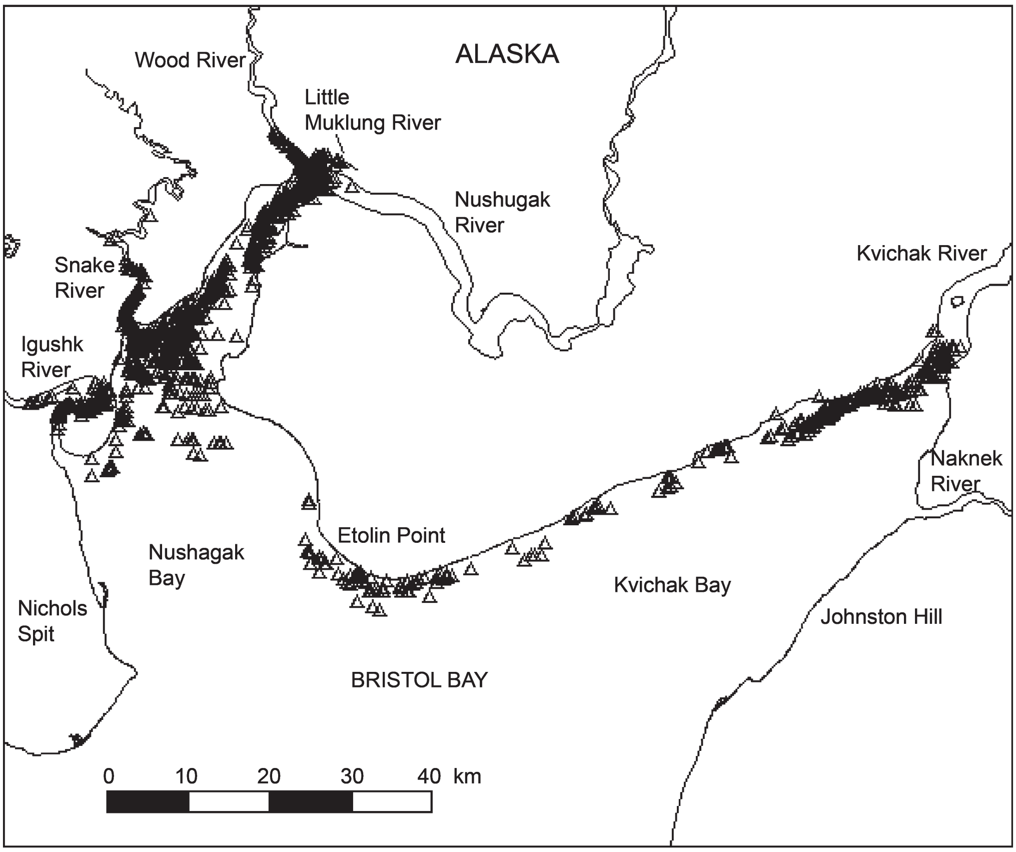

Fig. 2. Map of Bristol Bay showing all sightings of beluga whales made during aerial surveys in 1993-2005 (triangles).

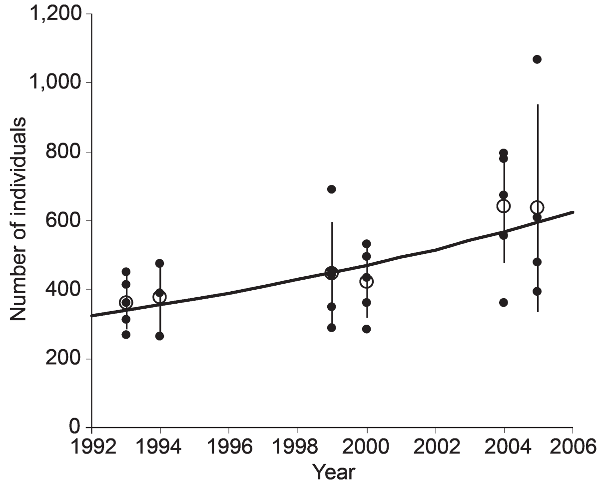

Fig. 3. Fit of the exponential model to Bristol Bay beluga count data (black dots=individual counts, white circles=mean counts, vertical bars=standard deviations, heavy black line=predicted counts).

Within year counts made in good and excellent sighting conditions in 1993-2005 were reasonably consistent and the coefficients of variation associated with those counts were relatively low. However, more whales were counted on some surveys than on others. Possible explanations include: (1) on some surveys some groups of belugas may not have been counted because they were absent from the area surveyed; (2) some whales at the surface within the survey area may have been missed because of sighting conditions or some other factor; and (3) belugas may have behaved differently at different times (e.g. spent more or less time at the surface). As discussed above, all available information indicates that Bristol Bay belugas are restricted to the surveyed area during June-July so the first possible explanation is unlikely. Undoubtedly some animals are missed during any given survey but no attempt was made to quantify that bias since the main objective was to estimate population trend. Missed animals should not bias our estimate of trend provided that the bias is consistent over time. Several measures were taken to ensure consistent bias, including eliminating surveys flown in fair or poor sighting conditions and using a limited pool of experienced observers and identical techniques in all the surveys. The third factor, variation in beluga behaviour, probably explains most of the variability in counts for two reasons. First, data collected on surfacing patterns of belugas in Bristol Bay show considerable variation, including long periods when whales rest or feed in water so shallow that they are at the surface essentially all the time (Frost *et al*., 1985). Second, substantial differences have been seen in counts of specific whale groups over relatively short periods of time. The best example of this was on 14 July 2005, the day when the highest count of the entire series was obtained. On that day a large group of whales was located in western Kvichak Bay at about 13:30 hours, with a best count of 638 animals. Approximately 2.5 hours later, during the second survey of the day, another count was made of the same group in the same area, and the best count that time was 163, just 25% of the number seen earlier on the same day. Sighting conditions were excellent during both counts, and when the second count was so much lower it was decided to fly additional lines covering the entire adjacent area but no more whales were found. This within-day difference was probably due entirely to behaviour, with the higher count occurring when essentially all animals were at the surface and the lower one when many of them were diving. Dive data from five satellite-tagged belugas in Bristol Bay corroborate that there is considerable variation in surfacing behaviour (J. Citta, pers. comm.).

Why has the Bristol Bay beluga population increased?
Other than the counts and estimates described above, there are no historical data on the number of belugas in Bristol Bay. However, the apparently steady increase in the population over the 12 years of our study indicates that it was below the environmental carrying capacity (K) during that period. For this to be true, either something was acting previously to keep the population from growing toward K, something changed causing K to increase, or both.

Human activities have caused both intentional and unintentional removals of belugas from this population. During 1954-1966, 127 belugas were killed for research (Brooks, 1957; Vania and Klinkhart, 1967) and 8 were live-captured for oceanaria (Lensink, 1958; Ray, 1962; Reeves and Leatherwood, 1984). Belugas are hunted by Alaska Natives for subsistence, and since 1987 the ABWC and the Bristol Bay Marine Mammal Council have compiled beluga harvest data for Bristol Bay. The reported average annual landed catch during 1987-2006 was 20 belugas. Average catch was highest for the 5-yr period 1987-1991 (mean=25, range=13-36) and somewhat lower during the subsequent three 5-yr periods (1992-1996 mean=20, range=6-35; 1997-2001 mean=17, range=6-31; 2002-2006 mean=20, range=13-23) (Frost and Suydam, in prep.). These figures do not include the number of whales that were struck but not retrieved, but that number is believed to be 'quite low' in this area (ABWC, pers. comm.).

Belugas also have been taken incidentally in gillnet fisheries for salmon but data on the numbers killed in Bristol Bay are incomplete. Frost *et al.* (1984) found 27 dead belugas during their studies in May-July 1983, at least 12 of which had died in nets. The Bristol Bay salmon gillnet fisheries have never been monitored by observers, but during the period between 1990 and 2000 fishermen reported one beluga death in 1990, one in 1991 and none thereafter (Angliss and Outlaw, 2005). It is uncertain whether the number killed in nets has declined since 1983; in general, self-reported data on incidental takes in fisheries are negatively biased (Credle *et al.*, 1994). There have been no major changes in fishing effort or methods in the Bristol Bay salmon fishery over the past 25 years that would be expected to result in a decline in the incidental take, although the total number of actively fished permits has declined by about 10% (Westing *et al.*, 2006). It is likely that some belugas die each year as a result of interactions with the gillnet fishery.

Killer whales (*Orcinus orca*) are natural predators of belugas. Frost *et al.* (1992) documented a number of killer whale sightings in Bristol Bay in 1989-1990, including instances where they chased and killed belugas. Those authors considered such events to be very unusual based on historical observations of biologists and local residents. However, Bristol Bay area residents report that killer whales have been seen quite frequently in the Nushagak side of Bristol Bay in recent years, and that they affect the distribution of belugas when they are present (Molly Chythlook, pers. comm.). Since there have been no directed studies of killer whales in this area it is impossible to assess whether or not they have had an influence on abundance of Bristol Bay belugas.

It is possible that the environmental carrying capacity for Bristol Bay belugas has changed due to an increase in availability of food for them. The prey species of belugas in Bristol Bay during the late spring and summer are relatively well known (Brooks, 1955; Frost *et al*., 1984; Lensink, 1961). In May and early June they feed mostly on smelt (*Osmerus mordax*) and red salmon smolt migrating out of the rivers. From mid-June through mid-August they feed primarily on salmon, with red salmon dominant but other species (chum, *Oncorhynchus keta*; pink, *O. gorbuscha*; and silver *O. kisutch*) becoming more important later in the season. From mid-August through September salmon are seldom eaten and prey items found in beluga stomachs have included flatfishes, sculpins, lampreys (*Lampetra japonica*) and shrimps (*Crangon* spp.). There are no published data on diet during the autumn and winter months.

There are no programmes for assessing or monitoring the abundance of potential beluga prey other than salmon, but there are extensive data for salmon (Fig. 4). Red salmon returns to Bristol Bay streams are strongly cyclical, with peaks in abundance about every five years and smaller runs in intervening years. From the late 1950s through the 1970s, Bristol Bay run cycles were regular and quite consistent, with an average run size of about 18 million (Hare and

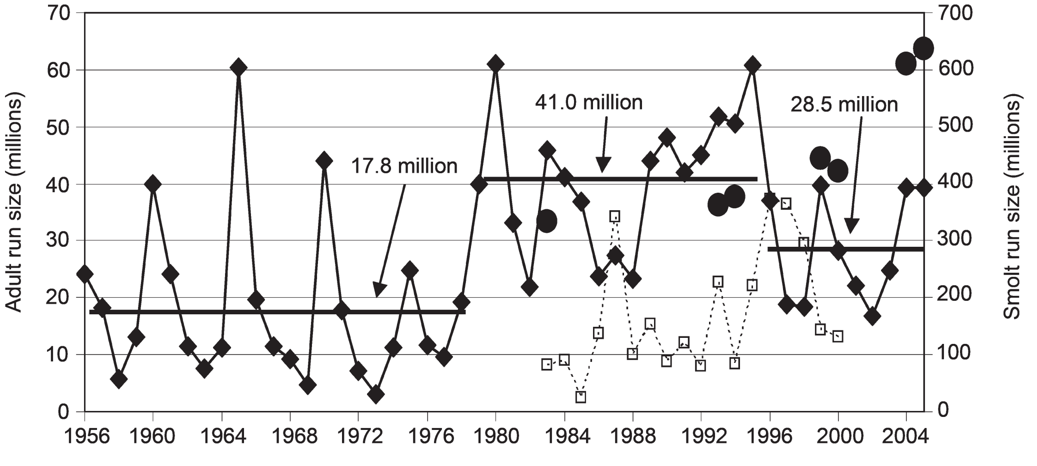

Fig. 4. Beluga counts from aerial surveys and estimated run size (in millions) of salmon smolts and adults in Bristol Bay, Alaska, 1956-2005. Horizontal lines and numbers on the figure show the average adult salmon run for three time periods.

Mantua, 2000; Hyun, 2002; Sands, 2006; Westing *et al*., 2006). Salmon abundance increased markedly during 1979-1995, with less regular cyclical peaks and average run size of 41 million. From 1996-2005, average runs declined to 28.5 million, but were still substantially higher than prior to 1980. While thresholds of prey abundance needed for belugas to thrive and increase are not understood, the larger size of red salmon runs before and during the period covered by the aerial surveys may be a partial explanation for the increased beluga numbers. Salmon stocks in Bristol Bay were greatly reduced by overfishing in the 1940s and early 1950s. By the mid-1950s, resource agencies were attempting to control salmon predators because of their perceived negative effect on salmon abundance (Brooks, 1955; Fish and Vania, 1971). Certain of the actions taken were specifically intended to reduce beluga predation on salmon smolts by displacing them from river mouths during the outmigration of smolt. Efforts began in 1956 and included harassment by motorboats and small dynamite charges (Lensink, 1961), followed later by acoustic devices that transmitted killer whale calls underwater (Fish and Vania, 1971). The 'beluga spooker' program was discontinued in 1978 and organised efforts to displace belugas no longer occur (Frost *et al*., 1984).

Although there is no clear single explanation for the apparent increase in beluga numbers in Bristol Bay since 1993, it is possible that several factors have played a role, either alone or in combination. These include recovery from research kills in the 1960s, a modest decline in the rate of subsistence harvest since the early 1990s and a delayed response to increases in salmon abundance in the 1980s. It is also conceivable that killer whale predation has lessened over this period although no there are no data currently to support or refute that possibility. In the absence of *inter alia* information on how the body condition of individuals in the population may have changed over time, it is not possible to make firm inferences as to why this population has increased. Simultaneous indexing of both population abundance and condition of individuals in the population may provide managers with a much more complete understanding of the status of a population compared to a situation where only information on absolute abundance or change in abundance is available (Gerrodette and DeMaster, 1990).

Management implications

In the United States, the Marine Mammal Protection Act requires use of the 'potential biological removal' (PBR) system to evaluate whether human 'taking' of marine mammals is excessive relative to the goals of that Act, and if it is steps must be taken to reduce the number of animals incidentally removed (killed or seriously injured) in commercial fisheries. The basic population data required to support the PBR system are a minimum estimate of population size and an estimate of the maximum net productivity rate (R_{max}). Using those data and an assumed recovery factor, an upper limit on removal levels consistent with management goals is calculated (Wade, 1998) for comparison with the number of animals being killed or seriously injured by various human activities. While PBR was first articulated in legislation passed in the US, it has also been used by managers in other countries to calculate safe removal levels for marine mammals (e.g. Butler *et al*., 2008; Marsh *et al*., 2004).

The PBR process produces an informative threshold for managers regarding anthropogenic removals (Read and Wade, 2000). However, the process requires a considerable amount of data that can be expensive and difficult to collect. Serious problems that may arise include: (1) inadequate resources to survey populations with sufficient precision and frequency (Taylor *et al*., 2007); (2) problems with developing methods to correct survey data for detection and availability biases; (3) a lack of data for estimating R_{max}; (4) a lack of data on total anthropogenic removals; and (5) insufficient data to adequately describe population structure. For R_{max} so little is known that in 2007 default values were used in the stock assessments produced by NMFS for 147 of the 156 stocks that were evaluated (*http://www.nmfs.noaa.gov/pr/sars/draft.htm*). The result is that in many situations, for example for 13 of the 36 Alaskan stocks evaluated in 2006 (Angliss and Outlaw, 2007), a valid PBR cannot be calculated. Furthermore, there are situations where populations have shown major declines in spite of the fact that estimated human takes have been well below PBR (Angliss and Outlaw, 2007) e.g. western stock of Steller sea lions, *Eumetopias jubatus*; eastern Pacific stock of northern fur seals, *Callorhinus ursinus*; southwest Alaska stock of northern sea otter, *Enhydra lutris kenyoni*. Presumably such situations mean that those populations are declining for reasons other than direct take by humans, and therefore the PBR system is poorly suited for evaluating actions needed to allow the populations to recover.

Compared to the cost of acquiring the data required to implement the PBR management regime for a given population (i.e. a conservative estimate of current absolute abundance, an estimate of R_{max}, and an estimate of current total anthropogenic removals), basing management actions on monitoring trends in abundance can be less expensive and more effective for some populations in some areas, providing adequate data can be collected. More importantly, a series of index counts may detect population responses in situations where factors other than direct taking by humans are impacting the status of a population, and also could account for cumulative effects.

We believe that is the case with this study, which has shown that the Bristol Bay beluga stock has increased at a rate of 2.1%-7.5% per year during 1993-2005. This evidence of increase in population size indicates that the aggregate effects of direct takes (e.g. subsistence hunting, entanglement in fishing gear), indirect interactions with humans (e.g. competition for food resources, habitat alteration), and other factors (e.g. predation, changing climate) over that 12-year period were 'acceptable' in the sense that they did not prevent the population from increasing. Therefore, we conclude that there is no need for a change in the *status quo* with regard to management of this stock.

There are two other beluga whale populations that are (or have been) of generally similar size to Bristol Bay and that occur in similar sub-arctic environments, Cook Inlet (Alaska) and the Saint Lawrence estuary (Canada). The trend in abundance is being closely monitored for both, with results showing that the Cook Inlet population is declining (Hobbs *et al*., 2000; Lowry *et al*., 2006) while the St. Lawrence population is most likely stable (Gosselin *et al*., 2001; Hammill *et al*., 2007). Results from this study suggest that once factors limiting those populations have been identified and mitigated, it would be reasonable to expect them to increase at a similar rate to the Bristol Bay population. To adequately protect and manage these small populations, it is essential that trend monitoring programmes be continued in those regions. In addition, to the extent possible, it would be useful to collect data on the body condition of animals in these populations. It is also important to continue to monitor human activities so that if changes in trends are detected, potential causes can be examined.

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REFERENCES

Adams, M., Frost, K.J. and Harwood, L.A. 1993. Alaska and Inuvialuit Beluga Whale Committee (AIBWC) – An initiative in at home management. *Arctic* 46(2): 134-37.

Angliss, R.P. and Outlaw, R.B. 2005. Alaska marine mammal stock assessments 2005. *NOAA Tech. Mem. NMFS-AFSC-* 161: 252pp.

Angliss, R.P. and Outlaw, R.B. 2007. Alaska marine mammal stock assessments 2006. *NOAA Tech. Mem. NMFS-AFSC-* 133: 224pp.

Brooks, J.W. 1955. Beluga investigations. *Annu. Rep. Alaska Dep. Fish. Game* 7: 98-106.

Brooks, J.W. 1957. Marine mammals in relation to commercial fisheries in Alaska. Paper presented at the Alaska Science Conference, September 1957, Anchorage, AK. [Available on request from ADF&G Div. Wildl. Conserv., 1300 College Road, Fairbanks, AK 99701, USA].

Butler, J.R.S., Middlemas, S.J., McKelvey, S.A., McMyn, I., Leyshon, B., Walker, I., Thompson, P.M., Boyd, I.L., Duck, C., Armstrong, J.D., Graham, I.M. and Baxter, J.M. 2008. The Moray Firth Seal Management Plan: an adaptive framework for balancing the conservation of seals, salmon, fisheries and wildlife tourism in the UK. *Aquat. Conserv*. 18: 1025-38.

Chythlook, M. and Coiley, P. 1994. The subsistence use of beluga whale in Bristol Bay by Alaska Natives, 1993. Tech Paper No. 231, Alaska Department of Fish and Game, Division of Subsistence, Juneau, Alaska, USA. 29pp.

Credle, V.R., DeMaster, D.P., Merklein, M.M., Hanson, M.B., Karp, W.A. and Fitzgerald, S.M. 1994. NMFS observer programs: minutes and recommendations from a workshop held in Galveston, Texas, November 10-11, 1993. *NOAA Technical Memorandum NMFS* OPR-94-1: 94pp.

DeMaster, D.P., Lowry, L.F., Frost, K.J. and Bengston, R.A. 2000. The effect of sea state on estimates of abundance for beluga whales (*Delphinapterus leucas*) in Norton Sound, Alaska. *Fish. Bull*. 99: 197-201.

Fernandez-Gimenez, M.E., Huntington, H.P. and Frost, K.J. 2006. Integration or co-optation? Traditional knowledge and science in the Alaska Beluga Whale Committee. *Environ. Conserv*. 33: 1-10.

Fish, J.F. and Vania, J.S. 1971. Killer whale, *Orcinus orca*, sounds repel white whales, *Delphinapterus leucas*. *Fish. Bull*. 69(3): 531-35.

Frost, K.J. and Lowry, L.F. 1990. Distribution, abundance and movements of beluga whales, *Delphinapterus leucas*, in coastal waters of western Alaska. *Can. Bull. Fish. Aquat. Sci*. 224: *Advances in research on the beluga whale,* Delphinapterus leucas: 39-57.

Frost, K.J., Lowry, L.F. and Nelson, R.R. 1984. Belukha whale studies in Bristol Bay, Alaska. pp.187-200. *Proceedings of the Workshop on Biological Interactions among Marine Mammals and Commercial Fisheries in the Southeastern Bering Sea. October 18-21 1983, Anchorage, AK*. Alaska Sea Grant Rep. 84-1.

Frost, K.J., Lowry, L.F. and Nelson, R.R. 1985. Radiotagging studies of beluga whales, *Delphinapterus leucas*, in Bristol Bay, Alaska. *Mar. Mammal Sci*. 1: 191-202.

Frost, K.J., Russell, R.B. and Lowry, L.F. 1992. Killer whales, *Orcinus orca*, in the southeastern Bering Sea: recent sightings and predation on other marine mammals. *Mar. Mammal Sci*. 8(2): 110-19.

Gerrodette, T. and DeMaster, D.P. 1990. Quantitative determination of optimum sustainable population level. *Mar. Mammal Sci*. 6(1): 1-16.

Gosselin, J.F., Lesage, V. and Robillard, A. 2001. Population index estimate for the beluga of the St. Lawrence Estuary in 2000. Department of Fisheries and Oceans Canada, Can. Sci. Advis. Sec. Res. Doc. 2001/049. 13pp.

Hammill, M.O., Measures, L.N., Gosselin, J.F. and Lesage, V. 2007. Lack of recovery in St. Lawrence Estuary beluga. Department of Fisheries and Oceans Canada, Can. Sci. Advis. Sec. Res. Doc. 2007/026. 23pp.

Hare, S.R. and Mantua, N.J. 2000. Empirical evidence for North Pacific regime shifts in 1977 and 1989. *Prog. Oceanogr*. 47(2-4): 103-46.

Hobbs, R.C., Rugh, D.J. and DeMaster, D.P. 2000. Abundance of belugas, *Delphinapterus leucas* in Cook Inlet, Alaska, 1994-2000. *Mar. Fish. Rev*. 62(3): 37-45.

Hyun, S.Y. 2002. In season forecasts of sockeye salmon returns to the Bristol Bay districts of Alaska. PhD dissertation, University of Washington. 167pp.

Lensink, C.J. 1958. Belugas. *Annual report for 1958*. Unpublished report for the Alaska Department of Fish and Game, Juneau, Alaska, USA. [Available on request from ADF&G Div. Wildl. Conserv., 1300 College Road, Fairbanks, AK 99701 USA].

Lensink, C.J. 1961. Status report: beluga studies. *Annual report for 1958*. Unpublished report for the Alaska Department of Fish and Game, Juneau, Alaska, USA. 38pp. [Available on request from ADF&G Div. Wildl. Conserv., 1300 College Road, Fairbanks, AK 99701 USA].

Lowry, L., O'Corry-Crowe, G. and Goodman, D. 2006. *Delphinapterus leucas* (Cook Inlet population). *In*: IUCN (eds). *2006 IUCN Red List of Threatened Species*. p.15.

Marsh, H., Lawler, I.R., Kwan, D., Delean, S., Pollock, K. and Alldredge, M. 2004. Aerial surveys and the potential biological removal technique indicate that the Torres Strait dugong fishery is unsustainable. *Anim. Conserv*. 7: 435-43.

O'Corry-Crowe, G., Suydam, R., Rosenberg, A., Frost, K. and Dizon, A.E. 1997. Phylogeography, population structure and dispersal patterns of the beluga whale, *Delphinapterus leucas*, in the western Neartic revealed by mitochondrial DNA. *Mol. Ecol*. 6: 955-70.

Ray, C. 1962. Three whales that flew. *Natl. Geogr. Mag*. 121(3): 346-59.

Read, A.J. and Wade, P.R. 2000. Status of marine mammals in the United States. *Conserv. Biol*. 14: 929-40.

Reeves, R.R. and Leatherwood, S. 1984. Live-capture fisheries for cetaceans in USA and Canadian waters, 1973-1982. *Rep. int. Whal. Commn* 34: 497-507.

Sands, T. 2006. Overview of the Bristol Bay salmon fishery, 2004-2006, a report to the Alaska Board of Fisheries. Alaska Department of Fish and Game, Special Publication No. 06-28, Anchorage, AK. [Available on request from ADF&G Div. Comm. Fish., PO Box 25526, Juneau, AK 99802, USA].

Taylor, B.L., Martinez, M., Gerrodette, T., Barlow, J. and Hrovat, Y.N. 2007. Lessons from monitoring trends in abundance of marine mammals. *Mar. Mammal Sci*. 23: 157-75.

Vania, J. and Klinkhart, E. 1967. Marine mammal report. *Federal aid in wildlife restoration annual project segment report vol. VIII. projects W-14-R-1 and 2*. 24pp. [Available on request from ADF&G Div. Wildl. Conserv., 1300 College Road, Fairbanks, AK 99701 USA].

Wade, P.R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mammal Sci*. 14(1): 1-37.

Westing, C., Sands, T., Morstad, S., Salomone, P., Fair, L., West, F., Brazil, C. and Weiland, K.A. 2006. Annual management report 2005 Bristol Bay area. Alaska Department of Fish and Game Fishery Management Report No. 06-37, Anchorage, Alaska, USA. [Available on request from ADF&G Div. Comm. Fish., PO Box 25526, Juneau, AK 99802, USA].

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A note on cetacean distribution in the Indian EEZ and contiguous seas during 2003-07

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ABSTRACT

Relatively little is known about the distribution of cetaceans in Indian seas due to lack of systematic surveys. For collecting data on species distribution, 35 opportunistic surveys were conducted onboard FORV *Sagar Sampada* between October 2003 and February 2007 in the Indian EEZ and contiguous seas. In 5,254 hours of sighting effort, a total of 473 cetacean records were made with 5,865 individuals. The occurrence of 10 species from three cetacean families was confirmed. The Indo-Pacific bottlenose dolphin was the most frequently sighted species, whereas the spinner dolphin was dominant in terms of abundance. Long-beaked common dolphins, Indo-Pacific hump-backed dolphin and sperm whales were also recorded at frequent intervals. Cetaceans were found to have a wide geographical distribution in the Indian EEZ and contiguous seas. High abundance and species richness were recorded in the Southeastern Arabian Sea and southern Sri Lankan waters. From the information collected during the present study, the platform of opportunity has proved to be a useful means for cetacean survey.

KEYWORDS: SURVEY-VESSEL; INDIAN EEZ; DISTRIBUTION; BLUE WHALE; SPERM WHALE; FALSE KILLER WHALE; SHORT-FINNED PILOT WHALE; RISSO'S DOLPHIN; STRIPED DOLPHIN; SPINNER DOLPHIN; LONG-BEAKED COMMON DOLPHIN; INDO-PACIFIC BOTTLENOSE DOLPHIN; INDO-PACIFIC HUMP-BACKED DOLPHIN

INTRODUCTION

India has an exclusive economic zone (EEZ) of about 2.02 million km^2. The southern peninsula extends into the tropical waters of the Indian Ocean, with the Bay of Bengal in the east and the Arabian Sea in the west. Knowledge on the cetaceans in the Indian Seas is generally confined to reports from stranded and accidentally caught specimens, although information from these reports can help in understanding a number of important biological features of the populations. Consolidating this information, Kumaran (2002) concluded that 25 species of cetaceans and one species of sirenian, *Dugong dugon*, occur in the Indian Seas. However, these data, to a large extent, cannot be extrapolated to provide information on the true distribution or abundance of cetaceans. This lack of information is significant, as the Indian EEZ is located within the Indian Ocean Sanctuary, which was established in 1979 (IWC, 1980, p.27) to encourage conservation and research on cetaceans in the area. In India, cetaceans are classified as endangered and protected under the Wildlife (Protection) Act, 1972.

Systematic vessel surveys have not been conducted for cetaceans in the Indian Seas. This paper presents the results of ship-based sighting surveys of cetaceans in the Indian EEZ and contiguous seas onboard FORV *Sagar Sampada*. The primary objective of the study was to record occurrence and relative abundance.

MATERIALS AND METHODS

The sighting surveys were conducted from October 2003 to February 2007 onboard the fisheries and oceanographic research vessel FORV *Sagar Sampada* (overall length: 71.4m). The cruises were multidisciplinary and hence the cruise tracks were determined by the needs of other projects; thus the vessel was used as a platform of opportunity. The study area included the coastal, continental shelf and oceanic waters of the Indian EEZ and a part of the southern Sri Lankan Sea (Fig. 1). The surveyed area extended between 5-23°N and 66-95°E with a depth range of 20-5,000m. The speed of the ship varied with sea conditions and also with the kind of fisheries and oceanographic work carried out. When not on station, the average speed was seven knots. The surveyed areas had marked temporal variations, with striking changes in sea state and thus sighting conditions. The major seasonal oceanographic changes and adverse weather conditions are likely to have affected not only the cetacean distribution but the ability of observers to detect animals. Some species, particularly of dolphins and porpoises, are easily overlooked at higher sea states (e.g. Clarke, 1982). The survey was carried out in sea conditions corresponding to Beaufort scale zero to five; effort was suspended at six or above.

During the survey, data were collected by a single observer positioned on the flying bridge of the vessel, which was about 17m above sea level. This enabled the observer to look down into the wave troughs and spot cetaceans that would typically remain hidden from lower elevations. The observer scanned the ocean area in a 180° arc ahead of the ship, out to the horizon or to the farthest limit of visibility. The surveys were carried out during daylight hours, taking breaks for about four hours for lunch or for rest. Although the observations commenced at 0600hrs and ended at 1800hrs (and sometimes at 1900hrs, depending on the light condition), the average time of observation per day was considered to be eight hours. The visual surveys were carried out by scanning with naked eye interspersed with scans with binoculars. A *Nikon* 10×50mm CFWP handheld binocular was used for close observation of the located animal. A *Nikon* F80 camera fitted with Nikor 70-300mm lens and a *Sony* DCR-HC46E handycam with 800× digital zoom were employed to capture images of cetaceans (e.g. blows, dorsal fin, flipper, upper body, flukes etc.) to assist in confirming species identity.

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On sighting a cetacean, a standard methodology was adopted to document the observation. Date, time, position, nearest landmark, distance from the shore, depth at the area of sighting, wind direction and force, sea state, visibility, sea surface temperature and salinity were recorded, along with details of the cetaceans including species, morphology, behaviour, group size and associated animals. The ship's position was recorded with the help of a *Simrad* GN33 GPS Navigator. For recording the depth at the area of sighting, a *Simrad* EK 60 echo-sounder of frequency 38kHz was employed. An *EMCON* SBE 9plus underwater unit provided SST and salinity data. Although angle and distance measurements to sightings with reference to the observer's position in the ship were obtained during some cruises, they were found inadequate to be presented in this paper.

The survey, being opportunistic, was conducted in 'passing mode' (i.e. the vessel did not divert to confirm species identification and school size) and cetaceans sighted were approached only rarely. The observed cetaceans were identified to the lowest taxonomic level possible, often based on a brief view of a splash, blow, dorsal fin, head, flipper or back, even though this needs lot of experience. Published pictures of the whole animal along with species description of body morphology, colouration and behaviours were compared with the observed characters for identification of the sighted individuals. Whenever necessary, the species identifications were validated later with the photographs taken onboard. 'Marine Mammals of the World' (Jefferson *et al*., 1993) and other published literature aided identification and confirmation of species. Only confirmed sightings at the species or generic level are documented here. Unconfirmed and possible sightings were downgraded to 'unidentified whales' and 'unidentified dolphins'.

Data analysis

The data collected were compiled in an *Excel* spreadsheet. SPSS and Primer were employed for statistical analysis. For distribution, relative abundance and diversity analysis, the survey area was divided into a 2°×2° grid, and the number of sightings/individuals correlated with oceanographic parameters. For analysing the observation effort and sightings the surveyed area was segregated into six zones: northeastern Arabian Sea; southeastern Arabian Sea; northern Bay of Bengal; southern Bay of Bengal; Andaman Sea; and southern Sri Lankan Sea. For correlating the time of the day with the sightings, the hours of observation of a day were divided in to five time strata, 0600-0900hrs, 0900-1200hrs, 1200-1500hrs, 1500-1800hrs and after 1800hrs; and the number of sightings in each time stratum was determined. To study seasonal distribution, the data were sorted by month. The sightings were plotted on maps using Ocean Data View Ver.3 software.

RESULTS

From October 2003 to February 2007, a total of 35 cruises were conducted. Each cruise lasted for 10 to 45 days. The cruise tracks are given in Fig.1. The number of days at sea was 657 and the observation effort was 5,254 hours. In all, 764.7 hours (14.5% of total observation) were spent in the northeastern Arabian Sea, 2,017.8 hours (38.4%) in the southeastern Arabian Sea, 636.0 hours (12.1%) in the northern Bay of Bengal, 843.0 hours (16.0%) in the southern Bay of Bengal, 595.5 hours (11.3%) in the Andaman Sea and 397.0 hours (7.6%) in the southern Sri Lankan Sea (Indian Ocean). Observation conditions were almost always moderate to excellent, ranging in Beaufort scale from zero to five. Of the total sightings, 33.4% was at Beaufort 0-2,

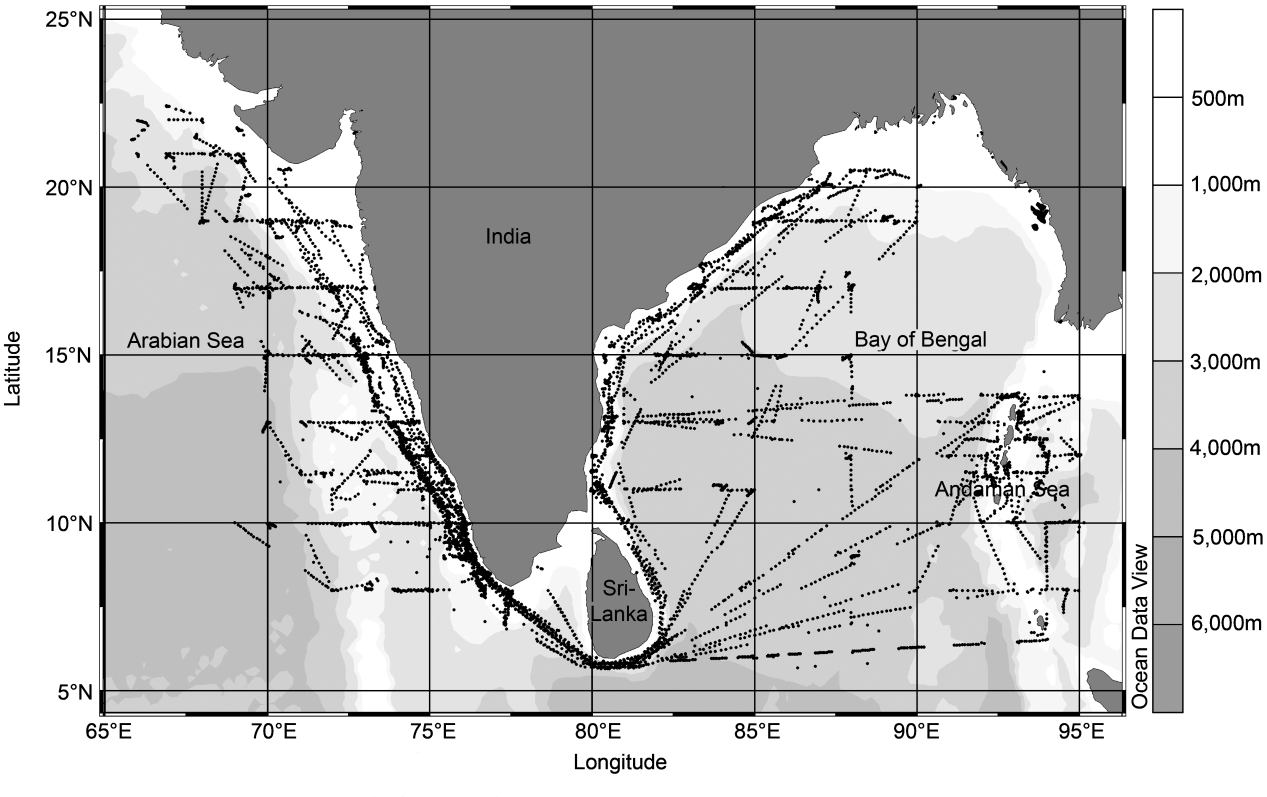

Fig. 1. Tracks of 35 cruises of FORV *Sagar Sampada* during October 2003-February 2007.

57.1% at Beaufort 3-4, and 9.5% at Beaufort five. Of 657 observation days cetaceans were sighted on 299 days. A total of 473 sightings, comprising 5,865 individuals were recorded. Of these, 26% (123 sightings; 1,619 individuals) were identified to species or genus. Almost one-third of sightings were made during the period 1500-1800 hours.

Geographical distribution

The surveys revealed that cetaceans are widely distributed in the Indian EEZ and contiguous seas (Fig. 2). A total of 124 sightings (26.2%) were over the continental shelf (<200m depth) and the remaining (73.8%) from oceanic waters (>200m depth). In the southern Sri Lankan Sea, the number of sightings and individuals per hour of observation were the highest at 0.22 and 1.78 respectively (Table 1). This was followed by the southeastern Arabian Sea with 0.10 sightings h^{-1} and 1.24 individuals h^{-1}.

Species distribution

Among the 10 species identified, two were whales and eight were small cetaceans. The two species of whales were the blue whale (*Balaenoptera musculus*) of the family Balaenopteridae and the sperm whale (*Physeter macrocephalus*) of the family Physeteridae. The small cetaceans recorded were false killer whale (*Pseudorca crassidens*), short-finned pilot whale (*Globicephala macrorhynchus*), Risso's dolphin (*Grampus griseus*), striped dolphin (*Stenella coeruleoalba*), spinner dolphin (*Stenella longirostris*), long-beaked common dolphin (*Delphinus capensis*), Indo-Pacific bottlenose dolphin (*Tursiops aduncus*) and Indo-Pacific hump-backed dolphin (*Sousa chinensis*), all belonging to the family Delphinidae.

The Southeastern Arabian Sea showed the highest species diversity with records of nine species followed by the Southern Sri Lankan Sea with six species (Table 2).

The Indo-Pacific bottlenose dolphin was the species recorded most often (26 sightings) whereas the spinner dolphin was the most abundant in terms of number of individuals (552). The long-beaked common dolphin and Indo-Pacific hump-backed dolphin were also relatively abundant. The group size of spinner dolphin was the largest and ranged from 5 to 110 individuals (mean: 33). Among the dolphins, the group size of Indo-Pacific hump-backed dolphin was the smallest (mean: 3.6). The sperm whale was the most frequently sighted species among large whales. Group size is shown in Table 3.

Table 1

Observation effort and sightings in the six areas during 2003-2007.

Area	No. of 2°x2° grids	Effort (h)	Sightings	Sightings/h	Individuals	Individuals/h
Northeastern Arabian Sea	12	764.7	45	0.059	411	0.54
Southeastern Arabian Sea	17	2,017.8	194	0.096	2,506	1.24
Northern Bay of Bengal	13	636	39	0.061	751	1.18
Southern Bay of Bengal	23	843	66	0.078	995	1.18
Andaman Sea	14	595.5	43	0.072	497	0.83
Southern Sri-Lankan Sea	7	397	86	0.22	705	1.78
Total	86	5,254	473	0.09	5,865	1.12

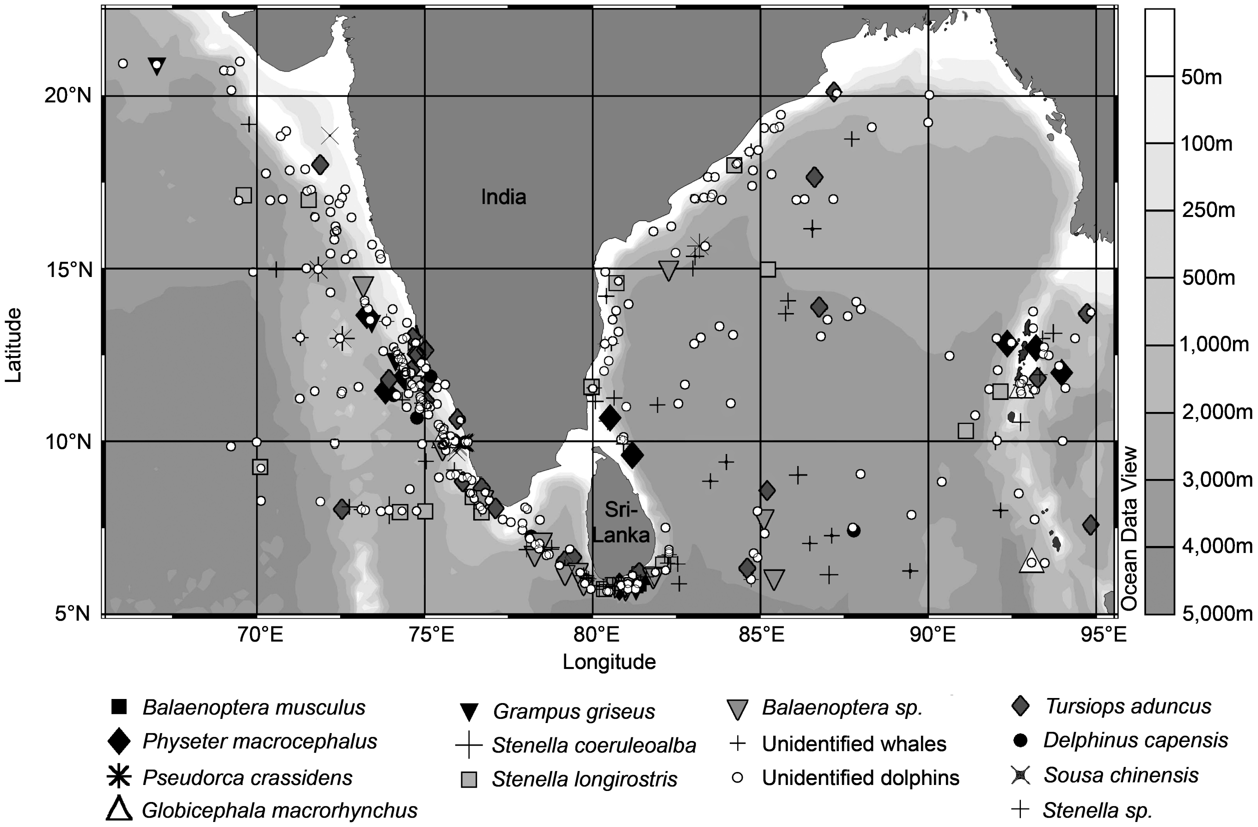

Fig. 2. Distribution map of cetacean species based on FORV *Sagar Sampada* sighting cruises during October 2003-February 2007.

Table 2

Number of sightings (s) and individuals (n).

Species	Northeastern Arabian Sea		Southeastern Arabian Sea		Northern Bay of Bengal		Southern Bay of Bengal		Andaman Sea		Southern Sri-Lankan Sea		Total	
	s	n	s	n	s	n	s	n	s	n	s	n	s	n
Balaenoptera musculus	0	0	0	0	0	0	0	0	0	0	4	13	4	13
Balaenoptera sp.	0	0	3	9	0	0	3	7	0	0	12	24	18	40
Physeter macrocephalus	0	0	3	18	0	0	2	7	3	15	1	1	9	41
Pseudorca crassidens	0	0	3	18	1	4	0	0	0	0	0	0	4	22
Globicephala macrorhynchus	0	0	1	2	0	0	0	0	2	17	0	0	3	19
Grampus griseus	1	15	2	32	0	0	0	0	0	0	1	25	4	72
Stenella coeruleoalba	0	0	1	5	0	0	0	0	0	0	0	0	1	5
Stenella longirostris	2	80	7	254	1	20	3	155	2	13	2	30	17	552
Stenella sp	0	0	1	6	2	220	2	23	4	50	2	40	11	339
Tursiops aduncus	1	4	14	208	2	10	2	30	3	35	4	32	26	319
Delphinus capensis	0	0	5	59	0	0	2	23	0	0	1	50	8	132
Sousa chinensis	1	6	17	59	0	0	0	0	0	0	0	0	18	65
Unidentified whales	1	1	13	17	3	4	14	25	2	3	33	90	66	140
Unidentified dolphins	39	305	124	1,819	17	493	38	725	27	364	26	400	284	4,106
Total	45	411	194	2,506	26	751	66	995	43	497	86	705	473	5,865

Table 3

Sightings and abundance in the Indian EEZ and contiguous seas.

Species	No. of sightings	No. of individuals	Group size	
			Range	Mean
Balaenoptera musculus	4	13	1-7	3
Balaenoptera sp.	18	40	1-10	2.2
Physeter macrocephalus	9	41	1-9	4.5
Pseudorca crassidens	4	22	1-11	5.5
Globicephala macrorhynchus	3	19	2-10	6.3
Grampus griseus	4	72	12-25	18
Stenella coeruleoalba	1	5	5	-
Stenella longirostris	17	552	5-110	32.5
Stenella sp.	11	339	3-200	30.8
Tursiops aduncus	26	319	1-75	12.3
Delphinus capensis	8	132	2-50	16.5
Sousa chinensis	18	65	1-20	3.6

Baleen whales

Of the 473 sightings, 22 records (4.7%) were baleen whales. Most of the sightings were off southern and southwestern Sri Lanka between 5-7°N and 78-82°E. On four occasions, the animals were identified as blue whales; the others were identified only to genus.

A total of four blue whale groups consisting of 13 individuals was encountered during the survey. All the sightings were from the southern Sri Lankan Sea between 5-7°N and 80-82°E. The school size varied from a single solitary animal to a group of seven individuals. In 18 sightings comprising 40 individuals, the animals could be identified as *Balaenoptera* sp. The sightings of balaenopterid whales were less frequent in Indian seas compared to off southern Sri Lanka. Groups of up to 10 individuals were recorded, but on most of the occasions it was either a solitary animal or a pair. The mean group size was 2.2.

Sperm whales

The sperm whale was sighted on nine occasions (7.3% of confirmed sightings) with a total of 41 individuals. The sperm whale showed a wide distribution with records from the Southeastern Arabian Sea, Southern Bay of Bengal, Andaman Sea and Southern Sri-Lankan Sea. The group size ranged from one to nine.

Delphinids

SPINNER DOLPHIN

A total of 17 sightings of spinner dolphins was recorded, which contributed 13.8% to the total confirmed sightings. They were sighted frequently in almost all parts of the survey area, showing a wide distribution in the Indian EEZ and contiguous seas. The spinner dolphin had the maximum number of individuals observed during the survey, 552 (34.1% of the total number in confirmed sightings) and they were usually seen in large active groups. The group size varied between five and 110 individuals (average 33).

INDO-PACIFIC BOTTLENOSE DOLPHIN

The Indo-Pacific bottlenose dolphin was encountered more frequently than any other cetacean during the survey. A total of 26 sightings of the species was recorded i.e. 21.1% of the total confirmed sightings. A total of 319 individuals was seen with group sizes ranging from 1-75, with an average of 12. Distribution was found to be wide with records from all the survey zones.

LONG-BEAKED COMMON DOLPHIN

Long-beaked common dolphins were recorded eight times, comprising 6.5% of the total confirmed sightings. A total of 132 individuals was observed. Six sightings were from the southeastern Arabian Sea while one sighting each was made in the Sri Lankan Sea and Andaman Sea. The number of individuals in the group varied between two and 50 with an average group size of 17 individuals.

INDO-PACIFIC HUMP-BACKED DOLPHIN

The Indo-Pacific hump-backed dolphin was sighted on 18 occasions, 14.6% of the total confirmed sightings, with a total of 65 individuals. This is a coastal species and 88.9% of the sightings were in Cochin backwaters and the Cochin bar-mouth area between 9°40'-9°59'N and 75°35'-76°18'E. The maximum group size was 20 individuals and the average was 3.6. Solitary individuals were also recorded.

Others

There were three sightings (2.4% of confirmed sightings) of short-finned pilot whales, which comprised 19 individuals. One sighting was in the southeastern Arabian Sea at 10°10'N and 75°58'E and the other two were in the

Andaman Sea. Group size ranged from 2-10 with an average of six individuals. Four sightings of false killer whales (22 individuals) were made during the survey, which contributed 3.3% to the total confirmed sightings. Three sightings were in the southeastern Arabian Sea at locations between 12-15°N and 71-73°E. The other was in the northern Bay of Bengal at 15°65’N and 83°18’E. Group size varied from 1-11 with an average of five individuals. Four sightings (3.3% of confirmed sightings) of Risso’s dolphin were made. A total of 72 individuals were observed. The group size ranged from 12 to 25 individuals with an average of 18. Three of the sightings were in the Arabian Sea, and one off southeast Sri Lanka. One record of the striped dolphin with five individuals was made at 8°N and 73°57’E off Minicoy. On eleven occasions (8.9% of confirmed sightings) the animals were recorded as *Stenella* sp. A total of 339 individuals were observed and group sizes ranged from 3-200 with an average of 31 individuals.

Unidentified cetaceans

Identification of 350 sightings (74.0% of the total number of sightings) could not be made to species or genus and these were recorded as ‘unidentified’. Of these, 60.0% (284 records) were small cetaceans with (a total of 2,788 individuals) and the remaining 14.0% (66 records) were whales (140 individuals). In this category more whale sightings were recorded from the Sri Lankan Sea, whereas occurrence of unidentified small cetaceans was common with records from all the six survey zones.

Seasonal distribution of different species

The survey cruises were conducted over 37 months, from October 2003-February 2007 (Table 4), although the number of days of observation was not equally distributed among the months, some effort and sighting records are available for all months and for three months (October, January and February) surveys occurred for all four years. The total maximum number of sightings was in February and the lowest in August whilst peak in sightings per day was maximum in November (1.13) and minimum in August (0.18).

Of the 37 months covered, the Indo-Pacific bottlenose dolphin was sighted in 18 months, spinner dolphin in 12 months and long-beaked common dolphin in eight months, and the sperm whale was sighted in seven months (Table 5). The Indo-Pacific bottlenose dolphin was sighted in all the months from January to December except in September. The Indo-Pacific hump-backed dolphin was sighted in 10 months (except in July and August), the spinner dolphin in eight months (except in April, May, August and October) and long-beaked common dolphin in seven months (except in March, July and August).

DISCUSSION

Ship-based visual survey is a conventional and widely practiced method to collect data on the relative and absolute abundance and distribution of marine mammals at the species level (Aragones *et al*., 1997). The use of platforms of opportunity has been shown to be cost-effective and to

Table 4

Number of sightings in each month (pooled for the years 2003-07).

Month	Observation effort		No. of days of sighting	No. of sightings	Sightings/day	Sightings/h
	Days	Hours				
Jan.	62	496	27	40	0.65	0.08
Feb.	61	488	33	62	1.02	0.13
Mar.	32	254	18	24	0.75	0.09
Apr.	58	464	27	36	0.62	0.08
May	42	336	27	37	0.88	0.11
Jun.	70	560	31	50	0.71	0.09
Jul.	65	520	27	38	0.58	0.07
Aug.	44	352	6	8	0.18	0.02
Sep.	53	424	31	46	0.87	0.11
Oct.	51	408	24	42	0.82	0.1
Nov.	47	376	24	53	1.13	0.14
Dec.	72	576	24	37	0.51	0.06

Table 5

Number of sightings by month during Oct. 2003-Feb. 2007.

Species	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.	Total	Frequency (months)
Balaenoptera musculus		2						2					4	2
Balaenoptera sp.		5			1	1	5		1	2	2	1	18	8
Physeter macrocephalus	3	3				2				1			9	4
Pseudorca crassidens				2			1			1			4	3
Globicephala macrorhynchus		2				1							3	2
Grampus griseus				1							3		4	2
Stenella coeruleoalba											1		1	1
Stenella longirostris	1	2	1			2	2		4		2	3	17	8
Stenella sp					1	4	4	1			1		11	5
Tursiops aduncus	4	3	1	2	1	1	4	1		1	5	3	26	11
Delphinus capensis	1	2		1	1				1	1		1	8	7
Sousa chinensis	2	2	1	1	1	2			3	2	1	3	18	10

contribute to the knowledge on cetaceans, especially when data are collected by trained personnel following standard procedures (Robbins *et al*., 2006; Robbins and Mattila, 2000).

The encounter rates recorded in the present study are lower than other records from the Indian Ocean (Table 6). In the northwest Indian Ocean and Sri Lankan waters, Alling (1986) reported 0.9 sightings per day. Sighting records as high as 6.4 per day have also been recorded (Ballance and Pitman, 1998).

The low sighting records in the present study may be due to a number of limitations:

(i) all the cruises were 'opportunistic' without a structured cruise programme, and the effort was not uniformly distributed temporally and spatially;
(ii) the vessel FORV *Sagar Sampada* is too large for sighting cruises and not easily maneuverable, making tracking an animal after sighting impossible;
(iii) the observation deck is 17m above sea level which may reduce the possibility of wrong identification of the species from that height – thus only confirmed identities are included here, which has resulted in the very high percentage (74.0%) of unidentified cetaceans;
(iv) only a single observer was used whereas two observers is more normal and would have improved the quality of observation – three observers were used in the opportunistic survey conducted by Ballance and Pitman (1998) and this resulted in a substantially higher encounter rate.

In spite of these limitations, the following conclusions could be arrived at:

(i) cetaceans were found to have a wide geographical distribution in the Indian EEZ and contiguous seas;
(ii) abundance and species richness are greatest in the southeastern Arabian Sea (off Kerala-Karnataka) and southern Sri-Lankan waters (these areas were reported to have rich cetacean faunas in earlier studies – Alling, 1986; de Silva, 1987; Ilangakoon, 1997);
(iii) among large whales, the sperm whale is the most abundant species with wide distribution. Baleen whales including the blue whale were relatively common in the Sri Lankan Sea which is known to be an important blue whale feeding area, even though the occurrence may be seasonal (Alling *et al*., 1990) – Jefferson *et al*. (2008) suggest that the blue whales in this area may be the pygmy blue whale subspecies (*B. musulus brevicauda*);
(iv) the distribution pattern observed in the present study agrees with historical records based on incidental capture.

The spinner dolphin was the most frequently recorded species during the last century in India (Kumaran, 2002). In the present survey, the spinner dolphin was dominant in terms of abundance, whereas Indo-Pacific bottlenose dolphin was the most dominant species in terms of number of records. These were followed by the long-beaked common dolphin and Indo-Pacific hump-backed dolphin. As the surveys were mostly in the fishing grounds, the four predominant delphinids observed in the study were among the most recorded species in incidental catches in fishing gear (Jayaprakash *et al*., 1995; Lal Mohan, 1985; Sathasivam, 2006).

Of the 26 known species of marine mammals in Indian waters (Kumaran, 2002), the identification of only 10 species was confirmed during the survey. Four species that could not be confirmed but recorded as possible were the fin whale (*Balaenoptera physalus*), sei whale (*B. borealis*), common minke whale (*B. acutorostrata*) and humpback whale (*Megaptera novaeangliae*). The sightings recorded as possible sei whales may also be the Bryde's whale (*Balaenoptera edeni*), as these two species are difficult to differentiate at sea and the occurrence of the sei whale in the area is still doubtful (Jefferson *et al*., 2008). Bryde's whale, pygmy sperm whale (*Kogia breviceps*), dwarf sperm whale (*K. simus*), Cuvier's beaked whale (*Ziphius cavirostris*), Irrawwady dolphin (*Orcaella brevirostris*), killer whale (*Orcinus orca),* melon-headed whale (*Peponocephala electra),* rough-toothed dolphin (*Steno bredanensis),* pantropical spotted dolphin (*Stenella attenuata*), finless porpoise (*Neophocaena phocaenoides*), Ganges river dolphin (*Plantanista gangetica gangetica*) and dugong (*Dugong dugon*) have been recorded from the region but were not sighted. Of these, the finless porpoise and pantropical spotted dolphin have been recorded as incidental catches in fishing gear (Yousuf *et al*., 2008) and a stranded Bryde's whale (Jayasankar *et al*., 2007) was recorded along the Gulf of Mannar (Southeast coast of India) when the sighting survey was under progress. In 2008, a rough-toothed dolphin was washed ashore near Karwar (southwest coast of India; Miriam Paul, pers. comm.). The finless porpoise and Irrawwady dolphin are distributed in shallow coastal waters where FORV *Sagar Sampada* could not operate. The Ganges river dolphin is a freshwater form and the dugong is a resident of the Gulf of Mannar and Palk Bay, which were not covered in the surveys. It is possible that the species which were not recorded in the present study may have been included in the unidentified sightings, or they may be rare in occurrence, even though many of these species were reported earlier from the study area (Chantrapornsyl *et al*., 1991; Kumaran, 2002; Leatherwood *et al*., 1991).

Table 6

Comparison of present study with other cetacean sighting surveys.

Area	Type of survey	Observation effort (hours/days)	No. of species	No. of sightings	Individuals	Sighting effort		Reference
						Per hour	Per day	
Northwest Indian Ocean and Sri Lanka	Dedicated	295 days	13	271	5,361	…	0.9	Alling (1986)
Western tropical Indian Ocean	Opportunistic	403.9h/92 days	21	589	31,136 (corrected value)	1.5	6.4	Ballance and Pitman (1998)
East Kalimantan, Indonesia	Dedicated	362h/80 days	9	112	868	0.3	1.4	Kreb and Budioao (2005)
Indian EEZ/contiguous seas	Opportunistic	5,254h/657 days	10	473	5,865	0.1	0.7	Present study

This platform of opportunity has provided a useful means for collection of cetacean sighting data. The data generated on species occurrence and distribution will be useful for assisting with the estimation of the abundance of marine mammals in the Indian Seas in the future. For this, the oceanic surveys onboard research vessels need to be supplemented with coastal surveys with smaller boats.

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REFERENCES

Alling, A. 1986. Records of odontocetes in the northern Indian Ocean (1981-1982) and off the coast of Sri Lanka (1982-1984). *J. Bombay Nat. Hist. Soc.* 83(2): 376-94.

Alling, A., Dorsey, E.M. and Gordon, J.C.D. 1990. Blue whales *Balaenoptera musculus* off the northeast coast of Sri Lanka: distribution, feeding and individual identification. pp.247-58. *In*: Leatherwood, S. and Donovan, G.P. (eds). *Cetaceans and Cetacean Research in the Indian Ocean Sanctuary*. United Nations Environmental Programme, Nairobi, Kenya.

Aragones, L.V., Jefferson, T.A. and Marsh, H. 1997. Marine mammal survey techniques applicable in developing countries. *Asian Mar. Biol.* 14: 15-39.

Ballance, L.T. and Pitman, R.L. 1998. Cetaceans of the Western Tropical Indian Ocean: distribution, relative abundance, and comparisons with cetacean communities of two other tropical ecosystems. *Mar. Mammal Sci.* 14(3): 429-59.

Chantrapornsyl, S., Kinze, C.C., Leatherwood, S. and Prematunga, W.P. 1991. Notes on the genus *Kogia* in the northern Indian Ocean. pp.79-88. *In*: Leatherwood, S. and Donovan, G.P. (eds). *Cetaceans and cetacean research in the Indian Ocean Sanctuary*. United Nations Environment Programme Marine Mammal Technical Report. 287pp.

Clarke, R. 1982. An index of sighting conditions for surveys of whales and dolphins. *Rep. int. Whal. Commn* 32: 559-61.

de Silva, P.H.D.H. 1987. Cetaceans (whales, dolphins and porpoises) recorded off Sri Lanka, India, from the Arabian Sea and Gulf, Gulf of Aden and from the Red Sea. *J. Bombay Nat. Hist. Soc.* 84(3): 505-25.

Ilangakoon, A. 1997. Species composition seasonal variation, sex ratio and body length of small cetaceans caught off west, south-west and south coast of Sri Lanka. *J. Bombay Nat. Hist. Soc.* 94: 298-306.

International Whaling Commission. 1980. Chairman's Report of the Thirty-First Annual Meeting. *Rep. int. Whal. Commn* 30:25-41.

Jayaprakash, A.A., Nammalwar, P., Krishna Pillai, S. and Elayath, M.N.K. 1995. Incidental bycatch of dolphins at Fisheries Harbour, Cochin, with a note on their conservation and management in India. *J. Mar. Biol. Assoc. India* 37: 126-33.

Jayasankar, P., Anoop, B., Reynold, P., Afsal, V.V. and Rajagopalan, M. 2007. Species of a whale and an unknown fish sample identified using molecular taxonomy. *Indian J. Fish.* 54(3): 339-43.

Jefferson, T.A., Leatherwood, S. and Webber, M.A. 1993. *FAO Species Identification Guide. Marine Mammals of the World.* FAO, Rome. 320pp.

Jefferson, T.A., Webber, M. and Pitman, R.L. 2008. *Marine Mammals of the World: a Comprehensive Guide to their Indentification.* Academic Press, London. 573pp.

Kreb, D. and Budioao. 2005. Cetacean diversity and habitat preferences in tropical waters of East Kalimantan, Indonesia. *Raffles Bull. Zool.* 53(1): 149-55.

Kumaran, P.L. 2002. Marine mammal research in India – a review and critique of the methods. *Current Science* 83(10): 1210-20.

Lal Mohan, R.S.L. 1985. Observations on the by-catch of dolphins *Stenella longirostris*, *Tursiops aduncus*, *Sousa chinensis* and *Delphinus delphis tropicalis* in the gill nets off Calicut coast, India. *Proc. Symp. Endangered Mar. Anim. Mar. Parks* 1: 78-83. MBAI, Cochin, 508pp.

Leatherwood, S., McDonald, D., Prematunga, W.P., Girton, P., Ilangakoon, A. and McBrearty, D. 1991. Records of the 'Blackfish' (killer, false killer, pilot, pygmy killer and melon-headed whales) in the Indian Ocean,1772-1986. pp.33-65. *In*: Leatherwood, S. and Donovan, G.P. (eds). *Cetaceans and cetacean research in the Indian Ocean Sanctuary*. United Nations Environment Programme. Marine Mammal Technical Report, Nairobi, Kenya. 287pp.

Robbins, J., Carlson, C. and Palazzo Jr, J. 2006. Preliminary worldwide index of cetacean data collection from opportunistic platforms. Paper SC/58/WW3 presented to the IWC Scientific Committee, May 2006, St. Kitts and Nevis, West Indies (unpublished). 1pp. [Paper available at the Office of this Journal].

Robbins, J. and Mattila, D.K. 2000. The use of commercial whalewatching platforms in the study of cetaceans: benefits and limitations. Paper SC/52/WW8 presented to the IWC Scientific Committee, June 2000, in Adelaide, Australia (unpublished). 7pp. [Paper available from the Office of this Journal].

Sathasivam, K. 2006. Status of marine mammals of India. *J. Bombay Nat. Hist. Soc.* 103(2-3): 270-73.

Yousuf, K.S.S.M., Anoop, A.K., Anoop, B., Afsal, V.V., Vivekanandan, E., Kumarran, R.P., Rajagopalan, M., Krishnakumar, P.K. and Jayasankar, P. 2008. Observations on incidental catch of cetaceans in three landing centres along the Indian coast. *J. Mar. Biol. Ass. UK.* [Biodiversity Records. Published online].

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Effect of pingers on harbour porpoise (*Phocoena phocoena*) bycatch in the US Northeast gillnet fishery

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ABSTRACT

Harbour porpoise (*Phocoena phocoena*) bycatch in the US Northeast gillnet fishery is managed under the Harbour Porpoise Take Reduction Plan (HPTRP), which was implemented on 1 January 1999. The HPTRP divides this fishery into management areas that are either completely closed to all gillnets or closed only to gillnets that do not use pingers. Questions about pingers that have arisen include: (1) would pingers be as effective in an operational fishery as in controlled scientific experiments; (2) would the fishery comply with these regulations; and (3) would harbour porpoises habituate to pingers? To investigate these questions, data from over 25,000 gillnet hauls observed by the Northeast Fisheries Observer Program after the implementation of the HPTRP, 1999-2007, were examined. In a 1994 controlled scientific experiment conducted in part of this fishery that used 15cm mesh gillnets, the bycatch rate in pingered nets was 92% less than that in nets without pingers. In contrast, in the operational fishery, the bycatch reduction in pingered nets was 50-70%, depending on the time, area and mesh size. In particular, there was no observed bycatch in pingered nets that used the same mesh size as used in the experiment. Thus, it seem that the apparent decrease in pinger effectiveness in the operational fishery was partially due to the type of gillnet used and lack of compliance. Pinger usage started out high in 1999 (the first year required), dropped substantially during 2003-05 and perhaps due to outreach activities increased beginning in 2006. During years of high pinger usage, 87% of the tested pingers were functional, while only 36% of the tested pingers were functional during years of low pinger usage. In general, as expected, observed bycatch rates in hauls without pingers were greater than bycatch rates in hauls with the required number of pingers. Unexpectedly, bycatch rates of observed hauls with an incomplete set of pingers were higher that in observed hauls without pingers. Confounding factors that could partially explain this apparently contrary result are discussed. There was no evidence for temporal trends in the bycatch rates, suggesting that harbour porpoises had not habituated to the pingers. In conclusion, in the US Northeast gillnet fishery, harbour porpoises do not appear to have habituated to pingers, and pingers appear to have reduced the bycatch rate, particularly when the required number of pingers were used and in nets using mesh sizes of 15cm or less.

KEYWORDS: NOISE; GILLNETS; INCIDENTAL CATCHES; CONSERVATION; CATCH PER UNIT EFFORT; MONITORING; ATLANTIC OCEAN; NORTH AMERICA; SHORT-TERM CHANGE; MANAGEMENT REGULATIONS; SAMPLING TECHNIQUES

INTRODUCTION

During the Kraus *et al*. (1997) controlled scientific experiment, a 92% reduction in bycatch of harbour porpoises (*Phocoena phocoena*) was documented in gillnets that used a 15cm (6in) stretched mesh size, within the US Northeast Atlantic Mid-Coast management area, in autumn (October to December) 1994 (0.0591 harbour porpoises per haul in control nets versus 0.0048 harbour porpoises per haul in pingered nets). Based on the success of this experiment, the US National Marine Fisheries Service developed the Harbour Porpoise Take Reduction Plan (HPTRP)[1] to use pingers as one of the mitigation tools for reducing harbour porpoise bycatch in gillnet fisheries in the US portion of the Northwest Atlantic Ocean. The HPTRP was implemented on 1 January 1999 (NOAA, 1998).

The Northeast gillnet fishery is prosecuted in US waters east of 72°W and north of 40°N (Fig. 1) and targets Atlantic cod (*Gadus morhua*), monkfish (*Lophius americanus*), pollock (*Pollachius virens*) and various flounder species. The HPTRP divides this region into management areas (MA) that are either completely closed to all gillnets or closed only to gillnets that do not use pingers (Table 1). The HPTRP specifies that, when pingers are required, an operating and functional pinger must be attached at the end of each gillnet string and at the bridle of each net within that string, where a net is usually 92m (300ft) long. Thus, 11 pingers are required on a 10 net string. The HPTRP defined a pinger as an instrument which, when immersed in water, broadcasts a 10kHz (±2kHz) sound at 132dB (±4dB) re 1µPa at 1m, lasting 300ms (±15ms), and repeating every 4s (±2s).

Several concerns about pingers were raised during the development of the HPTRP and during an IWC review of pingers (IWC, 2000); namely, effectiveness, compliance and habituation. There was some concern expressed that pingers may not consistently reduce the bycatch of harbour porpoises, particularly to the levels demonstrated in experiments such as in Kraus *et al*. (1997) i.e. that harbour porpoise bycatch rates in operational fisheries might not be as low as in controlled scientific experiments. Other concerns centred over issues that (1) pingers might not be used properly (e.g. not the required number and/or no replacement of broken pingers or used batteries) with the result that in operational fisheries harbour porpoise bycatch might increase due to declining compliance; and (2) concern that harbour porpoises might become habituated to the sounds made by the pingers with the result that in operational fisheries harbour porpoise bycatch rates might increase with time, as shown elsewhere by Cox *et al*. (2001) and Carlström *et al*. (2009). To investigate these concerns, this paper examines data collected by the Northeast Fisheries Observer Program (NEFOP) to document patterns in harbour porpoise bycatch rates, levels of compliance to the pinger regulations and possible indications of habituation.

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[1] *http://www.nero.noaa.gov/prot_res/porptrp*

* *Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543, USA.*
+ *Northeast Fisheries Science Center, 28 Tarzwell Dr., Narragansett, RI 02882, USA.*

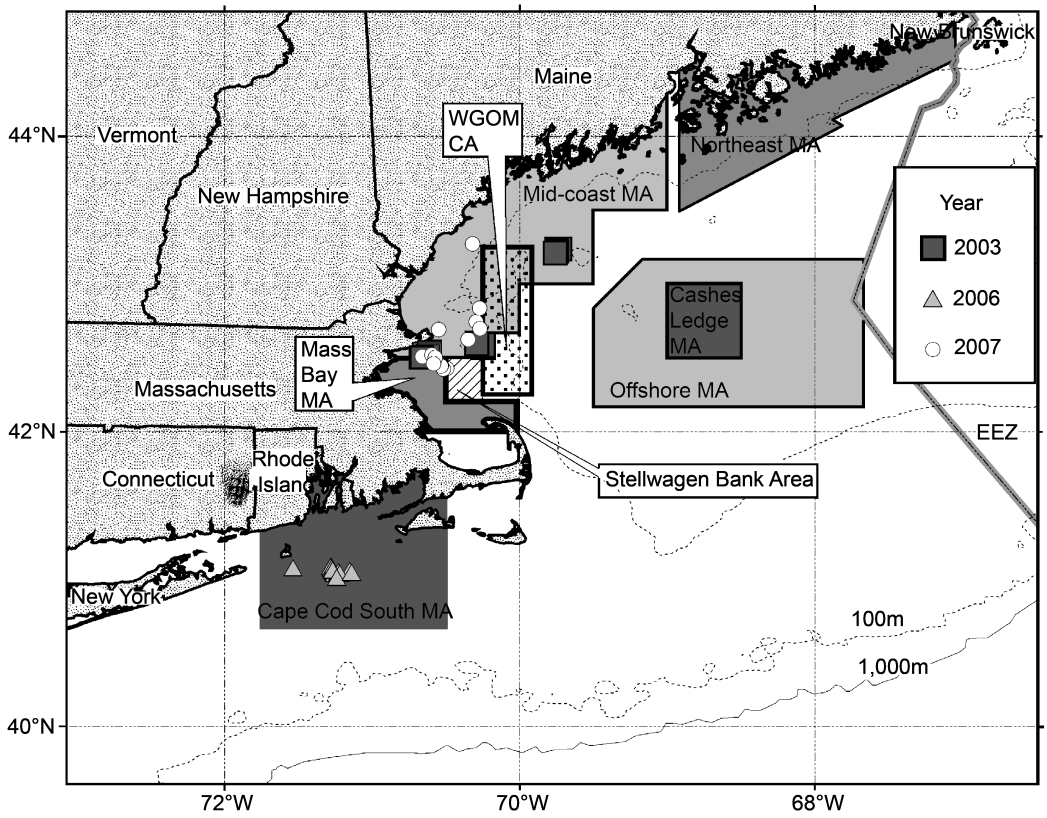

Fig. 1. Locations of the US Northeast gillnet fishery management areas (MA), closed area (CA) and other areas of interest. In addition, the locations of trips that had pinger performance tested are shown.

Table 1

Times and areas in the US Northeast gillnet fishery that are either closed to all gillnets (Closed) or else closed to all gillnets that do not use pingers (Closed – pingers allowed).

Area	Dates	Status of gillnets[1]
Northeast	15 Aug. – 13 Sep.	Closed
Mid-coast	15 Sep. – 31 May	Closed – pingers allowed
Massachusetts Bay	1 Dec. – 28/29 Feb.	Closed – pingers allowed
	1-31 Mar.	Closed
	1 Apr. – 31 May	Closed – pingers allowed
Offshore	1 Nov. – 31 May	Closed – pingers allowed
Cashes Ledge	1-28/29 Feb.	Closed
Western Gulf of Maine	All year round	Closed
Cape Cod South	1 Dec. – 28/29 Feb.	Closed – pingers allowed
	1-31 Mar.	Closed
	1 Apr.– 31 May	Closed – pingers allowed

[1] A pinger is defined as an acoustic deterrent device which, when immersed in water, broadcasts a 10kHz (±2kHz) sound at 132dB (±4dB) re 1 μPa at 1m, lasting 300ms (±15ms), and repeating every 4s (±2s).

DATA

Observers from the NEFOP collect data on characteristics of the trip, haul, gear, economic factors, catch and incidental bycatch. Trip characteristics include: vessel name and number; date sailed; date landed; home port; port fish landed; steam time; and number of crew. Haul characteristics include: weather conditions; wind speed and direction; wave height; depth range; latitude; longitude; time of the beginning and ending of the haul; soak duration; species targeted on each haul; presence and quantity of fish kept and discarded; and the number of incidental bycatch of cetaceans, seals, turtles and birds. Gear characteristics for gillnets include: mesh size; twine size; length of string; height of net; hang ratio; number of floats and weights; and length of tie downs. Economic factors related to the trip include: tons of ice used; fuel used; price of water, food, oil and bait; and damage costs. Observers identify both kept and discarded catch, and, on some trips, record the amount caught of each species. When an incidental bycatch occurs, information recorded includes: species identification; number of each species; condition of the body; body length; sex; tag number (if body is returned to the sea); and types of samples taken (body parts or whole animal). Only bycatches of harbour porpoises and observed hauls that have complete latitude-longitude information that were observed during 1 January 1999 to 31 May 2007 were used in this paper.

As interest arose on the use of pingers to deter marine mammal bycatch, the NEFOP modified their data collection protocols, logs, gear and training to include: whether active marine mammal deterrent devices (i.e. pingers) were used; how many were on the gear when set; frequency in kilohertz; whether it was salt-water activated; brand name of the pingers; number of pingers hauled back on the gear; and number of pingers lost as reported by the captain.

In addition to collecting the pinger information for all observed hauls, the NEFOP implemented a Pinger Tester Program in October 2003. NEFOP staff drafted a pinger tester datasheet, developed a tester and trained observers in the field to test whether or not pingers were functioning during observed trips. A dozen pinger testers were issued to observers during times and places where pingers were required to be used.

There are two types of sampling protocols that an observer may follow while on a gillnet trip, 'Limited' and 'Complete'. During a 'Complete' trip observers are more focused on fish sampling and discards whereas during a 'Limited' trip observers are focused exclusively on marine mammal incidental takes and pinger testing. If a gillnet trip within an area requiring pingers is a 'Limited' trip and the observer is equipped with a pinger tester, then the observer tests the performance of each pinger as the net is hauled on board by evaluating if a pinger is: (1) audible; (2) inaudible and tested; or (3) inaudible and not tested. If a gillnet trip is a 'Complete' trip, the observer primarily performs the fish sampling requirements; however, if a marine mammal take occurs, and the observer is equipped with a pinger tester, then the remaining pingers on that string are tested as they are hauled on board. In this paper, only the 'Limited' trips were used to investigate pinger performance. Both types of trips were used to investigate pinger usage, pinger effectiveness and habituation.

METHODS

Compliance

To document compliance with HPTRP complete closure regulations, the numbers of observed hauls within totally closed times and areas were summarised.

Pinger usage

To document compliance with the pinger usage regulations, percentages of observed hauls that used various quantities of pingers were summarised by year and management area. Pinger usage was grouped into four categories:

(1) all of the required number of pingers (given the number of nets per string);
(2) more than 50% but less than all of the required number of pingers;
(3) some pingers but less than or equal to 50% of the required number of pingers; and
(4) no pingers on a string.

The choice of categories (2) and (3) was to allow some investigation of the effect of various levels of incomplete pinger use whilst retaining sufficient sample sizes by category.

Pinger performance

To document compliance with the functional pinger regulation, pinger performance data that were collected by the NEFOP Pinger Tester Program (during 2003, 2006 and 2007) were evaluated by summarising the percentage of pingers that were audible and inaudible.

Pinger effectiveness

To document the effectiveness of pingers, in the times and areas where pingers were required, bycatch rates of harbour porpoises were estimated from hauls that used no pingers, some pingers and the required number of pingers. The bycatch rate was defined as the sum of observed dead harbour porpoises divided by the sum of the fishing effort on the observed hauls, where three proxies of fishing effort were investigated: metric tons (mtons) landed; hauls; and volume-soaked (the product of net length, net height and soak duration, in the units of km^2-days). Since the unit of effort required when estimating bycatch for the entire fishery (e.g. Belden, 2007; Belden and Orphanides, 2007) is mtons landed (Orphanides and Palka[2]), this unit of effort is the primary unit of effort presented in this paper, although the other units were also investigated.

The coefficient of variation (CV) of the bycatch rates were estimated using 1,000 bootstrapped iterations. First, for each iteration, the observer dataset was resampled by haul with replacement to result in a dataset with the same number of hauls as in the original dataset. Second, the bycatch rate from the new dataset was calculated. Finally, the CV was defined as the standard deviation of the 1,000 bycatch rates divided by the actual bycatch rate.

Reduction in bycatch rates of hauls with no pingers versus hauls with the required number of pingers was tested with a non-parametric one-tailed Mann-Whitney-Wilcoxon (MWW) two sample rank-sum test. The differences between bycatch rates of hauls with none, some and all of the required number of pingers were tested using the Kruskal-Wallis rank-sum test.

Habituation

To determine if habituation to pingers may have occurred in times and areas where pingers were required, the observer data were used to monitor bycatch rate trends. Long-term trends were defined as trends in the annual rates over the years since the implementation of the HPTRP. Short-term trends were defined as trends in the monthly rates within the time period that pingers were required within a management area. To determine if bycatch rates increased over time on a short- or long-term basis, bycatch rates were summarised and modelled using generalised linear and additive models (GLMs and GAMs), where the numbers of harbour porpoise takes in a haul were regressed against the year (or month), percentage of pinger usage, and an offset of the effort (mtons landed), using a quasi-Poisson model.

RESULTS

Since the implementation of the HPTRP (1 January 1999 to 31 May 2007), about 25,400 gillnet hauls have been observed in the Northeast gillnet fishery, of which about 20,750 hauls were in the Gulf of Maine and about 4,650 were south of Cape Cod (Table 2A). The Mid-Coast MA, Massachusetts Bay (Mass Bay) MA and the area outside all GOM management areas had the most observed hauls. The Cashes Ledge MA and Western Gulf of Maine (WGOM) Closed Area, both officially closed to fishing for specified

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[2] Though all three of the above units of effort are available for the observed hauls, mtons landed is the only reliable unit of effort available to expand the sample bycatch rate to the entire fishery. Mtons landed may be considered a non-standard measure of fishing effort, though in the case of harbour porpoise bycatch in the previously observed gillnet fishery, mtons landed is a valid unit of effort when using ratio estimation methods because the underlying assumptions of the methodology are valid. That is, as the mtons landed increases so does the number of observed dead harbour porpoises. For more details refer to: *Orphanides, C. and Palka, D. 2007. Landings: the unit of effort for bycatch rates in gillnet fishing gear. Presented to the Harbour Porpoise Take Reduction Team in December 2007. Available from the author.*

Table 2

By month and management area (MA): (A) number of harbour porpoise takes and observed hauls and (B) resulting bycatch rates (harbour porpoises per mtons landed). In addition, for each MA, the resulting bycatch rates using three different units of effort (C). Data used from 1 January 1999 to 31 May 2007. Dark shaded cells indicate the times/areas when pingers are required. Light shaded cells with bold numbers indicate the times/areas that are closed to all gillnets.

Month	Cashes Ledge	Mass Bay	Mid-coast	Offshore	WGOM	Stellwagen Bank	Outside GOM MAs	TOTAL GOM	CC South MA	Outside CC South MA	TOTAL south of Cape Cod
A. Number of observed takes (Number of observed hauls)											
1999 to 2007											
Jan.	0 (0)	0 (273)	1 (139)	0 (55)	**1 (67)**	3 (377)	0 (433)	5 (1,344)	2 (197)	1 (60)	3 (257)
Feb.	**0 (21)**	0 (193)	0 (177)	0 (79)	**2 (98)**	7 (396)	0 (281)	9 (1,245)	11 (179)	6 (111)	17 (290)
Mar.	0 (13)	**1 (61)**	3 (199)	0 (95)	**1 (153)**	2 (499)	1 (370)	8 (1,390)	**0 (3)**	7 (155)	7 (158)
Apr.	0 (13)	0 (0)	0 (30)	0 (145)	**0 (0)**	0 (0)	5 (649)	5 (837)	1 (270)	33 (303)	34 (573)
May	0 (0)	0 (326)	0 (5)	0 (77)	**0 (22)**	0 (200)	1 (614)	1 (1,244)	9 (739)	16 (452)	25 (1,191)
1999 to 2006											
Jun.	2 (24)	0 (721)	1 (195)	0 (124)	**0 (29)**	0 (286)	0 (639)	3 (2,018)	0 (0)	0 (335)	0 (335)
Jul.	0 (0)	0 (397)	2 (944)	0 (127)	**0 (60)**	0 (210)	0 (630)	2 (2,368)	0 (0)	0 (266)	0 (266)
Aug.	0 (30)	0 (488)	1 (888)	0 (82)	**0 (67)**	0 (214)	0 (746)	1 (2,515)	0 (0)	0 (158)	0 (158)
Sep.	0 (36)	2 (496)	4 (728)	0 (46)	**0 (67)**	0 (252)	0 (806)	6 (2,431)	0 (0)	0 (190)	0 (190)
Oct.	0 (53)	0 (38)	11 (790)	0 (150)	**0 (60)**	0 (5)	0 (722)	11 (1,818)	0 (0)	0 (357)	0 (357)
Nov.	0 (18)	2 (95)	26 (925)	0 (83)	**5 (74)**	0 (5)	1 (681)	34 (1,881)	0 (0)	0 (508)	0 (508)
Dec.	0 (13)	3 (466)	7 (325)	0 (76)	**1 (31)**	6 (280)	2 (476)	19 (1,667)	1 (277)	0 (85)	1 (362)
TOTAL	2 (221)	8 (3,554)	56 (5,345)	0 (1,139)	10 (728)	18 (2,724)	10 (7,047)	104 (20,758)	24 (1,665)	63 (2,980)	87 (4,645)
B. Bycatch rate (number of observed takes/observed mtons of landing)											
1999 to 2007											
Jan.	0	0	0.045	0	**0.082**	0.047	0	0.022	0.047	0.038	0.044
Feb.	**0**	0	0	0	**0.192**	0.192	0	0.056	0.558	0.069	0.16
Mar.	0	**0.267**	0.156	0	**0.047**	0.037	0.017	0.038	**0**	0.068	0.065
Apr.	0	0	0	0	**0**	0	0.069	0.04	0.025	0.170	0.145
May	0	0	0	0	**0**	0	0.01	0.005	0.051	0.084	0.068
1999 to 2006											
Jun.	0.332	0	0.032	0	**0**	0	0	0.007	0	0	0
Jul.	0	0	0.01	0	**0**	0	0	0.003	0	0	0
Aug.	0	0	0.006	0	**0**	0	0	0.002	0	0	0
Sep.	0	0.023	0.028	0	**0**	0	0	0.01	0	0	0
Oct.	0	0	0.066	0	**0**	0	0	0.023	0	0	0
Nov.	0	0.052	0.121	0	**0.145**	0	0.005	0.066	0	0	0
Dec.	0	0.043	0.071	0	**0.095**	0.079	0.018	0.044	0.013	0	0.010
TOTAL	0.023	0.016	0.052	0	0.056	0.04	0.005	0.022	0.066	0.041	0.062
C. Bycatch rates using different units of effort for entire time period											
Mtons landed	0.023	0.016	0.052	0	0.056	0.040	0.005	0.022	0.066	0.041	0.062
Hauls	0.009	0.002	0.010	0	0.014	0.007	0.001	0.005	0.014	0.021	0.019
Volume-soaked (km^2-days)	0.641	0.527	2.024	0	2.317	1.134	0.256	0.911	1.837	1.090	1.953

time periods, had the lowest number of observed hauls. There were no hauls observed in the Northeast MA at any time in the year because fishing in this area has nearly stopped; thus further discussions do not include the Northeast MA.

Compliance

Few hauls were observed (Table 2A) in the times and areas that were totally closed to gillnets due to the implementation of the HPTRP (Table 1). However, within the WGOM Closed Area (closed all year round for fish conservation reasons) there was observed gillnet fishing during nearly every month; observed hauls were usually very close to a border, in particular the western inshore border (Fig. 2); and harbour porpoise bycatch was observed from November through March (Table 2A).

Pinger usage

Pinger usage dropped substantially in 2003 and started increasing again in 2006 (Fig. 3). This pattern occurred in all management areas (Fig. 4). To allow for the situations where one pinger may have died or accidently fallen off during the time the net was under water, compliance in Fig. 4 was expressed as observed hauls with more than 90% of the required number of pingers, which is the result of the commonly used 10-net string missing one pinger.

Pinger performance

Of the 42 observers trained to use the pinger tester, eight actually collected data on pinger performance. Sixty-nine gillnet strings, with a total of 813 pingers, were tested (Fig. 1). Most of these trips were since 2006: 12 trips were in 2003; 15 trips in 2006; and 42 in 2007. Of the 813 pingers examined, 346 (43%) were audible by ear and thus not tested; 109 (13%) were not audible by ear and were not tested; 307 (38%) were not audible by ear, were tested and determined to be working properly; and 51 (6%) were not audible by ear, were tested and determined to be not working.

Thus, over all years at least 80% of the tested pingers were working and perhaps as many as 93% were working (as estimated by including only those pingers that were tested or audible by ear). During 2003, a year with low pinger usage (Figs 3 and 4), 113 pingers were investigated

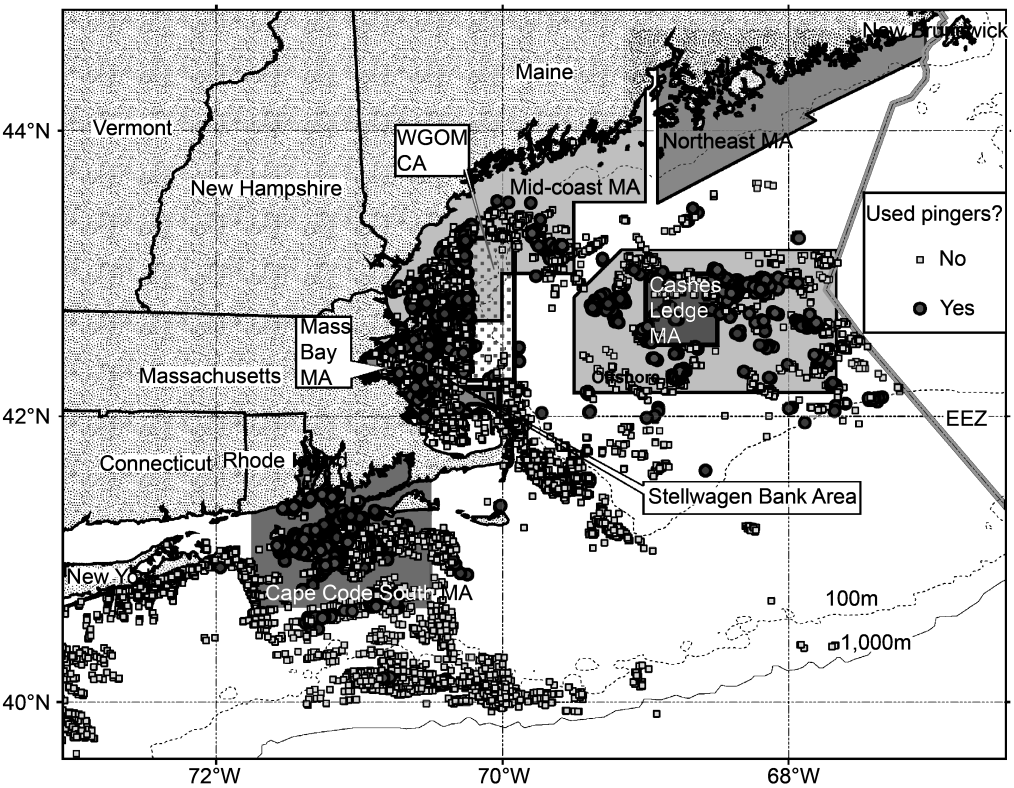

Fig. 2. Location of hauls that did not use pingers (small light squares) and hauls that did use pingers (large dark circles) from 1 January 1999 through 31 May 2007.

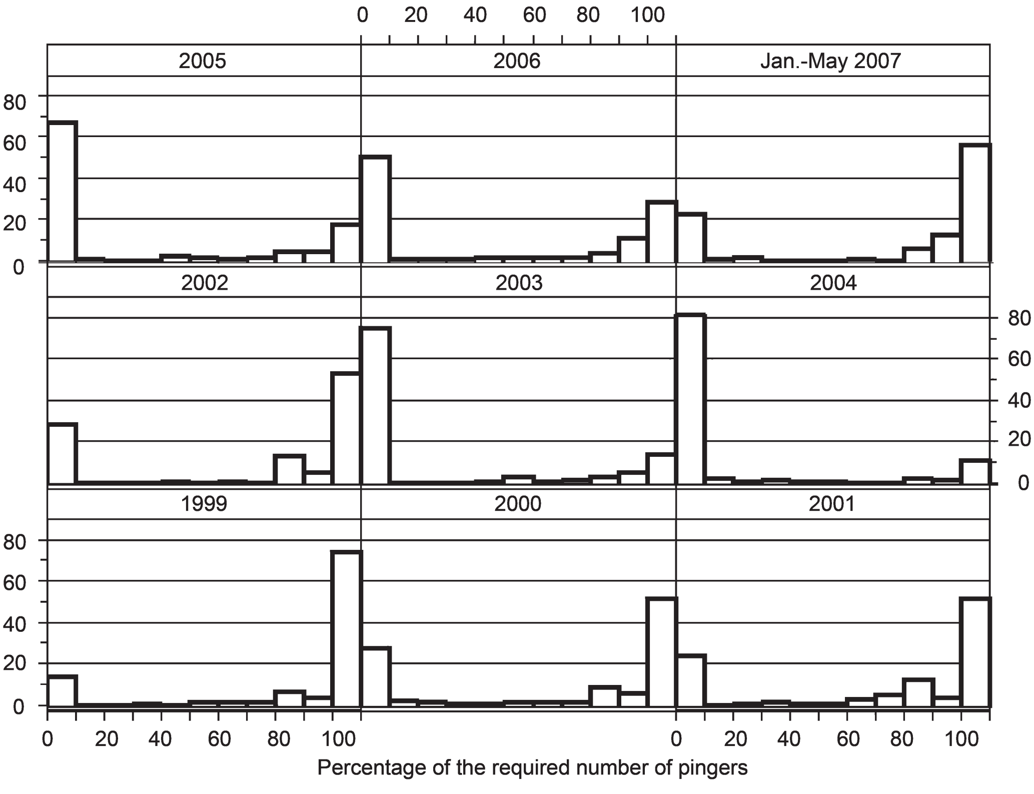

Fig. 3. The distribution, by year, of the number of pingers used per string in the US Northeast gillnet fishery during times and areas that pingers were required (1 January 1999 to 31 May 2007), where the number of pingers used was normalised by the number of pingers required for the length of that string. For example, the regulations state that if a gillnet string consists of 10 nets then 11 pingers are required. If that 10-net string was an observed haul and there were 11 pingers on the string, then it had 100% of the required number of pingers, while if that string did not have any pingers, then it had 0% of the required number of pingers.

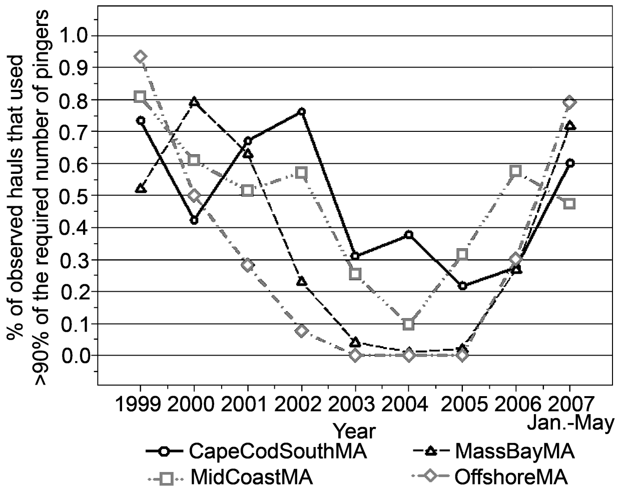

Fig. 4. During 1 January 1999 to 31 May 2007, by management area, the percentage of observed hauls that used more than 90% of the required number of pingers, during times pingers were required.

and only 36% were working. In contrast, in 2006-07 when pinger usage was high, 700 pingers were investigated and 87% were working.

Pinger effectiveness

Since the implementation of the HPTRP, harbour porpoise bycatch rates (Table 2B and 2C) differed by area. The highest bycatch rates (no matter which unit of effort was used in the definition of the bycatch rate) were in the Cape Cod South (CCSouth) MA, WGOM Closed Area and Mid-Coast MA. The next highest bycatch rates were within the Stellwagen Bank Area and the area south of Cape Cod but outside the CCSouth MA.

Pooling over all years and management areas since the implementation of the HPTRP for those times and areas for which pingers were required, the bycatch rate (harbour porpoises per mtons landed) of hauls without pingers was about two to three times the rate of hauls with the required number of pingers (Table 3A); this was a significant difference (p-value=0.0048). Significant differences were also observed (Table 3B) when the bycatch rate was defined as harbour porpoises per haul (p-value=0.0054) and harbour porpoises per volume-soaked (km^2-days; p-value=0.0052). A similar pattern was evident within each area when pooled over years (Fig. 5), and within most individual years when pooled over areas (Fig. 6).

Harbour porpoise bycatch rates of hauls with an incomplete set of pingers were usually two to three times the bycatch rates of hauls without pingers, for each area pooled over years (Fig. 5) and for each year pooled over areas (Fig. 6). These differences were significant for each area (Table 3A), except for the WGOM Closed Area (where pingers are not required), according to the MWW test. These differences were also significant when the bycatch rate was defined as harbour porpoises per haul or as harbour porpoises per volume-soaked (Table 3B) and when comparing the three levels using the Kruskal-Wallis rank-sum test.

To gain further insight about the characteristics of hauls with an incomplete set of pingers, the percent of required pingers, spatial distribution, and mesh size were explored. In nearly every year, bycatch rates of hauls with some pingers, but less than or equal to 50% of the required number of pingers, was greater than or equal to the bycatch rate of hauls with more than 50% but less than 100% of the required number of pingers (Fig. 6). Bycatch rates of hauls with an incomplete set of pingers was high in all areas when the data were pooled over the entire time period (Fig. 5), but the location of these hauls were not spatially clustered, even when looking at each year individually (figures not shown). That is, the locations of hauls with none or some pingers were not spatially aggregated. In 15cm mesh gillnets that were in the operational fishery (661 observed hauls), which is the same size mesh used in the Kraus *et al*. (1997) controlled scientific experiment, there was no observed harbour porpoise bycatch. The general pattern observed in the operational fishery was, as mesh size increased so did the bycatch rate, no matter how many pingers were on the gillnet; although bycatch rates in nets with pingers were still

Table 3

Comparison of bycatch rates (harbour porpoises per mtons landed) and number of observed hauls (n haul) with no pingers, some pingers and the required number of pingers for various areas in the US Northeast gillnet fishery after the implementation of the HPTRP (A). For all MAs, comparison of bycatch rates that are defined with three units of effort for hauls with no pingers, some pingers, and the required number of pingers (B).

A. Bycatch (Byc) rates using mtons landed as the unit of effort

	No pingers			Some pingers			Required number of pingers		
Area	Byc rate	%CV	n hauls	Byc rate	%CV	n hauls	Byc rate	%CV	n hauls
all MAs	0.053*	19.9	3,157	0.120+	20.9	1,065	0.024	35.1	2,407
Mid-coast MA	0.084*	25.6	1,287	0.130+	23.1	670	0.041	40.1	1,057
Mass Bay MA	0.009*	101.4	927	0.524+	63.7	39	0	0	353
CC South MA	0.075*	29.4	660	0.139+	53.4	262	0.023	71.9	743
Offshore MA	0	0	269	0	0	92	0	0	249
Cashes Ledge MA	0	0	14	0	0	2	0	0	5
Stellwagen Bank	0.074*	26.7	1,371	0.238+	72	68	0	0	118
WGOM CA	0.099	49.9	212	0.131+	42	149	0.034	1.0	122

B. Bycatch rates pooled over all MAs using different units of effort

	No pingers		Some pingers		Required number of pingers	
Unit of effort	Byc rate	% CV	Byc rate	% CV	Byc rate	% CV
Mtons landed	0.053*	19.9	0.120+	20.9	0.024	35.1
Number of hauls	0.0114*	19.7	0.0301+	21.5	0.0046	34.3
Volume soaked (km^2-day)	0.00016*	19.6	0.00041+	21.1	0.00008	34.8

*When comparing no pingers versus required number of pingers p-value < 0.05. +When comparing no pingers versus some pingers p-value < 0.05.

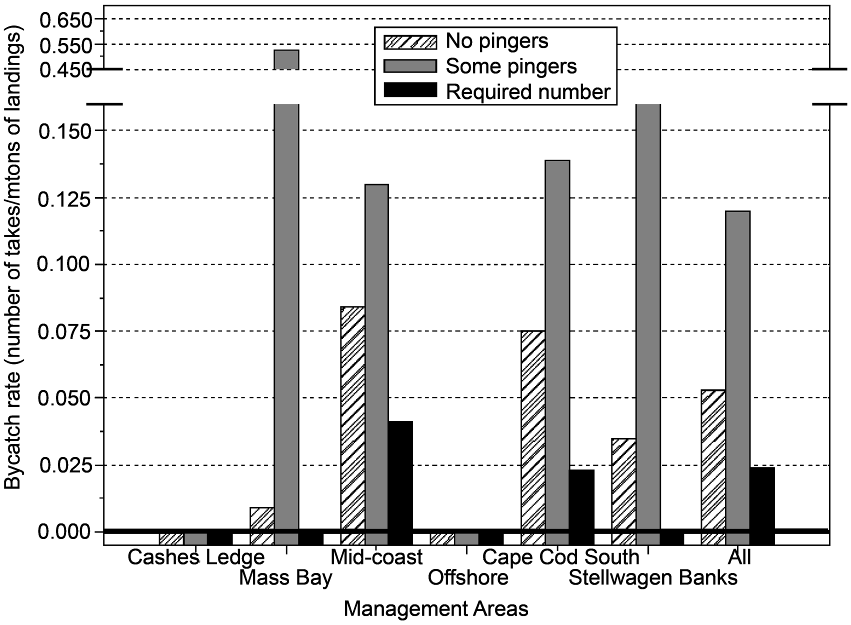

Fig. 5. Within each management area, pooled over all years, bycatch rates (harbour porpoises per mtons landed) of hauls that had no pingers (0%), some pingers (1-99%) and the required number of pingers (100%).

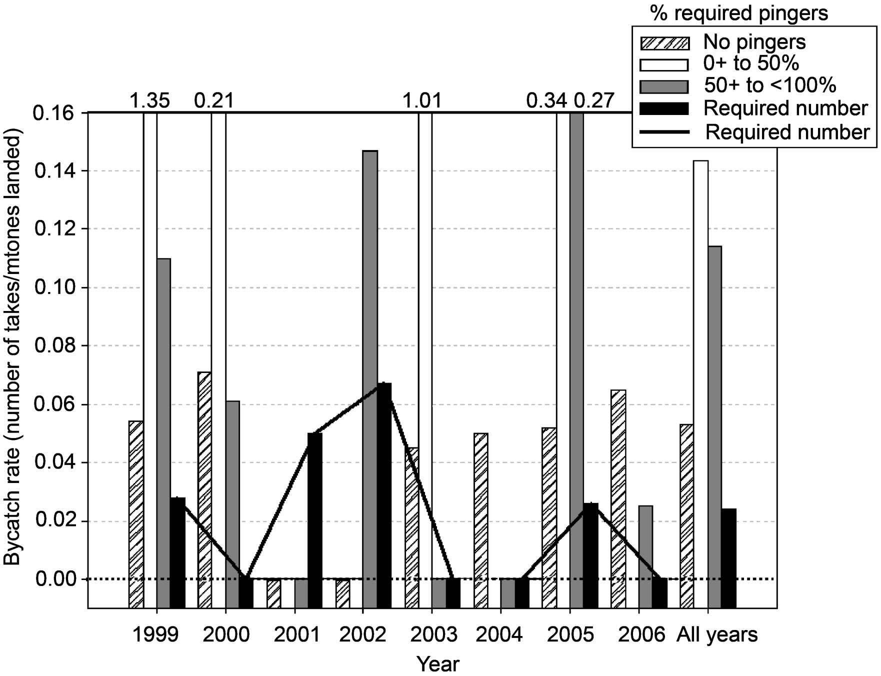

Fig. 6. For each year, pooled over all managed times and areas in the US Northeast gillnet fishery, bycatch rates (harbour porpoises per mtons landed) of hauls that had no pingers (0%), less than half of the required number of pingers (0+ to 50%), more than half of the required number of pingers (50+ to <100%) and the required number of pingers (100%). The black line connects the bycatch rates of hauls with the required number of pingers for each year.

less then that in nets without pingers, no matter what mesh size (Table 4 which uses the unit of effort (hauls) in the bycatch rate as reported in Kraus *et al.,* 1997).

Habituation

Harbour porpoise bycatch rates of hauls with pingers fluctuated from year to year in each management area (Fig. 5). There was no evidence of a long-term trend over years (line in Fig. 6) or a short-term trend over months (Fig. 7), using either landings or hauls as the unit of effort. When the bycatch rates were modelled with a GLM that included the percentage of required number of pingers as a covariate, there was no evidence of a significant slope over years (slope=0.064, SE=0.068, *t*-value=0.943) or over months (slope=-0.065, SE=0.065, *t*-value=-1.008). This same conclusion resulted when modelling just the hauls with all the required number of pingers or when using GAMs to model the bycatch rates.

Table 4

Comparison of bycatch rates (harbour porpoises per haul) and number of observed hauls (n haul) with no pingers, some pingers and the required number of pingers for gillnets with different mesh sizes.

	No pingers			Some pingers			Required number of pingers		
Mesh size (inches)	Byc rate	% CV	n hauls	Byc rate	% CV	n hauls	Byc rate	% CV	n hauls
3+ thru 6#	0	0	98	0	0	168	0	0	431
6+ thru 7	0.005*	35.2	1648	0.032+	24.8	508	0.003	49.9	1187
7+ thru 10	0.007*	51.6	733	0.047+	43.4	148	0	0	276
10+ thru 14	0.034*	26.1	678	0.037	52.9	241	0.014	47.3	512

Of the 697 observed hauls in the 3+ thru 6 mesh size category, 8 observed hauls used 3in, 7 used 5in, 3 used 5.25in, 18 used 5.5in and 661 observed hauls used 6in. *When comparing no pingers versus required number of pingers p-value < 0.05. +When comparing no pingers versus some pingers p-value < 0.05.

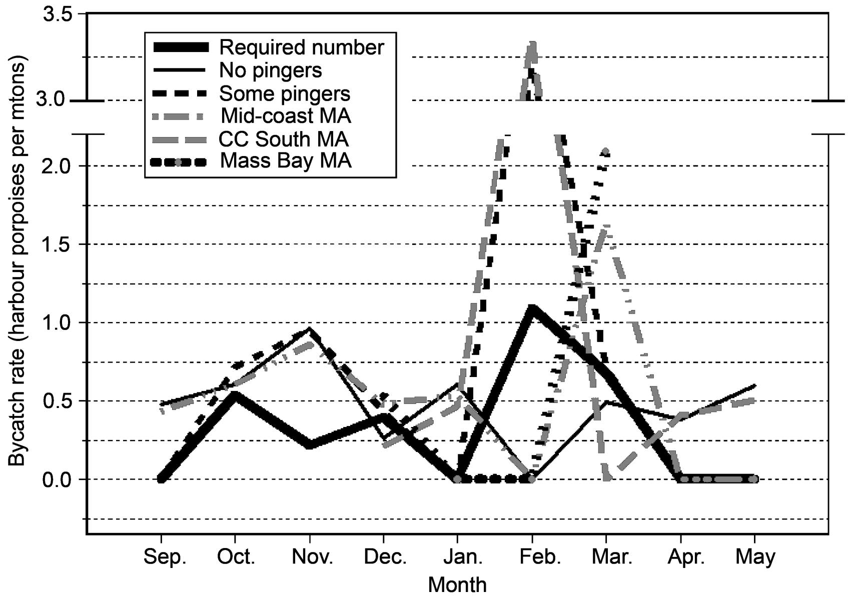

Fig. 7. Bycatch rates (harbour porpoises per mtons landed) during the times pingers were required by month for different percentages of pingers (all the required number of pingers; some pingers; and no pingers) and for the Mid-Coast MA, Cape Cod South MA and Mass Bay MA.

DISCUSSION

Compliance

Pinger usage

Pinger usage varied greatly from year to year. Perhaps one of the reasons why the pinger usage increased in 2006 and 2007 was that during October 2006 to January 2007 NOAA Fisheries Northeast Regional Office conducted an outreach program where they presented outreach materials in ports from New Jersey to Maine to remind industry of the HPTRP requirements and educate them on bycatch and pinger maintenance. When comparing the percentage of pinger usage by season, the overall fall rate of usage doubled from 20% in autumn (September to December) 2005 (before the outreach project) to 40% usage in autumn 2006 (after the outreach project); and the overall winter rate of usage jumped from 3% in winter (January to May) 2006 (before the outreach project) to 58% in winter 2007 (after the outreach project).

Pinger performance

Pinger tester data collection was sporadic because of challenges in implementing the Pinger Tester Program. At the beginning of the program, in 2003, many of the observers with testers were on fishing trips that did not use pingers, although pingers were required. During the development of the program, observers encountered the challenge of testing saltwater activated pingers that were not immersed in enough water to activate the signal. So, a field was added to the datasheet to specify whether or not the pingers were saltwater activated, and the observers were instructed to ensure that the pingers were wet when tested. In addition, the testers were not sufficiently robust under field conditions and often were non-functional when the observer had an opportunity to use it. In response, the carrying case for the tester was weather-proofed, observers were supplied with ample replacement batteries, and provided with a live pinger that they could use to determine if the tester was functioning properly. In subsequent versions of the tester, engineers added more padding to the internal wiring, which slightly improved the durability. After attempting to design more durable testers, four new testers are presently being used in the field. These modifications should provide more data on the performance of pingers from a variety of ports and seasons, thus providing a more representative sample of the fishery.

If pingers were tested on a random, representative sample of the fishery, it should be possible to estimate bycatch rates of hauls that had various percentages of functional pingers. This information could then be used to improve the bycatch estimates and might be used to determine an optimal number of pingers to reduce bycatch and minimise the overhead costs to the fishery.

Pinger effectiveness

Field studies worldwide concluded that pingers and 'acoustic harassment devices' (AHDs) can reduce bycatch of harbour porpoises and other small cetaceans (Barlow and Cameron, 2003; Culik *et al*., 2001; Gearin *et al*., 2000; Johnston, 2002; Johnston and Woodley, 1998; Kraus *et al*., 1997; Laake *et al*., 1998; Larsen and Krog, 2007; Lien *et al*., 1995; Morton and Symonds, 2002; Olesiuk *et al*., 2002; Trippel *et al*., 1999). The present paper provides additional support that pingers can reduce harbour porpoise bycatch, even in an operational fishery.

Since the implementation of the HPTRP, there were no observed takes in gillnets with mesh sizes of 15cm, the size used in the controlled scientific experiment. All of the observed bycatch was in nets using >15cm mesh sizes. Thus, the bycatch reduction documented in the controlled scientific experiment appears to also be true for the operational fishery. However, in the operational fishery, a variety of average mesh sizes are normally used, 13.3-35.5cm (5.25-14in) and the bycatch rates appear to depend on the mesh size. Thus, it appears that additional factors not tested in the controlled experiment also influence the bycatch rate. To lend further support to this conclusion, Palka *et al*. (In press) documented that the bycatch rate in the Gulf of Maine since the implementation of the HPTRP can be modelled using a quasi-Poisson distribution by the following variables: management area, sea surface temperature (SST), North Atlantic Oscillation (NAO) value, mesh size and lead line weight. That is, environmental factors and mesh size appear to influence the bycatch rate, in addition to the use of pingers.

Harbour porpoise bycatch rates in hauls with an incomplete set of pingers had a much higher bycatch rate than hauls without pingers or hauls with the required number of pingers, no matter what the mesh size, area or year. One possible reason for this pattern is that gillnets with an incomplete set of pingers may also have had more non-functioning pingers, but this information was not previously collected. However, this still would not fully explain why the rate in strings with some pingers was higher than the rate of hauls without any pingers. Two other possible reasons for this pattern, that are explored below, are gaps between the functional pingers and other gear/environmental factors.

The first hypothesis is that harbour porpoises may 'interpret' a gap in pingers to be a gap in the net, and thus try to swim into this 'gap' but rather become entangled in an unpingered portion of the gillnet. However Larsen and Krog (2007) found that harbour porpoise bycatch reduction was still evident when pingers were 455m and 585m apart in the Danish hake gillnet fishery, where the pingers had an harmonic energy bandwidth from 20-160kHz and a source level of 136-145dB re. 1μPa @ 1m. Although the wide spacings in the Danish fishery may perhaps mimic missing or non-functional pingers in the US Northeast fishery, the quite different pinger specifications in the US Northeast fishery (10kHz, no harmonics, 132dB and about 92m apart) preclude the application of the Danish results to the US Northeast fishery. In order to investigate this hypothesis in the US Northeast fishery, more detailed information must be collected on the functionality of pingers on nets surrounding a take and the distance between operational pingers. This type of information is now being collected by the NEFOP and can be explored in the future.

The second hypothesis is that, perhaps by chance, the hauls with an incomplete set of pingers may have different environmental/gear characteristics to those with none or all the required pingers. As stated earlier, Palka *et al*. (In press) illustrated that the bycatch rate in the Gulf of Maine since the implementation of the HPTRP can be modelled by the management area, SST, NAO value, mesh size and lead line weight. The bycatch rates of hauls with characteristics that relate to higher than average bycatch rates (Mid-Coast MA, ≤12.5°C SST, ≤0.15 NAO value and mesh sizes ≥ 17.8cm (7in)) were 0.0602 harbour porpoises per haul (from 216 hauls) in nets with no pingers, 0.0506 harbour porpoises per haul (from 178 hauls) in nets with some pingers and 0.0101 harbour porpoises per haul (from 199 hauls) in nets with the required number of pingers. The same pattern of bycatch rates occurred if the unit of effort in the bycatch rate was defined as mtons landed or volume soaked. So, for this subset of hauls with similar environmental/gear characteristics, there was an 83% bycatch rate reduction due to use of the required number of pingers, and the bycatch rate of hauls with some pingers was intermediate between the bycatch rate of hauls with no pingers and with all of the required number of pingers. Thus, it appears that the reason(s) for the bycatch rate reduction may not be fully understood until the mechanisms as to how pingers reduce bycatch and the relationship between bycatch rates and environmental/gear characteristics are more completely understood.

Habituation

The pattern of bycatch rates over months in management areas where pingers are required did not show an increasing trend, as would be expected if habituation occurred. Instead monthly bycatch rates appear to track the migration of the harbour porpoises, i.e. a northerly direction in the spring and a southerly direction in autumn.

In addition, there was no evidence of an increase in the annual bycatch rates since the implementation of pinger use, as would be expected had habituation occurred. This may reflect the fact that harbour porpoises in the area are not continuously exposed to pingers for the following reasons: (1) pingers are not used continuously in any one area (Table 1); (2) pingers do not emit sounds into a large region around each pinger; (3) during some times of the year harbour porpoises inhabit areas without pingers; and (4) even within a season, porpoises migrate through all these management areas, and move considerably within the Gulf of Maine and Canadian Bay of Fundy as shown by tracked animals (Read and Westgate, 1997).

However, it should be recognised that the NEFOP data do not provide a direct method to study habituation. A better approach may be to conduct a long-term study of the behaviour of animals around a pingered gillnet (Carlström *et al*., 2009; Cox *et al*., 2003; Culik *et al*., 2001; Laake *et al*., 1998) or around a single pinger (e.g. Cox *et al*., 2001), but such studies have not yet occurred in the US Northeast fishery. However, the NEFOP data do provide an indication that habituation has not occurred on a level that affects the bycatch estimate for an operational fishery.

ACKNOWLEDGEMENTS

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REFERENCES

Barlow, J. and Cameron, G.A. 2003. Field experiments show that acoustic pingers reduce marine mammal bycatch in the California drift gill net fishery. *Mar. Mammal Sci.* 19(2): 265-83.

Belden, D. 2007. Estimates of cetacean and pinniped bycatch in the 2005 northeast sink gillnet and mid-Atlantic coastal gillnet fisheries. *NEFSC Lab. Ref. Doc.* 07-08: 16pp. [Available from: *http://www.nefsc.noaa.gov/publications/crd/crd0708*].

Belden, D. and Orphanides, C.D. 2007. Estimates of cetacean and pinniped bycatch in the 2006 northeast sink gillnet and mid-Atlantic coastal gillnet fisheries. *NEFSC Lab. Ref. Doc.* 07-20: 18pp. [Available from: *http://www.nefsc.noaa.gov/publications/crd/crd0720*].

Carlström, J., Berggren, P. and Tregenza, N.J.C. 2009. Spatial and temporal impact of pingers on porpoises. *Can. J. Fish. Aquat. Sci.* 66: 72-82.

Cox, T.M., Read, A.J., Solow, A. and Tregenza, N. 2001. Will harbour porpoises (*Phocoena phocoena*) habituate to pingers? *J. Cetacean Res. Manage.* 3(1): 81-86.

Cox, T.M., Read, A.J., Swanner, D., Urian, K. and Waples, D. 2003. Behavioural responses of bottlenose dolphins, *Tursiops truncatus*, to gillnets and acoustic alarms. *Biol. Conserv.* 115: 203-12.

Culik, B.M., Koschinski, S., Tregenza, N. and Ellis, G.M. 2001. Reactions of harbour porpoises (*Phocoena phocoena*) and herring (*Clupea harengus*) to acoustic alarms. *Mar. Ecol. Prog. Ser.* 211: 255-60.

Gearin, P.J., Gosho, M.E., Laake, J., Cooke, L., Delong, R.L. and Hughes, K.M. 2000. Experimental testing of acoustic alarms (pingers) to reduce bycatch of harbour porpoise, *Phocoena phocoena*, in the state of Washington. *J. Cetacean Res. Manage.* 2(1): 1-10.

International Whaling Commission 2000. Report of the Scientific Committee. Annex I. Report of the Sub-Committee on Small Cetaceans. *J. Cetacean Res. Manage. (Suppl.)* 2:235-57.

Johnston, D.W. 2002. The effect of acoustic harassment devices on harbour porpoises (*Phocoena phocoena*) in the Bay of Fundy, Canada. *Biol. Conserv.* 108: 113-18.

Johnston, D.W. and Woodley, T.H. 1998. A survey of acoustic harassment device (AHD) use in the Bay of Fundy, NB, Canada. *Aquat. Mamm.* 24: 51-61.

Kraus, S.D., Read, A.J., Solow, A., Baldwin, K., Spradlin, T., Anderson, E. and Williamson, J. 1997. Acoustic alarms reduce porpoise mortality. *Nature* 388: 525.

Laake, J., Rugh, D. and Baraff, L. 1998. Observations of harbor porpoise in the vicinity of acoustic alarms on a set gill net. *NOAA Tech. Mem. NMFS-AFSC-* 84: 40.

Larsen, F. and Krog, C. 2007. Fishery trials with increased pinger spacing. Paper SC/59/SM2 presented to the IWC Scientific Committee, May 2007, Anchorage, USA (unpublished). 8pp. [Paper available from the Office of this Journal].

Lien, J., Hood, C., Pittman, D., Ruel, P., Borggaard, D., Chisholm, C., Wiesner, L., Mahon, T. and Mitchell, D. 1995. Field tests of acoustic devices on groundfish gillnets: assessment of effectiveness in reducing harbour porpoise bycatch. pp.1-22. *In*: Kastelein, R.A., Thomas, J.A. and Nachtigall, P.E. (eds). *Sensory Systems of Aquatic Mammals*. De Spil Publishers, Woerden.

Morton, A.B. and Symonds, H.K. 2002. Displacement of *Orcinus orca* (L.) by high amplitude sound in British Columbia, Canada. *ICES J. Mar. Sci.* 59: 71-80.

NOAA. 1998. Taking of marine mammals incidental to commercial fishing operations; Harbor porpoise take reduction plan regulations. *Federal Register Notice* 63(231): 464-90. [Department of Commerce, National Oceanic and Atmospheric Administration 50CFR Part 229].

Olesiuk, P.F., Nichol, L.M. and Ford, J.K.B. 2002. Effect of the sound generated by an acoustic harassment device on the relative abundance and distribution of harbour porpoises (*Phocoena phocoena*) in Retreat Passage, British Columbia. *Mar. Mammal Sci.* 18(4): 843-62.

Palka, D.L., Orphanides, C.D. and Warden, M.L. In press. Summary of harbor porpoise bycatch, covariates and levels of compliance in the northeast and mid-Atlantic gillnet fisheres after the implementation of the Take Reduction Plan: January 1 1999 through May 31 2007. *NOAA Technical Memorandum NMFS*. [Will be available at *http://www.nefsc.noaa.gov/nefsc/publications/tm/tmlist.htm*].

Read, A.J. and Westgate, A.J. 1997. Monitoring the movements of harbour porpoises (*Phocoena phocoena*) with satellite telemetry. *Mar. Biol.* 130: 315-22.

Trippel, E.A., Strong, M.B., Terhune, J.M. and Conway, J.D. 1999. Mitigation of harbour porpoise (*Phocoena phocoena*) by-catch in the gillnet fishery in the lower Bay of Fundy. *Can. J. Fish. Aquat. Sci.* 56: 113-23.

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Effect of acoustic deterrents on the behaviour of common dolphins (*Delphinus delphis*)

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ABSTRACT

Not all delphinids are similarly affected by acoustic deterrent devices (pingers). At-sea trials were carried out to assess a range of acoustic signals and deterrents on the behaviour of common dolphins. In initial tests two acoustic deterrent devices, which previously produced an evasive response by bottlenose dolphins, failed to elicit any similar behaviour in common dolphins. A new signal output device, which permitted a range of signals to be tested at various source levels and characteristics was subsequently developed but again no significant effects on the behaviour of common dolphins were observed. Two commercially available acoustic deterrents, which had deterred common dolphins in previous studies, produced an occasional mild evasive response. Significant modification of the signal type or source level may be more effective, but our results suggest that pingers, at their current state of development, may not provide a consistently effective deterrent signal for common dolphins.

KEYWORDS: ACOUSTICS; INCIDENTAL CATCHES; CONSERVATION; MANAGEMENT PROCEDURE; FISHERIES; GILLNETS; COMMON DOLPHIN; BOTTLENOSE DOLPHIN; NORTHERN HEMISPHERE

INTRODUCTION

Large numbers of dolphins and porpoises die in fishing gear worldwide, posing serious threats to several populations and species (Northridge, 1991; Perrin *et al*., 1994). This bycatch may also affect the structure and function of marine systems at the population, community and ecosystem levels (IWC, 2001). There have been a number of studies testing the usefulness of acoustic devices or `pingers' to deter small cetaceans from fishing nets, with mixed results (Barlow and Cameron, 2003; Jefferson and Curry, 1996). A number of studies (Culik *et al*., 2001; Johnston, 2002; Kastelein *et al*., 2000; Laake *et al*., 1998; Olesiuk *et al*., 2002) have tested the efficacy of pingers on set gillnets, targeting harbour porpoises (*Phocoena phocoena*) in particular. Reductions in bycatch of this species have been observed in controlled experiments with pingers on commercial gillnets (Kraus *et al*., 1997; Trippel *et al*., 1999). Cox *et al*. (2003) found that while bottlenose dolphins (*Tursiops truncatus*) approached a gillnet fitted with acoustic alarms more frequently when alarms were inactive, the alarms had much less of an effect on dolphins than had been observed for porpoises, suggesting they would be unlikely to reduce bycatch. Recently, a study by Kastelein *et al*. (2006) has shown very different reactions of a captive harbour porpoise and striped dolphin (*Stenella coeruleoalba*) to an acoustic alarm. Clearly, the responses of small cetaceans to pingers will vary among species, and perhaps among individuals. It is, therefore, not appropriate to generalise from the results of previous field tests on other species (IWC, 2000).

A variety of pelagic trawl fisheries in international and European waters incidentally catch a wide range of cetacean species (Fertl and Leatherwood, 1997; Morizur *et al*., 1999). However, there have been few published studies on the use of acoustic deterrent devices to reduce cetacean bycatch in these fisheries. In order to attempt to reduce the number of dolphins caught in the pelagic trawl fishery for albacore tuna (*Thunnus alalunga*), Bord Iascaigh Mhara (BIM; the Irish Sea Fisheries Board) have developed pingers which, it is hoped, will displace dolphins from the net opening during towing and thus reduce the risk of entanglement (BIM, 2000). Leeney *et al*. (2007) showed that these pingers were effective, at least in the short term, in eliciting avoidance behaviour by bottlenose dolphins. However, these pingers are intended to target primarily the short-beaked common dolphin (*Delphinus delphis*), the species most frequently entrapped in pelagic trawls in the Irish albacore tuna fishery (BIM, 2000). In the present study, similar field trials were conducted to determine the effectiveness of these pingers and other acoustic deterrents on this species.

MATERIALS AND METHODS

Six different acoustic devices were tested. Two prototype devices from BIM (Continuous Pinger and Responsive Pinger) were tested on five occasions and a modified version of the RP (RP2) on five occasions. An RP was also modified into a multiple signal output device and was tested on 15 occasions using different signals. Finally the CETASAVER and the Dolphin Deterrent Device (DDD) were tested on five and ten occasions each, making a total of 45 trials carried out in this study.

Deterrent devices and acoustic signals

The Continuous Pinger (CP; prototype pelagic trawl deterrent, Loughborough University/*Aquatech* UK), produces a short duration (<1s) continuous, high intensity sound source emitted at varying intervals of between 5 and 20s. The sound frequency was modulated between 20 and 160kHz with a peak source level of 157dB re 1µPa@1m. The *Aquatec* Interactive or Responsive Pinger (RP;

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Aquamark interactive pinger, *Aquatec* UK) logs and stores dolphin echolocation clicks as well as logging when the pinger is activated and for what duration. The RP acoustic signal is only activated when an internal hydrophone receives clicks from a dolphin between 10-150m from the pinger. This is due to technical considerations. As a dolphin approaches a unit, the length of the decoded echolocation clicks in the pinger increases due to the electronics. Using the present settings, 10m is the computed minimum distance for the unit; any closer than this the unit will not decode the echolocation clicks and not activate the pinger. The peak sound output source level is also 157dB re 1µPa@1m, with sound frequency modulation between 35 and 160kHz, and harmonics up to 160kHz. The output of the RP can be adjusted to produce different types and lengths of signals. For most of Trial 1, a standard setting involving a 300ms alarm was used (termed RP1). However towards the end of Trial 1, the duration of the acoustic signal from the RP on activation was increased from <1s to 10s (termed RP2). Both the CP and RP were used in Trial 1.

A multiple signal output device was developed by BIM, which permitted acoustic characteristics to be altered in real time permitting a range of signals at various source levels, frequencies, lengths and output levels to be tested. The device was connected to a laptop via a RS232 communications cable enabling the remote operation and resetting of this device. This device was used during Trial 2; settings for each test are shown in Table 1.

The CETASAVER (03 version) is manufactured by IFREMER and it has two types of signal. The first is a frequency modulated signal between 30-150kHz of 1s duration (random time and frequency organised sweeps of base square wave). The peak intensity is 190dB re 1µPA@1m and a pulse at 178µPA@1m. The signal is repeated at a minimum of every 2s, maximum of 5.5s with an average of 4s. The second signal is a click train at 90kHz of 0.1s duration, with constant click time and repetition. The Dolphin Deterrent Device (model DDD02F) is manufactured by STM Products, Italy and has three signal types: a starting sequence; a frequency modulated signal; and click trains. The starting sequence is a complex of sound patterns of frequency-modulated signals and identification patterns including a low frequency contribution. The frequency modulated signal ranges from 5 to 250kHz with a duration between of 0.5 and 9s with random time and frequency organised sweeps of base square wave similar to the CETASAVER. Its peak intensity is 174 dB re 1µPa@1m and a pulse at 165-170µPa@1m. Both devices were tested during Trial 3.

A licence was obtained from the National Parks and Wildlife Service of the Department of Environment, Heritage and Local Government to emit sounds which could potentially disturb dolphins, following the submission of a risk assessment. An abundance of 11,141 (CV=0.61) common dolphins in the Celtic Sea was estimated in July 2005 (SCANS II 2008). During the winter, common dolphins range widely and are likely to be more abundant in the Celtic Sea than in the summer (Brereton *et al*., 2005). Thus only a very small proportion of the common dolphin population in the Celtic Sea was likely to be exposed to these sounds.

Experimental design

Three trials were carried out from an 11m catamaran (MV *Holly Jo*); Trial 1 between 31 January and 4 February 2006, Trial 2 on 29 January 2007 and Trial 3 on the 19 April 2007, all off the south coast of Ireland between Castletownshend and Youghal, County Cork (Fig. 1).

Acoustic deterrent devices were attached singularly to a 7m long modified scaffold pole, which was attached to the vessel at mid-ships around 7m from the bow, with devices at a depth of 2-3m. The pole and pinger could easily be turned by hand through 90^0 to port and starboard if necessary to ensure the deterrent was emitting sound towards the dolphins in the vicinity of the vessel. The vessel travelled daily from shore to approximately 15km offshore in search of common dolphin groups. In Trial 1, once a group of common dolphins were located within 50m of the boat, the pole was lowered over the side with no acoustic deterrent device attached, and secured to the side of the vessel.

At least two, two-minute samples (controls of dolphin behaviour) were recorded (as per Leeney *et al*., 2007) before the pole was removed, an acoustic output device attached and then re-deployed. A further two behavioural samples were then recorded if possible. A HP30 hydrophone (MAGREC, UK) was used as an independent method of determining whether dolphins were echolocating and whether the RP had been activated. The HP30 hydrophone was attached to the pole with cable ties and the cable fed into an amplifier box.

Table 1

Signal type and characteristics for tests carried out with the frequency modified signal output device during Trial 2, Jan. 2007.

Signal no.	Signal type	Duration (mins)	Frequency (kHz)	Signal length (ms)	Signal interval (s)
1	Control	-	-	-	-
2	Frequency modulation	1	30-130	200-1,000, random (fixed)	Random 2-6
3	Random clicks	1	~ 60	1,500 (fixed)	Random 2-6
4	Tonal frequency	2	20	300	Random 2-6
5	Up sweep	2	20-80	300	Random 2-6
7	Up sweep	2	21-80	300	Random 2-6
8	Down sweep	2	80-20	300	Random 2-6
13	Frequency modulation	1	30-130	200-1,000, random (fixed)	1
14	Random clicks	1	~ 60	1,500 (fixed)	2
15	Tonal frequency	1	20-80	1,000	1
17	Up sweep	1	80-20	200-1,000, random (fixed)	1
18	Down sweep	1	80-20	1,000	1
23	Frequency modulated	1	31-130	200-1,000, random (fixed)	2
24	Frequency modulated	1	31-130	200-1,000, random (fixed)	2
25	Random clicks	1	~ 60	1,500 (fixed)	2

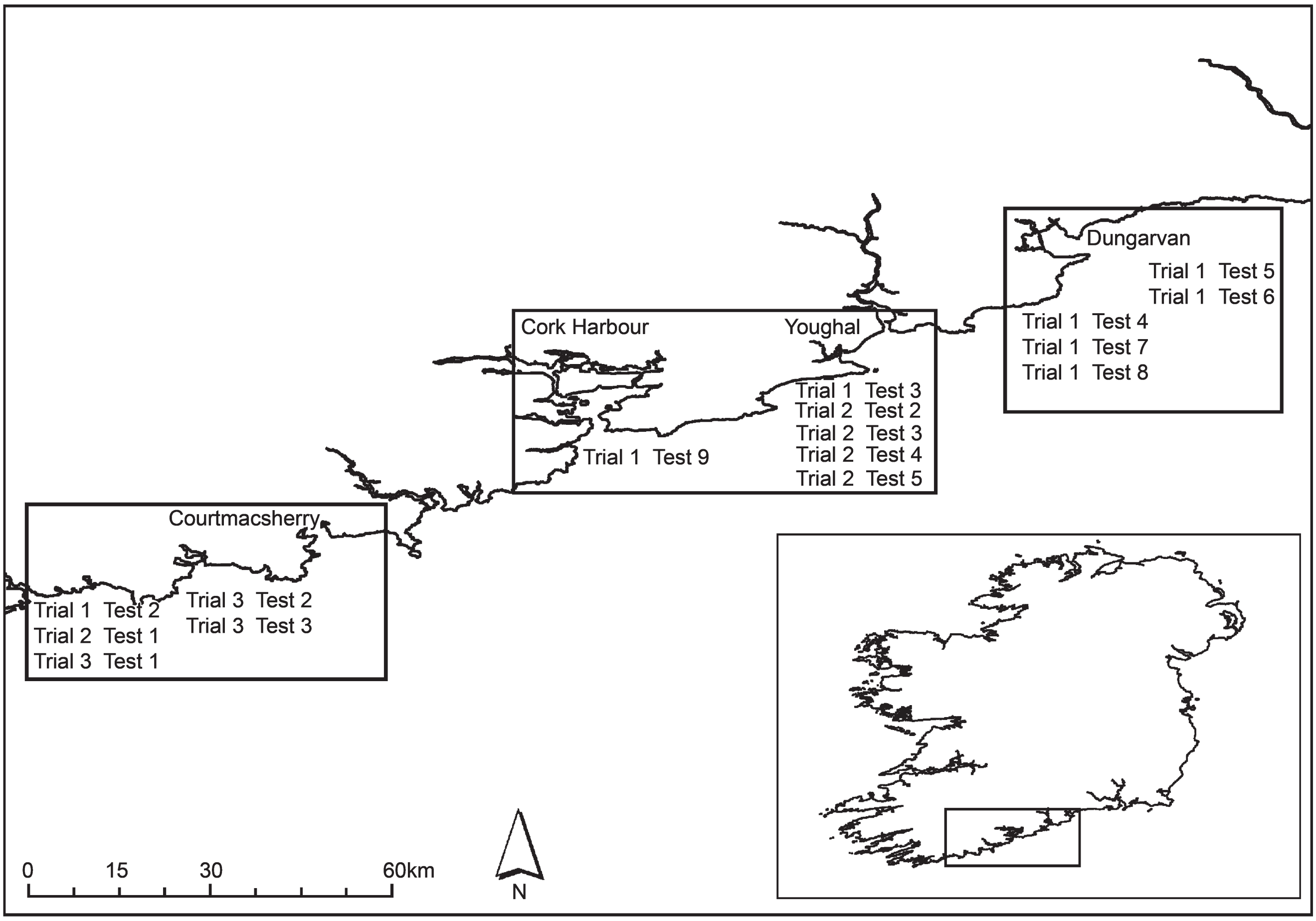

Fig. 1. Map showing location of common dolphin trials along the south coast of Ireland.

During Trial 2, the multiple signal output device remained fixed on the scaffold pole throughout. Initial behavioural sampling occurred in the absence of a signal; a test signal was then produced and further behavioural samples were taken.

The tests in each trial were observer 'blind', in that the person recording the behaviour had no knowledge of the type or status of the pinger being deployed, or signal output. The behaviour recorder was stationed at the bow of the boat, whilst another team member prepared and deployed each signal output device, noting the time of deployment, the type of device and its status (on or off). Device type and status were allocated to each encounter in no particular order. A bat box III (*Stag Electronics*) was tuned to the relevant frequency to test whether the acoustic deterrent devices were emitting a signal, prior to deployment.

For Trial 3, a 6m XS Rigid Inflatable Boat (RIB) with 115hp outboard engine was used in order to test two acoustic commercially developed deterrents, the CETASAVER and the Dolphin Deterrent Device (DDD). A group of around 20 common dolphins was located and approached by the *Holly Jo*. As the dolphins started to bowride, their behaviour was recorded from the *Holly Jo*. The RIB was stationery at a distance of several hundred metres from the bow of the *Holly Jo* at the start of each test. As the *Holly Jo* travelled towards the RIB at a velocity of 7.5km hr^{-1} the distance between the vessels was recorded every 5-10s from the RIB using a Leica LRF 1200 Rangemaster, which is accurate to ±1m up to a 400m range. An acoustic deterrent was deployed from the RIB to a depth of 2m but observers on the *Holly Jo* were unaware of the time of deployment, the type of device used or the status of the device (on/off). No controls were deployed and the behaviour prior to deployment was used to compare behaviour pre- and post- deployment of an acoustic deterrent. The DDD had two components to it, a short transmission of around 30s duration at the start of its deployment, followed by a regular transmission. Other trials (Anon., 2007) have suggested that the opening sequence may have a greater deterrent effect than the main deterrent signal. The first two tests carried out omitted the initial sequence by starting the device in the boat prior to deployment. Three subsequent tests were carried out using the full signal including the opening sequence from an initial distance of more than 100m but as no changes in behaviour were recorded, these were carried out with the full signal when dolphins were close (<30m) to the deterrent. The final test with the CETASAVER (Test 5) involved deploying the device when the dolphin group was less than 50m from the RIB. Dolphin behaviour during these close approach tests was also recorded from the adjacent *Holly Jo*, which was within 200m of the RIB with recorders observing the dolphins through 8×40 binoculars. At the end of each test the deterrents were tested aurally to ensure they were working.

Behavioural sampling

Behavioural sampling followed the methods detailed in Leeney *et al*. (2007). Behaviour was recorded in seven behavioural categories modified from Bearzi *et al*. (1999) via scan sampling (Altmann, 1974). Focal groups/schools were sampled (see Mann, 1999), rather than individuals, since common dolphins are usually found in groups and the gross changes in behaviour that might be associated with a deterrent reaction were of interest in the study. The group-follow protocol may under-record those behaviours that are less obvious or associated with a few individuals, but here behaviours were used that could be reliably and consistently recorded following the recommendations of Mann (1999). The data were then analysed to look for a combination of broad changes in behavioural categories, which might constitute some disturbance effect. The behavioural response observed was categorised into three levels of reaction intensity, based on observations of the effects of disturbance on behaviour in small cetaceans (Lusseau, 2003; Nowacek *et al*., 2001):

Level 0 = no detectable change of the behaviour.

Level 1 = avoidance (change of direction by 90°) or change of activity rhythms (increase of surfacing interval, tightening of group formation or increase of swimming speed).

Level 2 = significant change of behaviour: combination of rapid change of swimming direction (opposite direction from the source), increase of swimming speed and co-ordinated surfacing behaviour.

A wide area was covered in order to maximise the possibility of encountering multiple groups of common dolphins. Photographs were taken of the dorsal fins of as many individuals of each group as possible to facilitate recognition and avoidance of groups, which had already been exposed to tests. Adults with calves were avoided, although calves were present in some of the larger groups that tests were carried out on (see Tables 2 and 4).

RESULTS

A total of 10 encounters with apparently different groups of common dolphins occurred during Trial 1. Sixteen tests were carried out on group sizes ranging from 4 to 24 dolphins. After the first two deployments of active pingers, which solicited no reaction, two tests were carried out per group. An ongoing assessment of the reactions of dolphins to various changes in pinger settings was made. If no visible change in behaviour was recorded after the first test, another test was carried out using different settings, providing the dolphins remained in the vicinity of the vessel. This maximised the number of trials that could be carried out within a short time frame. All dolphins were bow-riding the survey vessel during each trial.

Five groups of common dolphins were located on 29 January 2007 during Trial 2. The structure of these groups was dynamic and numerous individuals may therefore have been involved in more than one test. If no reaction to a pinger frequency was detected in the initial test, a second signal was immediately tested.

A total of five tests were carried out with the CETASAVER and nine with the DDD during Trial 3, all on the same group of dolphins. Both deterrents were confirmed to be working at the end of each test. The first tests were with CETASAVER, followed by eight with the DDD. The first two of the DDD excluded the opening sequence. Two further tests with CETASAVER were followed by one with the DDD.

Behavioural responses

Trial 1

The behaviour of common dolphins recorded during deployment of the pole (BEFORE) and after the deployment of the pole and a pinger (AFTER) during Trial 1 is shown in Table 2. Dolphins were always fast swimming on the bow. After deployment of the pinger, fast swimming on the bow was still the most frequently recorded behaviour. No behaviour that could be described as evasive was recorded for any trial. No change in surfacing mode or group formation, indicative of evasive behaviour, was observed. When recorded, the distance from vessel after deployment increased on seven occasions (54%, mean increase of 18m), stayed the same on four occasions (31%) and decreased on two occasions (15%, mean decrease of 5m). On two occasions, both whilst testing an active CP, dolphins were observed making an obvious movement away from the bow of the vessel immediately after the pinger was deployed, but

Table 2

Description of dolphin behaviour before and after deployment of a pinger during Trial 1. Only changes in behaviour are shown.

				Before					After					
Test no.	Pinger type	Pinger status	Group size[1]	Group form[2]	Surf mode[3]	Speed[4]	Dist <than	Behav.[5]	Group form[2]	Surf mode[3]	Speed[4]	Dist <than	Behav.[5]	Evasive behaviour
1	CP	On	8A	T-Lo	Li	F	10m	BR		Li/OR		+30m		NO
2	RP	On	1A 1J	T-Lo	Li/OR	F	25m	BR				+10m		NO
3	RP	On	10A 2C	T-Lo	Li	F	15m	BR	Lo-D					NO
4	CP	On	10A 2C	Lo-D	Li	F	15m	BR				+5m		NO
5	RP	On	4A	T-Lo	Li	F	25m	BR		Li/OR				NO
6	CP	On	6A 1 J	T-Lo	Li/OR	F	25m	BR	Lo			-5m		NO
7	RP	Off	6A	T-Lo	Li/OR	F	25m	BR		P		-5m		NO
8	RP	On	7A	Lo	Li	F	10m	BR	Lo-D			+15m		NO
9	RP2[6]	On	5A 24-6A	Lo	Li	F	20m	BR		Li/OR		+50m	BR/BRE	NO
10	CP	On	6A	Lo	Li/OR	F	10m	BR/BRE		Li	N	+10m	BR	NO
11	RP2[6]	On	7A	Lo	Li	F	15m	BR						NO
12	RP	Off	7A 1C	Lo	Li	F	15m	BR		Li/OR		+5m	BR/CO	NO
13	RP2[6]	On	4-12	Lo	Li	F	15m	BR						NO
14	CP	Off	4A	Lo	Li	F	15m	BR				-5m		NO
15	RP2[6]	On												NO
16	RP2[6]	On												NO

[1] Dolphin age categories: A=adult, J=juvenile, C=calf. [2] Group formation: T=tight, Lo=loose, D=dispersed. [3] Surfacing mode: Li=lively, OR=occasional races. [4] Speed: F=fast, N=normal, S=slow. [5] Behaviour: NS=normal surfacing, BR=bow-riding, OR=occasional races, BRE=breaching, CO=courtship. [6] RP2 refers to the new setting with pinger duration increased to 10 seconds.

this reaction was short-lived and could not be described as evasive. Overall, observations suggested little change in the behaviour of dolphin groups after the deployment of pingers (both CP or RP and either active or inactive).

The results from the click train detection function of the RP during trials in 2006 are shown in Table 3. Although the RP showed that dolphin clicks were detected and logged, the dolphin must be at least 10m from the RP to activate the pinger. According to the RP log, the pinger was activated on at least six occasions, three occasions when the original settings were used (RP1) and three occasions when the signal duration was increased to 10 seconds (RP2). A hydrophone was used as an independent measure of click detection. Dolphin click activity was detected on the hydrophone on all of the CP deployments and all RP2 deployments (Table 3). On two occasions (Tests 9 and 11), the pinger was activated according to the hydrophone but was not logged by the RP. This suggests that the RP did not always log its own activation. During three tests, there were no recordings on the hydrophone despite pinger activation being logged by the RP. This may be due to the emission of high frequency signals, beyond the detection range of the hydrophone. Nonetheless, these data show that the dolphins were echolocating and did activate the RP on a number of occasions, but did not show any evasive behaviour.

Trial 2

The behaviour of common dolphins exposed to signals from the multiple signal output device is shown in Table 4. Common dolphins were generally observed to be foraging prior to these trials, with bowriding only recorded on four occasions during controls. This contrasts with Trial 1, in which dolphins were mainly bow-riding prior to the deployment of pingers. There were no consistent changes in group formation, surfacing mode, speed of travel or mean distance from the vessel after deployment of the signal output device.

Trial 3

In tests with the CETASAVER, the distance between the dolphins and the *Holly Jo* increased during Test 1 from less than 10m to around 20m when the vessel was around 250m

Table 3

Results from the acoustic logs on the Responsive Pinger (RP) in Trial 1.

Test no.	Pinger type[1]	Pinger status	Time activated	Dolphin clicks detected	Pinger activation detected	Detected on hydrophone
2	RP	On	13:48:34	Y	Y	
3	RP	On	11:05:43	Y	Y	-
5	RP	On	14:58:23	Y	Y	-
5	RP	On	15:04:35	Y	N	-
5	RP	On	15:07:30	Y	N	-
8	RP	On	10:05:02	Y	N	-
9	RP2	On	10:53:30	Y	N	Y
11	RP2	On	11:07:54	Y	N	Y
13	RP2	On	10:36:14	Y	Y	Y
15	RP2	On	11:45:43	Y	Y	Y
15	RP2	On	15:24:42	Y	Y	Y
16	RP2	On	15:25:06	Y	Y	Y

[1] RP2 refers to the new setting with pinger duration increased to 10 seconds.

Table 4

Dolphin group size and behaviour before and after deployment of pinger during Trial 2. Only changes in behaviour are shown.

		Before					After					
Group composition[1]	Signal type	Group form[2]	Surf mode[3]	Speed[4]	Dist <than	Behav.[5]	Group form[2]	Surf mode[3]	Speed[4]	Dist <than	Behav.[5]	Evasive behaviour
10-15A, 2C	1	T	Li-OR	N-F	10m	BR		Q-OR		+20m	NS	NO
20-25A, 2C	2	T	Li-OR	N-F	10m	NS	Lo	Q	N		BR	NO
20A, 1C	3	T	Q-OR	N-F	30m	NS	T-Lo		N	-10m		NO
20A, 1C	4	T	Q-OR	N-F	30m	-	Lo		N	-10m	NS	NO
25A, 2J, 3C	5	T	N	N	50m	NS	Lo	Q-Li	N-F	-20m	BRE	NO
15-20A	7	T	N-OR	N	30m	NS	Lo	Q-OR	N-F	-10m	BR	NO
15-20A	8	T	N	N	30m	NS	T-Lo	N-OR	N-F	+20m	NS-BR	NO
10-12A	13	T	Li	F	10m	BR		N	N	+5m	NS-BR	NO
10-12A	14	T	N	N	30m	NS	Lo	Q		-15m	NS-OR	NO
10-12A	15	T	N-OR	N	10m	NS-OR		Q-OR	N-F			NO
7A, 2J	17	T-Lo	N	N-F	30	NS-BR		N-Li	N-F	+20m	OR	NO
25A, 4C	18	T-Lo	N	S	10m	BR-BRE	Lo	Li	F		BR	NO
15+	23	T	N-Li	N-F	10m	NS/OR	T-Lo	N	N	+20m	NS	NO
4A, 1J	23	T	N	N	10m	NS	-	-	-	-	-	-
4A, 1J	24	T	N	S	10m	BR					NS	NO
4A	25	T	Q	N	10m	NS		N-OR	N-F	+15m		NO

[1] Dolphin age categories: A=adult, J=juvenile, C=calf. [2] Group formation: T=tight, Lo=loose, D=dispersed. [3] Surfacing mode: Q=quiet, Li=lively, OR=occasional races, N=normal. [4] Speed: F=fast, N=normal, S=slow. [5] Behaviour: NS=normal surfacing, BR=bow-riding, OR=occasional races, BRE=breaching.

from the deterrent, which was deployed from the stationery RIB. In the next two tests, no changes in behaviour were observed. In Test 4 the dolphins began to move away from the *Holly Jo* when they got to within 30m of the deterrent. Their behaviour changed from 'bow-riding' to 'travelling'. In the final test (Test 5), when the deterrent was placed in the water within 50m of dolphins, there was no change in the behaviour of dolphins over a 20s period. However, all these reactions were considered a Level 0 reaction.

In the first test with the DDD, dolphin behaviour changed as they approached the deterrent. Their distance from the *Holly Jo* increased from 10 to 30m over the first 50s of the trial and from 30 to more than 50m when within 300m of the deterrent. Swimming direction also changed and 'occasional leaps and races' were recorded together with 'travel'. Thus, a mild change in behaviour (Level 1) occurred. However, when the trial was repeated there was no change in any of the behavioural categories recorded. After a short period without tests, allowing the dolphins to resume their foraging behaviour, further trials were carried out with the DDD. The DDD starting sequence, which has a 30s duration, was tested seven times, from distances 5-100m to the dolphins. It solicited a Level 1 evasive reaction on three occasions (43%). In Test 4 this occurred from a distance of around 180m, Test 8 from less than 5m and Test 9 from within 30m. However in a test when the DDD was deployed within 20m of the dolphins, no change in behaviour was recorded.

CONCLUSIONS

A total of 45 tests were carried out to determine the effects of various potential acoustic deterrent signals on the behaviour of common dolphins. No responses that could be described as evasive, such as escape behaviour, a rapid change of swimming direction or increase in swimming speed, were consistently observed. Although the same group of dolphins were sometimes subjected to a number of consecutive tests, up to 14 different dolphin groups, ranging in size and in composition, including adults, juveniles and calves were exposed to pingers or acoustic deterrent signals over the course of the study. It is likely, therefore, that the reactions to acoustic deterrent signals described here are typical of common dolphins off the south coast of Ireland. Although a dolphin's motivation and thus its response threshold to a deterrent signal, may be elevated during bow-riding (Anon., 2007), dolphins tested in this study were engaged in a number of different behaviours prior to pingers being deployed. Thus the reaction to acoustic deterrent signals described here is likely not associated with any specific behaviour.

From an experimental point of view, the constraint for the RP that dolphins must be greater than 10m away was not ideal for bowriding experiments as dolphins spent most of their time less than 10m from the device. However one would have expected some echolocation clicks between 10 and 150m as dolphins approached the vessel and therefore some reaction if the signal was effective. The CP did not have the same constraints. Comparing dolphin responses between the two devices should demonstrate if the <10m effective distance was an issue. As there were no evasive responses to either device it was not possible to carry out a meaningful comparison of responses and controlling for the 10-150m range in the RP was not required.

Common dolphins were shown to be echolocating during these trials and did activate the RP. On two occasions, dolphins were observed making a slight movement away from the bow of the vessel immediately after the pinger was deployed. Whilst this reaction could not be described as evasive, it does suggest that the sound was detected by the dolphins.

The range of frequencies, signal lengths and signal intervals tested using the multiple signal output device did not elicit any strong reactions. Similarly, no major changes in dolphin behaviour were observed in response to any of the five CETASAVER deployments. The experimental design and the person deploying the equipment was the same as in previous trials eliminating the possibility that this may have contributed to the different results obtained. Mild changes in behaviour (Level 1) were observed during four out of nine (44%) deployments with the DDD device. None of these responses could be categorised as evasive behaviour (Level 2). No change in behaviour was observed during five deployments, including three cases when the full DDD signal was deployed at less than 100m from approaching animals.

Although the controlled exposure experiments presented here are in stark contrast to the noisy, complex environment around an active fishing trawl, the implications of these results for bycatch mitigation cannot be ignored. The lack of consistent behavioural changes and absence of any evasive behaviour from the group of common dolphins encountered suggests that the DDD did not have a major deterrent effect on common dolphins and would certainly not be capable of consistently displacing animals from the mouth of a pelagic trawl. These results are in contradiction to those described by IFREMER who found a strong deterrent effect by common dolphins in the Bay of Biscay for later models of the CETASAVER and to the starting sequence of the DDD (Anon., 2007). Although both devices were only tested on one dolphin group in the present study, the contrasting results suggest that intra-specific differences occur in the reaction of common dolphins to acoustic stimuli, which may be due to differences in spatial, temporal or other variables. The lack of consistent deterrent effects on all groups of animals in all locations raises questions about the efficacy of these devices in pelagic trawls. Reductions in bycatch have, however, been observed using these devices in some pelagic trials, although the reasons for these reductions are not fully understood (Anon., 2007). One possible explanation could be that acoustic devices permit animals to associate an escape route with the acoustic signal at the mouth of the trawl (Anon., 2007). This theory has yet to be proven, however, as it is currently not possible to effectively determine the presence of animals in trawls while the gear is deployed.

These results are in stark contrast to similar trials with the same CP and RP pingers tested here but carried out on bottlenose dolphins in the Shannon Estuary (Leeney *et al*., 2007), in which strong evasive behaviour was recorded in 75% of tests. Kastelein *et al*. (2006) suggested two reasons for the observed inter-species differences in reaction to acoustic alarms in their study, namely individual differences and species differences. As they only sampled one individual from two species, it was not clear how representative each study animal was for its species. The study presented here incorporated a wide range of individuals and groups, thus it is likely that the observed lack of reaction to the signals tested is characteristic of common dolphins in this region. Kastelein *et al*. (2006) also suggested that the need to flee from a sound may depend on the animal's perceived chances of being predated. Rapid habituation has been reported in recent acoustic deterrent trials with common dolphins (Anon., 2007). This study also suggested that deterrent effect declines with increased signal repetition, and increases with longer signal length. No such relationship was found in the present study as no deterrent effect was recorded for any combination of signal length, frequency or repetition.

The difference in responses of bottlenose dolphins and common dolphins to the same deterrent signals may be due to different acoustic sensitivities or thresholds of each species. Short-beaked common dolphins produce echolocation click trains at between 23-67kHz (Richardson, 1995) and whistles between 5-20kHz (Ansmann *et al.*, 2007). Bottlenose dolphins are sensitive to sounds between 1-200kHz and produce echolocation clicks around 110-130kHz (Richardson, 1995). There are no data available on the hearing sensitivity of common dolphins but Kastelein and Hagedoorn (2003) recorded the audiogram of a striped dolphin and showed that maximum sensitivity (42dBµPa@1m) occurred at 64kHz. The range of the most sensitive hearing was from 23 to 123kHz and became less sensitive below 32kHz and above 120kHz. Assuming common dolphin sensitivities are similar then as the CP and RP generated modulated frequencies between 20-160kHz, the multiple signal output device from 20-130kHz and both the CETASAVER and the DDD, covered this auditory range it seems unlikely that differences in dolphin auditory sensitivities can explain the different reactions to these deterrents.

More research is required to explore whether the results presented here are consistent at different locations and with other common dolphin populations. The interactive RP pinger developed by BIM was successful to some extent in that it responded consistently to dolphin vocalisations and a functioning deterrent device of this nature is desirable to reduce the input of noise into the marine environment and it may also delay the potential effects of habituation. A consistently effective deterrent signal for common dolphins will be required if this device is to prevent animals from entering a pelagic trawl.

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REFERENCES

Altmann, J. 1974. Observational study of behavior: sampling methods. *Behaviour* 49: 227-67.

Anon. 2007. Nephrops and cetacean species selection information and technology. EU Report NECCESSITY. 501605. Final report submitted to EC (restricted).

Ansmann, I.C., Goold, J., Evans, P.G.H., Simmonds, M. and Keith, S.G. 2007. Variation in the whistle characteristics of short-beaked common dolphins, *Delphinus delphis*, as two locations around the British Isles. *J. Mar. Biol. Assoc. U.K.* 87: 19-26.

Barlow, J. and Cameron, G.A. 2003. Field experiments show that acoustic pingers reduce marine mammal bycatch in the California drift gill net fishery. *Mar. Mammal Sci.* 19(2): 265-83.

Bearzi, G., Politi, E. and Notarbartolo di Sciara, G. 1999. Diurnal behavior of free-ranging bottlenose dolphins in the Kvarneric (Northern Adriatic Sea). *Mar. Mammal Sci.* 15(4): 1065-97.

Bord Iaschaigh Mhara. 2000. Diversification trials with alternative tuna fishing techniques including the use of remote sensing technology. Final report of EU Contract 98/101, Bord Iascaigh Mhara, Dun Laoghaire, Co. Dublin.

Brereton, T., Williams, A. and Martin, C. 2005. Ecology and status of the common dolphin *Delphinus delphis* in the English Channel and Bay of Biscay 1995-2002. *ECS Newsletter* 45 (Special Issue July 2005): 13-22. Proceedings of the ECS 18th Annual Conference, Kolmarden Djurpark, Kolmarden, Sweden, 1st April 2004.

Cox, T.M., Read, A.J., Swanner, D., Urian, K. and Waples, D. 2003. Behavioural responses of bottlenose dolphins, *Tursiops truncatus*, to gillnets and acoustic alarms. *Biol. Conserv.* 115: 203-12.

Culik, B.M., Koschinski, S., Tregenza, N. and Ellis, G.M. 2001. Reactions of harbour porpoises (*Phocoena phocoena*) and herring (*Clupea harengus*) to acoustic alarms. *Mar. Ecol. Prog. Ser.* 211: 255-60.

Fertl, D. and Leatherwood, S. 1997. Cetacean interactions with trawls: a preliminary review. *J. Northwest Atl. Fish. Sci.* 22: 219-48.

Commission, I.W. 2000. Report of the Scientific Committee. Annex I. Report of the Sub-Committee on Small Cetaceans. *J. Cetacean Res. Manage. (Suppl.)* 2:235-57.

International Whaling Commission. 2001. Report of the Scientific Committee. *J. Cetacean Res. Manage. (Suppl.)* 3:1-76.

Jefferson, T.A. and Curry, B.E. 1996. Acoustic methods of reducing or eliminating marine mammal-fishery interactions: do they work? *Ocean Coast. Manage.* 31(1): 41-70.

Johnston, D.W. 2002. The effect of acoustic harassment devices on harbour porpoises (*Phocoena phocoena*) in the Bay of Fundy, Canada. *Biol. Conserv.* 108: 113-18.

Kastelein, R.A., de Hann, D., Vaughn, N., Staal, C. and Schooneman, N.M. 2000. The influence of three acoustic alarms on the behaviour of harbour porpoises (*Phocoena phocoena*) in a floating pen. *Mar. Environ. Res.* 52: 351-71.

Kastelein, R.A. and Hagedoorn, M. 2003. Audiogram of a striped dolphin (*Stenella coeruleoalba*). *J. Acoust. Soc. Am.* 113(2): 1130-37.

Kastelein, R.A., Jennings, N., Verboom, W.C., De Hann, D. and Schooneman, N.M. 2006. Differences in the response of a striped dolphin (*Stenella coeruleoalba*) and a harbour porpoise (*Phocoena phocoena*) to an acoustic alarm. *Mar. Environ. Res.* 61: 363-78.

Kraus, S.D., Read, A.J., Solow, A., Baldwin, K., Spradlin, T., Anderson, E. and Williamson, J. 1997. Acoustic alarms reduce porpoise mortality. *Nature* 388: 525.

Laake, J., Rugh, D. and Baraff, L. 1998. Observations of harbor porpoise in the vicinity of acoustic alarms on a set gill net. *NOAA Tech. Mem. NMFS-AFSC-* 84: 40.

Leeney, R., Berrow, S., McGrath, D., O'Brien, J., Cosgrove, R. and Godley, B.J. 2007. Effects of pingers on the behaviour of bottlenose dolphins. *J. Mar. Biol. Assoc. U.K.* 87: 129-33.

Lusseau, D. 2003. Effects of tour boats on the behaviour of bottlenose dolphins: using Markov chains to model anthropogenic impacts. *Conserv. Biol.* 17(6): 1785-93.

Mann, J. 1999. Behavioral sampling methods for cetaceans: A review and critique. *Mar. Mammal Sci.* 15: 102-22.

Morizur, Y., Berrow, S.D., Tregenza, N.J.C., Couperus, A.S. and Pouvreau, S. 1999. Incidental catches of marine-mammals in pelagic trawl fisheries of the northeast Atlantic. *Fish. Res.* 41: 297-307.

Northridge, S. 1991. An updated world review of interactions between marine mammals and fisheries. *FAO Fish Tech. Paper* 251, Suppl. 1, 58pp.

Nowacek, S.M., Wells, R.S. and Solow, A.R. 2001. Short-term effects of boat traffic on bottlenose dolphins, *Tursiops truncatus*, in Sarasota Bay, Florida. *Mar. Mammal Sci.* 17: 673-88.

Olesiuk, P.F., Nichol, L.M. and Ford, J.K.B. 2002. Effect of the sound generated by an acoustic harassment device on the relative abundance and distribution of harbour porpoises (*Phocoena phocoena*) in Retreat Passage, British Columbia. *Mar. Mammal Sci.* 18(4): 843-62.

Perrin, W.F., Donovan, G.P. and Barlow, J. 1994. Gillnets and cetaceans. *Rep. int. Whal. Commn (special issue)* 15: 629.

Richardson, W.J. 1995. Marine mammal hearing. pp.205-40. *In*: Richardson, W.J., Greene, C.R., Malme, C.I. and Thomson, D.H. (eds). *Marine Mammals and Noise*. Academic Press, San Diego. xvi+576.

Trippel, E.A., Strong, M.B., Terhune, J.M. and Conway, J.D. 1999. Mitigation of harbour porpoise (*Phocoena phocoena*) by-catch in the gillnet fishery in the lower Bay of Fundy. *Can. J. Fish. Aquat. Sci.* 56: 113-23.

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Barium sulphate modified fishing gear as a mitigative measure for cetacean incidental mortalities

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ABSTRACT

Incidental mortality from entanglements in fishing gear is threatening cetacean populations worldwide. In eastern Canadian waters, entanglement deaths of the critically endangered transboundary North Atlantic right whale (*Eubalaena glacialis*) are a key conservation concern and incidental mortalities of harbour porpoise (*Phocoena phocoena*) in gillnets are a major source of mortality. Since the 1990s, a number of mitigation techniques to reduce mortalities in both species have been tested and the use of some in the US commercial fishery have been legislated. Despite this, the North Atlantic right whale population remains in a precarious state and entanglement deaths of harbour porpoise have been increasing in recent years. Further, mitigation devices, such as acoustic alarms, carry with them concerns about habituation, noise pollution, maintenance requirements and cost. The modifying of the physical characteristics of commercial fishing gear has shown some promise at reducing entanglement mortalities in initial testing while avoiding many of the drawbacks of other mitigation methods. In this study the current state of development and effectiveness of mitigation techniques through the addition of barium sulphate to fishing gear rope and twine were investigated. The development of a neutrally buoyant groundline, through the addition of barium sulphate, was undertaken in order to reduce the probability of large whale entanglements in lobster pot gear. The resulting product maintained a much lower profile in the water column relative to traditional polypropylene groundline, however, it was found unsuitable for hard-bottom areas as it was susceptible to chaffing and breaking. In order to reduce mortalities once large whales are entangled, a weak rope was developed again with the addition of barium sulphate. The breaking strength of this product was found to be 1,065lb which meets the US legislated limits (1,100lb), as opposed to traditional polypropylene rope which had a breaking strength of over 2,400lb. To meet the challenge of harbour porpoise entanglements, a gillnet twine was developed to have an increased acoustic profile and a more stiff form through the addition of barium sulphate. In field testing trials, the barium sulphate modified gillnets reduced harbour porpoise bycatch and had minimal effects on targeted groundfishes. Although they are in an early state of development, barium sulphate modified fishing gear shows promise at reducing entanglement deaths of cetaceans.

KEYWORDS: GILLNETS; INCIDENTAL CATCHES; CONSERVATION; NORTH ATLANTIC RIGHT WHALE; HARBOUR PORPOISE; FISH; ATLANTIC OCEAN; NORTH AMERICA; SUSTAINABILITY; NORTHERN HEMISPHERE; ECHOLOCATION; FISHERIES

INTRODUCTION

Incidental mortality of cetaceans due to bycatch and entanglement in commercial fishing gear is a conservation concern worldwide (Perrin *et al*., 1994). It is estimated that the North Atlantic right whale (*Eubalaena glacialis*), an endangered species (IUCN, 2008) with recent estimates of a population size of 350 (Kraus *et al*., 2005), owes over 10% of its mortality and over half of its serious injuries to entanglement in lobster pot and gillnet gear lines (Knowlton and Kraus, 2001; Moore *et al*., 2004). Further, in the North Atlantic, bycatch mortality of harbour porpoise (*Phocoena phocoena*) in gillnets has recently been identified as possibly exceeding sustainable levels in many areas, including the Gulf of Maine, Bay of Fundy and in the North, Celtic and Baltic seas (Read *et al*., 2004; Trippel *et al*., 1999; Vinther, 1999). Although recent management measures, such as time-area closures and the use of acoustic alarms on gillnets have been introduced to some areas in order to reduce cetacean mortality levels (NOAA, 1998), deaths from incidental capture in commercial fishing gear remains a concern and in the northeast USA, recent estimates of harbour porpoise bycatch (NMFS, 2006) show yearly increases in the last three years of observations (2002-04).

In the northwestern Atlantic, North Atlantic right whales become entangled in many major fishing gears but most often in lobster pot and gillnet gear buoy lines and buoyant groundlines that are suspended in the water column (Johnson *et al*., 2005). Entanglement most commonly occurs at the mouth or tail and most fatally when a whale becomes entangled with multiple body parts (Knowlton and Kraus, 2001). Incidental mortalities of harbour porpoises have occurred since the development of a gillnet fishery in the Bay of Fundy in the 1960s (Gaskin, 1992) and have since occurred throughout their range (Fontaine *et al*., 1994; Gaskin, 1984; NMFS, 2006; Perrin *et al*., 1994; Read and Gaskin, 1988; Trippel *et al*., 1996; Vinther, 1999). The mechanism of entanglement of porpoises includes entanglement of flukes and/or other body parts followed by an inability to surface for air from the bottom-set gear.

In US waters, attempts to mitigate whale entanglements in fishing gear have included area closures, gear modifications and requirements regarding how gear is fished (Lyman and McKiernan, 2005; McKiernan, 2002). Some of these regulations are common to lobster pot and gillnet gear, for example, no portion of buoy lines can be floating at the surface[1]. Others are more gear specific, such as the general prohibition of straight set gillnets at night in southeast USA waters[1]. However, area closures do not fully cover movements of right whales and gear modifications have not yet reduced entanglement rates (Kraus *et al*., 2005). In the 1980s, 52% of North Atlantic right whales showed signs of previous entanglements, while more recent estimates show almost three quarters of right whales have signs of having been entangled at least once (Knowlton *et al*., 2003).

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[1] Atlantic large whale take reduction plan regulations. Code of Federal Regulations Title 50, Part 229.32.

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A number of mitigation techniques have been used in an attempt to reduce incidental mortality of small cetaceans in gillnet fisheries. Time-area closures, in which areas of high porpoise density are closed to gillnet fishing during certain times of the year, and the required use of acoustic alarms or 'pingers' are in place for a number of areas in the northeast USA[2]. During initial trials, pingers were shown to be effective in reducing harbour porpoise bycatch (Kraus *et al.*, 1997; Trippel *et al.*, 1999) and their routine use in the northeast USA sink gillnet fishery began in 1999. However, the apparent utility of pingers as a mitigation tool has not translated into reduced bycatch in the commercial gillnet fishery in which mortalities have been increasing since 2001 (NMFS, 2006). Beyond this, numerous concerns exist surrounding the use of acoustic deterrent devices, such as habituation to sounds, mechanical failure, monitoring, noise pollution and habitat displacement (Culik *et al.*, 2001; Gearin *et al.*, 2000; Kastelein *et al.*, 2000).

Newer mitigation techniques in development involve altering the physical and/or chemical makeup of either the rope used for buoy and groundlines or the twine used to construct gillnet mesh material (Larsen *et al.*, 2007; Mooney *et al.*, 2007; Mooney *et al.*, 2004; Trippel *et al.*, 2003). These include the addition of barium sulphate[3] or other material to gillnet twine to reduce small cetacean bycatch by increasing its echolocation signature. Also, attempts have been made to reduce the breaking strength and decrease the buoyancy of gear rope through the addition of barium sulphate. Here the current status of development and effectiveness of mitigation techniques through the addition of barium sulphate to fishing gear rope and net twine are reviewed. Fishing products partly made of barium sulphate, which are relatively inexpensive to manufacturer, may serve to augment or replace current management measures that are in place to reduce the incidental mortality of cetaceans.

MATERIALS AND METHODS

Neutrally buoyant groundline

In October-November 2005 and March 2006, a field study was conducted to evaluate the behaviour of a barium sulphate modified lobster pot groundline. Standard polypropylene groundline is positively buoyant and presumably remains high in the water column causing increased risk of entanglement to large whales such as the North Atlantic right whale. The barium sulphate groundline of the same diameter was designed by Atlantic Gillnet Supply Ltd. (now Better Gear Inc.), to be neutrally or slightly negatively buoyant (density=1.04-1.06kg/m^3) after a call by the US government for industry to develop both neutrally buoyant and sinking groundlines. The relatively dense barium sulphate particles caused increased weight in the groundline and it was expected it would lead to it remaining lower in the water column, and in theory, lead to decreased risk of entanglement to large whales.

The behaviour of three types of 0.5" groundline rope were evaluated in natural conditions: (1) standard polypropylene rope that is positively buoyant and tends to float; (2) weighted, negatively buoyant rope; and (3) barium sulphate modified rope that was designed to be neutrally buoyant (20% barium sulphate by weight). The behaviour of each rope in the water column was monitored using depth sensors, DST milli, manufactured by Star-Oddi, Reykjavík, Iceland, which record depth (pressure) and temperature every five minutes. The dimensions of each cylindrical sensor were: length 4cm, diameter 1.2cm, weight 9g and with plastic housing each totalled 20g.

In 2005, a 150ft line was used composed of all three types of rope. Depth sensors were placed at the ends of each rope and on each rope at three stations evenly spaced between the anchors (Fig. 1a). The line was set approximately 500m from the wharf of the Biological Station in St. Andrews, New Brunswick in an area 20-25m deep on 5 October 2005 and left for 44 days, before being retrieved on 18 November 2005. In 2006, a 360ft line was used that was composed of two types of rope: standard polypropylene; and barium sulphate modified. The line was anchored at each end and at two other locations, at 120ft and 240ft (Fig. 1b). Between each set of anchors, three depth sensors were placed on each rope every 30ft (each rope within a line was of equal length). The line was set near the St. Andrews Biological Station in an area approximately 25m deep on 2 March 2006 and left for 7 days, being retrieved on 10 March 2006. To examine the potential effect of depth sensors on rope buoyancy, a 5m segment of 0.5" diameter floating polypropylene rope was placed in a tank at the Biological Station with a depth sensor and observed no effect on rope buoyancy at the point of sensor attachment.

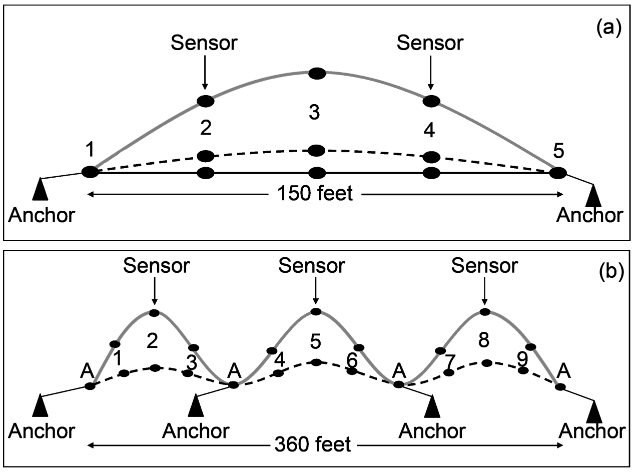

Fig. 1. Diagram of test gear configuration from the 2005 (a) and 2006 (b) study of mitigative groundline: weighted groundline – black, neutrally buoyant barium sulphate modified groundline – dotted, standard nylon groundline – grey. Numbers indicate stations (sites) at which depth sensors (black ovals) were secured. Stations 1 and 5 indicate anchor sensor stations in 2005 and 'A' indicates anchor sensor stations in the 2006 study.

Samples of the barium sulphate modified rope were also given to three fishermen for use in their lobster fishing gear configuration. One of these fishes from Metaghan, Nova Scotia in an area of rough bottom referred to as the McDormond Patch and used the sample provided in the winter of 2005/06. Two others, who fish from Grand Manan, New Brunswick, were given the barium sulphate modified rope in the summer of 2005. One of the Grand Manan fishermen used the sample provided on his gillnet gear for the rope segment from the anchor to one end of the net, whereas the other used it in lobster gear. All three participants fish in an area of extremely high tides (Bay of Fundy, Canada), which may affect net configuration and

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[2] Harbor Porpoise Take Reduction Plan Implementing Regulations. Code of Federal Regulations Title 50, Parts 229.32 and 229.33.

[3] Barium sulphate is a white crystalline solid that has a very low solubility in water. It is used extensively as a radiocontrast agent in medical applications, as a white pigment for paint and as a high temperature oxidiser in pyrotechnics.

rope profiles. Samples of the rope being used by fishermen were collected and sent to Seaside Inc. in Warren, Maine for testing of breaking strength.

Weak rope

For over 20 years polypropylene rope, which is easy and inexpensive to produce, has been used in gillnets as 'head rope' or 'float line' i.e. the rope across the top of the net. However, observations of whale entanglements in gillnets have noted that it is this headrope that represents a danger to whales; in short, an entangled whale could break the netting and the sink rope at the bottom of the net but not the headrope. Based on these observations, the National Marine Fisheries Service (NMFS) called for a headrope that would break at 1,100lb in order to reduce the risk of fatal whale entanglements in US waters[4]. While there has been no call for specific gear changes in Canadian waters, entanglements in fixed fishing gear have been identified as a major threat to the population (COSEWIC, 2003) and a reduction of the impacts of encounters with fishing gear is a key aspect of the recovery strategy of the North Atlantic right whale in Atlantic Canadian waters (Brown *et al.*, 2009).

There was no precedent for making a weaker head rope and several companies had previously tried to solve the problem through changing the draw ratio of the polypropylene (draw ratio is a measure of the degree of stretching during the orientation of a fibre or filament, expressed as the ratio of the cross-sectional area of the undrawn material to that of the drawn material). Normally the draw ratio is around seven, but by dropping it to two, a weaker product can be made. However this type of product has elasticity too great to be of use as a rope for gillnets. The approach applied in this study was to maintain the higher draw ratio to keep the elasticity low and to fill the fibres with 'foreign' materials that would not contribute to their strength[5]. Several means of doing this were tested. Starch was added, but this caused clogging of the extrusion die and the resulting rope fibres were inappropriate for use. A very fine grade of sodium chloride was evaluated but unless it was very dry there was clogging of the filtering screens (bridging), which shut down the extruder. Both of these, had they been successful, would have dissolved/biodegraded out of the fibres to leave a product with the density of polypropylene. Barium sulphate was investigated finally because it has a very low tendency to bridge in an extruder and gave a reproducible product that had good handling qualities.

Approximately 9.5km of 3/8" diameter barium sulphate modified head rope was made at Seaside Inc., Warren, Maine. This weak rope was considerably negative in buoyancy (50% barium sulphate by weight) and was dyed light purple in colour to distinguish it from other ropes. The weak rope was distributed for use among three fishermen in the lower Bay of Fundy, Canada (Fig. 2) in autumn 2003 and spring of 2004. Initially it was intended solely for the float lines of gillnets but was also used for gillnet end lines (end lines equal the height of a gillnet, 3-4m, and connect the bottom and float lines at the end of each gillnet mesh panel which typically measure around 100m in Canada) and as connecting lines in single and paired lobster pots which enabled a broader test of its possible application. A questionnaire was developed with both general and detailed questions to learn first-hand about how the rope was used and its appraisal by fishermen. Lengths of used weak rope

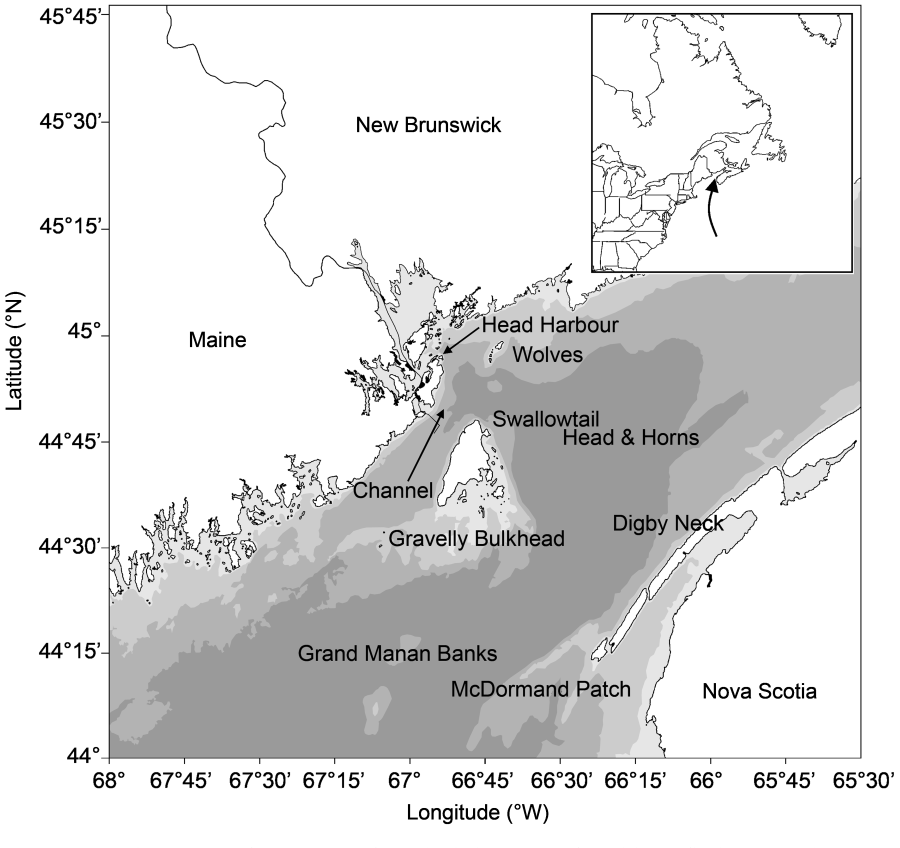

Fig. 2. Map of lower Bay of Fundy listing areas of traditional fishing grounds.

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[4] Atlantic large whale take reduction plan regulations. Code of Federal Regulations Title 50, Part 229.32.

[5] United States Patent Application 20050155271.

were sent to Seaside Inc. and in association with a local company undertook breaking strength tests of various segments of rope (16ft sample lengths) on a US government certified machine.

To gather more objective information on the weak rope's performance in field conditions, an outdoor experiment was set up at the St. Andrews Biological Station. The barium sulphate modified rope was exposed to sunlight and seawater over a four month period. The rope was suspended above the high tide level, below the high tide level but above the low tide level (intertidal) and below the low tide level. Standard polypropylene rope was also suspended above the high tide level. Breaking strength was measured at the start of the study and once a month, for four months. In order to determine longer-term performance, breaking strengths of ropes held above the high tide level and intertidal samples were measured after 24 months. It is extremely difficult to conduct a control-impact type study with weak rope. This is due to the relative rarity of right whale entanglements that would occur in an experimental setting over for example a two-year study period coupled with the limited amount of rope made available to fishermen. This is in contrast to harbour porpoise bycatch gillnet experiments in which entanglements are frequent enough to evaluate mitigative gear in a short period.

Acoustically reflective gillnets

From 1998 until 2001, Fisheries and Oceans Canada and others were part of an effort to develop and test the effectiveness of a barium sulphate modified gillnet mesh in reducing harbour porpoise bycatch (Trippel *et al.*, 2003). Two types of nylon monofilament mesh gillnets (strand diameter 0.57-0.60mm) were used, one in which the strands contained fine barium sulphate particles (3% by volume; 10% by weight) and another in which the strands were made of 100% nylon (used regularly worldwide). As barium is a heavy metal, it was assumed its presence would make gillnet mesh more reflective to echolocation signals produced by small cetaceans. The barium sulphate modified net was dyed pale blue to mask the white opaque colour of barium sulphate, whereas the standard nylon net was colourless and far more transparent. Gillnet strings used were 300m long (three 100m panels (mean was 3.02 panels ± 0.003 SE), 4m deep, had a stretched mesh size of 15cm, and were set at a depth of approximately 60m (mean depth was 59.1m ± 0.11 SE). Whenever possible, the strings were fished for 24 hours and retrieved daily (mean soak time was 26.2hr ± 0.29 SE).

As a part of this study, observers were placed onboard gillnet fishery vessels in the lower Bay of Fundy, Canada, in order to quantify porpoise and groundfish catches in both standard nylon mesh nets and barium sulphate mesh nets. Observer coverage in 1998 was augmented by fishing vessels that operated without an observer but who participated in a voluntary reporting programme on the effectiveness of the two types of gillnet mesh. In 1999, a number of fishing trips used mixed strings comprised of both standard nylon mesh panels and barium sulphate modified gillnet panels. Field coordination of the observer programme was provided by the Grand Manan Fishermen's Association (in 1998, 2000 and 2001) and Javitech Ltd (in 1999). Fishing took place from July-September and participation in the observer programme included six vessels in 1998 and three vessels in 2000 and 2001. Trained observers were used in each year except 1998. Observers were trained by Fisheries and Oceans Canada on detection and description of porpoise entanglement, identification of fish species, and recording of details of the characteristics of gillnet set location and duration. In 1998, voluntary reporting of this information was made by fishermen. As voluntary reports were only made in 1998 its inclusion as a model variable over all years would not be appropriate. However, the mean porpoise catch in voluntary reports (0.009 porpoise string^{-1}) was similar to that of observers (0.007 porpoise string^{-1}).

Typically, count data (e.g. the number of animals captured) are most appropriately modelled using discrete probability distributions such as the Poisson distribution when equi-dispersed (variance equals the mean), or the negative binomial distribution when over-dispersed (variance is greater than the mean). In the case of very rare count data, neither of these distributions adequately account for an excessive amount of zero observations. One way to approach the problem of zero-inflation is to use models that are a mixture of both the distribution of observed counts and an excess number of zero observations. A zero-inflated Poisson model (ZIP) was introduced by Lambert (1992) to account for an excess of zeros in counts of defects introduced during manufacturing processes. Since then, zero-inflated models have been slowly gaining popularity in ecology and have been used in applications such as species survey counts (Potts and Elith, 2006) and bycatch of rare species (Minami *et al.*, 2007).

For a ZIP model, given set of observed species counts $y_\imath$ = 1,...*n* and

$$y_i = 0, \text{ with a probability } 1 - \pi(x)$$

$$y_i \sim \mathit{Poisson}[\lambda(z)], \text{ with a probability } \pi(x) \text{ such that}$$

$$P(Y = 0 \mid x, z) = 1 - \pi(x) + \pi(x)e^{-\lambda(z)} \text{ and}$$

$$P(Y = r \mid x, z) = \frac{\pi(x)e^{-\lambda(z)}\lambda(z)^r}{r!}, \; r = 1, 2, \ldots$$

where $\pi(x)$ is the probability of the number of animals at a sample location that has a Poisson distribution and $\lambda(z)$ is the mean number of animals at the location. Both π and λ may depend on the same or a different group of covariates x and z, respectively (Cunningham and Lindermayer, 2005). In simple terms, species counts are modelled using a mixture of a logistic regression (when the counts equals zero) and a Poisson regression (when the counts are greater than zero).

Using a ZIP model in a generalised linear modelling framework, the effect of gillnet mesh type (standard nylon and barium sulphate modified), month and year on bycatch rate of harbour porpoise and fish were examined. In a generalised model framework, the zero mass is modelled using logistic regression while the observed catches are modelled using a Poisson error structure with a log link. All analyses were conducted in R v2.4.1, an open source statistical package (*http://www.r-project.org*). The analysis was limited to the Swallowtail area (Fig. 2) since other areas were not observed in all years and bycatch at Swallowtail comprised 92% of the total. The appropriateness of a Poisson model and a zero-inflated negative binomial model were also investigated however parameter estimates were very similar to those from the ZIP model and the latter was more parsimonious than the Poisson or zero-inflated negative binomial models as measured by Akaike's information criterion.

The effect of gillnet mesh type, month and year on fish catch rates were also investigated. Fish are relatively common in the gillnet catches and their distributions lacked an excessive probability mass at zero, thus zero-inflated models were not used. Instead a generalised linear model with a negative binomial model error structure and a log link was used, which is often appropriate for over-dispersed count data. The data were again limited to the Swallowtail area. The year 1999 was excluded from the statistical analysis due to the use of mixed mesh type strings. This is the first time the Bay of Fundy barium sulphate gillnet trial data have been published collectively and undergone external peer review. Previous analyses focused on using non-parametric methods to examine differences between catch rates in different mesh types for a limited data set (Trippel *et al.*, 2003; Trippel and Shepherd, 2004). This analysis uses more advanced techniques to more accurately model the data and error distribution, and simultaneously examines multiple factors such as year and month. This allows the effect of each factor on catch rate to be resolved rather than just examining a single factor.

RESULTS

Neutrally buoyant groundline

The weighted and barium sulphate modified ropes showed smooth, cyclic changes in depth during the first 40 hours of being set which corresponded to changes in water height due to daily tidal cycles (Figs 3 and 4). The depth of the standard polypropylene rope showed a similar cycle as well as the addition of short-cycle variations, likely due to water turbulence from currents and waves (Figs 3 and 4). The barium sulphate modified rope was consistently higher in the water column than the weighted rope, while the standard polypropylene rope was consistently the highest in the water column and showed a higher variation in depth, again likely due to the short-cycle variation from water turbulence (Table 1). Exhibiting 40 hours of recording of the rope profiles over approximately three tidal cycles permitted examination of the depth variation that could occur during a typical single set of a lobster trawl. At times, polypropylene rope had floated quite high in the water column such that it was 15m higher than barium sulphate modified and weighted ropes (Figs 3 and 4), though the mean difference commonly ranged from 3 to 6m (Table 1). Over the 7 days of the 2006 study, there was no discernible pattern in mean depth change compared to the first 40 hours for either the polypropylene rope or the barium sulphate modified rope (Table 1).

Comments on the barium sulphate modified rope were available from the fishermen from Metaghan, Nova Scotia and Grand Manan, New Brunswick. The general impression of a fisherman from Nova Scotia was that the composition of the barium sulphate modified rope could not withstand the wear and tear in the rough area he fishes, the McDormond Patch, off Southwest Nova Scotia. He reported excessive fraying and chaffing in the rope, which broke twice while gear was being hauled. After two weeks of use, he abandoned the barium sulphate modified rope completely out of fear of losing his fishing gear. Normally, he uses 0.5” Polysteel® groundline that lasts 3-4 years in the rough area he fishes and has a breaking strength of 5,100lb. The lobster fisherman from Grand Manan expressed similar concerns. After three weeks of fishing over rough grounds, chaffing had occurred to the rope and he had lost some gear.

Weak rope

The breaking strength of unused barium sulphate modified rope was 1,065lb (SE=13.7) compared to 2,471lb (SE=42.0) for polypropylene rope (Table 2). In the controlled study of rope strength changes over the first four months there was little indication of weakening and no apparent effect of exposure location (Table 2). Breaking strengths of

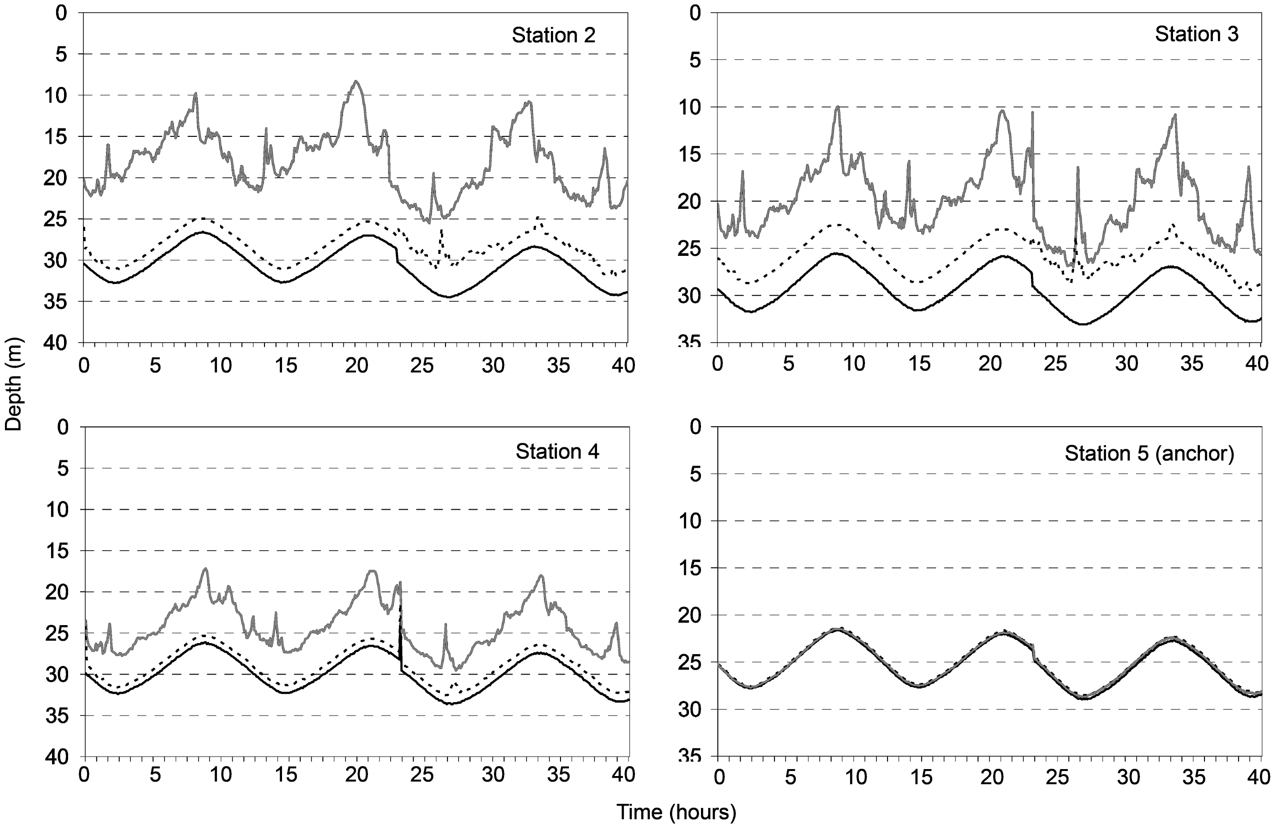

Fig. 3. Time series of depth over the first 40 hours of the 2005 neutrally buoyant groundline study: weighted rope – black, barium sulphate modified rope – dotted, standard polypropylene rope – grey. The sensor on station 1 (anchor) failed during deployment.

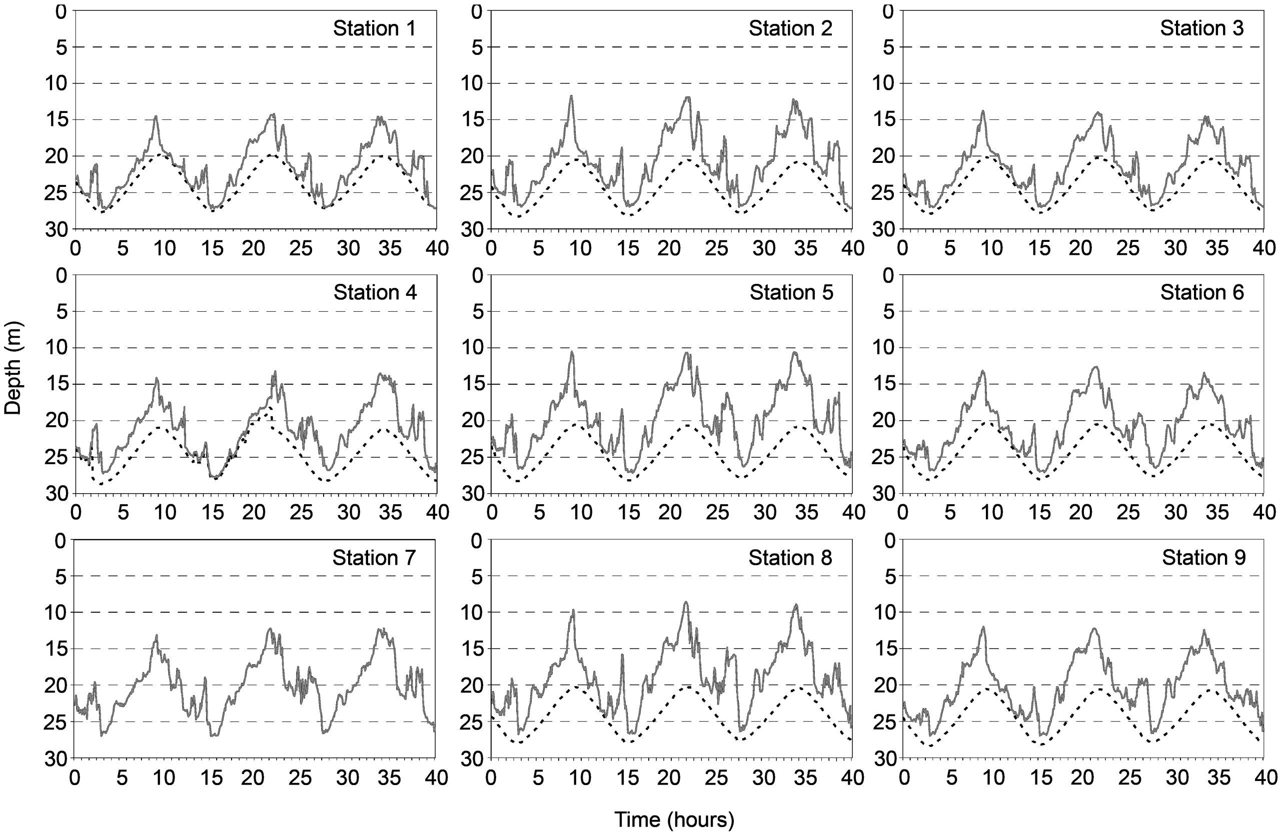

Fig. 4. Time series of depth over the first 40 hours of the 2006 neutrally buoyant groundline study: barium sulphate modified rope – dotted, standard polypropylene rope – grey. The station 7, barium sulphate modified rope sensor failed during deployment.

Table 1

Mean depth (standard error in parentheses) of each 0.5” rope type, over the first 40h and 7 days for the 2005 and 2006 neutrally buoyant groundline study. Stations designated with an ‘A’ are anchor sites and $BaSO_4$ represents barium sulphate modified rope.

2005			2006			
		40 hours			40 hours	7 days
Station	Rope	Depth (m)	Station	Rope	Depth (m)	Depth (m)
2	Weighted	26.1 (0.02)	A1	N/A	23.7 (0.11)	23.7 (0.05)
	$BaSO_4$	24.3 (0.02)	A2	N/A	24.8 (0.11)	24.8 (0.05)
	Polypropylene	16.2 (0.03)	A3	N/A	24.7 (0.11)	24.6 (0.05)
3	Weighted	24.9 (0.02)	A4	N/A	24.6 (0.11)	24.4 (0.05)
	$BaSO_4$	21.9 (0.02)	1	$BaSO_4$	23.9 (0.12)	23.9 (0.05)
	Polypropylene	18.3 (0.03)		Polypropylene	21.7 (0.16)	20.9 (0.07)
4	Weighted	25.3 (0.02)	2	$BaSO_4$	24.5 (0.11)	24.4 (0.05)
	$BaSO_4$	24.5 (0.02)		Polypropylene	20.8 (0.18)	19.6 (0.08)
	Polypropylene	21.0 (0.02)	3	$BaSO_4$	24.1 (0.11)	24.1 (0.05)
5	Weighted	20.8 (0.02)		Polypropylene	21.5 (0.16)	20.6 (0.08)
	$BaSO_4$	20.5 (0.02)	4	$BaSO_4$	24.5 (0.12)	24.8 (0.05)
	Polypropylene	19.8 (0.04)		Polypropylene	21.5 (0.17)	20.5 (0.08)
			5	$BaSO_4$	24.5 (0.11)	24.4 (0.05)
				Polypropylene	19.8 (0.20)	18.7 (0.09)
			6	$BaSO_4$	24.3 (0.11)	24.2 (0.05)
				Polypropylene	20.8 (0.17)	20.1 (0.07)
			7	$BaSO_4$	N/A	N/A
				Polypropylene	20.3 (0.17)	19.4 (0.08)
			8	$BaSO_4$	24.2 (0.11)	24.0 (0.05)
				Polypropylene	18.9 (0.19)	17.7 (0.09)
			9	$BaSO_4$	24.5 (0.11)	24.3 (0.05)
				Polypropylene	20.4 (0.18)	19.6 (0.07)

commercially deployed barium sulphate modified rope used by one fisherman showed similar breaking strengths (1,043lb and 1,039lb) to the barium sulphate rope in the controlled study (Table 2). The standard polypropylene rope used by this fisherman had a breaking strength of 2,851lb. A second fisherman used the barium sulphate modified rope in lobster pot lines over a period of three months. The rope was hauled approximately 75 times over hard rock bottom during that time period. After three months of use and three months of outdoor storage, the breaking strength of the barium sulphate rope had dropped to 932lb (SE=22.4; Table 2). Samples of standard rope used by the second fisherman varied from a maximum of 3,196 to 1,960lb. Both the barium sulphate modified rope and the standard polypropylene rope exposed to ambient, non-submerged conditions for 24 months lost a significant amount of their strength (Table 2). The intertidal barium sulphate modified rope retained most of its strength over the same 24 months.

In their questionnaires, the fishermen were generally positive about the performance of the weak rope, giving it an average of 8.3/10 on a subjective ranking scale (full questionnaire results available from authors upon request). One fisherman liked the smaller diameter of the weak rope, which he believed caused it to be less affected by currents. The second fisherman reported the weak rope broke once when it was caught on the bottom but otherwise reported no problems and was interested in using it again. The third fisherman was positive but did not complete the questionnaire.

Acoustically reflective gillnets

Over the entire four year period, 52 porpoise captures were observed in three areas: Swallowtail, the Wolves and Gravelly Bulkhead (Table 3; Fig. 5), although minimal observer coverage occurred elsewhere. All but four porpoise captured were observed at Swallowtail and two of the porpoise catches at the Wolves were in standard nylon mesh panels within strings using a combination of standard nylon

Table 2

Mean breaking strengths (standard error in parentheses) of 3/8” barium sulphate ($BaSO_4$) modified ‘weak’ rope and standard polypropylene ropes from both a controlled exposure study and after commercial applications.

Controlled study					Commercial application					
Rope	Depth	Time (months)	Ropes tested	Breaking strength (lb)	Rope	Source	Age	Ropes tested	Breaking strength (lb)	Description
$BaSO_4$	N/A	0	6	1,065 (13.7)	$BaSO_4$	1	16d	2	1,043 (7.0)	Head line
	0	1	3	1,048 (12.8)				3	1,039 (18.9)	End line
		2	3	1,072 (15.6)	Polypropylene	1	6-10y	3	2,851 (6.5)	Head and end lines
		3	3	1,041 (9.9)	$BaSO_4$	2	3mo	3	932 (22.4)	Lobster gear
		4	3	1,036 (12.1)	Polypropylene	2	8y	3	1,960 (49.4)	3/8in prickly twisted
		24†	4	385 (6.2)			5y	3	3,196 (29.5)	3/8in twisted
	Intertidal	1	6	1,059 (21.0)			9y	3	2,025 (34.4)	5/16in hard lay
		2	3	1,028 (13.5)			8y	3	2,062 (261.8)	Twisted
		24	3	961 (11.6)						
	Submerged	1	6	1,084 (9.4)						
		2	3	1,032 (22.2)						
Polypropylene	N/A	0	2	2,471 (42.0)						
	0	1	2	2,415 (130.0)						
		2	2	2,165 (18.0)						
		3	2	2,058 (2.5)						
		4	2	2,426 (16.0)						
		24†	4	587 (37.1)						

†24 month samples of zero depth ropes were continuously exposed to ambient, non-submerged conditions by being placed on a roof of a nearby building.

Table 3

Summary of vessel participation, fishing effort and harbour porpoise catches in a barium sulphate ($BaSO_4$) modified gillnet study near Swallowtail, Grand Manan, New Brunswick.

		Nylon mesh nets				$BaSO_4$ mesh nets			
Season	Vessels observed	Strings fished	Porpoise captured	Porpoise per string when captured	Probability of not capturing at least one porpoise	Strings fished	Porpoise captured	Porpoise per string when captured	Probability of not capturing at least one porpoise
1998									
Jul. 01-15	6	136	3	3.00	0.9926	25	0	-	1.000
Jul. 16-31	4	42	0	-	1.0000	23	0	-	1.000
Aug. 01-15	3	20	1	1.00	0.9500	5	0	-	1.000
Aug. 16-31	1	7	0	-	1.0000	2	0	-	1.000
Sep. 01-15	0	0	0	-	1.0000	0	0	-	1.000
2000									
Jul. 01-15	0	0	0	-	1.0000	0	0	-	1.000
Jul. 16-31	3	58	2	1.00	0.9655	33	0	-	1.000
Aug. 01-15	3	90	3	1.00	0.9667	51	0	-	1.000
Aug. 16-31	3	80	0	-	1.0000	50	0	-	1.000
Sep. 01-15	0	0	0	-	1.0000	0	0	-	1.000
2001									
Jul. 01-15	3	63	7	1.17	0.9048	64	3	1.00	0.9531
Jul. 16-31	3	110	9	1.13	0.9273	104	6	1.00	0.9423
Aug. 01-15	3	68	1	1.00	0.9853	82	2	1.00	0.9756
Aug. 16-31	3	86	3	1.00	0.9651	92	1	1.00	0.9891
Sep. 01-15	3	55	3	1.00	0.9455	59	4	1.00	0.9322

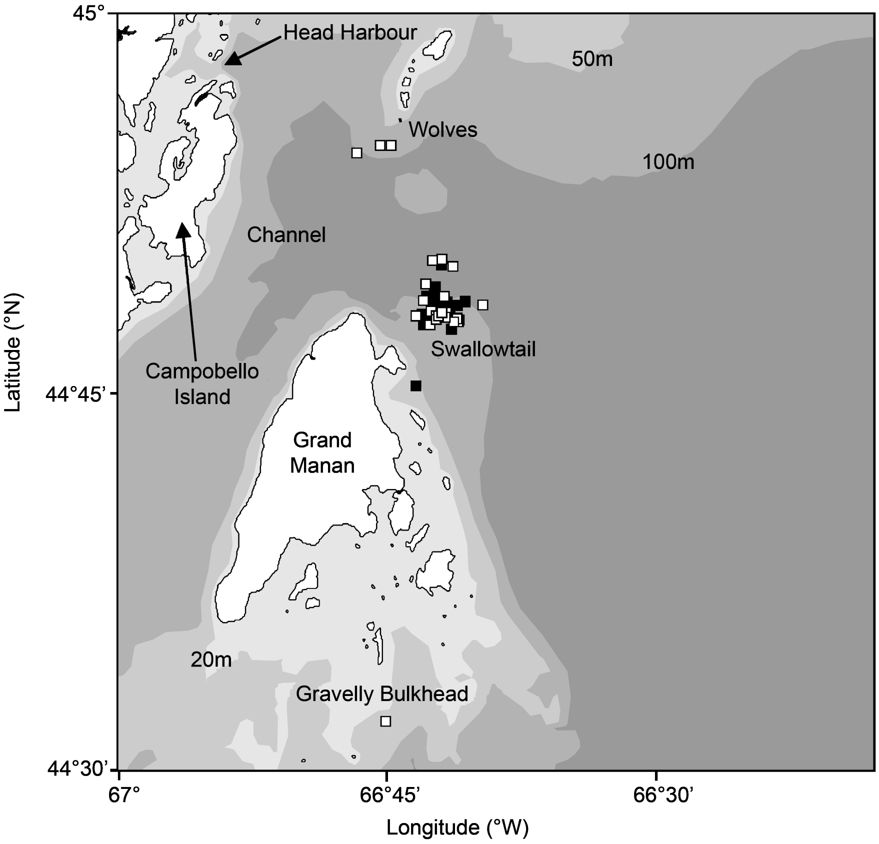

Fig. 5. Map of all observed porpoise mortalities in the lower Bay of Fundy demersal gillnet fishery from 1998-2001 (*n*=52). White squares represent mortalities in 100% nylon-mesh nets while solid squares represent mortalities in barium sulphate nets.

and barium sulphate modified panels. For the Swallowtail region, porpoise were not captured in the barium sulphate modified gillnets while four and five were caught in the standard nylon nets in 1998 and 2000, respectively (Table 3). In 2001, 16 porpoise were captured in barium sulphate modified nets (401 strings fished) while 23 were captured in standard nylon nets (382 strings fished).

The ZIP model for porpoise bycatch showed significant effects of year and mesh type on harbour porpoise bycatch (Table 4). On a monthly basis, harbour porpoise catch rates in the barium sulphate gillnets were consistently lower than in nylon mesh gillnets with the exception of September 2001 (Fig. 6). For the Poisson portion of the model, which evaluates catch rates when at

Table 4

Results of models testing the effect of gillnet mesh type on catch rates of harbour porpoise, cod, pollock, haddock and spiny dogfish. For the zero-inflated Poisson model, the Poisson regression part models the positive catches only (catch rate when at least one porpoise was captured) while the logistic regression part models the probability of catching no porpoise. Statistical significance levels for model estimates are * ≤0.05, **≤0.01, ***≤0.001, otherwise non-significant. SE=standard error.

Harbour porpoise zero-inflated Poisson model

Poisson regression part			Logistic regression part		
Factor	Coefficient	SE	Factor	Coefficient	SE
Nylon	1.83*	0.769	Nylon	1.95	1.316
Year 2000	-4.06***	0.806	Year 2000	-17.03	579.58
Year 2001	-1.23	0.797	Year 2001	-3.06**	0.956
August	-0.94	0.645	August	0.23	0.970
September	-0.31	0.8774	September	-0.30	1.163
Intercept	-1.08	1.097	Intercept	2.92*	1.402

Finfish negative binomial models

Factor	Cod	SE	Haddock	SE	Pollock	SE	Dogfish	SE
Nylon	0.02	0.059	0.42*	0.162	0.190	0.118	0.03	0.099
Year 2000	0.88***	0.099	1.72***	0.240	0.666**	0.252	0.38*	0.162
Year 2001	1.22***	0.086	-0.32	0.247	1.489***	0.216	0.31*	0.140
August	-0.43***	0.064	-0.16	0.171	-0.364**	0.129	0.30**	0.110
September	-0.98***	0.115	-1.51*	0.686	-0.139	0.196	0.44*	0.188
Intercept	1.49***	0.085	-0.83***	0.233	-1.508***	0.219	3.28**	0.138

least one porpoise is captured, the year 2000 was significantly lower than 1998 (Table 4). The logistic portion of the model, which evaluates the probability of not capturing at least one porpoise, indicated that the probability of not capturing any porpoises was significantly lower in 2001 than in 1998 (Table 4). The model results are supported by the observed data wherein catch rates of porpoise, when at least one was captured, was 1.0 porpoise per string in 2000, while the probability of not capturing at least one porpoise was less than one for all cases in 2001 (Table 3).

This resulted in a significant effect of mesh type where catch rates of harbour porpoise in nylon mesh gillnets were significantly higher than in barium sulphate mesh gillnets (*p*=0.017). Only haddock (*Melanogrammus aeglefinus*) catch rates were affected by mesh type (*p*=0.010; Table 4) where standard nylon nets were 1.5 times (95%CI=1.1-2.1 times) more efficient than barium sulphate modified nets. On a monthly basis, catch rates of haddock were lower in barium sulphate modified nets except August 2001 (Fig. 6). Mesh type did not have a significant effect on catch rates of Atlantic cod (*Gadus morhua*) (*p*=0.726), pollock (*Pollachius virens*) (*p*=0.109) or spiny dogfish (*Squalus acanthias*) (*p*= 0.727; Table 4).

DISCUSSION

Given the precarious population status of many marine mammals worldwide, effective protective measures must be put into place to mitigate external sources of mortality. Barium sulphate modified fishing gear shows considerable promise as a mitigation tool to reduce bycatch in Northwest Atlantic gillnet and lobster pot fisheries. The addition of barium sulphate to rope can result in both a lower profile in the water column and a lower breaking strength. The advantages of this are two fold. First, the probability of large whales becoming entangled in negatively buoyant lines should be reduced over lines that maintain a high profile and remain suspended in the water column. If by chance, a large whale did become entangled in barium sulphate modified rope such as in the headrope of a gillnet, it would be able to break free much easier as its breaking strength was found to

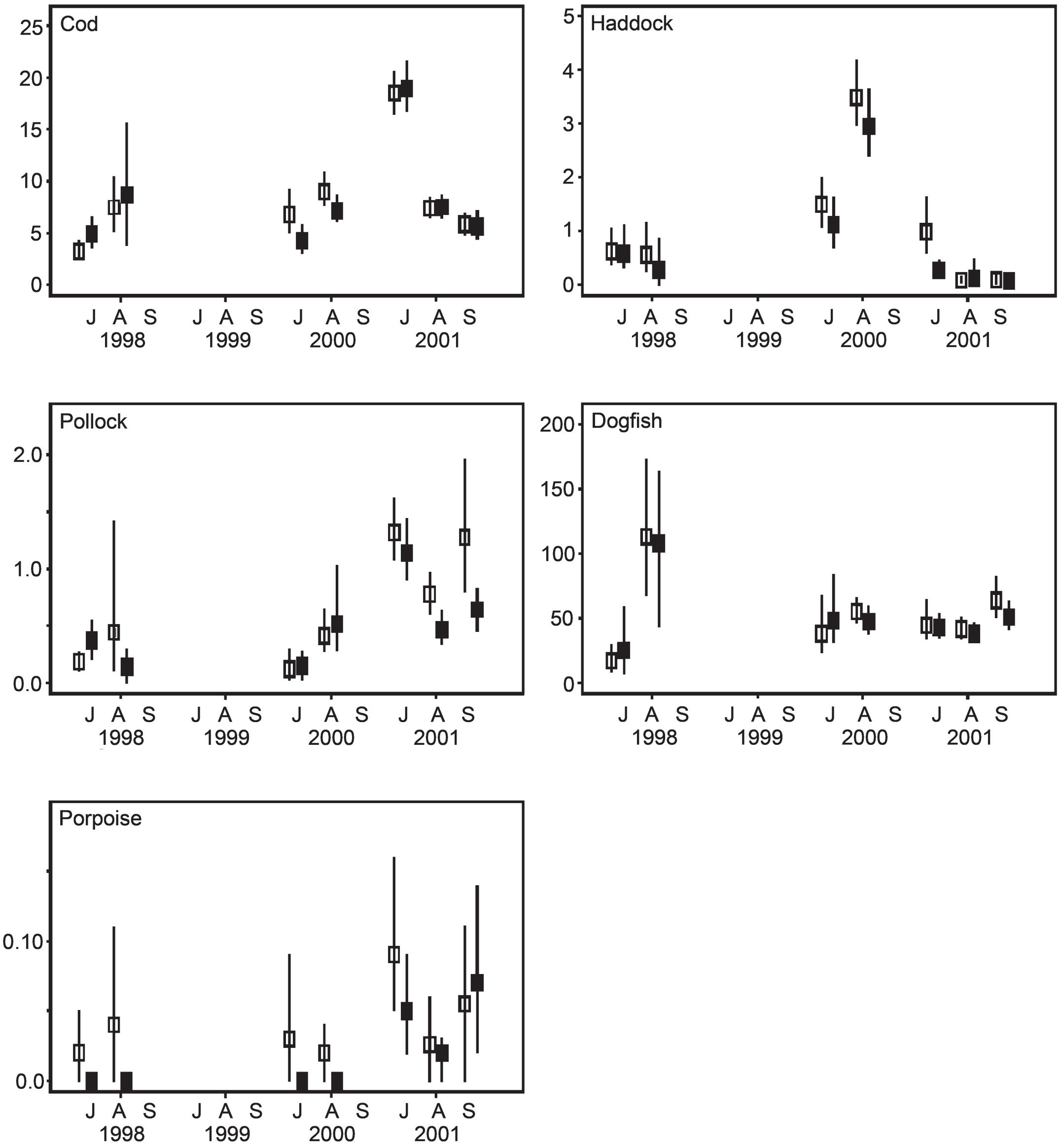

Fig. 6. Observed harbour porpoise and groundfish catch rates in gillnets (white: nylon mesh, black: barium sulphate mesh) in the Swallowtail area of Grand Manan, New Brunswick from 1998 to 2001 during the months of July (J), August (A) and September (S). 1999 results are not shown as mixed-mesh panel gillnets were used in this year. Vertical bars represent 95% confidence intervals.

be less than half of that of traditional polypropylene rope (Table 2). Additionally, in all cases when the breaking strength of barium sulphate modified rope was measured, it was below legislated limits required for fishing gear in many US waters.

Barium sulphate modified gillnets were also found to be effective in reducing harbour porpoise bycatch. The premise under which this gillnet material was designed was that the addition of a heavy metal salt to the nylon monofilaments would increase the echolocation signature or target strength of the nets. While this at first seems intuitive, others have found that harbour porpoise echolocation behaviour did not change in the presence of barium sulphate modified gillnets and have suggested reductions in bycatch were instead associated with the increased stiffness of the nets (Cox and Read, 2004). In a later study, Koschinski *et al.* (2006) found the distribution of harbour porpoise click intervals shifted to longer intervals when approaching a barium sulphate modified gillnet. They were able to estimate that harbour porpoise could detect the barium sulphate nets 4.4m in advance of regular nylon monofilament gillnets. However, they also found only 30% of individuals were actively echolocating when near the net and recommended the use of 2.5kHz warning tones to cause increased echolocation activity and thus an increased probability of a porpoise detecting the net. No gillnet field trials to date have been conducted that integrated barium sulphate nets with warning sounds.

The acoustic properties of barium sulphate modified gillnets have been evaluated in a number of studies (Koschinski *et al*., 2006; Larsen *et al*., 2007; Mooney *et al*., 2007; Mooney *et al*., 2004; Trippel *et al*., 2003). Using 200kHz multibeam sonar, Trippel *et al*. (2003) found that barium sulphate gillnets were approximately three times (2.6-3.3) more acoustically reflective than standard, nylon mesh gillnets. While most of the energy of the harbour porpoise echolocation click is between 140-160kHz (Au *et al*., 1999), they argued that these frequencies the acoustic reflectivity will be only slightly less than at 200kHz. In addition, sonar tests in the frequency range of 110-190kHz showed that the target strength of barium sulphate modified gillnets was 7.2dB higher than the target strength of standard nylon mesh nets at 150kHz (Koschinski *et al*., 2006). In another study, Mooney *et al.* (2004) evaluated the acoustic reflectivity of barium sulphate modified gillnets using signals of 80μs in duration, with a peak frequency of 120kHz and a 3dB bandwidth of 35kHz. After measuring reflectivity at a number of incident angles, they found target strength, and thus acoustic reflectivity, was higher for barium sulphate gillnets as compared to standard nylon mesh gillnets (Mooney *et al*., 2007; Mooney *et al*., 2004). However, they argued that given the relatively small increase in detection distances achieved by the barium sulphate gillnets and the swimming speed of small cetaceans, it is not clear if they could detect the modified gillnets in time to avoid entanglement.

The reduction in harbour porpoise bycatch found for barium sulphate modified gillnets may also be partially due to increased stiffness. Beyond their increased acoustic reflectivity, these nets have been shown to have an increased stiffness over regular nylon mesh gillnets (Mooney *et al*., 2007). Increased stiffness would be expected to reduce catches of both the echolocating porpoise and fish. In this study, reduced catches of haddock were found in barium sulphate modified gillnets, which perhaps could be explained through increased stiffness. This hypothesis has been previously suggested as the reason for reduced porpoise entanglements in chemically modified gillnets (Cox and Read, 2004). Given the modified behaviour of porpoise in the vicinity of barium sulphate modified gillnets (Koschinski *et al*., 2006) and the increased acoustic target strength of these nets (Mooney *et al*., 2007; Mooney *et al*., 2004), it is likely reduced bycatch results from both the increased acoustic reflectivity and stiffness. The original intention of using barium sulphate was to simply increase target strength, though the increase in stiffness was an additional benefit that was unexpected.

Barium sulphate is inexpensive and commercially available and was therefore considered a good candidate substance to be used to explore modifications that aim to reduce cetacean bycatch; other products that may have a similar effect include lead and stainless steel, though these would be confined to use as added strands within rope to achieve negative buoyancy. The barium sulphate is purchased as 'blanc fixe', a white solid with a particle size under 1μm. Barium sulphate has the chemical code of CAS No. 7727-43-7; the Material Safety Data Sheet reveals that while the pure powder should not be ingested, the skin contact is listed as 'no adverse effects expected'. No adverse effects are expected when barium sulphate is encased in nylon. Due to the low solubility of barium sulphate in seawater, the concentration of barium in solution in seawater cannot rise high enough to represent a toxic risk to marine organisms (Neff, 2002).

Barium sulphate modified fishing gears are not without their drawbacks. Most notably, the 20% barium sulphate 0.5" rope when used with lobster gear appears to be susceptible to chaffing and subsequent failure when used over rough fishing grounds. This was a concern of more than one fisherman. In many applications, the gear would be fished over sandy bottoms so this would not be a major concern, however within the Bay of Fundy, fishing over rocky bottoms is common. Field testing over rocky bottoms of other types of neutrally buoyant rope for groundlines of lobster trawl is required and is underway.

In controlled field trials, acoustic alarms have been shown to be more effective than barium sulphate modified gillnets in reducing porpoise entanglements (Kraus *et al*., 1997; Trippel *et al*., 1999). However, the relatively high catch rates in the barium sulphate modified gillnets in 2001 may be related to an unusually high abundance of harbour porpoise in the area that year as measured from herring (*Clupea harengus*) weir entrapments (Trippel and Shepherd, 2004). Despite the possibility of being less efficient than acoustic alarms in controlled tests, barium sulphate gillnets are silent, do not require an external power source and are otherwise operationally identical to standard nylon gillnets. As such, they avoid many of the disadvantages associated with the long-term deployment of pingers (non-compliance, loss or breakage, low battery power or non-replacement of batteries, and the need to maintain an additional piece of gear). Further, compared to regular nylon mesh gillnets, the cost of barium sulphate modified nets should be comparable and they are expected to have a 10-15% longer lifespan due to the presence of barium sulphate and the manufacturing process (this is based upon the properties of nylon in other manufacturing operations where adding a solid makes the nylon more durable). Thus, barium sulphate modified nets may be more readily adopted by fishermen than pingers and hold considerable promise in not only reducing harbour porpoise entanglements but are also worth exploring for other small cetacean bycatch problems.

Cetaceans in the Northwest Atlantic are facing continued and increasing human-induced mortalities despite considerable restrictions on fishing activity (Kraus *et al*., 2005; NMFS, 2006). Suggested measures to reduce entanglement mortalities include reductions in pot gear fishing effort (Kraus *et al*., 2005; Myers *et al*., 2007). Other mitigative solutions have suggested the use of alternative rope types to reduce entanglement deaths (Kraus *et al*., 2005). Barium sulphate modified rope, which is negatively buoyant and at the same time, weaker than standard rope, provides such an alternative. Barium sulphate modified nets appear to be a possible method to reduce harbour porpoise bycatch in the gillnet fishery, which has seen increasing mortalities in the US Northwest Atlantic since 2001 (NMFS, 2006). We believe the use of mitigative solutions, such as barium sulphate modified fishing gear, will be able to play an important role in future management measures aimed at reducing cetacean mortalities.

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REFERENCES

Au, W., Kastelein, R., Rippe, T. and Schooneman, N. 1999. Transmission beam pattern and echolation signals of a harbor porpoise (*Phocoena phocoena*). *J. Acoust. Soc. Am.* 106(6): 3699-705.

Brown, M.W., Fenton, D., Smedbol, K., Merriman, C., Robichaud-Leblanc, K. and Conway, J.D. 2009. Recovery strategy for North Atlantic right whale (*Eubalaena glacialis*) in Atlantic Canadian waters. Species at Risk Act recovery strategy series, Fisheries and Oceans, Canada. vi+66pp.

COSEWIC. 2003. *COSEWIC assessment and update status report on the North Atlantic right whale Eubalaena glacialis in Canada*. Committee on the Status of Endangered Wildlife in Canada, Ottawa, Canada. 28pp.

Cox, T.M. and Read, A.J. 2004. Echolocation behaviour of harbor porpoises *Phocoena phocoena* around chemically enhanced gill nets. *Mar. Ecol. Prog. Ser.* 279: 275-82.

Culik, B.M., Koschinski, S., Tregenza, N. and Ellis, G.M. 2001. Reactions of harbour porpoises (*Phocoena phocoena*) and herring (*Clupea harengus*) to acoustic alarms. *Mar. Ecol. Prog. Ser.* 211: 255-60.

Cunningham, R.B. and Lindermayer, D.B. 2005. Modelling count data of rare species: some statistical issues. *Ecology* 86: 1135-42.

Fontaine, P.M., Barrette, C., Hammill, M.O. and Kingsley, M.C.S. 1994. Incidental catches of harbour porpoises (*Phocoena phocoena*) in the Gulf of St. Lawrence and the St. Lawrence River Estuary, Quebec, Canada. *Rep. int. Whal. Commn (special issue)* 15: 159-63.

Gaskin, D.E. 1984. The harbour porpoise, *Phocoena phocoena* (L.): regional populations, status and information on direct and indirect catches. *Rep. int. Whal. Commn* 34: 569-86.

Gaskin, D.E. 1992. Status of the harbor porpoise, *Phocoena phocoena*, in Canada. *Can. Field-Nat.* 106(1): 36-54.

Gearin, P.J., Gosho, M.E., Laake, J., Cooke, L., Delong, R.L. and Hughes, K.M. 2000. Experimental testing of acoustic alarms (pingers) to reduce bycatch of harbour porpoise, *Phocoena phocoena*, in the state of Washington. *J. Cetacean Res. Manage.* 2(1): 1-10.

IUCN. 2008. The IUCN Red List of Endangered Species. [Available from *http://www.iucnredlist.org/*].

Johnson, A., Salvador, G., Kenney, J., Robbins, J., Kraus, S., Landry, S. and Clapham, P. 2005. Fishing gear involved in entanglements of right and humpback whales. *Mar. Mammal Sci.* 21(4): 635-45.

Kastelein, R.A., Rippe, H.T., Vaughan, N., Schooneman, N.M., Verboom, W.C. and De Haan, D. 2000. The effects of acoustic alarms on the behavior of harbor porpoises (*Phocoena phocoena*) in a floating pen. *Mar. Mammal Sci.* 16(1): 46-64.

Knowlton, A.R. and Kraus, S.D. 2001. Mortality and serious injury of northern right whales (*Eubalaena glacialis*) in the western North Atlantic Ocean. *J. Cetacean Res. Manage. (special issue)* 2: 193-208.

Knowlton, A.R., Man, M.K., Pettis, H.M., Hamilton, P.K. and Kraus, S.D. 2003. Analysis of scarring on North Atlantic right whales (*Eubalaena glacialis*): monitoring rates of entanglement interaction. Final report to the US National Marine Fisheries Service. 18pp. [Available from the New England Aquarium, Central Wharf, Boston, MA 02110].

Koschinski, S., Culik, B.M., Trippel, E.A. and Ginzkey, L. 2006. Behavioural reactions of free-ranging habour porpoises *Phocoena phocoena* encountering standard nylon and BaSO4 mesh gillnets and warning sound. *Mar. Ecol. Prog. Ser.* 313: 285-94.

Kraus, S.D., Brown, M.W., Caswell, H., Clark, C.W., Fujiwara, M., Hamilton, P.K., Denney, R.D., Knowlton, A.R., Landry, S., Mayo, C.A., McLellan, W.A., Moore, M.J., Nowachek, D.P., Pabst, D.A., Read, A.J. and Rolland, R.M. 2005. North Atlantic right whales in crisis. *Science* 309: 561-62.

Kraus, S.D., Read, A.J., Solow, A., Baldwin, K., Spradlin, T., Anderson, E. and Williamson, J. 1997. Acoustic alarms reduce porpoise mortality. *Nature* 388: 525.

Lambert, D. 1992. Zero-inflated Poisson regression, with an application to defects in manufacturing. *Technometrics* 34: 1-14.

Larsen, F., Eigaard, O.R. and Tougaard, J. 2007. Reduction of harbour porpoise (*Phocoena phocoena*) bycatch by iron-oxide gillnets. *Fish. Res. (Amst.)* 85: 270-78.

Lyman, E.G. and McKiernan, D.J. 2005. Scale modeling of fixed-fishing gear to compare and quantify differently configured buoyline and groundline profiles: an investigation of entanglement threat. Massachusetts Division of Marine Fisheries, Technical Report TR-22. 46pp.

McKiernan, D.J. 2002. A study of the underwater profiles of lobster trawl ground lines. Massachusetts Division of Marine Fisheries, Contract no. 50EANF-1-00048. 18pp.

Minami, M., Lennert-Cody, C., Gao, W. and Roman-Verdesoto, M. 2007. Modeling shark bycatch: the zero-inflated negative binomial regression model with smoothing. *Fish. Res. (Amst.)* 84: 210-21.

Mooney, T.A., Au, W.W.L., Nachtigall, P. and Trippel, E.A. 2007. Acoustic and stiffness properties of gillnets as they relate to marine mammal bycatch. *ICES J. Mar. Sci.* 64: 1324-32.

Mooney, T.A., Nachtigall, P. and Au, W.W.L. 2004. Target strength of a nylon monofilament and an acoustically enhanced gillnet: predictions of biosonar detection ranges. *Aquat. Mamm.* 30: 220-26.

Moore, M.J., Knowlton, A.R., Kraus, S.D., McLelland, W.A. and Bonde, R.K. 2004. Morphometry, gross morphology and available histopathology in North Atlantic right whale (*Eubalaena glacialis*) mortalities (1970 to 2002). *J. Cetacean Res. Manage.* 6(3): 199-214.

Myers, R.A., Boudreau, S.A., Kenney, R.D., Moore, M.J., Rosenberg, A.A., Sherrill-Mix, S.A. and Worm, B. 2007. Saving endangered whales at no cost. *Curr. Biol.* 17: R10-R11.

Neff, J.M. 2002. *Bioaccumulation in Marine Organisms*. Elsevier Press, New York. 451pp.

NMFS. 2006. Harbour porpoise (*Phocoena phocoena*): Gulf of Maine/Bay of Fundy stock. Draft Stock Status Report, NMFS, Woods Hole, MA.

NOAA. 1998. Taking of marine mammals incidental to commercial fishing operations; Harbour porpoise take reduction plan regulations. *Federal Register Notice* 63(231): 464-90. Department of Commerce, National Oceanic and Atmospheric Administration 50CFR Part 229.

Perrin, W.F., Donovan, G.P. and Barlow, J. 1994. *Report of the International Whaling Commission (Special Issue 15). Gillnets and Cetaceans*. International Whaling Commission, Cambridge, UK. 629pp.

Potts, J.M. and Elith, J. 2006. Comparing species abundance models. *Ecol. Modelling* 199: 153-63.

Read, A.J., Drinker, P. and Northridge, S. 2004. Bycatch of marine mammals in US and global fisheries. *Conserv. Biol.* 20(1): 163-69.

Read, A.J. and Gaskin, D.E. 1988. Incidental catch of harbor porpoises by gill nets. *J. Wildl. Manage.* 52(3): 517-23.

Trippel, E.A., Holy, N.L., Palka, D.L., Shepherd, T.D., Melvin, G.D. and Terhune, J.M. 2003. Nylon barium sulphate gillnet reduces porpoise and seabird mortality. *Mar. Mammal Sci.* 19: 240-43.

Trippel, E.A. and Shepherd, T.D. 2004. Bycatch of harbour porpoise (*Phocoena phocoena*) in the lower Bay of Fundy gillnet fishery, 1998-2001. *Can. Tech. Rep. Fish. Aquat. Sci.* 2521: iv and 33pp.

Trippel, E.A., Strong, M.B., Terhune, J.M. and Conway, J.D. 1999. Mitigation of harbour porpoise (*Phocoena phocoena*) by-catch in the gillnet fishery in the lower Bay of Fundy. *Can. J. Fish. Aquat. Sci.* 56: 113-23.

Trippel, E.A., Wang, J.Y., Strong, M.B., Carter, L.S. and Conway, J.D. 1996. Incidental mortality of harbour porpoise (*Phocoena phocoena*) by the gillnet fishery in the lower Bay of Fundy. *Can. J. Fish. Aquat. Sci.* 53: 1294-300.

Vinther, M. 1999. Bycatches of harbour porpoises (*Phocoena phocoena*, L.) in Danish set-net fisheries. *J. Cetacean Res. Manage.* 1(2): 123-35.

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Observations of western gray whales by ship-based whalers in the 19th century

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ABSTRACT

Animals belonging to the small, endangered population of western gray whales (*Echrichtius robustus*) are observed today primarily during the summer open-water season in feeding areas off the northeastern coast of Sakhalin Island, Russia. The migration route(s) and wintering area(s) used by this population are largely unknown. Gray whales once had a fairly extensive distribution in the Sea of Okhotsk but little detailed information has been published on when and where they occurred. Open-boat, ship-based whalers from the United States and a few other countries conducted an intensive hunt for bowhead whales (*Balaena mysticetus*) and North Pacific right whales (*Eubalaena japonica*) in the Sea of Okhotsk from the 1840s to 1870s. According to entries in voyage logbooks, the American whalers regularly encountered (and sometimes hunted) gray whales in the far northeastern corner of the Okhotsk Sea (Shelikhov Bay, Gizhiginskaya Bay and Penzhinskaya Gulf) between early May-late August. They also observed gray whales in summer along the northern coast of the sea (especially Tauskaya Bay), around the Shantar Islands, in Sakhalin Bay, off Cape Elizabeth at the northern tip of Sakhalin Island and along the west coast of the Kamchatka peninsula. No evidence was found in the logbooks studied of gray whales (and indeed of whaling effort) off northeastern Sakhalin Island where most observations of gray whales occur in the present day.

KEYWORDS: GRAY WHALE; WHALING-HISTORICAL; SEA OF OKHOTSK; NORTHERN HEMISPHERE; BOWHEAD WHALE; NORTH PACIFIC RIGHT WHALE

INTRODUCTION

Recent and current interest in gray whales (*Eschrichtius robustus*) from the western Pacific (Korean-Okhotsk) population has centred on that population's endangered status and the ongoing threats to its survival and recovery (e.g. IWC, 2004). Specifically, there is concern about: (a) the small number of whales in the population; (b) environmental degradation and disturbance from oil and gas development on the northeastern Sakhalin Island shelf, the main area where the population is presently known to congregate in summer to feed; and (c) mortality of gray whales in Japanese waters, mainly in set nets.

The history of this population has been reviewed by Mizue (1951), Nishiwaki and Kasuya (1970), Brownell and Chun (1977), Omura (1984; 1988), Weller *et al*. (1999; 2002), Kasuya (2002), Uni and Kasuya (2002) and Kato and Kasuya (2002). In addition to those reviews, Henderson (1972; 1984; 1990) made reference to ship-based whaling on the western population during the 19th century, primarily by American and French whalers. In our recent studies of whaling history in the North Pacific (e.g. Josephson *et al*., 2008), we have examined a sample of American voyage logbooks that contain substantial unpublished data on western gray whales. The present paper uses those data to describe where and when western gray whales were observed by ship-based whalers during the 19th century.

The data confirm that in the past gray whales used various parts of the Okhotsk Sea, probably as feeding grounds and as routes to and from such grounds. Given the rapid proliferation of offshore oil and gas operations around the perimeter of the Okhotsk Sea as well as on the entire Sakhalin Shelf (e.g. Reeves *et al*., 2005), understanding the historical (and thus both current and potential) summer range is essential.

BACKGROUND

Offshore or ship-based whaling for gray whales along the Asian coast from southern China to Japan and in the Sea of Okhotsk has been less well documented than shore-based whaling there. Among the reasons for this difference is the death in June 1999 of David Henderson, who authored classic studies of the American pre-modern, ship-based fishery for *eastern* Pacific gray whales (1972, 1984) and was working on a similarly exhaustive study of the Okhotsk fishery for *western* Pacific gray whales (e.g. see Henderson, 1984, p.176, note 14; Kugler, 1984, p.157, note 6). Henderson (1972) provided only limited information on western gray whales although on his Map I, he offered an intriguing sketch of their distribution (Fig. 1). Henderson (1984, pp.176-7) indicates that gray whales were not hunted by the American whalers in the Sea of Okhotsk until sometime in the 1840s and that catches of 6-7 whales were being made by some ships by the 1850s. He judged that the total kill of gray whales in the Okhotsk Sea by the American fleet was probably similar to that in the Bering Sea and Arctic Ocean, i.e. a few hundreds (his estimate of total kill in the latter areas between 1845-1874, adjusted for hunting loss, was 539); (Henderson, 1984, p.169). Some American whaling for gray whales continued in the Sea of Okhotsk until at least the mid-1880s (Henderson, 1984, p.177), by which time most of the remaining Arctic fleet was committed to the 'hazardous, though profitable, whaling in the Arctic [i.e. Bering, Chukchi and Beaufort seas]' (Clark, 1887, p.19). The Sea of Okhotsk was also a frequent destination of French whalers from the mid 1840s until perhaps the mid 1860s (Du Pasquier, 1982, pp.183, 192, 245-9; Kugler, 1984, p.152) and they probably took at least some gray whales although we have not found any direct evidence for this (Du Pasquier, 1986, p.274). At least three

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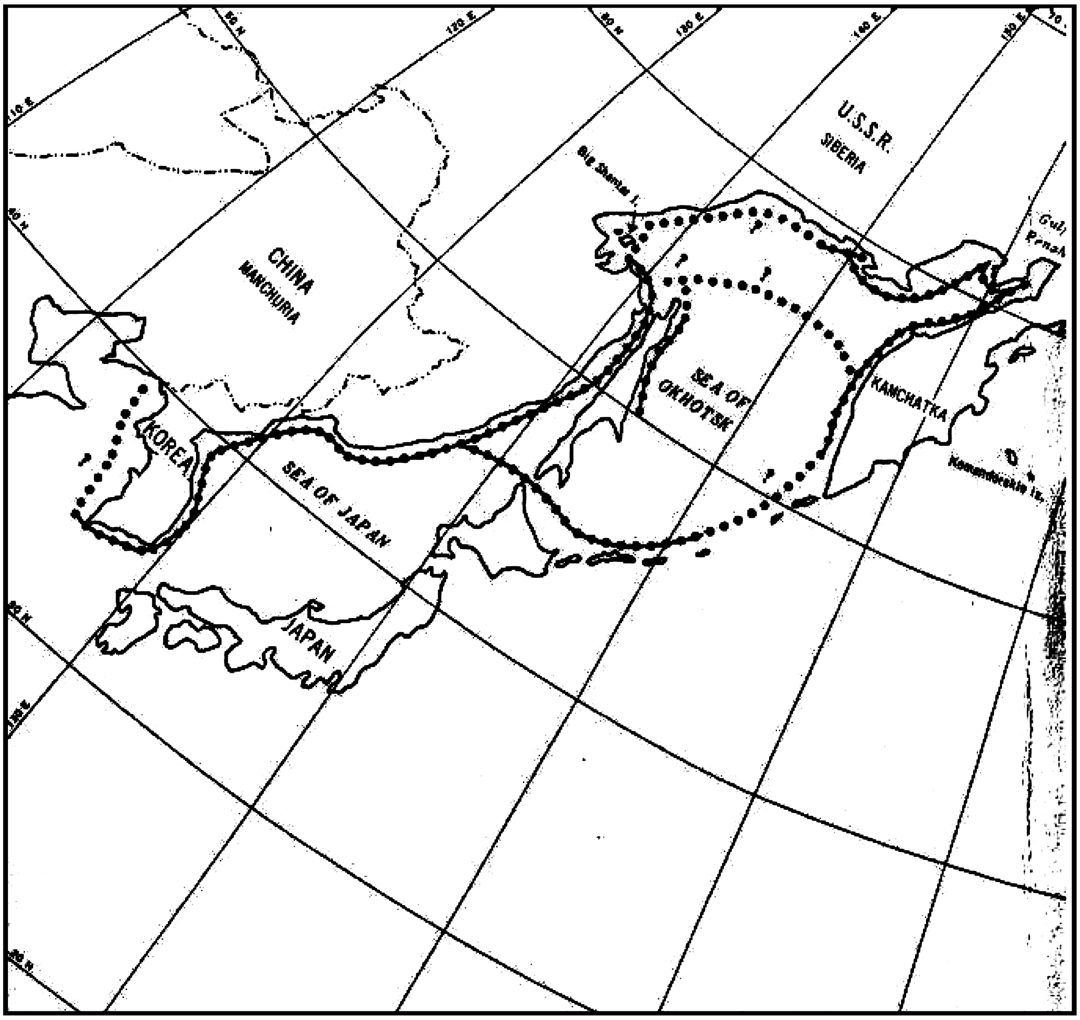

Fig. 1. Excerpt from 'Distribution, Migration Routes, and Calving Grounds of Pacific Gray Whales' in Henderson (1972). Cited sources used by Henderson as the basis for this map include Mizue (1951), Tomilin (1957) and 'Scammon's and other historical records'.

Russian whaleships (all originating from Finland) also visited the Okhotsk Sea in the 1850s (Clark, 1887, pp.206-7).

The Okhotsk Sea fishery for balaenids was most intensive from 1847-1867, with nearly 1,400 vessel-seasons, 90% of them from the United States and the rest from France, Bremen, Russia and the Hawaiian Islands, and occasionally Great Britain, Norway and Chile (Kugler, 1984, p.153). As an example of the intensity of this fishery, one ship's logbook reported that 65 other whaling vessels were in sight on 27 August 1854 in Shantar Bay (*Good Return*). In that same month, Lindholm (1863) counted 82 ships in the bay and on one day 363 whaleboats were in sight from his ship (*Storfursten Constantin*). Bowhead whales (*Balaena mysticetus*) and North Pacific right whales (*Eubalaena japonica*) were the principal targets and, according to Henderson's preliminary analyses (as reported by Kugler, *op cit.)*, more than 15,000 bowheads and 2,400 right whales were killed and processed by the Okhotsk whalers in those first 20 years. The fishery continued until the 1890s.

Henderson's only publication containing data on western Pacific gray whales (apart from the few references to Okhotsk whaling in his 1972 book and his 1984 book chapter) was a short article on American whaling in southern China in the 1860s (Henderson, 1990). There he recounted two voyages to the 'Chinese gray whale ground' in the winter of 1868-69 (by the New Bedford ships *Cornelius Howland* and *Onward*). Although they failed to strike any, the crews of these vessels sighted gray whales near the Chinese mainland coast at c.25-26°N off the island of 'Hatan Ho Tan' (Haitan, Pingtan Dao, or Tao on modern maps, according to Henderson), in the middle of Taiwan (Formosa) Strait and off the northern coast of Taiwan. Ellis (1991) cited two references to Chinese whaling, one a report by John Nieuhoff in 1673 describing the hunt for a species of baleen whale near Hainan Island, the other a more detailed account of Chinese coastal whaling in a September 1844 issue of *The Friend*, a Honolulu newspaper. The newspaper story noted that whales with young congregated 'in great numbers' south of Hailing Shan in January and February and were hunted with harpoons in very shallow water. Although the author of the article suspected these were right whales, the description is consistent with their being gray whales – 'covered with barnacles', breaching frequently, occurring over shallow sandbanks and yielding an average of about 50 bbl of oil (not unusually high for female gray whales that are near term or in early lactation). Just as Scammon (1874, p.23) concluded that a report of gray whales on the coast of China and about the shores of the island of Formosa 'needs confirmation', a certain degree of uncertainty still surrounds the topic of whether western gray whales were heavily hunted on their wintering grounds in the nineteenth century.

MATERIALS AND METHODS

In addition to a search of the literature, whaling voyage logbooks from the Kendall Whaling Museum and Old Dartmouth Historical Society collections, both available at the New Bedford Whaling Museum library in New Bedford, Massachusetts, were sampled. Okhotsk Sea logbooks were identified using library finding aids, Starbuck (1878) and Sherman *et al*. (1986) and through references in logbooks to other vessels sighted or 'spoken' on the grounds. Data from the logbooks, including date, position, details concerning whale observations, and other vessels spoken, were entered into a Microsoft Access database and plotted using ArcMap.

Frequently, the exact location could not be determined from the logbook and therefore it was necessary to estimate positions by interpolation, extrapolation and reference to landmarks. A particular problem encountered while working with this material was that place names used by the American whalers did not always correspond to the Russian names. For that reason, a gazetteer was compiled in the course of logbook reading as a research tool. Sketch maps prepared by the whalers themselves were useful in that regard (Fig. 2). A composite map of the region was developed from a variety of sources (Fig. 3). When places are mentioned throughout the text of this paper, alternative names have been indicated in brackets.

The American whalemen used several different terms to refer to gray whales, including ripsacks (rip sacks), musseldiggers (mussel diggers), devil fish, scrags, scamper downs, California grays, gray backs and California whales. Henderson (1972, pp.34-5; (1984, pp.163-4) included mud digger, digger, California ranger and hard head in the list of names used for the gray whale by American whalemen, but these terms were not encountered in the limited sampling of logbooks for this study. Another term, rock hopper, may have been used although it was not confirmed in the reading conducted for this study. Scammon (1874, p.24) listed only hard-head, mussel-digger, devil-fish, gray-back and rip-sack as the names he and other whalemen used besides California or gray whale (or California gray whale).

The American whalemen called bowheads polar whales (great polar whales according to Scammon, 1874), steeple tops, bowheads (bow heads) or often just whales (in most but not all instances, the species can be inferred from the context; see later). The term bunchback was applied to some bowheads, referring to a low dorsal hump on the caudal peduncle, and young bowheads in the Sea of Okhotsk were sometimes called poggies (Scammon, 1874, pp.60-1). Blue whales (*Balaenoptera musculus*) were consistently called sulphur (sulfur) bottoms. Right whales, humpback whales (*Megaptera novaeangliae*) and 'finback' whales (probably mainly fin whales, *B. physalus*, although this name also may have been applied to sei whales, *B. borealis*) were referred to as such. Scammon (1874, p.67) stated that American whalers called the North Pacific right whale 'north-west whale' to distinguish it from southern right whales but that term was not found in any of the Okhotsk logbooks examined for the present study. Scammon also made reference to a ''scrag' Right Whale' in the North Pacific (note that Mead and Mitchell, 1984 associated the term 'scrag' with the gray whale). Roys (in Maury, 1851, p.198) referred to the right whale in the western North Pacific as the Kamchatka whale but seemed to realise it was the same species as the north-west whale. No reference was found in any of the logbooks read for this study to sperm whales (*Physeter macrocephalus*) in the Sea of Okhotsk.

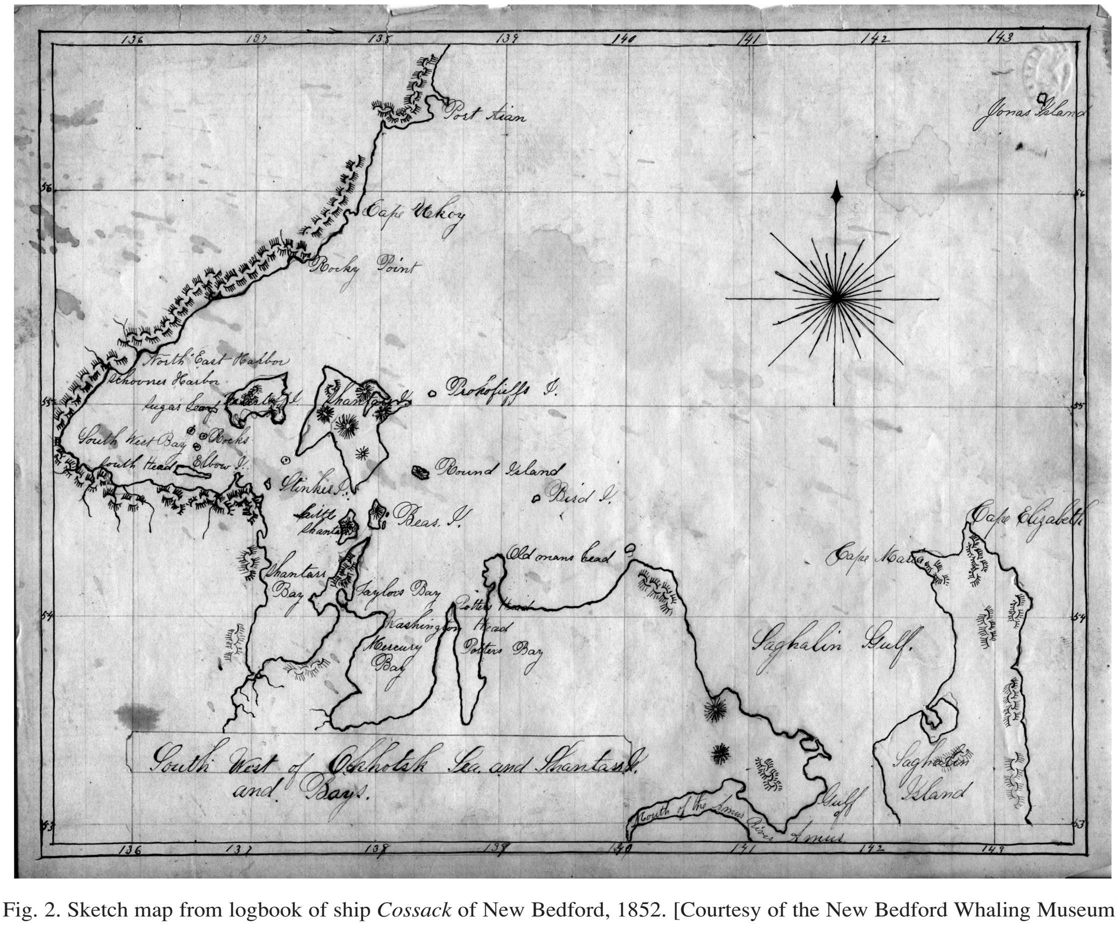

Fig. 2. Sketch map from logbook of ship *Cossack* of New Bedford, 1852. [Courtesy of the New Bedford Whaling Museum]

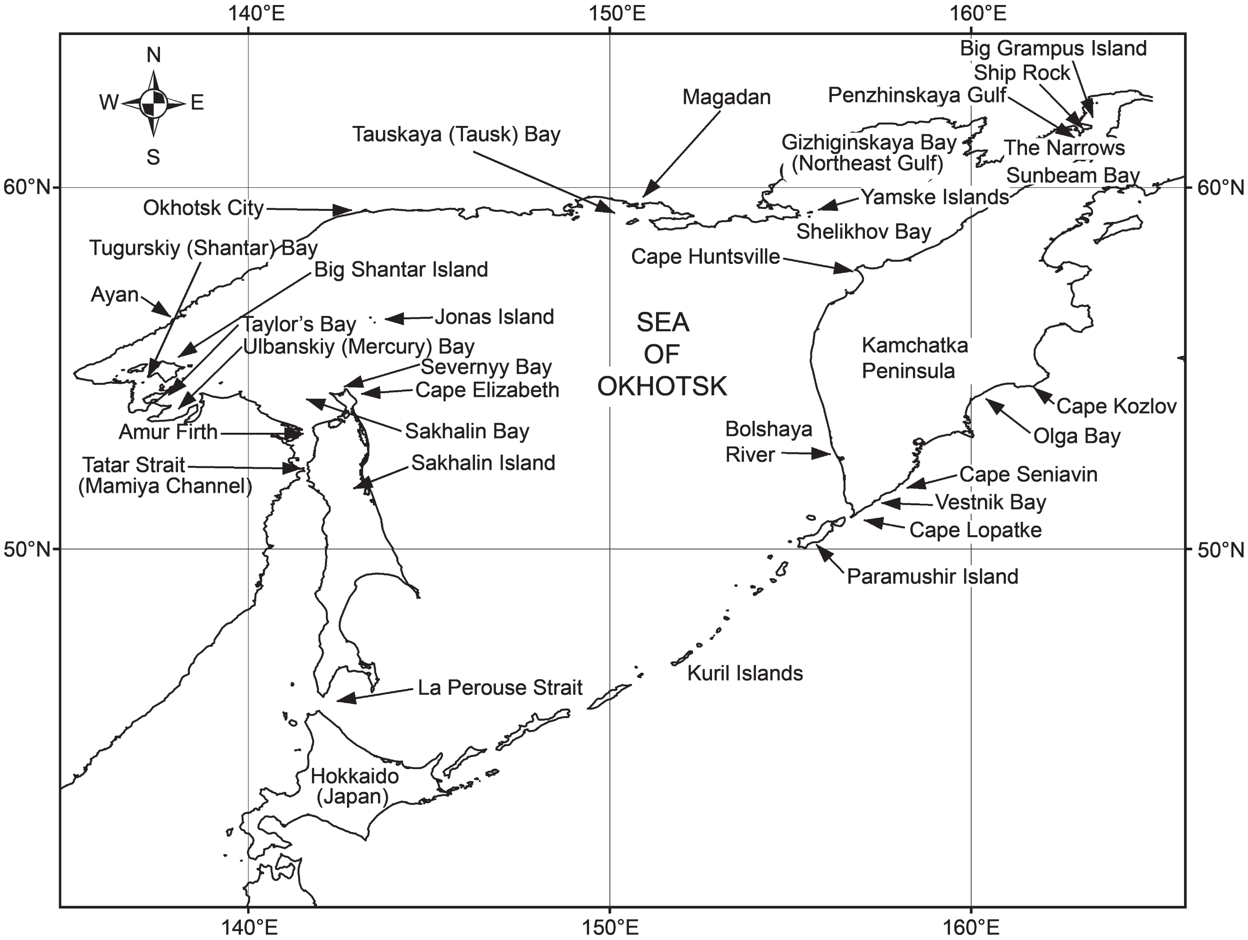

Fig. 3. Places mentioned in text.

RESULTS

The sample of logbooks examined for this study included 21 multiyear voyages that visited the Okhotsk Sea one or more times, for a total of 43 'vessel-seasons' in the Okhotsk. The ships were in the region for more than 5,000 vessel-days and daily positions were determined or estimated for most of those days (Fig. 4). The sampled voyages spanned the period from 1847-1885 though the majority of them took place in the 1850s and 1860s. Logbooks of 14 voyages, including 24 vessel-seasons in the Okhotsk, contained references to sightings of gray whales. There was a total of 160 daily entries with observations of gray whales, including sightings, chases, strikes and captures (Fig. 5).

Occurrence

Citing early Russian literature, Yablokov and Bogoslovskaya (1984) described gray whales (and humpback whales) as having been 'very common and even abundant in the coastal waters of the northern part of the Okhotsk Sea and off the western shores of the Kamchatka Peninsula'. They surmised (as did Tomilin, 1957, p.314) that gray whales migrated into the Okhotsk Sea from the Sea of Japan via both Tatar Strait (Tatarskiy Proliv) and La Perouse Strait. Indeed, American logbook data imply that gray whales at least occurred in or near La Perouse Strait in mid-June (e.g. some were sighted and chased there by the *Cicero*, 16 June 1859, c. 46°N, 142°E). Vladimirov (2004) questioned the feasibility of gray whales entering (or leaving) the Okhotsk Sea via Tatar Strait 'due to the small depths of the Amur Firth (2-3m) … which must be a natural obstacle for migrating animals'. In contrast, Rice and Wolman (1971, p.20), citing Mizue (1951), suggested that all gray whales passed through Tatar Strait 'as none was ever seen in La Perouse Strait'. According to Mizue (1951, p.79): '… it is reported by Mr. Tago that they reach Hokkaido or the western coast of Sakhalin in May or June and then through the Mamiya [Tatar] Channel go to the northern part of the sea of Okhotsk, where they seem to spend their summer. On their southwards migration they seem to take the same course as they come up north. It is not probable that grey whales pass through the Soya [La Perouse] Channel to the farther north, for fin and hump-back whales are captured there from the landstation in Hokkaido but not grey whales'. However, during the late 19th century the gray whale was considered the most frequently encountered baleen whale off the Sea of Japan coast of northern Hokkaido (i.e. Teshio) in late spring and early summer (T. Kasuya, pers. comm.). A total of 149 gray whales (13-29/yr) was taken by Japanese whalers off Teshio in the years 1889-1896 and some additional gray whales were taken off southern Sakhalin in the same period (Uni and Kasuya, 2002). It would seem, then, from the available evidence that at least in the past gray whales moved through both Tatar and La Perouse straits into, and possibly out of, the Sea of Okhotsk.

Many voyages to the Okhotsk Sea originated in Hawaii and therefore the ships entered via the northern Kuriles (usually in the vicinity of Paramushir Island) and then worked northwards along the western shore of Kamchatka. According to Henderson (1972, p.87, citing reports from *Fortune*, 6-8 June 1855; *Mary and Susan*, 19-30 August

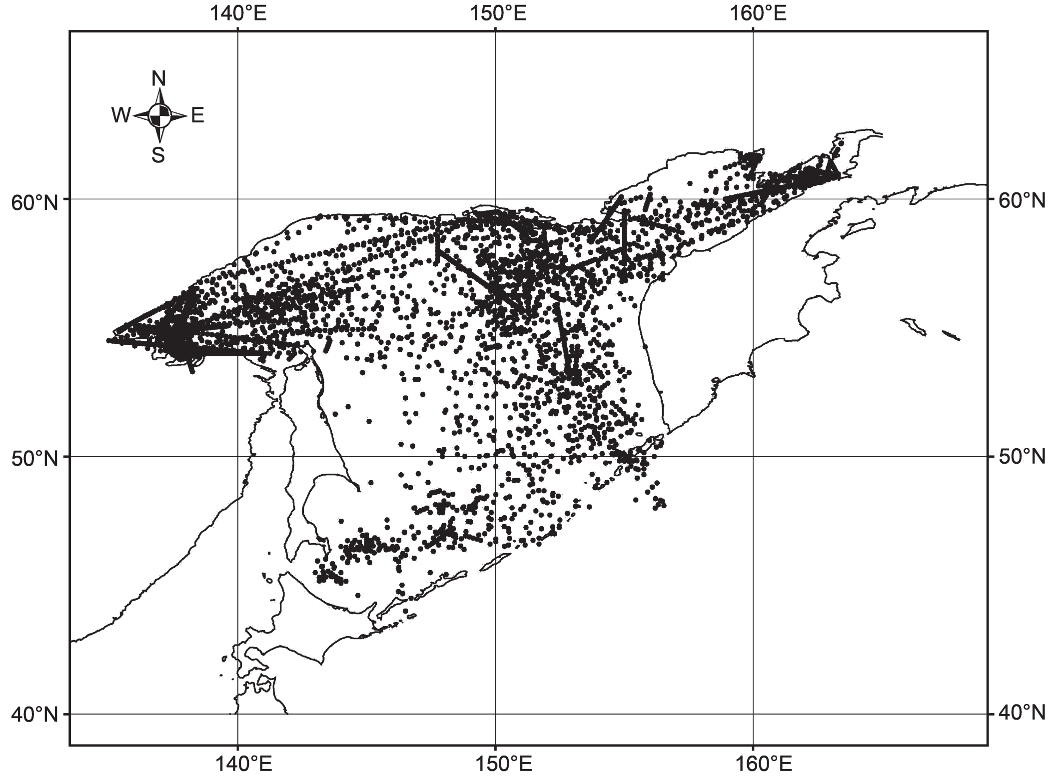

Fig. 4. Approximate positions of whaling vessels in and immediately outside the Sea of Okhotsk based on logbook records of 21 voyages (43 vessel-seasons). Note the nearly complete absence of search effort along the northeastern coast of Sakhalin Island.

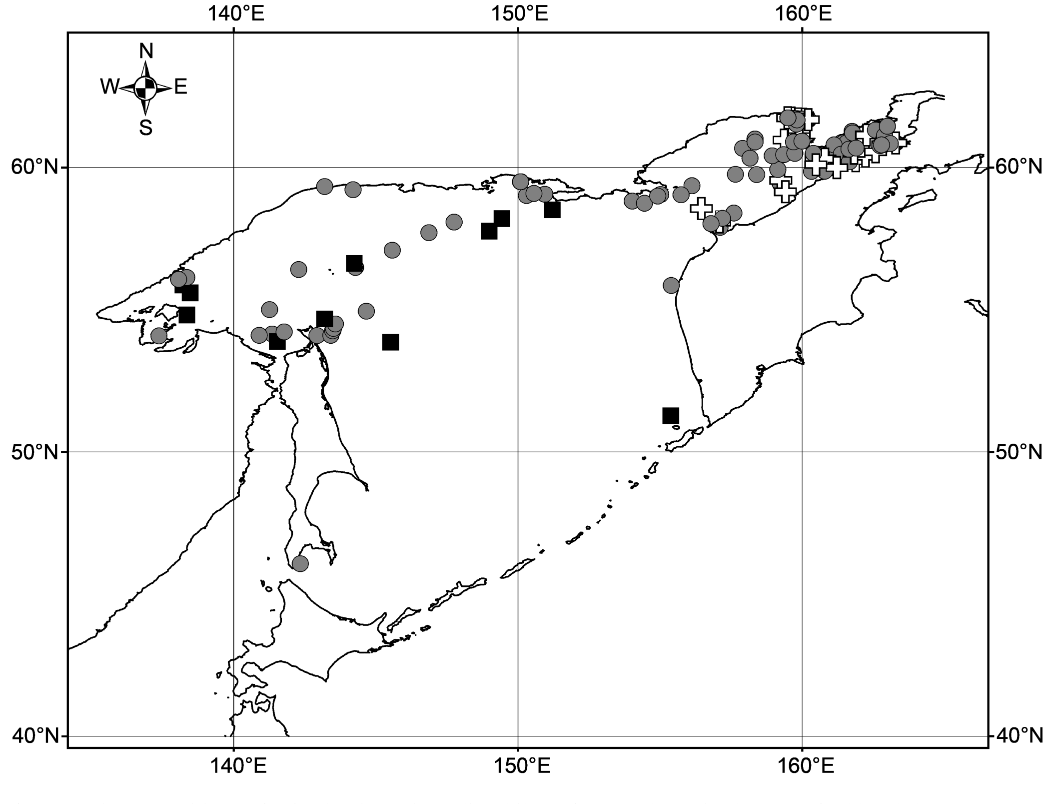

Fig. 5. Approximate positions of 160 sightings (including kills) of gray whales in the Sea of Okhotsk by 19th century American whalers. Note that some symbols are overprinted in areas with many observations. Crosses: May; circles: June-August; squares: September-October. Also note that many positions are based on extrapolation or interpolation from ship positions reported on days before or after that of the gray whale sighting. All data are from logbooks; see text.

1848; and *Montezuma*, 30 May and 29 September 1860), gray whales were observed mainly along the northern coast of the Sea of Okhotsk, and this is consistent with the information in Yablokov and Bogoslovskaya (1984).

Gray whales were observed consistently by the American whalers in the southeastern portion of Shelikhov Bay (Zaliv Shelikhova) and in Penzhinskaya Gulf (Penzhinskaya Guba) from early May (earliest record 6 May 1885; *Mary and*

Helen II) to the end of August (approximately 43 observations in the present sample; Fig. 5). They were seen in Gizhiginskaya Bay (North-east Gulf) between mid-May and late August (approx. 46 observations, including a report of 'thousands of devilfish but no bowheads' in the logbook of the *Benjamin Cummings* on 3 June 1869) and in the vicinity of Magadan along the north central coast of the Sea of Okhotsk (i.e. in Tauskaya Gulf/Guba, generally called Tausk or Taousk Bay) from at least early June to early July and from mid-August to mid or late September (approximately 19 observations). Gray whales were seen occasionally just east of Okhotsk City in the northwestern Sea of Okhotsk (*Lancaster*, 12 September 1860; *Oliver Crocker*, 29 August 1861). There is little evidence that they still occur in any of those northern areas (Maminov and Blokhin, 2004) although it is unclear how much search effort has been expended there at the appropriate times. It should also be noted that humpback whales were sometimes seen in large numbers – e.g. on 23 July 1854 between the Yamske Islands and Jonas Island (*Gov. Troup* 1853-56).

Only a handful of references to gray whales in the vicinity of the Shantar Islands were found in the logbooks: lowered for them on 13 August 1855 (*Gov. Troup*), saw 'a few rip sacks' northeast of Big Shantar Island on 5 June 1860 (*Oliver Crocker*), saw 'muscle diggers' on 7 September 1855 (*Nassau*) and saw 'Devilfish or Ripsack whales' in Taylor's Bay, off Ulbanskiy Bay, on 13 August 1885 (*Mary and Helen II*). Another sighting was recorded about midway between Ayan and the Shantar Islands on 14 June 1860 (*Lancaster*). Tomilin (1957, p.314) cited Russian observations in Akademii and Ulbanskiy bays. Also, in recent years a few observations have been reported in September in Tugurskiy and Ulbanskiy bays (Maminov and Blokhin, 2004). Humpback whales were chased occasionally in the Shantars as well (18 August 1854, *Gov. Troup*) but this was primarily a bowhead whaling area.

The American whalers observed gray whales, sometimes in sizeable concentrations (e.g. 20-30 seen in a day, references to 'plenty' being seen), in Sakhalin Bay (Sakhalinskiy Zaliv) and off Cape Elizabeth (Mys Yelizavety) at the northern tip of Sakhalin Island (*Mary and Susan*, various entries between 18 August-9 September 1848; *Gov. Troup*, 11 and 14 June 1855). A sighting of two gray whales in summer 2005 in Severnyy Bay, just southwest of Cape Elizabeth (Tyurneva *et al*., 2006), demonstrates the continued use of that area by the extant population. In addition, gray whales were seen in southern Sakhalin Bay in August 2000 (Maminov and Blokhin, 2004).

Finally, American whalers sighted gray whales at least occasionally near the northwestern (Cape Huntsville, 58°05'N, 157°06'W; *Europa*, 17 May 1869), central-western (vicinity of Moroshechnoye, 55°51'N, 155°52'E; *Mary and Susan*, 13 July 1849) and southwestern coasts of Kamchatka (51°16'N, 155°23'W; *Europa*, 14 September 1868). Although Blokhin (1996) reported that gray whales had not been seen along the western coast of Kamchatka in many years, one was reported in August 2000 at the mouth of the Bolshaya River (Vertyankin *et al*., 2004). In that regard, the logbook of *Mary and Helen II* refers to arrival at the Bolshaya mouth on 12 September 1885, the vessel having departed the Shantar Islands on 23 August bearing east 'for 'Bolshaya River' for Right Whales'. The 13 September logbook entry states: 'I am bound back to the west end of the Sea north of the Shantar Islands for I am convinced there are no whales to be seen about this locality or where I have been expecting to find 'Ripsack Whales' off the mouth of the 'Bolshaya River''. Having found neither right nor gray whales off Kamchatka, the *Mary and Helen II* relocated to the Sea's central-northern coast at Tauskaya Bay, with the top of the logbook page for 19-20 September declaring, 'In 'Tausk Bay' among the 'Ripsack Whales''. Within a few days thereafter, the vessel had again gone south to the centre of the Midas Ground at 53°02'N, 152°30'E (28 September) in pursuit of right whales.

In recent years, sightings of gray whales have become regular in Olga Bay (between Cape Seniavin and Cape Kozlov) and Vestnik Bay (at c. 51°35'N just north of Cape Lopatke) on the southeastern coast of Kamchatka (Vertyankin *et al*., 2004) and at least some of the individuals seen there have been photographically matched to individuals observed off northeastern Sakhalin (Yakovlev *et al*., 2007).

No evidence was found, beyond that presented by Henderson (1990), that American whalers observed or took gray whales in Chinese waters where sightings and a few strandings and kills were reported in the 20th century (Blokhin and Blokhin, 2006; Wang, 1984; 1993; 1999; Zhu, 1998; 2002). However, the present logbook sample included little coverage of whaling in those waters.

Removals

Our reading of logbooks thus far has not revealed any major inconsistencies with Henderson's (1984) findings as summarised above. Gray whales clearly were secondary targets and they were pursued mainly at times when the preferred bowheads and right whales were unavailable. For example, in 1859 the ship *Oliver Crocker* arrived in Tauskaya Bay at the end of May, with its first sighting of 'mussel diggers' reported on 3 June. Within a few days the boats were engaged in the pursuit of bowheads in the bay. Three were bomb-lanced and/or harpooned, but lost, between 6-10 June. By 15 June, the run of bowheads had ended and after a few more days the hunt for gray whales began. Between 19-30 June the *Crocker*'s boats were lowered daily and no fewer than nine gray whales were taken and processed. In addition at least two were lost when the harpoon drew or the line was cut. Two of the processed whales had sunk initially and, in both instances, were only recovered two days later. Three other ships were spoken in the bay during the second half of June – *L.C. Richmond*, *Robert Morrison* and *Cambria* – but it is unclear whether any of them took gray whales (the *Richmond* was seen taking a bowhead on 13 June; the *Cambria* reportedly had taken three bowheads that season as of 30 June). On 1 July, the *Crocker* sailed towards the Shantar Islands where bowheads were plentiful and hunting for them resumed on 11 July. There is no further mention of gray whales that year in the *Crocker* logbook. In the next two years, even though the *Crocker* followed a broadly similar itinerary, there is nothing in the logbook to suggest another episode of intensive gray whaling. Two were taken in North-east Gulf on 3 August 1861, the only day during the 1861 season when the logbook indicates the boats were lowered for 'ripsacks'.

The ship *Europa* arrived in Gizhiginskaya Bay (North-east Gulf) early in the 1868 season, with its first bowhead sighting on 19 May. From then until 28 June, when the first bowhead was taken, only one more bowhead was seen by the crew (on 7 June), whereas gray whales (and 'finbacks') were seen often and the boats were lowered on at least three occasions with the explicit intention of hunting gray whales (none was taken by the *Europa* until 20 June). The logbook records that on 5 June 1868: 'Lowered for Ripsacks for the purpose of trying our new whaling guns, tried four shots and the irons would not enter the blubber'. In 1869, the *Europa* followed the same itinerary but had more success finding and taking bowheads, and there was only one brief lull (11-12 June) when attempts were made to take both gray whales and a 'finback'. Later that season (26 July), the *Europa* 'picked up' and processed a dead gray whale found floating in Penzhinskaya Gulf between Ship Rock and Big Grampus Island and the log refers frequently to 'ripsacks' being observed (e.g. 17 August near Ship Rock, '…not seeing anything but Ripsacks') but with only one more desultory attempt made to hunt them (27 August).

In 1885, the *Mary and Helen II* reached the northeastern part of the Okhotsk Sea by the end of April and gray whales were sighted on 6 May and again on 20 May, by which time no bowheads had yet been seen. In fact, only one brief sighting of a bowhead (21 May) was made before mid-June. Even though gray whales were seen many more times, the boats were not lowered to chase them (except on 21 June and 11 July 'by mistake') until the episode in Tauskaya Bay in mid-September (mentioned above) when at least 4 gray whales were taken.

The following gray whale catch information was found in the documents examined for this study: 1 struck/lost by *Gov. Troup* in 1854; 9 (plus 2 struck/lost) by *Oliver Crocker* in 1859 and 3 in 1861 (see above); 1 by *Florida* (15 bbl) (Williams, 1964, p.185) and 1 by *South Boston* in 1861 (Williams, 1964, p.186); 4 by *California* in 1863 (Henderson, 1972, p.87); 1 each by *Endeavour*, *Rainbow* and *Europa* (plus 1 salvaged) in 1867 (*Europa* 1867); 1 by *Europa* (salvaged) in 1869; and 4 by *Mary and Helen II* in 1885.

The estimated total landed catch of western gray whales by modern whaling and Japanese net whaling between 1890-1966 was 1,800-2,000 (Kato and Kasuya, 2002). No other catches are known to have taken place during that time or since 1966 although some unreported catches may have been made by catcher boats from the People's Republic of China or the Republic of Korea, neither of which belonged to the International Whaling Commission until 1980 and 1978, respectively. It is also possible that some catches were made by whalers from the Democratic People's Republic of Korea or from Taiwan.

DISCUSSION AND CONCLUSIONS

Occurrence

Mizue (1951) analysed catch dates and positions for 545 gray whales in the 'East Sea Area' of Korea (i.e. in the Sea of Japan). The catches all occurred between November and May with a strong peak in December (63%) and January (22%). The infrequency of catches in other months from September through March apparently was not related to effort because, as Mizue (1951, p.76) points out, the same area was an important whaling ground for fin whales and 'many catcher-boats work during the season, from September to March next year' and so the whalers 'would have certainly caught grey whales if they had seen them in the months of October, November, February and March'.

The speculation by Mizue (1951) that gray whales migrated northwards from Korea through the Mamiya Channel (Tatar Strait) in May or June and thence to summering grounds in the northern Sea of Okhotsk is not inconsistent with some of the American whalers' observations. It is unclear, however, whether Mizue was correct in his belief (shared by Andrews, 1916, p.210) that gray whales calved and mated in Korean waters. There is no reason to doubt that at least some of the whales migrated to as far south as 25°N along the Chinese mainland and moved though Taiwan (Formosa) Strait (Henderson, 1990). It is also important to consider that Mizue's reasoning that gray whale females with large foetuses taken in mid-December off Korea were 'immediately before birth' and that 'delivery is made among the islands at the southern extremity of the Korean Peninsula' was without the benefit of Rice's (1983) analysis showing a 'prenatal diapause' in eastern Pacific gray whales such that foetal growth 'virtually ceases' during the final month of pregnancy and birth occurs between early January and mid-February (median 27 January). It is therefore plausible that the strong peak in occurrence of female gray whales in Korean waters from early December to early January (Mizue, 1951, his table 5) represents primarily a movement of migrating animals towards a destination farther south for parturition.

Based on the American whalers' observations summarised in this paper, gray whales were consistently observed in specific portions of the Sea of Okhotsk during the middle decades of the 19th century. Although most of the observations reported in the logbooks occurred on grounds where bowhead whales were the primary targets, some observations were also made on right whale grounds (e.g. on the Okhotsk side of Paramushir Island at 51°16'N, 155°23'W, 14 September 1868, *Europa* 1867-1868). While no sightings were reported on the northeastern Sakhalin Shelf where western gray whales are observed most often today, none of the logbooks read for this study contained evidence of the American whalers visiting that area (Fig. 4). The ship positions of whaling voyages into the Okhotsk Sea between 1844-1852, as plotted by Josephson *et al.* (2008), indicate some effort off the far northern, east-central and southern shores of Sakhalin but almost none along the northeastern coast on or near the present-day feeding areas. Lindholm (1863), a whaleman with much experience in the southern part of the Okhotsk Sea (particularly around the Shantar Islands), reported that gray whales were 'found in large numbers close to Cape Elizabeth [northern end of Sakhalin Island] and in the northern section of the sea during the summer'.

It is important to emphasise that the spatial and seasonal coverage of the Sea of Okhotsk represented by the logbooks was dictated primarily, if not solely, by the whalers' interest in catching bowhead and right whales although it was also influenced by sailing conditions, ice coverage and day length. It is fair to question whether the relative concentration of sightings of gray whales in the far northeastern reaches of the sea (Gizhiginskaya Bay and Penzhinskaya Gulf; Fig. 5) reflects relative density or is instead the result of a strong bias in search effort. It was not possible to address this issue rigorously given the biased nature of the effort data (the whalers went where they hoped to find bowhead whales and right whales) and the uncertainty about how consistently observations of non-target or secondary-target species like the gray whale were reported in the logbooks.

Removals

As indicated earlier, Henderson (1984) suggested that commercial ship-based whalers in the Sea of Okhotsk took about as many gray whales between the 1840s-1880s as were taken over roughly the same period in the Bering Sea and Arctic Ocean. The findings presented here are consistent with Henderson's suggestion. It is clear from the logbooks read for this study that gray whales were of little interest to the whalemen during the early years of the Okhotsk fishery when bowheads and right whales were available in good numbers. In the 1840s and early 1850s, few masters bothered to lower the boats when gray whales were observed on or *en route* to the bowhead or right whale grounds. This seems to have begun changing by the mid 1850s when logbooks record gray whales being chased more often (e.g. *Gov. Troup*, 9 and 17 July 1854 near the Jamskiye Rocks [Yamske Islands]; *Cicero*, 16 June 1859 in La Perouse Strait).

Within a given voyage, particularly from the 1860s onwards, it was not unusual for the crew to pursue *eastern* gray whales in the Mexican lagoons or alongshore Baja California and California in the winter, and *western* gray whales in the Sea of Okhotsk in the summer. Kugler (1984, p.153-4) referred to these as 'loop voyages', with the southern and northern components separated by visits to Hawaii and perhaps the Sea of Japan in the spring. Charles M. Scammon, for example, on the San Francisco ship *William C. Nye* during his last year as a whaling captain, sailed to the Okhotsk Sea for bowhead whaling in summer 1862 and then to Magdalena Bay for gray whaling in the following winter (Henderson, 1972, pp.86, 271).

The gray whale's reputation as a 'devilfish' is borne out by occasional statements in the logbooks. For example, when boats from the ship *Europa* (1866-1867) attacked a 'ripsack' in North-east Gulf on 19 August 1867, the bow boat 'got stove' after making the first strike, and then the other boats made the kill.

Historical vs current distribution of gray whales

In the sample of logbooks examined, there was no evidence that American 19th century whalers visited the northeastern coast of Sakhalin Island where gray whales have been studied intensively over the past decade. However, gray whales were observed regularly in certain other parts of the Sea of Okhotsk where the ship-based whalers hunted bowhead whales during the summer. They observed and hunted gray whales relatively often in the northeastern corner of the sea, especially in Gizhiginskaya Bay (North-east Gulf) and Penzhinskaya Gulf (Fig. 5). Gray whales were present in Gizhiginskaya Bay as early as mid-May when (or at least very soon after) the whalers arrived and gray whales continued to be observed there through the end of August. Good numbers also were observed in early June and as late as the third week of September in Tauskaya Bay (Magadan coast) and in late summer (mid-August to mid-September) in Sakhalin Bay off the northwestern coast of Sakhalin Island.

The early-season sightings of gray whales in the far northeastern part of the Okhotsk Sea are especially noteworthy. For example, when the steam bark *Mary and Helen II* reached the entrance of Penzhinskaya Gulf in the first week of May 1885, having encountered relatively heavy ice from 57°30'N and northwards, two 'ripsack whales' were observed on 6 May. On that same date the logbook records: 'Ice all along the west side of the Gulf and seemingly packed on the land as far to the north as we can see while on the east side it's perfectly clear'. If, as has been generally assumed, the gray whales that occupy the summer feeding areas off northeastern Sakhalin Island enter the Sea of Okhotsk in spring (early May to early June) via either La Perouse Strait or Tatar Strait (Vladimirov, 2004; Yablokov and Bogoslovskaya, 1984), the question arises whether some whales over-winter in the Okhotsk Sea or, alternatively, enter it *via* an inter-island route in the Kurile chain, having moved northwards along the Pacific coast of Japan. Once in the Okhotsk, they would continue northwards along western Kamchatka to arrive in the northeastern gulfs of the Sea of Okhotsk by early May.

The historical distribution of gray whales in the Sea of Okhotsk appears to have been much more extensive than it is at present although increased search effort at appropriate times in areas of historical occurrence is needed to confirm their absence from such areas. Although not definitive, the information on western gray whales obtained from American 19th century whaling logbooks is of potential value in the following ways:

(a) to inform the timing and spatial coverage of modern survey effort;
(b) to support, in principle if not also in planning as to time and location, a satellite tagging and tracking programme to learn more about the movements of western gray whales;
(c) to contribute to an accurate reconstruction of catch history for input to population models; and
(d) to provide a basis for formulating hypotheses regarding sub-stock structure.

Other species

Another topic related to the present study that deserves further investigation is the occurrence and distribution of other whale species in the Sea of Okhotsk. For example, interest has been expressed in clarifying 'possible confusion between right and bowhead whales in the Townsend data' (IWC, 2009). Scammon (1874), who stated that bowheads were first taken by American whalers in the Pacific in 1843 off Kamchatka, and in 1847 in the Okhotsk Sea, provides a starting point for such an effort. He cites (1874, p.60) Tchantar Bay (the Shantar Islands area), Taousk Gulf (Tauskaya Gulf) and Penjinsk Gulf (Penzhinskaya Gulf) as 'noted whaling-grounds [for bowheads], as well as several other points about the coasts'. He also claims (1874, p.68) that right whales were found 'toward the northern borders in the early part of the season; later, the ships cruise [for them] in the southern quarter, about the Kurile Islands'. Published correspondence between various whalemen and M.F. Maury (1851), fuelled by the momentous 1848 voyage of Thomas Roys to the Bering Strait region where he 'discovered' bowheads, illustrates the considerable extent to which the whaling fraternity was itself still trying in the late 1840s to resolve the distinctions in morphology, distribution and ecology between the North Pacific right whale and the bowhead whale.

Judging by preliminary results of this study, clarification of the respective historical ranges of the two species is not likely to be achieved without detailed examination of a substantial number of logbooks, particularly for the early years of the Okhotsk Sea fishery when both right whales and bowheads would have been much more numerous than they were in the later years. For example, in the logbook of the Stonington ship *Mary and Susan* (1847-1850), it is reported that on 29 June 1849 '10 or 12 R Whales and 1 Steeple top' were seen in the northeastern Sea of Okhotsk at 57°11'N, 152°56'E, suggesting that both right whales and bowheads were observed on the same day in the same area. Earlier in the season, the crew had unsuccessfully chased a 'Polar whale' on 7 June and then secured one of two 'Polar whales' observed on 8 June, both encounters in the vicinity of 57°N, 151°E. Many whales are reported as seen and chased between the first and last weeks of June but the species is specified only for one sighting – a 'small Right whale' on 15 June at 57°53'N, 152°32'E. During the previous year's voyage to the same ground (bounded approximately by 56°30’ to 57°45’’N × 150° to 153°E), the *Mary and Susan* reportedly found and hunted ‘right’ whales on many days from 28 May to 23 June, with no reference in the logbook to ‘polar whale’ or any other name that could be interpreted as referring to the bowhead.

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REFERENCES

Unpublished (logbooks and journals)

With Observations of Gray Whales

Benjamin Cummings 1866-1871. Logbook of the ship Benjamin Cummings of New Bedford, Charles Halsey, Master. 27 September 1866 – 29 November 1869. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 462.

Benjamin Cummings 1866-1871. Logbook of the ship Benjamin Cummings of New Bedford, Charles Halsey, Master. 27 September 1866 – 1 May 1871. New Bedford Free Public Library, New Bedford, Massachusetts.

Betsy Williams 1851-1854. Logbook of the ship Betsy Williams of Stonington, Gilbert Pendleton, Jr., Master. 23 April 1851 – 20 April 1854. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM 370.

Cicero 1856-1860. Logbook of the ship Cicero of New Bedford, Charles Courtney, Master. 12 August 1856 – 16 May 1860. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 377.

Europa 1866-1870. Logbook of the ship Europa of Edgartown, Thomas Mellen, Master. 29 August 1866 – 21 August 1870 Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 663.

Euphrates 1857-1861. Logbook of the ship Euphrates of New Bedford, William H. Heath, Master. 15 October 1857 – 7 April 1861. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 384.

Good Return 1851-1855. Logbook of the ship Good Return of New Bedford, Benjamin Franklin Wing, Master. 2 September 1851 – 6 April 1855. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 794.

Governor Troup 1853-1856. Logbook of the ship Governor Troup of New Bedford, Anthony Milton, Master. 1 August 1853 – 15 April 1856. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 829.

Josephine 1863-1866. Logbook of the ship Josephine of New Bedford. James L. Chapman, Master. 14 April 1863 – 12 June 1867. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM 122.

Lancaster 1859-1862. Logbook of the ship Lancaster of New Bedford. Thomas N. Russell, Master. 5 December 1859 – 8 April 1862. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM 127.

Mary and Helen II 1884-1885. Logbook of the steam bark Mary and Helen II of San Francisco. Frederick A. Barker, Master. 30 December 1884 – 4 November 1885. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM 937.

Mary and Susan 1847-1850MS. Logbook of the ship Mary and Susan of Stonington. Gilbert Pendleton, Jr., Master. 20 October 1847 – 23 March 1850. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM 369.

Midas 1861-1865. Logbook of the bark Midas of New Bedford. Henry A. Howland, Master. 15 November 1861 – 29 August 1865. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 483.

Nassau 1853-1856. Logbook of the Ship Nassau of New Bedford. Henry Clay Murdock, Master. 9 October 1853 – 12 July 1856. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 612.

Oliver Crocker 1858-1863. Logbook of the ship Oliver Crocker of New Bedford. David Cochran, Master. 26 October 1858 – 1 March 1863. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM 162.

Without Observations of Gray Whales

Cicero 1860-1865. Logbook of the ship Cicero of New Bedford, John R. Stivers, Master. 9 October 1860 – 26 May 1865. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 018.

Coral 1886-1887. Logbook of the bark Coral of San Francisco, Rodolphus Delano Wicks, Master. 6 November 1886 – 2 November 1888. Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM 470.

Erie 1847-1850MS. Logbook of the ship Erie of Fairhaven, Ichabod Norton, Master. 1 September 1847 – 11 April 1850, Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 382.

Fortune 1850-1854. Logbook of the bark Fortune of New Bedford, David Evans Hathaway and William Davis, Jr., Masters. 21 October 1850 – 18 May 1854, Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 385B.

Henry Kneeland 1851-1854. Logbook of the ship Henry Kneeland of New Bedford, Willian H. Vinal, Master. 3 February 1852 – 12 October 1853, Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # ODHS 620.

John and Winthrop 1885-1888. Logbook of the bark John and Winthrop of San Francisco, William Henry Poole, Master. 27 October 1885 – 17 December 1888, Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM 118.

Splendid 1854-1858. Logbook of the ship Splendid of Edgartown, John Sprague Smith, Master. 2 November 1854 – 18 May 1858, Kendall Institute, New Bedford Whaling Museum, New Bedford, Massachusetts, Log # KWM187.

Published literature

Andrews, R.C. 1916. *Whale Hunting with Gun and Camera: a Naturalist’s Account of the Modern Shore-whaling Industry, of Whales and their Habits, and of Hunting Experiences in Various Parts of the World*. D. Appleton and Co., New York. 333pp.

Blokhin, A.S. and Blokhin, S.A. 2006. Some results of search for probable winter grounds of the western gray whale (*Eschrichtius robustus*) in the South China coastal waters. pp.76-80. *Marine Mammals of the Holarctic: Collection of Scientific Papers*. 4th International Conference, St. Petersburg, 10-14 September 2006.

Blokhin, S.A. 1996. Distribution, abundance and behavior of gray whales (*Eschrichtius robustus*) of American and Asian populations in regions of their summer location nearshore of the Far East. *Izv. TINRO* 121: 36-53. [In Russian].

Brownell, R.L., Jr. and Chun, C. 1977. Probable existence of the Korean stock of the gray whale (*Eschrichtius robustus*). *J. Mammal*. 58(2): 237-9.

Clark, A.H. 1887. History and present condition of the fishery. pp.3-281 of part XV, ‘The Whale Fishery’, *In*: G.B. Goode (ed.), *The Fisheries and Fishery Industries of the United States* Section V History and methods of the fisheries, Vol. II. Gov. Print. Off., Washington, DC.

Du Pasquier, T. 1982. *Les Baleiniers Français au XIXe Siècle (1814-1868)*. Terre et Mer 4 Seigneurs, Grenoble. 256pp. [In French].

Du Pasquier, T. 1986. Catch history of French right whaling mainly in the South Atlantic. *Rep. int. Whal. Commn (special issue)* 10: 269-74.

Ellis, R. 1991. *Men and Whales*. Alfred A. Knopf, New York. xv+542pp.

Henderson, D.A. 1972. *Men and Whales at Scammon’s Lagoon*. Dawson’s Book Shop, Los Angeles. 313pp.

Henderson, D.A. 1984. Nineteenth century gray whaling: grounds, catches and kills, practices and depletion of the whale population. pp.159-86. *In*: Jones, M.L., Swartz, S.L. and Leatherwood, S. (eds). *The Gray Whale*, Eschrichtius robustus. Academic Press, Inc., Orlando, Florida. xxiv+600pp.

Henderson, D.A. 1990. Gray whales and whalers on the China coast in 1869. *Whalewatcher* 24(4): 14-16.

International Whaling Commission. 2004. Report of the Workshop on the Western Gray Whale: Research and Monitoring Needs, 22-25 October 2002, Ulsan, Korea. *J. Cetacean Res. Manage. (Suppl.)* 6:487-500.

International Whaling Commission. 2009. Report of the Scientific Committee. Annex F. Report of the sub-committee on bowhead, right and gray whales. *J. Cetacean Res. Manage. (Suppl.)* 11:169-92.

Josephson, E.A., Smith, T.D. and Reeves, R.R. 2008. Depletion within a decade: the American 19th century North Pacific right whale fishery. pp.133-47. *In*: Starkey, D.J., Holm, P. and Barnard, M. (eds). *Oceans Past: Management Insights from the History of Marine Animal Populations*. Earthscan, London.

Kasuya, T. 2002. Japanese whaling. pp.655-62. *In*: Perrin, W.F., Wursig, B. and Thewissen, J.G.M. (eds). *Encyclopedia of Marine Mammals*. Academic Press, San Diego, California.

Kato, H. and Kasuya, T. 2002. Some analyses on the modern whaling catch history of the western North Pacific stock of gray whales (*Eschrichtius robustus*), with special reference to the Ulsan whaling ground. *J. Cetacean Res. Manage*. 4(3): 277-82.

Kugler, R.C. 1984. Historical survey of foreign whaling: North America. pp.149-57. *In*: s'Jacob, H.K., Snoeijing, K. and Vaughan, R. (eds). *Arctic Whaling: Proceedings of the International Symposium, Arctic Whaling, February 1983*. Arctic Center, University of Groningen, Netherlands. 181pp.

Lindholm, O. 1863. Whales and how tides and currents in the Okhotsk Sea affect them. (From an unidentified Journal with news from the far east of Russia, pp.42-43. Translated from Russian in 1965 by L.A. Hutchinson for the University of California San Diego, La Jolla). 12pp.

Maminov, M.K. and Blokhin, S.A. 2004. Gray whales (*Eschrichtius robustus*) in coastal waters of southern Far East. pp.362-68. *Marine mammals of the Holarctic: collection of scientific papers, 3rd International Conference, Koktebel, Crimea, Ukraine, 11-17 October 2004*.

Maury, M.F. 1851. *Explanations and Sailing Directions to Accompany the Wind and Current Charts, approved by Commodore Lewis Warrington*. Secretary of the Navy, Washington DC.

Mead, J.G. and Mitchell, E.D. 1984. Atlantic gray whales. pp.33-53. *In*: Jones, M.L., Swartz, S.L. and Leatherwood, S. (eds). *The Gray Whale,.* Eschrichtius robustus. Academic Press, Orlando, FL. xxiv+600pp.

Mizue, K. 1951. Grey whales in the East Sea area of Korea. *Sci. Rep. Whales Res. Inst., Tokyo* 5: 71-79.

Nishiwaki, M. and Kasuya, T. 1970. Recent record of gray whale in the adjacent waters of Japan and a consideration on its migration. *Sci. Rep. Whales Res. Inst., Tokyo* 22: 29-37.

Omura, H. 1984. History of gray whales in Japan. pp.57-77. *In*: Jones, M.L., Swartz, S.L. and Leatherwood, S. (eds). *The Gray Whale,* Eschrichtius robustus. Academic Press Inc., Orlando Florida. xxiv+600pp.

Omura, H. 1988. Distribution and migration of the western Pacific stock of the gray whale. *Sci. Rep. Whales Res. Inst., Tokyo* 39: 1-9.

Reeves, R.R., Brownell, R.L., Burdin, A., Cooke, J.C., Darling, J.D., Donovan, G.P., Gulland, F.M.D., Moore, S.E., Nowacek, D.P., Ragen, T.J., Steiner, R.G., VanBlaricom, G.R., Vedenev, A. and Yablokov, A.V. 2005. Report of the Independent Scientific Review Panel on the Impacts of Sakhalin II Phase 2 on Western North Pacific Gray Whales and Related Biodiversity. IUCN, Gland, Switzerland. 123pp. [Available from *http://www.iucn.org/wgwap*].

Rice, D.W. 1983. Gestation period and fetal growth of the gray whale. *Rep. int. Whal. Commn* 33: 539-44.

Rice, D.W. and Wolman, A.A. 1971. *The Life History and Ecology of the Gray Whale* (Eschrichtius robustus). American Society of Mammalogists, Special Publication No. 3, Stillwater, Oklahoma. viii+142pp.

Scammon, C.M. 1874. *The Marine Mammals of the North-western Coast of North America, Described and Illustrated: Together with an Account of the American Whale-Fishery*. John H. Carmany and Co., San Francisco. x+319+vpp. [Reprinted in 1968 by Dover Publications, Inc., New York].

Sherman, S.C., Downey, J.M., Adams, V.M. and Pasternack, H. 1986. *Whaling Logbooks and Journals 1613-1927: An Inventory of Manuscript Records in Public Collections*. Garland Publications, New York and London. 469pp.

Starbuck, A. 1878. History of the American whale fishery from its earliest inception to the year 1876. Appendix A. pp.1-768+6pls. *Report of the US Commissioner of Fish and Fisheries, Part 4, 1875-76*. Government Printing Office, Washington, D.C. [Reprinted in 1964 in two volumes by Argosy Antiquarian Ltd, New York].

Tomilin, A.G. 1957. *Zveri SSSR i Prilezhasfchikh Stran. Zveri Vostochnoi Evropy i Severnoi Azii*. Vol. IX. Kitoobraznye. Izdatel'stvo Akademi Nauk SSSR (Soviet Academy of Sciences Publishers), Moscow. 756pp. [Translated in 1967 as *Mammals of the USSR and Adjacent Countries. Mammals of Eastern Europe and Adjacent Countries. Vol. IX. Cetacea* by the Israel Program for Scientific Translations, Jerusalem, 717pp.][In Russian].

Tyurneva, O.Y., Maminov, M.K., Shvetsov, E.P., Fadeev, V.I., Selin, N.I. and Yakovlev, Y.M. 2006. Seasonal movements of gray whales (*Eschrichtius robustus*) between feeding areas on the northeast shelf of Sakhalin Island. pp.530-35. *Marine Mammals of the Holarctic: Collection of Scientific Papers*. 4th International Conference, St. Petersburg, 10-14 September 2006.

Uni, Y. and Kasuya, T. 2002. Review of historical records of gray whales in Japanese waters. Paper SC/02/WGW15 presented to the IWC/SC Western Gray Whale Workshop, October 2002, Ulsan, Korea (unpublished). 6pp. [Paper available at the Office of this Journal].

Vertyankin, V.V., Nikulin, V.S., Bednykh, A.M. and Kononov, A.P. 2004. Sightings of grey whale (*Eschrichtius robustus*) near southeastern Kamchatka. pp.126-28. *Marine mammals of the Holarctic: collection of scientific papers, 3rd International Conference, Koktebel, Crimea, Ukraine, 11-17 October 2004*.

Vladimirov, V.A. 2004. Comtemporary state of our knowledge of the Okhotsk-Korean gray whale population. pp.129-32. *Marine Mammals of the Holarctic: Collection of Scientific Papers*, 3rd International Conference, Koktebel, Crimea, Ukraine, 11-17 October 2004.

Wang, P. 1984. Distribution of the gray whale (*Eschrichtius gibbosus*) off the coast of China. *Acta Ther. Sinica* 4(1): 21-26. [In Chinese with English summary].

Wang, P. 1993. Fauna of marine mammals in China. *Acta Oceanologica Sinica* 12(2): 273-78.

Wang, P. 1999. *Chinese Cetaceans*. Ocean Enterprises Ltd. 325pp. [In Chinese].

Weller, D.W., Burdin, A.M., Würsig, B., Taylor, B.L. and Brownell, R.L., Jr. 2002. The western gray whale: a review of past exploitation, current status and potential threats. *J. Cetacean Res. Manage*. 4(1): 7-12.

Weller, D.W., Wrsig, B., Bradford, A.L., Burdin, A.M., Blokhin, S.A., Minakuchi, H. and Brownell, R.L., Jr. 1999. Gray whales (*Eschrichtius robustus*) off Sakhalin Island, Russia: seasonal and annual patterns of occurrence. *Mar. Mammal Sci*. 15(4): 1208-27.

Williams, H., Ed. 1964. *One Whaling Family*. Houghton Mifflin, Boston. 401pp.

Yablokov, A.V. and Bogoslovskaya, L.S. 1984. A review of Russian research on the biology and commercial whaling of the gray whale. pp.465-85. *In*: Jones, M.L., Swartz, S.L. and Leatherwood, S. (eds). *The Gray Whale* Eschrichtius robustus. Academic Press Inc., Orlando, Florida. xxiv+600pp.

Yakovlev, Y., Tyurneva, O. and Vertyankin, V. 2007. Photographic identification of the Korean-Okhotsk gray whale (*Eschrichtius robustus*) offshore northeastern Sakhalin Island and southeastern Kamchatka, Russia, 2006. Final report for Russian Academy of Sciences, Far Eastern Branch, Institute of Marine Biology, Vladivostok.

Zhu, Q. 1998. Strandings and sightings of the western Pacific stock of the gray whale *Eschrichtius robustus* in Chinese coastal waters. Paper SC/50/AS5 presented to the IWC Scientific Committee, April 1998 (unpublished). 4pp. [Paper available from the Office of this Journal].

Zhu, Q. 2002. Historical records of western Pacific stock of gray whale *Eschrichtius robustus* in Chinese coastal waters from 1933 to 2002. Paper SC/02/WGW13 presented to the Workshop on the Western Gray Whale, Ulsan, Korea, 22-25 October 2002 (unpublished). 4pp. [Paper available from the Office of this Journal].

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