Report

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A review of current information on global cetacean stocks within the Areas of Relevance

Summary

Task 1 of this project delivers a report reviewing information on cetacean stocks within areas of relevance (AOR) considered important by the JIP. These AORs were described as text by the JIP using country boundaries. All spatial representations have been produced by SMRU Ltd. This report summarises information for each AOR in turn.
For each of the AOR, information on species and associated stocks that are present is listed with an indication of the amount of data that exist. For species where quality information on abundance exists within an AOR, the possibility of trend analysis is discussed.
This report will be provided to the JIP for guidance on those cetacean stocks that will be taken forward for consideration in Task 2.

1 Introduction

Reviewing information on cetacean populations that overlap with E&P activities is important to allow an objective assessment of any relationships between cetacean stock trends and a range of external influencing factors. Cetacean species are found globally and many studies have been undertaken to monitor population numbers over the past decades. Despite the surveys that have taken place, monitoring species, subspecies, populations and stocks of cetaceans is not straight forward. Many cetacean species are wide-ranging and not easily observed at sea. Additionally, many undertake migrations that lead to seasonal differences in distribution. These differences in distribution may be on a population level (e.g. Humpback whales migrating between feeding and breeding areas; see Clapham 2000 for review) or on a sex differentiation level (e.g. sperm whales, where females remain in low latitudes and males migrate to feed in higher latitudes, Whitehead and Weilgart, 2000). To deal with these difficulties, researchers have developed different means of monitoring populations and analysing data including line-transect surveys for all species present in an area (e.g. SCANS surveys; http://biology.st-andrews.ac.uk/scans2/), individual species accounts of stock structures and movements (http://www.nmfs.noaa.gov/pr/sars/species.htm); photo-identification surveys of species, to assess numbers within restricted geographic boundaries (e.g. Parra *et al*., 2006 and Smith *et al*., 1999) and counts of animals passing geographic points (Buckland and Breiwick, 2002 and Zeh *et al*., 1991).

Although all these methods can provide good means of monitoring populations, a further problem of differentiating biologically important species, subspecies, units or stocks for management purposes still exists. Although many cetacean species have been split into subspecies and some have been further split into stocks or management units, many others remain undefined. Units at the species and subspecies levels are on different evolutionary trajectories and therefore represent important evolutionary potential (Taylor 2005). However, stocks are units whose population dynamics are essentially independent of neighbouring stocks but may or may not represent important evolutionary potential (Taylor 2005). The internal dynamics of the groups of animals that constitute a stock are important in maintaining the stock structure, but the connections between stocks may be important in determining the overall population structure of the species. Marine mammal species, subspecies and stocks are not clearly defined and new species and subspecies are still being identified e.g. the Northern right whale has recently been split into two species, one in the North Atlantic (*Eubalaena glacialis)* and one in the North Pacific (*Eubalaena japonica)*, (Rosenbaum *et al*., 2000) based on genetic data. Similarly, the Irrawaddy dolphin has recently been split into two species, with the Asian species retaining the name Irrawaddy dolphin (*Orcaella brevirostris*) and the Australian species being termed the Australian snubfin dolphin *(Orcaella heinsohni)* (Beasley *et al*., 2005). In this case both genetic and morphological data were available to confirm the designation. Historically morphological data, such as skull measurements, are used to corroborate genetic data but this data are not always easily accessible for wide ranging long lived marine mammals. To complement genetic or morphological data, information on geographical ranges and behaviour can be used as evidence to determine subspecies, but should not be used as a primary form of evidence due to uncertainty about the degree to which geographical distribution and behaviour actually reflect genetic divergence (Taylor 2005). A further problem is that of historical distribution, as much of the information on cetacean distributions comes from surveys in the past few decades and as such trying to look at historical trends for anything beyond a species level and over any length of time is problematic.

The aim of this task was to review the primary literature to identify the occurrence of populations and stocks of cetaceans within each Area of Relevance and for each AOR provide a list and maps of cetacean populations for which quality data exist. For each AOR, we provide a list of all species present, and, for each species, outline whether abundance data are available and whether stock structure is known. Additionally, survey areas of existing data are shown with respect to each AOR and an annotated bibliography summarises survey methodologies and extent of the survey areas. The potential for attempting trend analysis is discussed with reference to the available data for each AOR.

2 Methodology

2.1 Literature review, data collation and defining areas of relevance (AOR)

An extensive literature review was carried out to identify data sources from the primary literature. In order to determine changes in cetacean stock trends over time, quality data on abundance must be assessed. Furthermore to detect long-term changes in population abundance or density of a species in any given area, multiple comparable surveys must have

been completed. Data from ship-based and or aerial line transect surveys, where the analysis has accounted for biases inherent when surveying at sea, was considered the most scientifically robust at this stage. However, for many areas, this type of data does not exist and in these cases other survey data were considered. This included data from photo-identification surveys, shore based counts and acoustic surveys. Data were collated, to individual species, on a global scale and put into context of 7 areas of relevance (AOR) (Table 2.1) based on text descriptions supplied by the JIP. Data included surveys from the past four decades and where survey blocks could be identified these were digitised in ArcGis.

Table 2.1: Area of Relevance (AOR) ID's and associated descriptions.

Area of Relevance ID	Description
AOR_001	West Africa
AOR_002	East Africa
AOR_003	Australia
AOR_004	Alaska
AOR_005A	North America – West Coast
AOR_005B	North America – East Coast
AOR_006A	Europe
AOR_006B	Sakhalin
AOR_007	Brazil, South America

A wide range of information relating to each survey was entered into the database including survey method, abundance estimation method, abundance estimate, confidence interval around abundance estimate and year and season of survey. Existence of abundance data was determined for all species known to be present within the AOR for the past four decades.

2.2 Determining species presence

For each AOR it was necessary to determine which cetacean species may be present within the boundaries. In the literature, distribution of marine mammals is often inferred from current range or range inferred from suitable habitat. In line with this, we used a suitable habitat approach, namely RES (Relative Environmental Suitability) predictions (Kaschner *et al*., 2006). For each AOR, a spatial grid with resolution of 0.5-degree latitude by 0.5-degree longitude was applied. This grid resolution represents a standardized unit and has been used as part of an ecological niche model by Kaschner *et al*., (2006) to generate an index of the RES of each cell for a given species by relating known habitat usage to local environmental conditions. Predictions generated by the RES model have been validated (Kaschner *et al*., 2006) and can therefore be regarded as a reliable way to indicate species presence, particularly at very large scales. Species presence was determined based on the predicted habitat suitability for each species in each AOR using an assumed threshold of species presence of predicted suitability > 0.2, to account for any over prediction of species occurrence. The output from this method is a list of all species predicted to occur within each AOR. However, to account for the AOR boundary placement being based on country boundaries, any species that was predicted to occur in less than 100 cells was excluded from

the species list. The only exceptions to this were if known research projects for that species existed in the area. This list was used as the basis for each AOR table in section 3.

2.3 Stock structure information

To look at changes in stock trends, information about stocks and populations of marine mammals was collated. To enable coverage of all species, primary references sources were outlined and used as the main resource. These references sources were considered the most comprehensive for this task. The primary sources used were;

1. The IUCN assessment status of cetacean subspecies and subpopulations spreadsheet, drafted on April 13th 2007 (permission for reference given by P Hammond) and the IUCN website (http://www.iucnredlist.org/search/search-basic).
2. The NOAA Fisheries, office of protected resources, Marine Mammal Stock Assessment Reports (SARs) by Species/Stock (http://www.nmfs.noaa.gov/pr/sars/species.htm).
3. The review of small cetaceans: Distribution, Behaviour, Migration and Threats, complied for the Convention on Migratory Species (CMS) ().
4. The Marine Mammals of the World webpage (http://nlbif.eti.uva.nl/bis/marine_mammals.php).
5. Taylor, B.L. (2005) Identifying units to conserve. In Marine Mammal Research: Conservation beyond crisis. Reynold J.E., Perrin W.F., Reeves R.R., Montgomery, S. Ragen, T.J. (Eds) John Hopkins University press. 149-164
6. The North Atlantic Marine Mammal Commission (NAMMCO) website (http://www.nammco.no/).
7. The peer-reviewed publications listed in section 6.1 that outline stocks with associated abundance estimates.

Where possible stocks were determined down to the lowest classification, but where information was not within these documents, stocks were classified as unknown. The given classifications could be based on management stocks or populations restricted to given ranges and for many species stock classification was not possible and only subspecies could be identified. For AORs where stocks are stated, it is important to note other stocks may also be present but not identified in the primary literature sources used. Hence the reader should consider the stock information given in tables 3.1.1 – 3.9.1 non exhaustive.

2.4 Spatial representation of data

For each AOR, spatial survey information, if available, was collated and displayed. For each AOR, all surveys that fell partially or totally within the defined AOR area have been digitised using ArcGIS. For instances where multiple blocks were associated with one survey the total survey block is displayed. The scale of surveys is variable within each AOR and, as such, some AORs have coverage in only a small part of the total area and others have coverage across the majority of the AOR. By digitising the survey area and hence associating an abundance estimate with a spatial area, the ability to determine densities is possible.

2.5 Annotated bibliography

For each survey that is cited spatially, an annotated bibliography has been produced. Each citation is followed by a brief descriptive and evaluative paragraph to inform the reader of the relevance, accuracy, and quality of the sources cited.

3 Results

3.0 Areas of Relevance

In the original tender document, the areas of relevance were described using continent and country boundaries by the JIP. To allow spatial representation of these areas, shapefiles using these continents and countries as guides were constructed. In total 9 different AOR were defined based on the original 7 areas described by the JIP. These nine areas are represented in Table 2.1 and Figure 3.0.

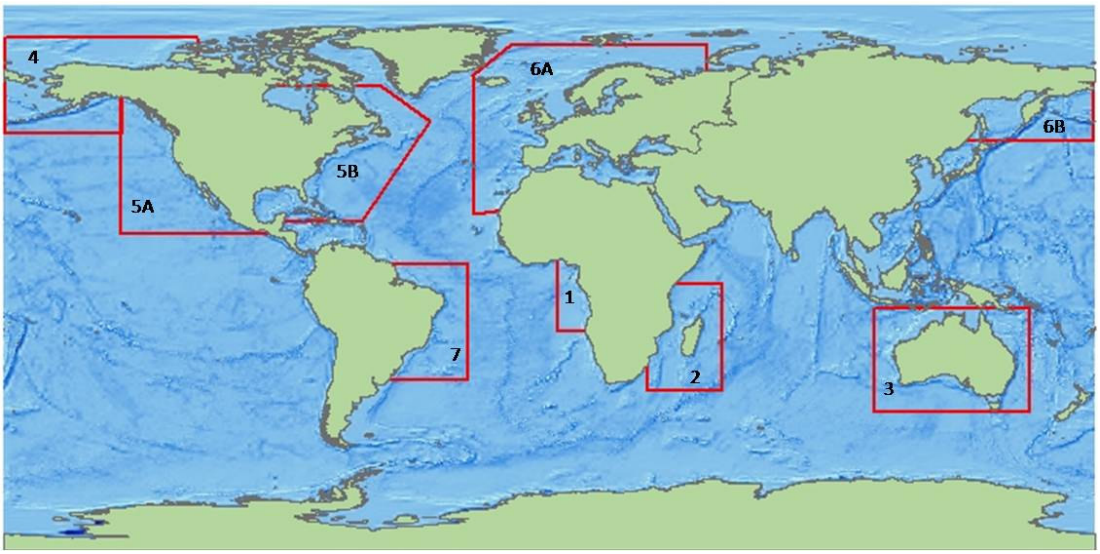

Figure 3.0: The nine Areas of Relevance identified for investigation of cetacean abundance data availability. Numbers correspond to numeric part of Area of Relevance ID in table 2.1.

3.1 Area of Relevance 1: West Africa

The West Africa area of relevance extends from the western border of Nigeria in the north to the southern border of Angola in the south. The coastlines of Cameroon, Equatorial Guinea, Gabon, the Republic of Congo and the Democratic Republic of Congo all fall within this region.

Table 3.1.1: Summary table of the abundance data available for cetaceans in AOR_001

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. Dwarf minke whale	No						
Southern right whale	Unknown	No						
Antarctic minke whale	Unknown	No						
Sei whale	1. Southern hemisphere subspecies	No						
Bryde's whale	Unresolved[1]	No						
Blue whale	1. Southern Ocean/Pygmy subspecies (*B. m. brevicauda*) 2. Antarctic subspecies (*B. m. intermedia*)	No						
Fin whale	1. Southern hemisphere subspecies	No						
Pygmy right whale	Unknown	No						
Long-beaked common dolphin	Unknown	No						
Short-beaked common dolphin	Unknown	No						
Pygmy killer whale	Unknown	No						
Short-finned pilot whale	Unknown	No						
Risso's dolphin	Unknown	No						
Southern bottlenose whale	Unknown	No						
Pygmy sperm whale	Unknown	No						
Dwarf sperm whale	Unknown	No						
Fraser's dolphin	Unknown	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Southern right whale dolphin	Unknown	No						
Andrews' beaked whale	Unknown	No						
Blainville's beaked whale	Unknown	No						
Gervais' beaked whale	Unknown	No						
Gray's beaked whale	Unknown	No						
Hector's beaked whale	Unknown	No						
Strap-toothed whale	Unknown	No						
True's beaked whale	Unknown	No						
Humpback whale	1. Breeding stock B1 (Gabon) 2. Breeding stock B2 (Angola)	Yes			✓	AOR_001_001	3.1.1	72
Killer whale	Unknown	No						
Melon-headed whale	Unknown	No						
Sperm whale	Unknown	No						
False killer whale	Unknown	No						
Atlantic humpbacked dolphin	Unknown	No						
Pantropical spotted dolphin	Unknown	No						
Rough-toothed dolphin	Unknown	No						
Clymene dolphin	Unknown	No						
Striped dolphin	Unknown	No						
Atlantic spotted dolphin	Unknown	No						
Spinner dolphin	Unknown	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Bottlenose dolphin	Unknown	No						
Cuvier's beaked whale	Unknown	No						

[1] Three populations in occur in the southern African region - it's unclear which relates to AOR_001. Best (2001).

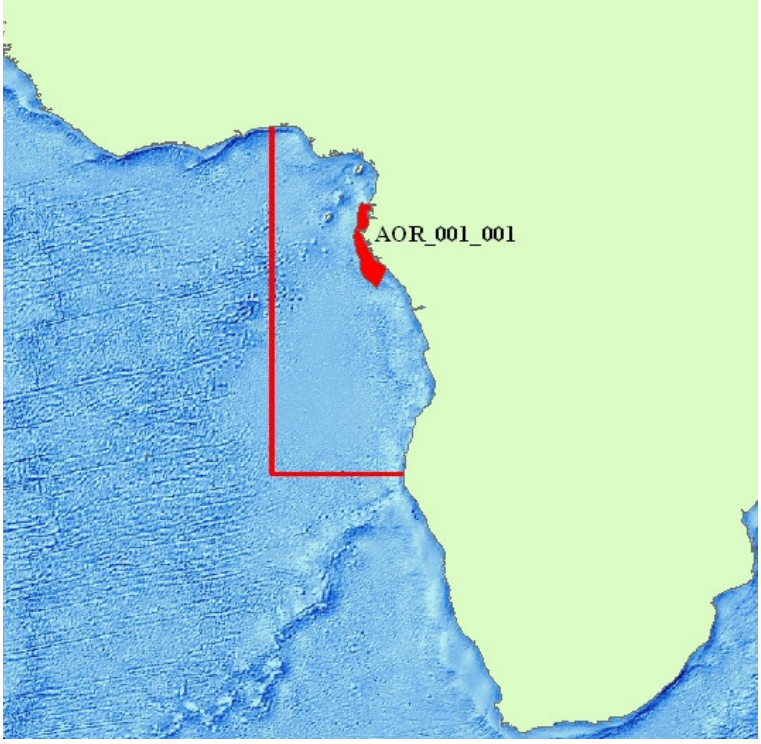

Figure 3.1.1: Surveys conducted in West Africa

3.1.1 Summary of available data for AOR 1

AOR1 lies within the South Atlantic Ocean and thirty-nine cetacean species were predicted to be present within its boundaries. Of these thirty nine species, only five; the common Minke whale; Fin whale; Humpback whale; Blue whale and Sei whale have associated stock/ subspecies information (Table 3.1.1.). This information is only to subspecies resolution for everything but the Humpback whale. For the humpback whale, breeding stock information is known, as breeding stock B is shown to migrate to breed off southern West Africa. This breeding stock is further divided into sub-breeding stocks with Stock B1 found off Gabon and B2 found off west South Africa, Namibia and Angola (Johnstone and Butterworth, 2007). Only one systematic survey, (AOR_001_001) to assess abundance of humpback whales was found to fall within AOR 1 (Rosenbaum *et al*., 2004). This survey lasted for five days in 2002 and covered the entire coastline of Gabon and part of the Congolese coast (Figure 3.1.1). Although two strata were surveyed and estimates given for both, the study area is small relative to the breeding grounds of humpback whales off West Africa, so the number of whales using this area is likely to be much larger than indicated in this study.

It is impossible to look for population trends of humpback whales with this data, as no comparable surveys within this area exist.

3.2 Area of Relevance 2: East Africa

This area of interest extends from the northern border of Kenya in the north to the southern border of Mozambique, therefore including the coastlines of Tanzania and Madagascar.

Table 3.2.1: Summary table of the abundance data available for cetaceans in AOR_002

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. Dwarf minke whale	No						
Southern right whale	Unknown	No						
Antarctic minke whale	Unknown	No						
Sei whale	1. Southern hemisphere subspecies	No						
Bryde's whale	Unresolved[1]	No						
Blue whale	1. Southern Ocean/Pygmy subspecies (*B. m. brevicauda*) 2. Antarctic subspecies (*B. m. intermedia*)	Yes		✓		AOR_002_004	3.2.1	4
Fin whale	1. Southern hemisphere subspecies	No						
Pygmy right whale	Unknown	No						
Long-beaked common dolphin	Unknown[2]	No						
Short-beaked common dolphin	Unknown	No						
Pygmy killer whale	Unknown	No						
Short-finned pilot whale	Unknown	No						
Risso's dolphin	Unknown	No						
Southern bottlenose whale	Unknown	No						
Longman's beaked whale	Unknown	No						
Pygmy sperm whale	Unknown	No						
Dwarf sperm whale	Unknown	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Hourglass dolphin	Unknown	No						
Fraser's dolphin	Unknown	No						
Southern right whale dolphin	Unknown	No						
Andrews' beaked whale	Unknown	No						
Blainville's beaked whale	Unknown	No						
Ginkgo-toothed beaked whale	Unknown	No						
Gray's beaked whale	Unknown	No						
Hector's beaked whale	Unknown	No						
Strap-toothed whale	Unknown	No						
True's beaked whale	Unknown	No						
Humpback whale	1. Breeding stock C	Yes		✓	✓	AOR_002_002 AOR_002_003	3.2.1 3.2.1	30 5
Killer whale	Unknown	No						
Melon-headed whale	Unknown	No						
Sperm whale	Unknown	No						
False killer whale	Unknown	No						
Indo-Pacific humpback dolphin	Unknown	Yes		✓	✓	AOR_002_001	3.2.1	80
Pantropical spotted dolphin	Unknown	No						
Rough-toothed dolphin	Unknown	No						
Striped dolphin	Unknown	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Spinner dolphin	Unknown[3]	No						
Indian Ocean bottlenose dolphin	Unknown	Yes		✓	✓	AOR_002_001	3.2.1	80
Bottlenose dolphin	Unknown	No						
Cuvier's beaked whale	Unknown	No						

[1] Could belong to Southwest Indian Ocean stock or South African Inshore stock.

[2] Known subspecies are long-beaked common dolphin and Indo-Pacific common dolphin.

[3] Subspecies *S. l. longirostris* is found in tropical shelf waters.

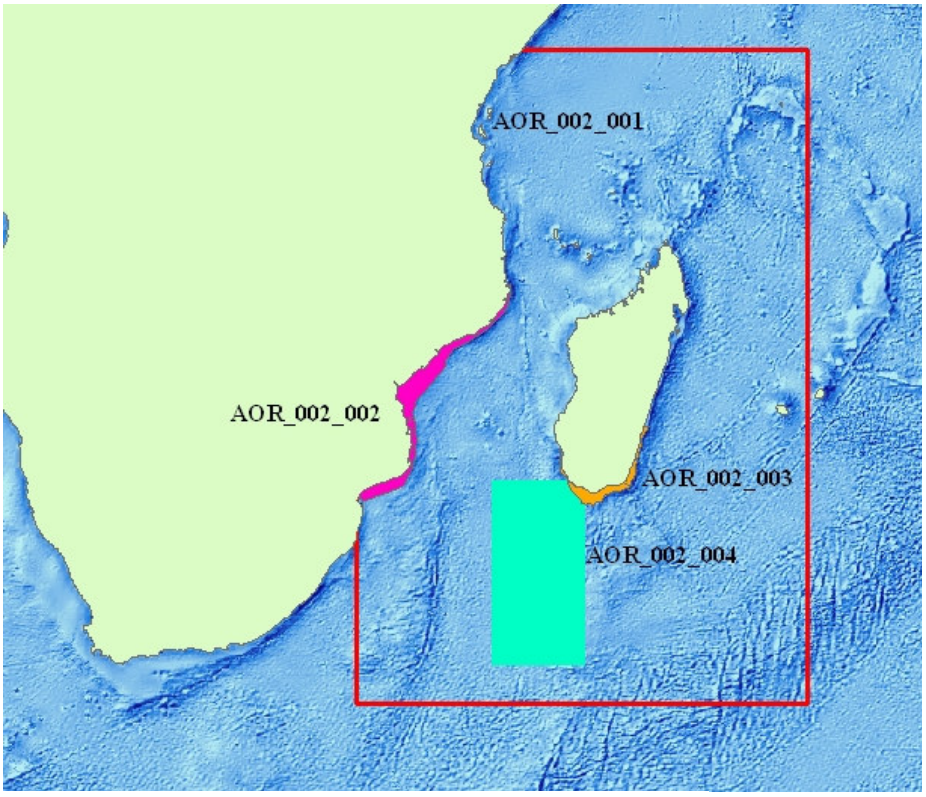

Figure 3.2.1: Surveys conducted in East Africa

3.2.1 Summary of available data for AOR 2

AOR2 lies within the Indian Ocean and forty cetacean species were predicted to be present within its boundaries. Of these forty species, only five; the common Minke whale; Fin whale; Humpback whale; Blue whale and Sei whale have associated stock/ subspecies information (Table 3.2.1). This information is to subspecies resolution for everything but the Humpback whale. For the humpback whale, breeding stock information is known, as breeding stock C is shown to migrate to breed off southern East Africa.

Abundance information exists for four species in AOR2. Boat-based line-transect surveys were used to estimate abundance for humpback whales and blue whales during three separate surveys. Humpback whale abundance was estimated, in two different survey areas, in 1994 (AOR_002_003; Best *et al*., 1996) and in 2003 (AOR_002_002; Findley *et al*., 2004) (Figure 3.2.1). The study areas do not cover the entire breeding grounds of humpback whales off East Africa, so the number of whales using this area is likely to be larger than indicated in this study. Blue whale abundance was estimated in 1996 (AOR_002_002; Best *et al*., 2003) (Figure 3.2.1) in an area estimated to cover approximately a third of the range of this population.

Indo-Pacific humpback dolphin and Indian Ocean bottlenose dolphin abundance was estimated in a small-scale photo-identification study using data collected from 1999 to 2002 (AOR_002-001; Stensland *et al*., 2006) (Figure 3.2.1). Mark-recapture methods were used to estimate abundance for each species and a large number of individuals were resighted between years suggesting that most of the population was photographed and that these species are at least seasonally resident in these waters. Abundance estimates for humpback

dolphins for 1999 (58) 2001 (65) and 2002 (63) and for Indo-Pacific bottlenose dolphins from 1999 (150), 2000 (153), 2001 (179) and 2002 (136) exist.

It is impossible to look for population trends of humpback whales and blue whales with the existing data, as only one estimate exists for each survey area and hence there are no comparable figures for trend analysis. For Indo-Pacific humpback dolphin and Indian Ocean bottlenose dolphin multiple estimates exist for survey area AOR_001_002 and there may be some potential for trend analysis between 1999 and 2002.

3.3 Area of Relevance 3: Australia

The Australian area of relevance covers the waters around the entire coast of mainland Australia and including Tasmania. The Northern boundary of the AOR is determined by the presence of the nearest land mass and the southern Boundary extends to include Tasmania.

Table 3.3.1: Summary table of the abundance data available for cetaceans in AOR_003

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. Dwarf subspecies 2. O stock	No						
Southern right whale	Unknown	Yes	✓	✓		AOR_003_005[1]	3.3.1	1
Antarctic minke whale	Unknown	No						
Sei whale	1. Southern hemisphere subspecies	No						
Bryde's whale	1. "Ordinary" stock 2. Pygmy (*B. edeni*) stock	No						
Blue whale	1. Southern Ocean/Pygmy subspecies (*B. m. brevicauda*) 2. Antarctic subspecies (*B. m. intermedia*)	No						
Fin whale	1. Southern hemisphere subspecies	No						
Arnoux's beaked whale	Unknown	No						
Pygmy right whale	Unknown	No						
Long-beaked common dolphin	1. Indo-Pacific subspecies	No						
Short-beaked common dolphin	Unknown	No						
Pygmy killer whale	Unknown	No						
Short-finned pilot whale	Unknown	No						
Long-finned pilot whale	1. Southern hemisphere subspecies	No						
Risso's dolphin	Unknown	No						
Southern bottlenose whale	Unknown	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Longman's beaked whale	Unknown	No						
Pygmy sperm whale	Unknown	No						
Dwarf sperm whale	Unknown	No						
Hourglass dolphin	Unknown	No						
Fraser's dolphin	Unknown	No						
Dusky dolphin	Unknown	No						
Southern right whale dolphin	Unknown	No						
Andrews' beaked whale	Unknown	No						
Blainville's beaked whale	Unknown	No						
Ginkgo-toothed beaked whale	Unknown	No						
Gray's beaked whale	Unknown	No						
Hector's beaked whale	Unknown	No						
Strap-toothed whale	Unknown	No						
True's beaked whale	Unknown	No						
Humpback whale	1. Breeding stock D (Western Australia) 2. Breeding stock E (Eastern Australia)	Yes	✓	✓	✓	AOR_003_002 AOR_003_003	3.3.3 3.3.4	71 66
Snubfin dolphin	Unknown	Yes		✓	✓	AOR_003_001	3.3.2	70
Killer whale	Unknown	No						
Melon-headed whale	Unknown	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Spectacled porpoise	Unknown	No						
Sperm whale	Unknown	No						
False killer whale	Unknown	No						
Indo-Pacific humpback dolphin	Unknown	Yes		✓	✓	AOR_003_001	3.3.2	70
Pantropical spotted dolphin	Unknown	No						
Rough-toothed dolphin	Unknown	No						
Striped dolphin	Unknown	No						
Spinner dolphin	Unresolved[2]	No						
Tasman or Shepherd's beaked whale	Unknown	No						
Indian Ocean bottlenose dolphin	Unknown	No		✓		AOR_003_004	3.3.4	23
Bottlenose dolphin	Unknown	No						
Cuvier's beaked whale	Unknown	No						

[1] These data are count data, not estimates of abundance

[2] Two subspecies occur in Australian waters; *S. l. longirostris* and *S. l. roseiventris*

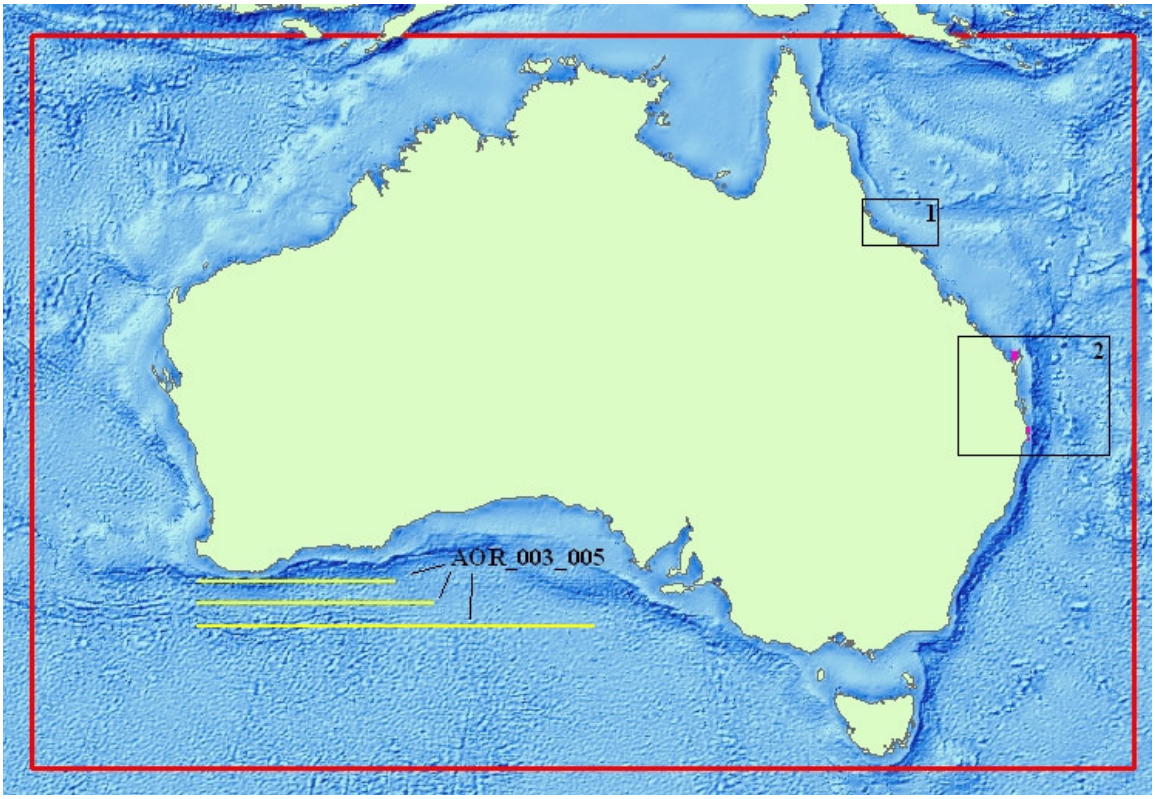

Figure 3.3.1: Overview of surveys conducted in Australian waters.

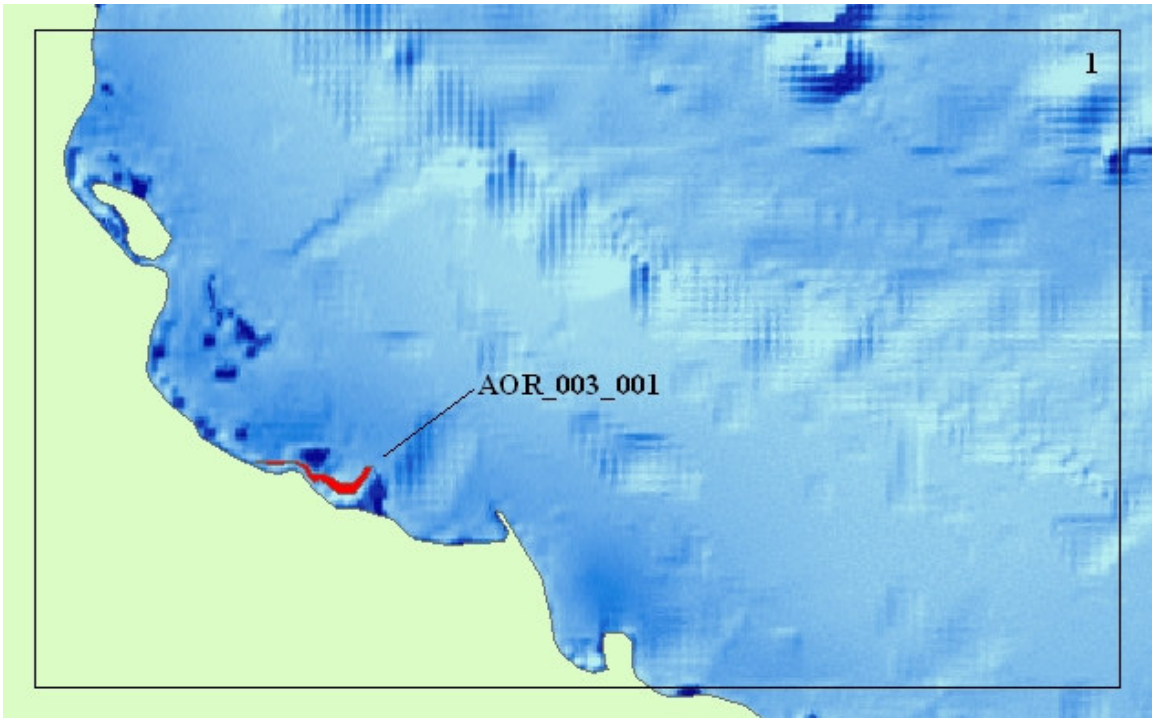

Figure 3.3.2: Survey area AOR_003_001 in Cleveland Bay, northeast Queensland.

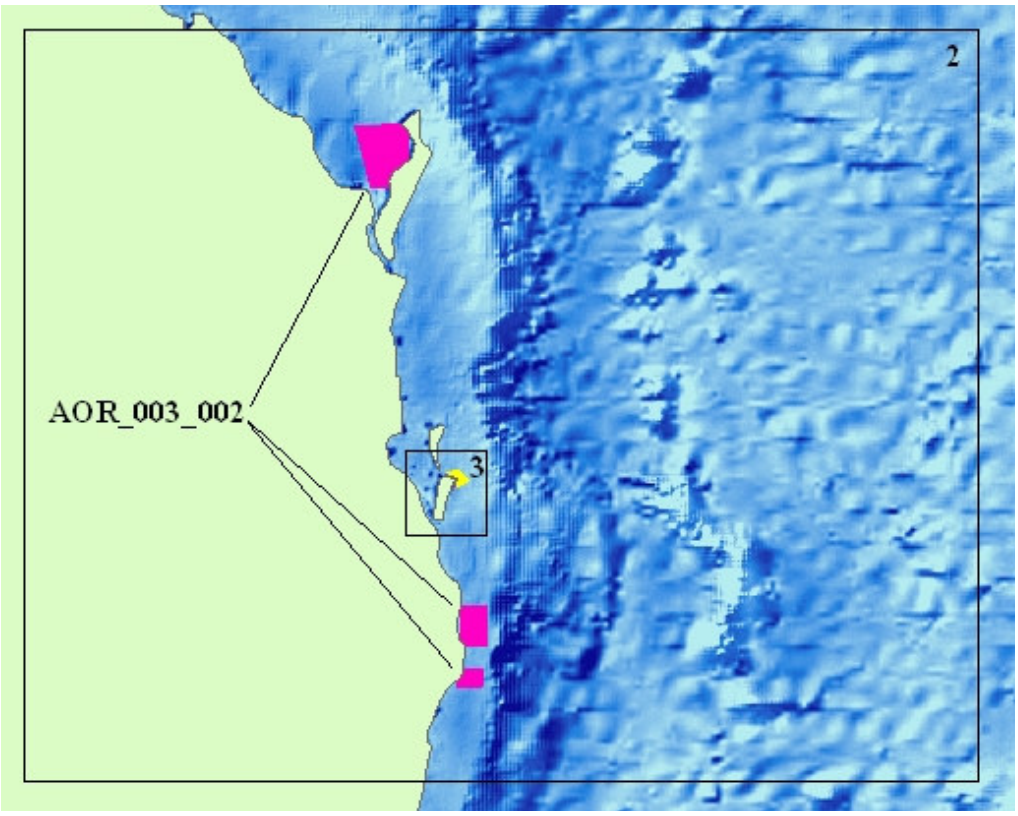

Figure 3.3.3: Survey area of AOR_003_002 conducted on the east coast of Australia

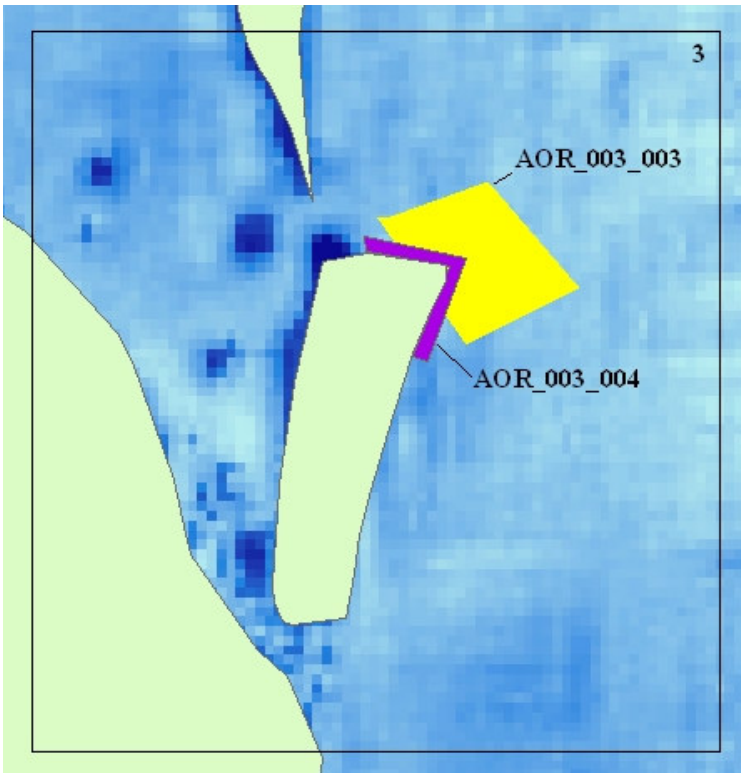

Figure 3.3.4: Surveys AOR_003_003 and AOR_003_004 conducted from Stradbrook Island

3.3.1 Summary of available data for AOR 3

AOR3 falls within both the Indian Ocean and the South West Pacific and forty-six cetacean species were predicted to be present within its boundaries. Of these forty-six species, eight; the common Minke whale; Fin whale; Humpback whale; Blue whale, Sei whale, Bryde's whale, long-finned pilot whale and long-beaked common dolphin have associated stock/ subspecies information (Table 3.3.1). This information is only to subspecies resolution for everything but the Humpback whale, common Minke whale and Bryde's whale. For the humpback whale, breeding stock information is known, as breeding stocks D and E are shown to migrate to breed off western Australia and eastern Australia respectively. For the Minke whale and Bryde's whale, stocks and subspecies are both found within the region.

Only one line transect survey was found for AOR3 and this survey resulted in annual counts of southern right whales off the southwest coast of Australia between 1983 and 1997. Southern right whale abundance was only estimated for the period 1995-97. However, abundance information exists for four species from photo-identification and land-based studies. The abundance of snubfin dolphins and Indo-Pacific humpback dolphins was estimated using photo-identification in Cleveland Bay, northeast Queensland between 1999 and 2002 (AOR_003_001; Parra *et al*., 2006) (Figure 3.3.2). Similarly the abundance of Indian Ocean bottlenose dolphins was estimated off North Stradbroke Island, using photo-identification in 1998 and 1999 (AOR_003_004; Chilvers and Corkeron, 2003) (Figure 3.3.4).

Abundance estimates of group E breeding stock humpback whales have been estimated using photo-identification methods, in three different locations on the east coast of Australia in 2005 (AOR_003_002; Paton *et al*., 2006) (Figure 3.3.3). A further study using a land-based survey from Point Lookout on North Stradbroke Island, provides both an absolute abundance of the population and relative abundance for 1981, 1982, 1986, 1987, 1991, 1993, 1996, 2000 and 2004 (AOR_003_003 Noad *et al.,* 2006) (Figure 3.3.4).

For snubfin dolphins and Indo-Pacific humpback dolphins there may be potential for trend analysis between 1999 and 2002 within AOR_003_001. For eastern Australian humpback whales multiple estimates exist for breeding stock E and there is the potential for analysis of relative abundance from the land based surveys.

3.4 Area of Relevance 4: Alaska

The Alaska Area of Relevance includes the Bering, Chukchi and Beaufort Seas. In order to include the majority of the Beaufort Sea the border of the area of relevance falls east of the Alaska-Canada border.

Table 3.4.1: Summary table of the abundance data available for cetaceans in AOR_004

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. North Pacific stock 2. Alaska stock	Yes		✓	✓	AOR_004_007 AOR_004_008 AOR_004_009 AOR_004_010	3.4.2 3.4.2 3.4.3 3.4.3	61 62 83 & 61 88 & 89
Sei whale	1. Northern hemisphere stock 2. Eastern North Pacific stock	No						
North Pacific right whale	1. Eastern North Pacific stock	No						
Blue whale	1. North Pacific subspecies (*B. m. musculus*)	No						
Bowhead whale	1. Western Arctic-Bering sea stock 2. Bering-Chukchi-Beaufort stock	Yes	✓		✓	AOR_004_011	3.4.3	40 & 86
Fin whale	1. Northern hemisphere subspecies	Yes		✓	✓	AOR_004_007 AOR_004_008 AOR_004_009 AOR_004_010	3.4.2 3.4.2 3.4.3 3.4.3	61 62 83 & 61 88 & 89
Baird's beaked whale	1. Alaska stock	No						
Beluga or white whale	1. Beaufort Sea stock 2. Bristol Bay stock 3. Cook Inlet stock 4. Eastern Bering Sea stock 5. Eastern Chukchi Sea stock	Yes		✓	✓	AOR_004_002 AOR_004_003 AOR_004_004 AOR_004_005 AOR_004_006	3.4.1 3.4.1 3.4.1 3.4.1 3.4.2	49 55 57 51 50 & 56
Gray whale	1. Eastern North Pacific stock	No						
Risso's dolphin	Unknown	No						
Pacific white-sided dolphin	1. North Pacific stock	Yes	✓			AOR_004_001	Not shown	13
Northern right whale dolphin	Unknown	Yes	✓			AOR_004_001	Not shown	13

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Hubb's beaked whale	Unknown	No						
Humpback whale	1. Central North Pacific stock 2. Western North Pacific stock	Yes		✓	✓	AOR_004_008 AOR_004_009 AOR_004_010	3.4.2 3.4.3 3.4.3	62 83 & 61 88 & 89
Stejneger's beaked whale	1. Alaska stock	No						
Narwhal	Unknown	No						
Killer whale	1. ATI transient stock 2. Eastern North Pacific (ENP) Alaska Resident Stock 3. ENP Transient Stock 4. Gulf of Alaska 5. Aleutian Isands and Bering Sea transient stock 6. West Coast transient stock	Yes			✓	AOR_004_009 AOR_004_010	3.4.3 3.4.3	83 & 61 88 & 89
Dall's porpoise	1. Alaska stock	Yes	✓	✓	✓	AOR_004_001 AOR_004_007 AOR_004_009	Not shown 3.4.2 3.4.3	13 61 83 & 61
Sperm whale	1. North Pacific stock	No						
Harbour porpoise	1. Eastern Northern Pacific stock 2. Bering Sea stock 3. Gulf of Alaska stock 4. SE Alaska stock	Yes		✓	✓	AOR_004_007 AOR_004_009	3.4.2 3.4.3	61 83 & 61
Cuvier's beaked whale	1. Alaska stock	No						

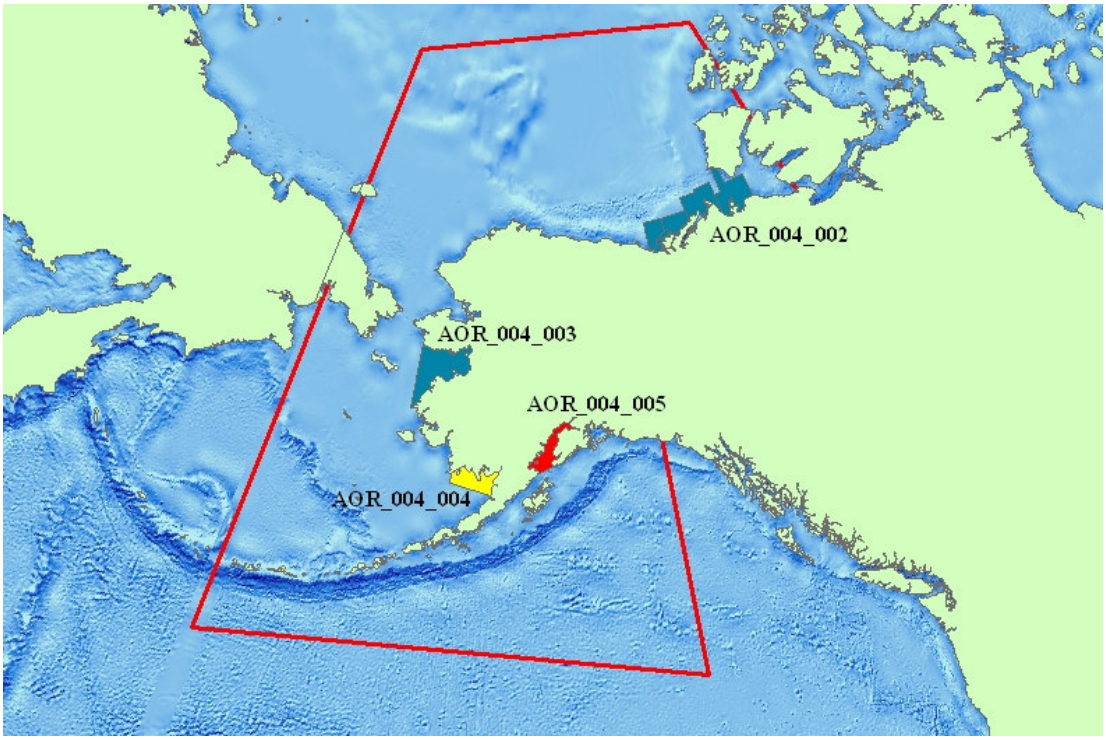

Figure 3.4.1: Beluga whale surveys conducted in Alaskan waters.

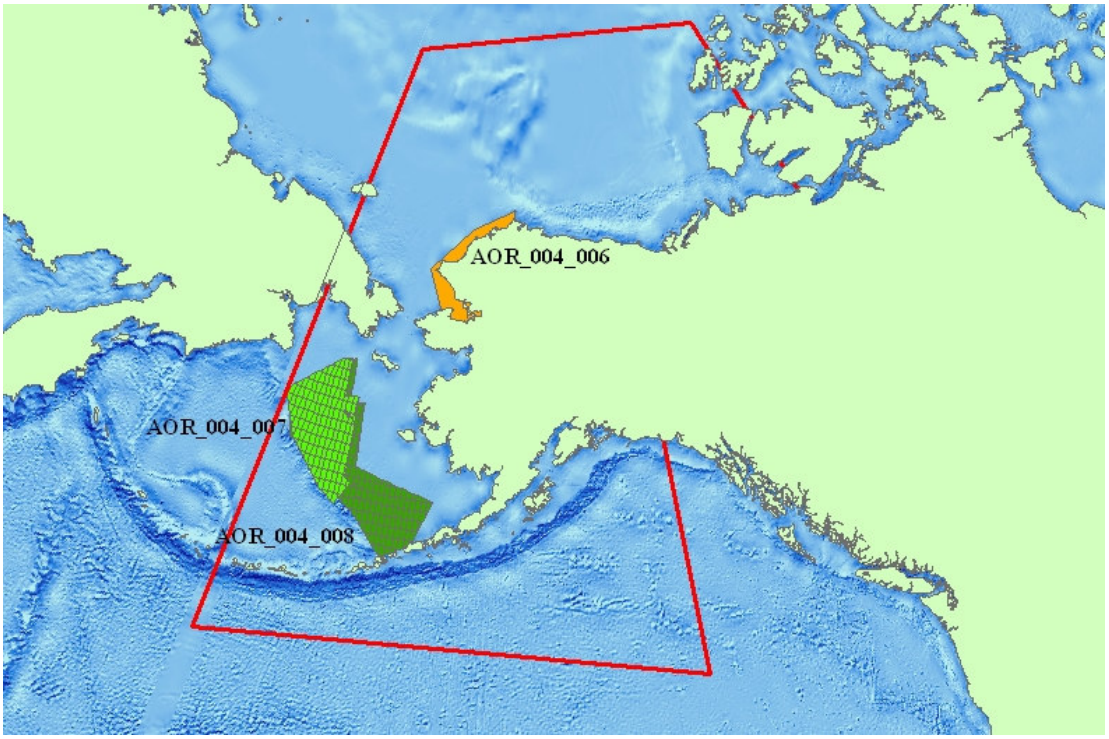

Figure 3.4.2: Beluga whale (AOR_004_006) and multi-species surveys (AOR_004_007 and AOR_004_008) conducted in Alaska.

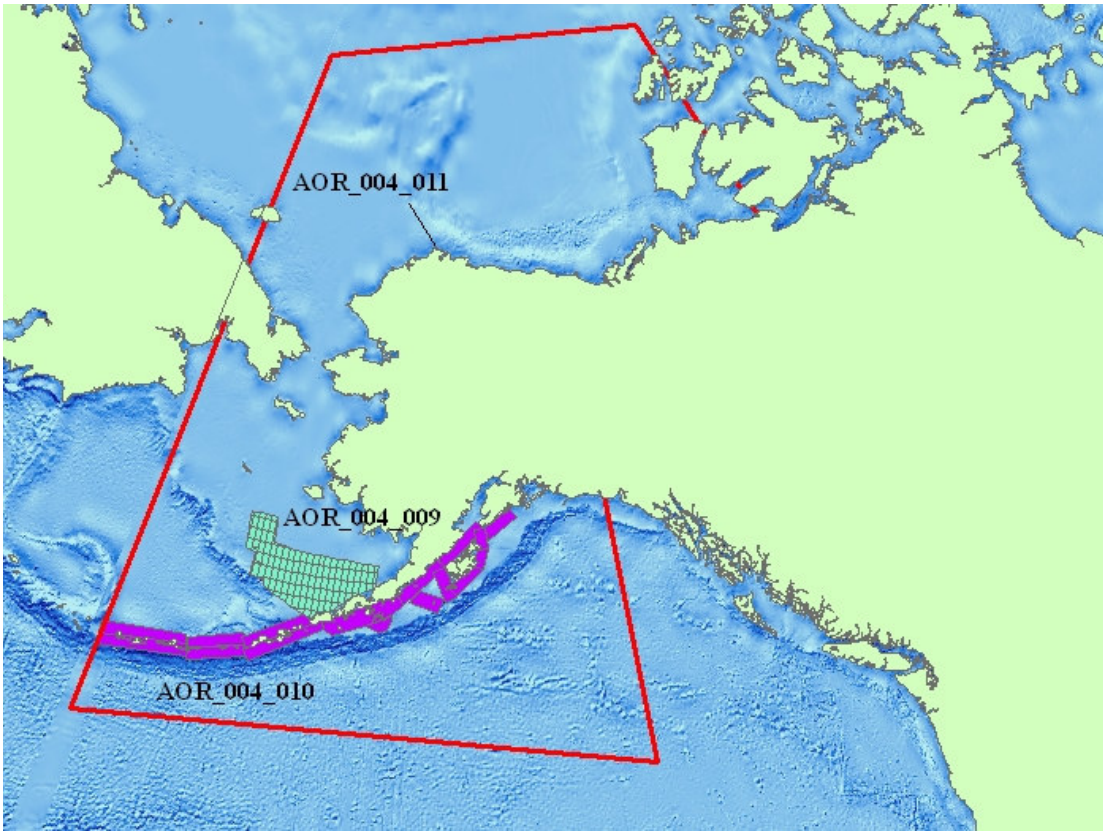

Figure 3.4.3: Multi-species surveys (AOR_004_009 and AOR_004_010) and bowhead whale surveys undertaken in Alaskan waters.

3.4.1 Summary of available data for AOR 4

AOR4 falls within both the North east Pacific and the Beaufort Sea and twenty-one cetacean species were predicted to be present within its boundaries. Of these twenty-one species, seventeen have associated stock information (Table 3.4.1). Alaska is a well studied area and as such resolution about stocks is well known.

Abundance information exists for ten species in AOR4 (Table 3.4.1; Figures 3.4.1-3.4.3). For most of these species the National Marine Fisheries Service have multiple stock abundance estimates and as such the potential for trend analysis exists for multiple stocks within this area.

3.5 Area of Relevance 5A: West Coast of North America

Area of relevance 5A reaches from the Alaska-Canada border in the north to the Mexico-Central America border in the south.

Table 3.5.1: Summary table of the abundance data available for cetaceans in AOR_005A

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. North Pacific stock 2. California-Oregon-Washington stock	Yes			✓	AOR_005A_001 AOR_005A_006 AOR_005A_007 AOR_005A_008	3.5.2 3.5.4 3.5.4 3.5.4	85 2 & 16 2, 16 & 36 16 & 36
Sei whale	1. Eastern North Pacific stock	Yes		✓		AOR_005A_007	3.5.4	2, 16 & 36
Bryde's whale	1. Eastern Tropical Pacific stock	Yes	✓	✓	✓	AOR_005A_007 AOR_005A_012 AOR_005A_013	3.5.4 3.5.5 3.5.5	2, 16 & 36 43 42 & 43
North Pacific right whale	1. Eastern North Pacific stock	Yes		✓		AOR_005A_008	3.5.4	16 & 36
Blue whale	1. Eastern North Pacific stock	Yes		✓	✓	AOR_005A_006 AOR_005A_007 AOR_005A_008 AOR_005A_009 AOR_005A_012	3.5.4 3.5.4 3.5.4 3.5.4 3.5.5	2 & 16 2, 16 & 36 16 & 36 16 43
Fin whale	1. California-Oregon-Washington stock 2. Northeast Pacific stock 3. Gulf of California subspecies	Yes		✓	✓	AOR_005A_001 AOR_005A_006 AOR_005A_007 AOR_005A_008	3.5.2 3.5.4 3.5.4 3.5.4	85 2 & 16 2, 16 & 36 16 & 36
Baird's beaked whale	1. California-Oregon-Washington stock	Yes		✓	✓	AOR_005A_006 AOR_005A_007	3.5.4 3.5.4	2 & 16 2, 16 & 36
Long-beaked common dolphin	1. California stock	Yes		✓	✓	AOR_005A_007 AOR_005A_008[1] AOR_005A_012[1] AOR_005A_013	3.5.4 3.5.4 3.5.5 3.5.5	2, 16 & 36 16 & 36 43 42 & 43

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Short-beaked common dolphin	1. California- Oregon-Washington stock	Yes	✓	✓	✓	AOR_005A_006 AOR_005A_007 AOR_005A_008[1] AOR_005A_012[1] AOR_005A_013	3.5.4 3.5.4 3.5.4 3.5.5 3.5.5	2 & 16 2, 16 & 36 16 & 36 43 42 & 43
Gray whale	1. Eastern North Pacific stock	Yes		✓		AOR_005A_008 AOR_005A_016	3.5.4 3.5.3	16 & 36 10
Pygmy killer whale	Unknown	Yes	✓			AOR_005A_012	3.5.5	43
Short-finned pilot whale	1. California- Oregon-Washington stock	Yes	✓	✓	✓	AOR_005A_007 AOR_005A_012 AOR_005A_013	3.5.4 3.5.5 3.5.5	2, 16 & 36 43 42 & 43
Risso's dolphin	1. California-Oregon-Washington stock	Yes	✓	✓	✓	AOR_005A_006 AOR_005A_007 AOR_005A_008 AOR_005A_012	3.5.4 3.5.4 3.5.4 3.5.5	2 & 16 2, 16 & 36 16 & 36 43
Longman's beaked whale	Unknown	No						
Pygmy sperm whale	1. California-Oregon-Washington stock	No						
Dwarf sperm whale	1. California- Oregon-Washington stock	Yes	✓			AOR_005A_012	3.5.5	43
Fraser's dolphin	Unknown	Yes	✓			AOR_005A_012	3.5.5	43
Pacific white-sided dolphin	1. North Pacific stock 2. California-Oregon-Washington stock with North and South forms	Yes		✓	✓	AOR_005A_001 AOR_005A_006 AOR_005A_007 AOR_005A_008	3.5.2 3.5.4 3.5.4 3.5.4	85 2 & 16 2, 16 & 36 16 & 36
Northern right whale dolphin	1. California-Oregon-Washington stock	Yes		✓	✓	AOR_005A_006 AOR_005A_007 AOR_005A_008	3.5.4 3.5.4 3.5.4	2 & 16 2, 16 & 36 16 & 36

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Hubb's beaked whale	Unknown	No						
Blainville's beaked whale	Unknown	No						
Ginkgo-toothed beaked whale	Unknown	No						
Humpback whale	1. Central North Pacific stock 2. Eastern North Pacific stock	Yes		✓	✓	AOR_005A_001 AOR_005A_002 AOR_005A_006 AOR_005A_007 AOR_005A_008	3.5.2 3.5.2 3.5.4 3.5.4 3.5.4	85 17 2 & 16 2, 16 & 36 16 & 36
Pygmy beaked whale	Unknown	No						
Perrin's beaked whale	Unknown	No						
Stejneger's beaked whale	Unknown	No						
Killer whale	1. Eastern North Pacific (ENP) Northern resident stock 2. ENP Offshore stock 3. ENP Transient Stock 4. ENP Southern resident stock 5. West coast transient stock	Yes	✓	✓	✓	AOR_005A_001 AOR_005A_006 AOR_005A_007 AOR_005A_008 AOR_005A_012	3.5.2 3.5.4 3.5.4 3.5.4 3.5.5	85 2 & 16 2, 16 & 36 16 & 36 43
Melon-headed whale	Unknown	Yes	✓			AOR_005A_012	3.5.5	43
Dall's porpoise	1. California-Oregon-Washington stock	Yes		✓	✓	AOR_005A_001 AOR_005A_002 AOR_005A_006 AOR_005A_007 AOR_005A_008	3.5.2 3.5.2 3.5.4 3.5.4 3.5.4	85 17 2 & 16 2, 16 & 36 16 & 36

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Sperm whale	1. North Pacific stock 2. California-Oregon-Washington stock	Yes	✓	✓	✓	AOR_005A_006 AOR_005A_007 AOR_005A_008 AOR_005A_011 AOR_005A_012 AOR_005A_013	3.5.4 3.5.4 3.5.4 3.5.5 3.5.5 3.5.5	2 & 16 2, 16 & 36 16 & 36 3 43 42 & 43
Harbour porpoise	1. Monterey Bay stock 2. Morro Bay stock 3. Washington Inland waters stock 4. Northern California-Southern Oregon stock 5. Oregon-Washington coastal stock 6. San Francisco-Russian River stock	Yes		✓	✓	AOR_005A_001 AOR_005A_003 AOR_005A_004 AOR_005A_005 AOR_005A_008	3.5.2 3.5.3 3.5.3 3.5.3 3.5.4	85 35 21 & 35 19 & 20 16 & 36
Vaquita	Unknown	No						
False killer whale	Unknown	Yes	✓			AOR_005A_012	3.5.5	43
Pantropical spotted dolphin	1. Coastal stock 2. Northeastern offshore stock 3. Southern offshore stock	Yes	✓	✓[2]	✓	AOR_005A_010 AOR_005A_012 AOR_005A_013 AOR_005A_014 AOR_005A_015	3.5.4 3.5.5 3.5.5 3.5.6 3.5.6	41 & 43 43 42 & 43 41 & 43 41 & 43
Rough-toothed dolphin	Unknown	Yes	✓		✓	AOR_005A_012	3.5.5	43
Striped dolphin	1. California-Oregon-Washington stock	Yes	✓	✓	✓	AOR_005A_006 AOR_005A_007 AOR_005A_012 AOR_005A_013	3.5.4 3.5.4 3.5.5 3.5.5	2 & 16 2, 16 & 36 43 42 & 43

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Spinner dolphin	1. Eastern stock 2. Central American stock 3. Whitebelly stock	Yes	✓	✓[3]	✓	AOR_005A_012 AOR_005A_013	3.5.5 3.5.5	43 42 & 43
Bottlenose dolphin	1. California coastal stock 2. California- Oregon-Washington stock	Yes	✓	✓	✓	AOR_005A_007 AOR_005A_008 AOR_005A_012	3.5.4 3.5.4 3.5.5	2, 16 & 36 16 & 36 43
Cuvier's beaked whale	1. California-Oregon-Washington stock	Yes	✓	✓	✓	AOR_005A_007 AOR_005A_012	3.5.4 3.5.5	2, 16 & 36 43

[1] Common dolphin abundance estimate was not species specific

[2] Abundance of northeastern offshore stock not estimated in 1990's

[3] Abundance of eastern stock not estimated in 1990's

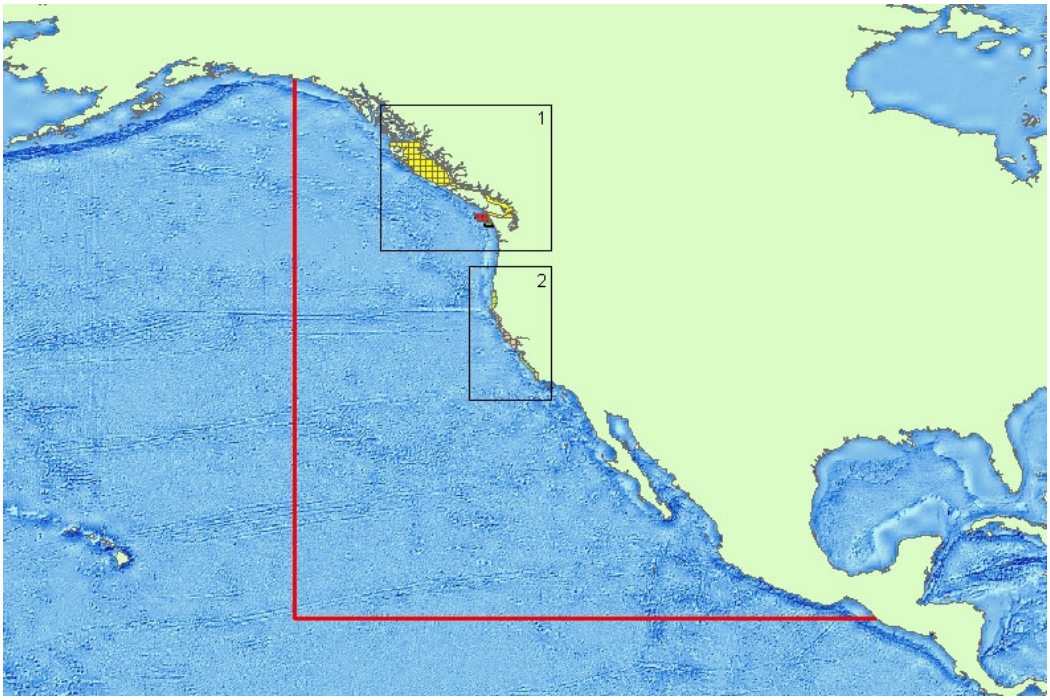

Figure 3.5.1: Small-scale coastal cetacean surveys from the west coast of Canada, USA and Mexico.

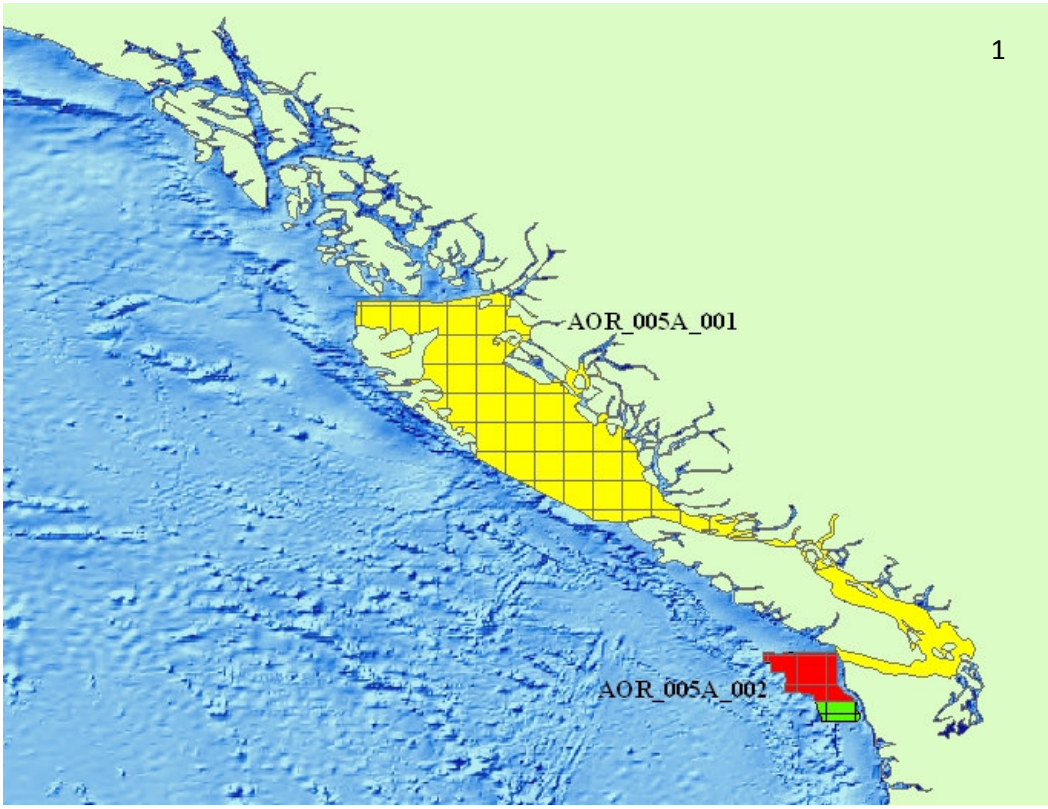

Figure 3.5.2: Box 1 - Small-scale coastal surveys conducted in Canadian waters.

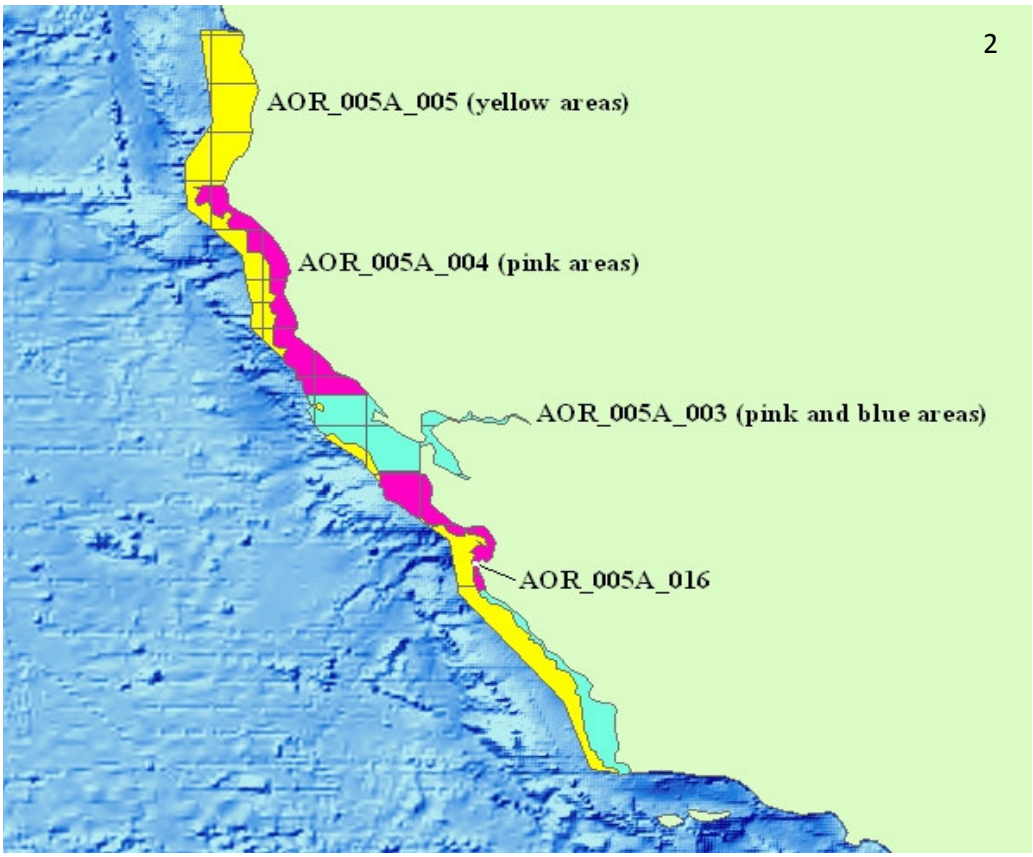

Figure 3.5.3: Box 2 - Small-scale coastal surveys conducted off the coast of California.

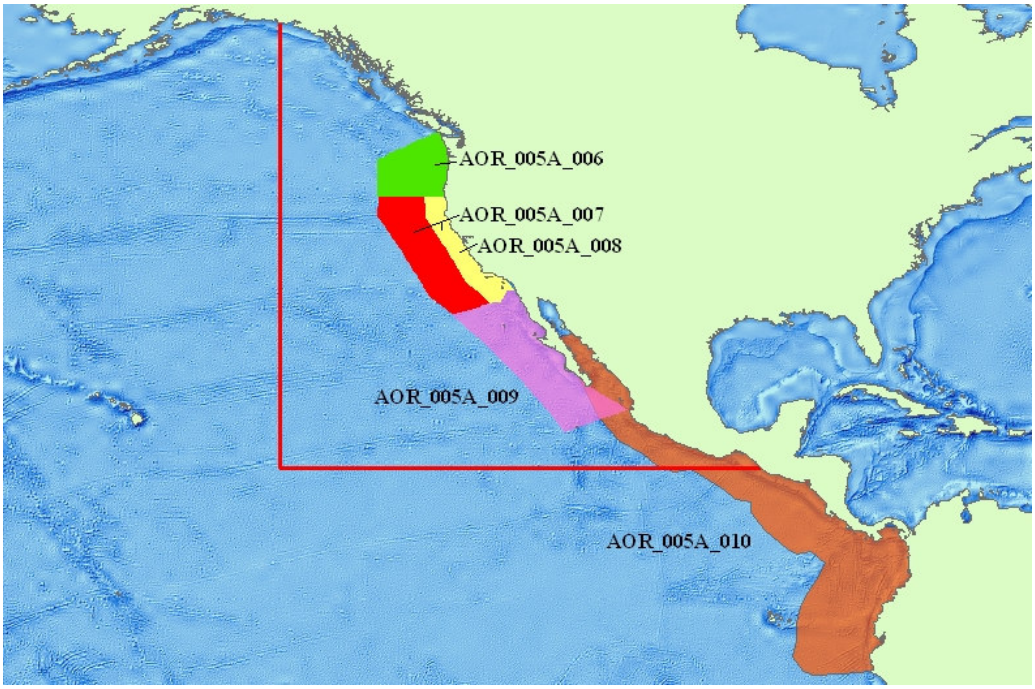

Figure 3.5.4: Medium-scale coastal surveys conducted off the west coast of Canada, USA and Mexico.

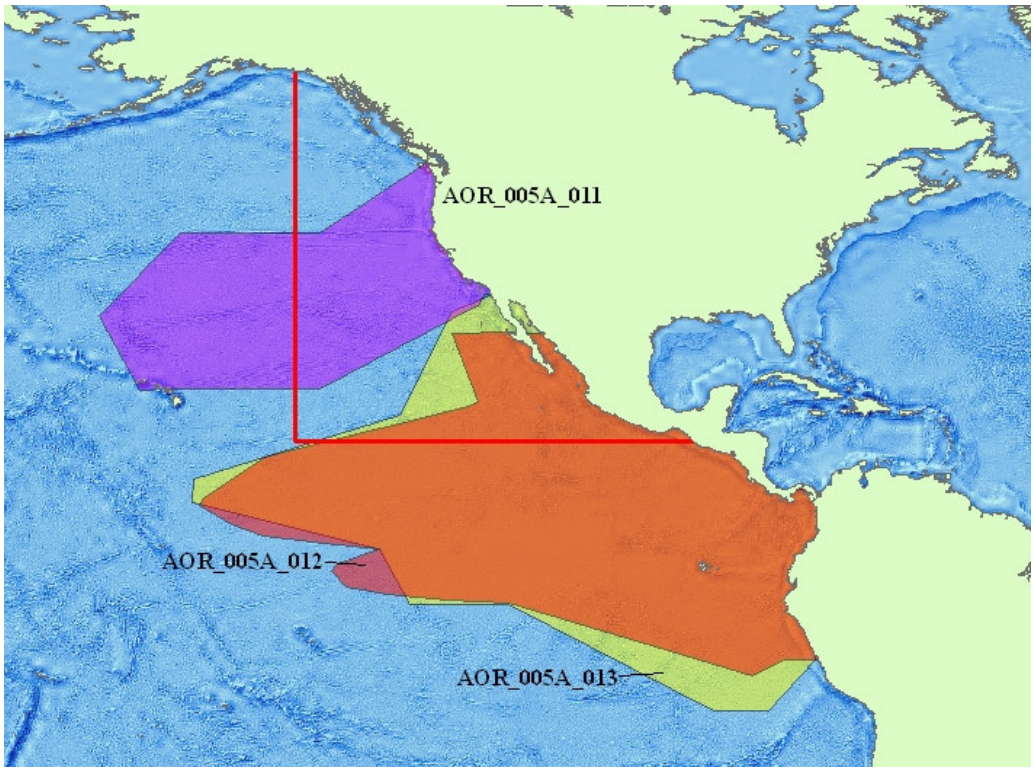

Figure 3.5.5: Large-scale surveys conducted in the eastern Pacific.

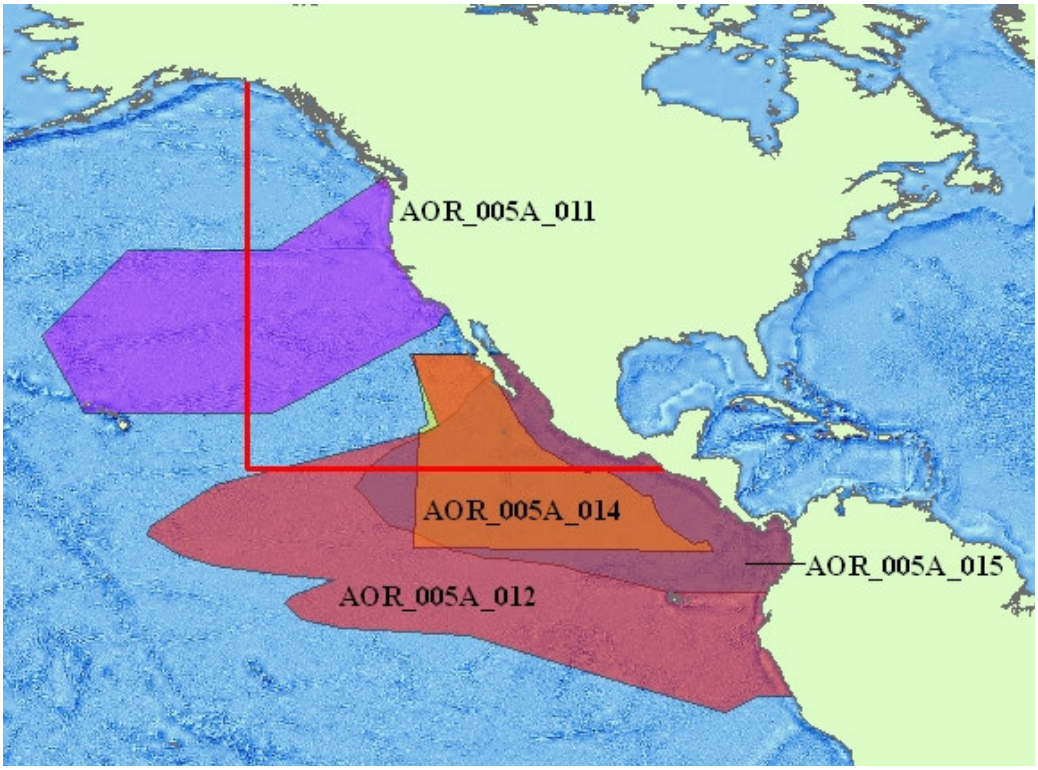

Figure 3.5.6: Large-scale surveys conducted off the west coast of Canada, USA and Mexico showing additional strata boundaries of the Eastern Tropical Pacific surveys.

3.5.1 Summary of available data for AOR 5A

AOR5A falls within the Northeast Pacific Ocean and thirty-nine cetacean species were predicted to be present within its boundaries. Of these thirty-nine species, twenty-six have associated stock information (Table 3.5.1). The West coast of the USA is a well studied area and as such resolution about stocks is well known.

Abundance information exists for thirty species in AOR5A (Table 3.5.1; Figures 3.5.1-3.5.6). For most of these species the National Marine Fisheries Service have multiple stock abundance estimates and as such the potential for trend analysis exists for multiple stocks within this area.

3.6 Area of Relevance 5B: East Coast US

Area of Relevance 5B includes the east coast of Canada and the USA as well as the Gulf of Mexico.

Table 3.6.1: Summary table of the abundance data available for cetaceans in AOR_005B

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. Canadian Eastern coastal stock	Yes		✓	✓	AOR_005B_001 AOR_005B_004 AOR_005B_009	3.6.1 3.6.1 3.6.2	53 69 64
Sei whale	1. Novia Scotia stock	Yes		✓	✓	AOR_005B_004	3.6.1	69
Bryde's whale	1. Northern Gulf of Mexico stock	Yes		✓		AOR_005B_007 AOR_005B_011 AOR_005B_012	3.6.1 3.6.2 3.6.2	27, 28 & 48 27 37, 65 & 84
North Atlantic right whale	1. Western stock	Yes		✓	✓	AOR_005B_004	3.6.1	69
Blue whale	1. Western North Atlantic stock	No						
Bowhead whale	1. Hudson Bay- Foxe Basin stock 2. Baffin Bay- Davis Strait stock[1]	No						
Fin whale	1. Western North Atlantic stock	Yes		✓	✓	AOR_005B_001 AOR_005B_004 AOR_005B_005 AOR_005B_009	3.6.1 3.6.1 3.6.1 3.6.2	53 69 39 64
Long-beaked common dolphin	Unknown	No						
Short-beaked common dolphin	Unknown	Yes		✓	✓	AOR_005B_004 AOR_005B_005	3.6.1 3.6.1	69 39
Beluga or white whale	1. Eastern Hudson Bay stock 2. Ungava Bay stock 3. St Lawrence River stock	No						
Pygmy killer whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓		AOR_005B_007 AOR_005B_012	3.6.1 3.6.2	27, 28 & 48 37, 65 & 84

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from 1980's	1990's	2000's	Survey ID	Figure	Reference
Short-finned pilot whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_005 AOR_005B_007 AOR_005B_009 AOR_005B_012	3.6.1 3.6.1 3.6.1 3.6.2 3.6.2	69 39 27, 28 & 48 64 37, 65 & 84
Long-finned pilot whale	1. Western North Atlantic stock	Yes		✓	✓	AOR_005B_001 AOR_005B_004 AOR_005B_005 AOR_005B_009	3.6.1 3.6.1 3.6.1 3.6.2	53 69 39 64
Risso's dolphin	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_011 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2 3.6.2	69 27, 28 & 48 64 27 37, 65 & 84
Northern bottlenose whale	1. Western North Atlantic stock 2. Gully (Newfoundland) stock	Yes		✓	✓	AOR_005B_002 AOR_005B_004	3.6.1 3.6.1	44 69
Pygmy sperm whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_011 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2 3.6.2	69 27, 28 & 48 64 27 37, 65 & 84
Dwarf sperm whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_011 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2 3.6.2	69 27, 28 & 48 64 27 37, 65 & 84
Atlantic white-sided dolphin	1. Western North Atlantic stock	Yes		✓	✓	AOR_005B_001 AOR_005B_004	3.6.1 3.6.1	53 69
White-beaked dolphin	1. Western North Atlantic stock	Yes		✓		AOR_005B_001	3.6.1	53

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from 1980's	1990's	2000's	Survey ID	Figure	Reference
Fraser's dolphin	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓		AOR_005B_007 AOR_005B_011 AOR_005B_012	3.6.1 3.6.2 3.6.2	27, 28 & 48 27 37, 65 & 84
Sowerby's beaked whale	1. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2	69 27, 28 & 48 64 37, 65 & 84
Blainville's beaked whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_012 AOR_005B_013 AOR_005B_014	3.6.1 3.6.1 3.6.2 3.6.2 3.6.3 3.6.3	69 27, 28 & 48 64 37, 65 & 84 26 63
Gervais' beaked whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2	69 27, 28 & 48 64 37, 65 & 84
True's beaked whale	1. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2	69 27, 28 & 48 64 37, 65 & 84
Humpback whale	1. Gulf of Maine stock	Yes		✓	✓	AOR_005B_001 AOR_005B_004 AOR_005B_008 AOR_005B_015	3.6.1 3.6.1 3.6.2 Not shown	53 69 25 79
Narwhal	1. Baffin Bay stock 2. Hudson Strait-Repulse Bay stock	No						
Killer whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓		AOR_005B_007 AOR_005B_012	3.6.1 3.6.2	27, 28 & 48 37, 65 & 84

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from 1980's	1990's	2000's	Survey ID	Figure	Reference
Melon-headed whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓		AOR_005B_007 AOR_005B_012	3.6.1 3.6.2	27, 28 & 48 37, 65 & 84
Sperm whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_005 AOR_005B_006 AOR_005B_007 AOR_005B_009 AOR_005B_011 AOR_005B_012	3.6.1 3.6.1 3.6.1 3.6.1 3.6.2 3.6.2 3.6.2	69 39 7 & 48 27, 28 & 48 64 27 37, 65 & 84
Harbour porpoise	1. Gulf of Maine- Bay of Fundy stock 2. Gulf of St Lawrence stock 3. Newfoundland and Labrador stock	Yes		✓	✓	AOR_005B_001 AOR_005B_003 AOR_005B_004	3.6.1 3.6.1 3.6.1	53 68 69
False killer whale	1. Northern gulf of Mexico stock	Yes		✓		AOR_005B_007 AOR_005B_011 AOR_005B_012	3.6.1 3.6.2 3.6.2	27, 28 & 48 27 37, 65 & 84
Pantropical spotted dolphin	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_011 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2 3.6.2	69 27, 28 & 48 64 27 37, 65 & 84
Rough-toothed dolphin	1. Northern Gulf of Mexico stock	Yes		✓		AOR_005B_007 AOR_005B_009 AOR_005B_011 AOR_005B_012	3.6.1 3.6.2 3.6.2 3.6.2	27, 28 & 48 64 27 37, 65 & 84
Clymene dolphin	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓		AOR_005B_007 AOR_005B_009 AOR_005B_012	3.6.1 3.6.2 3.6.2	27, 28 & 48 64 37, 65 & 84

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Striped dolphin	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_005 AOR_005B_007 AOR_005B_009 AOR_005B_012	3.6.1 3.6.1 3.6.1 3.6.2 3.6.2	69 39 27, 28 & 48 64 37, 65 & 84
Atlantic spotted dolphin	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓		AOR_005B_004 AOR_005B_005 AOR_005B_007 AOR_005B_009 AOR_005B_010 AOR_005B_011 AOR_005B_012	3.6.1 3.6.1 3.6.1 3.6.2 3.6.2 3.6.2 3.6.2	69 39 27, 28 & 48 64 45 27 37, 65 & 84
Spinner dolphin	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓		AOR_005B_004 AOR_005B_007 AOR_005B_011 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2	69 27, 28 & 48 27 37, 65 & 84

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from 1980's	1990's	2000's	Survey ID	Figure	Reference
Bottlenose dolphin	1. Eastern Gulf of Mexico coastal stock 2. Gulf of Mexico Bay Sound and estuarine stock 3. Gulf of Mexico Continental shelf and slope stock 4. Gulf of Mexico outer continental shelf stock 5. Northern Gulf of Mexico coastal stock 6. Western Gulf of Mexico coastal stock 7. Western North Atlantic coastal stock 8. Western North Atlantic offshore stock	Yes		✓	✓	AOR_005B_004 AOR_005B_005 AOR_005B_007 AOR_005B_009 AOR_005B_010 AOR_005B_011 AOR_005B_012	3.6.1 3.6.1 3.6.1 3.6.2 3.6.2 3.6.2 3.6.2	69 39 27, 28 & 48 64 45 27 37, 65 & 84
Cuvier's beaked whale	1. Northern Gulf of Mexico stock 2. Western North Atlantic stock	Yes		✓	✓	AOR_005B_004 AOR_005B_007 AOR_005B_009 AOR_005B_012	3.6.1 3.6.1 3.6.2 3.6.2	69 27, 28 & 48 64 37, 65 & 84

[1] Whether this is one stock or two is currently being disputed.

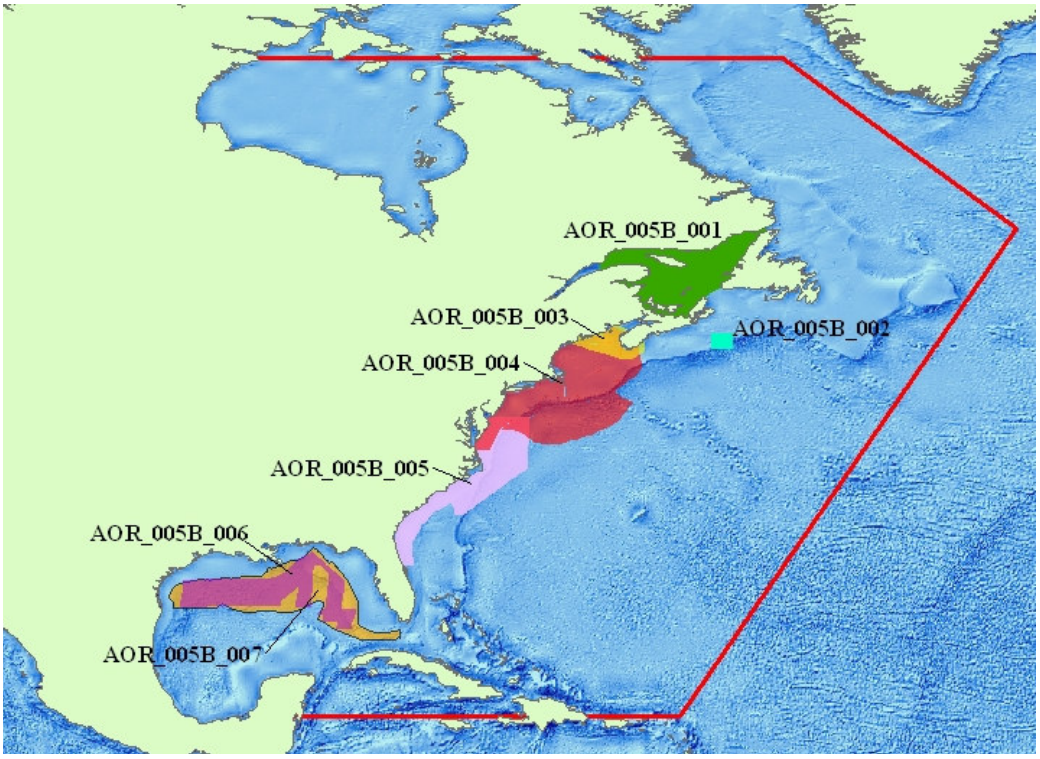

Figure 3.6.1: Cetacean surveys conducted on the east coast of Canada and America and in the Gulf of Mexico.

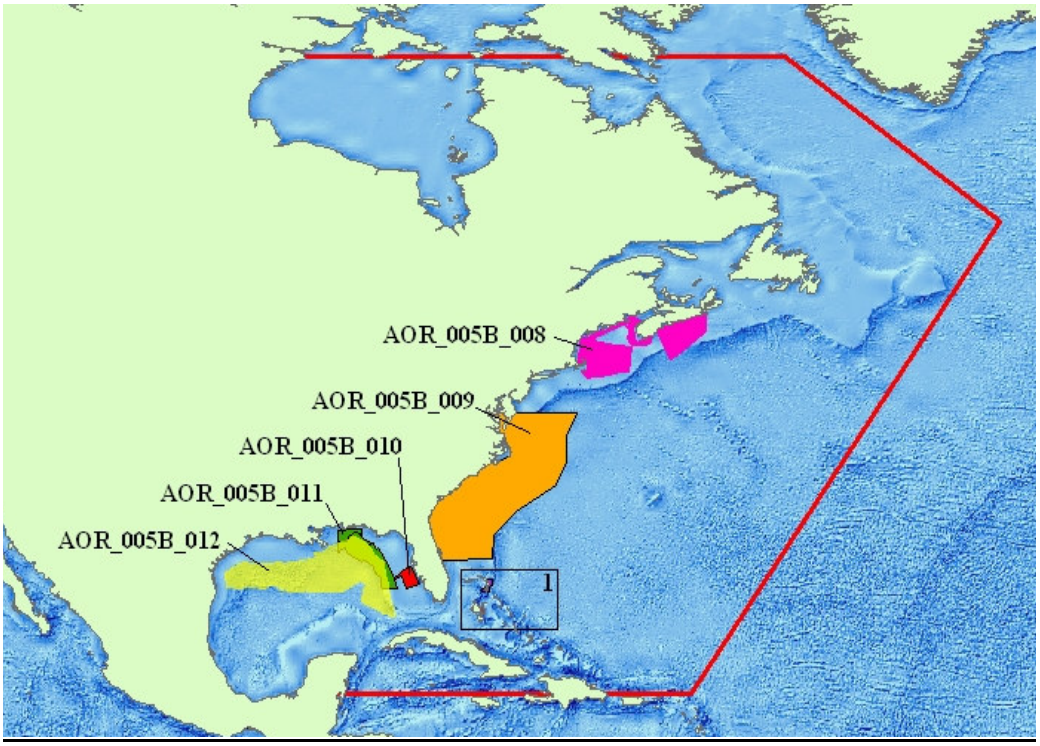

Figure 3.6.2: Cetacean surveys conducted on the east coast of Canada, America and Mexico.

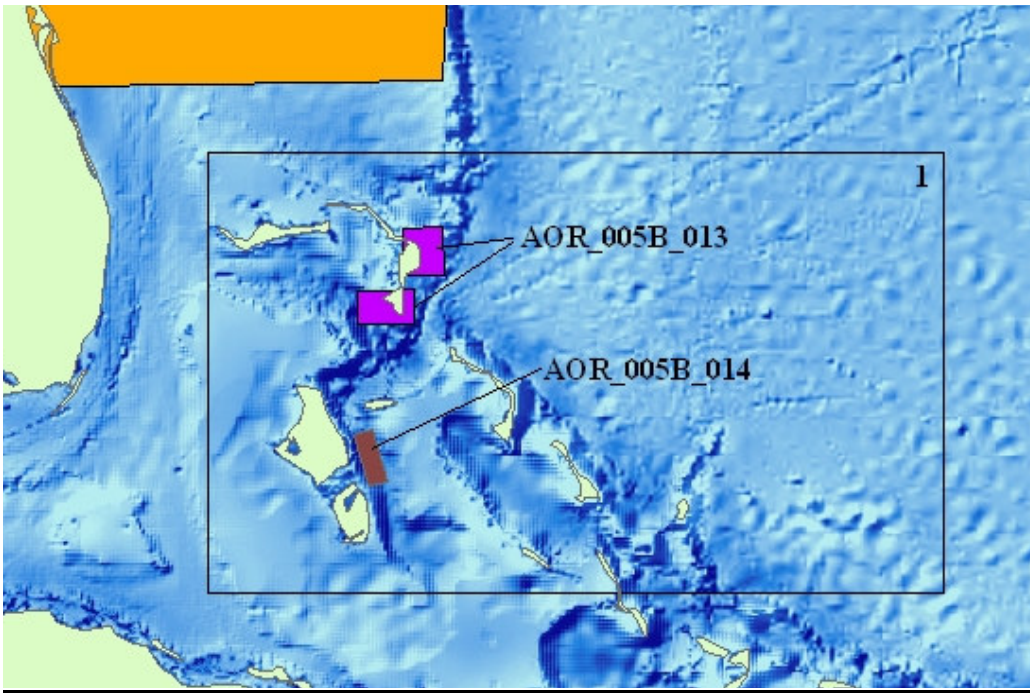

Figure 3.6.3: Box 1 - small-scale cetacean surveys conducted in the Bahamas.

3.6.1 Summary of available data for AOR 5B

AOR5B falls within the North east Pacific Ocean and thirty-nine cetacean species were predicted to be present within its boundaries. Of these thirty-nine species, thirty-seven have associated stock information (Table 3.6.1). The East coast of the USA is a well studied area and as such resolution about stocks is well known.

Abundance information exists for thirty-four species in AOR5B (Table 3.6.1; Figures 3.6.1-3.6.3). For most of these species the National Marine Fisheries Service have multiple stock abundance estimates and as such the potential for trend analysis exists for multiple stocks within this area.

3.7 Area of Relevance 6A: Europe

This area has been divided in two. AOR_006A includes the Northeast Atlantic, North Sea, Barents Sea and the Mediterranean, while AOR_006B consists of the Sakhalin Island region of Russia.

Table 3.7.1: Summary table of the abundance data available for cetaceans in AOR_006A

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. Central North Atlantic stock 2. Northeastern Atlantic stock	Yes	✓	✓	✓	AOR_006A_009 AOR_006A_011 AOR_006A_012	3.7.2 3.7.3 3.7.4	24, 74 & 78 9, 11, 22, 46 & 76 47 & 14
Sei whale	1. North Atlantic stock	Yes	✓	✓		AOR_006A_001 AOR_006A_011	3.7.1 3.7.3	58 & 59 9, 11, 22, 46 & 76
Bryde's whale	Unknown	No						
Blue whale	1. North Atlantic stock (*B. m. musculus*)	Yes	✓			AOR_006A_011	3.7.3	9, 11, 22, 46 & 76
Bowhead whale	1. Spitsbergen stock	No						
Fin whale	1. North Norway stock 2. West Norway-Faroe Islands stock 3. East Greenland- Iceland stock 4. West Greenland stock 5. British Isles, Spain and Portugal stock 6. Mediterranean stock	Yes	✓	✓		AOR_006A_001 AOR_006A_007 AOR_006A_009 AOR_006A_011	3.7.1 3.7.1 3.7.2 3.7.3	58 & 59 34 24, 74 & 78 9, 11, 22, 46 & 76
Long-beaked common dolphin	Unknown	No						
Short-beaked common dolphin	1. Mediterranean subpopulation (*D. capensis*) 2. Black sea subpopulation (*D. c. ponticus*)	Yes		✓	✓	AOR_006A_002 AOR_006A_012	3.7.1 3.7.4	15 14 & 47
Beluga or white whale	1. 'Karskaya' (Russian Arctic) stock 2. Barents Sea and Svalbard stock	No						
Pygmy killer whale	Unknown	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Short-finned pilot whale	Unknown	Yes	✓			AOR_006A_011	3.7.3	9, 11, 22, 46 & 76
Long-finned pilot whale	1. Northeastern Atlantic stock 2. Mediterranean stock	Yes	✓			AOR_006A_011	3.7.3	9, 11, 22, 46 & 76
Risso's dolphin	1. Mediterranean stock	No						
Northern bottlenose whale	Unknown	Yes	✓			AOR_006A_011	3.7.3	9, 11, 22, 46 & 76
Pygmy sperm whale	Unknown	No						
Dwarf sperm whale	Unknown	No						
Atlantic white-sided dolphin	Unknown	Yes		✓	✓	AOR_006A_001 AOR_006A_002 AOR_006A_012	3.7.1 3.7.1 3.7.4	58 & 59 15 14 & 47
White-beaked dolphin	1. Eastern North Atlantic stock	Yes	✓	✓	✓	AOR_006A_011 AOR_006A_012	3.7.3 3.7.4	9, 11, 22, 46 & 76 14 & 47
Fraser's dolphin	Unknown	No						
Sowerby's beaked whale	Unknown	No						
Blainville's beaked whale	Unknown	No						
Gervais' beaked whale	Unknown	No						
True's beaked whale	Unknown	No						
Humpback whale	1. North Atlantic stock	Yes	✓	✓		AOR_006A_009 AOR_006A_011 AOR_006A_014	3.7.2 3.7.3 Not shown	24, 74 & 78 9, 11, 22, 46 & 76 79
Narwhal	1. Greenland Sea and eastward stock	No						
Killer whale	1. Strait of Gibraltar stock	Yes	✓	✓	✓	AOR_006A_011	3.7.3	9, 11, 22, 31, 46 & 76

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Melon-headed whale	Unknown	No						
Sperm whale	1. Mediterranean stock 2. North Atlantic stock	Yes	✓	✓	✓	AOR_006A_011 AOR_006A_013	3.7.3 Not shown	9, 11, 22, 46 & 76 38
Harbour porpoise	1. North Atlantic stock 2. Norwegian west coast stock 3. Ireland stock 4. British North Sea stock 5. Danish North Sea stock 6. inland waters of Denmark stock 7. Baltic Sea stock 8. Black Sea subspecies (*P. p. relicta*)	Yes	✓	✓	✓	AOR_006A_003 AOR_006A_004 AOR_006A_008 AOR_006A_011 AOR_006A_012 AOR_006A_015	3.7.1 3.7.1 3.7.1 3.7.3 3.7.4 3.7.1	73 75 6 9, 11, 22, 46 & 76 14 & 47 81
False killer whale	Unknown	No						
Pantropical spotted dolphin	Unknown	No						
Rough-toothed dolphin	Unknown	No						
Clymene dolphin	Unknown	No						
Striped dolphin	1. Mediterranean stock	Yes		✓	✓	AOR_006A_007 AOR_006A_010 AOR_006A_012	3.7.1 3.7.2 3.7.4	34 32 14 & 47
Atlantic spotted dolphin	Unknown	No						
Spinner dolphin	Unknown	No						
Bottlenose dolphin	1. Mediterranean stock 2. Moray Firth stock 3.Cardigan Bay stock	Yes		✓	✓	AOR_006A_005 AOR_006A_006 AOR_006A_008 AOR_006A_012	3.7.1 3.7.1 3.7.1 3.7.4	18 33 6 14 & 47
Cuvier's beaked whale	1. Mediterranean stock	No						

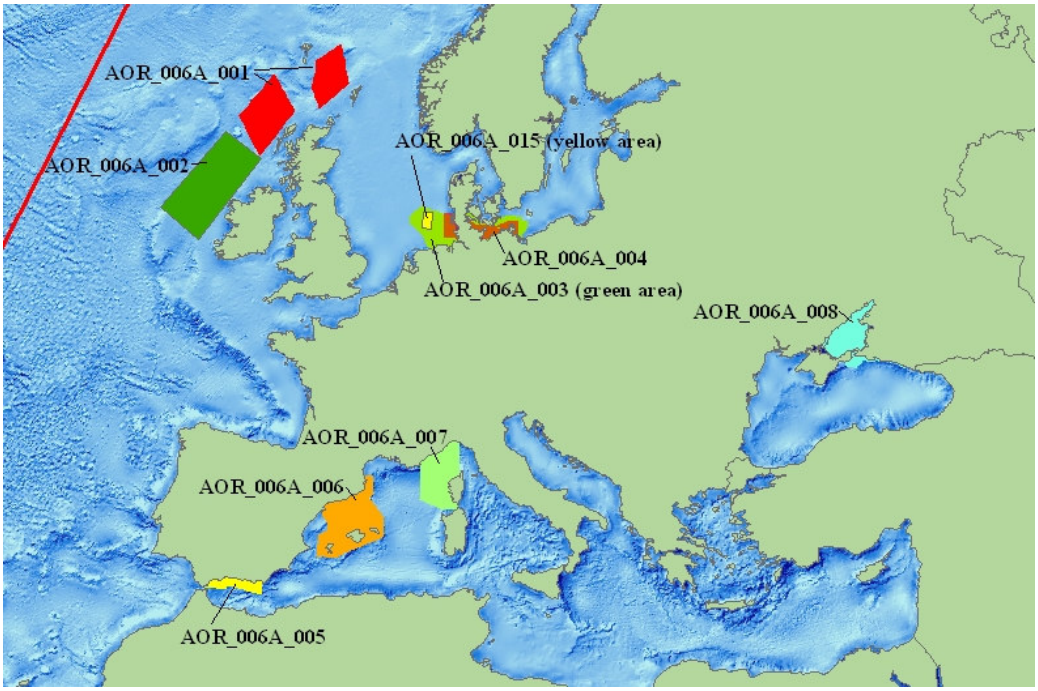

Figure 3.7.1: Small scale cetacean surveys conducted in European waters.

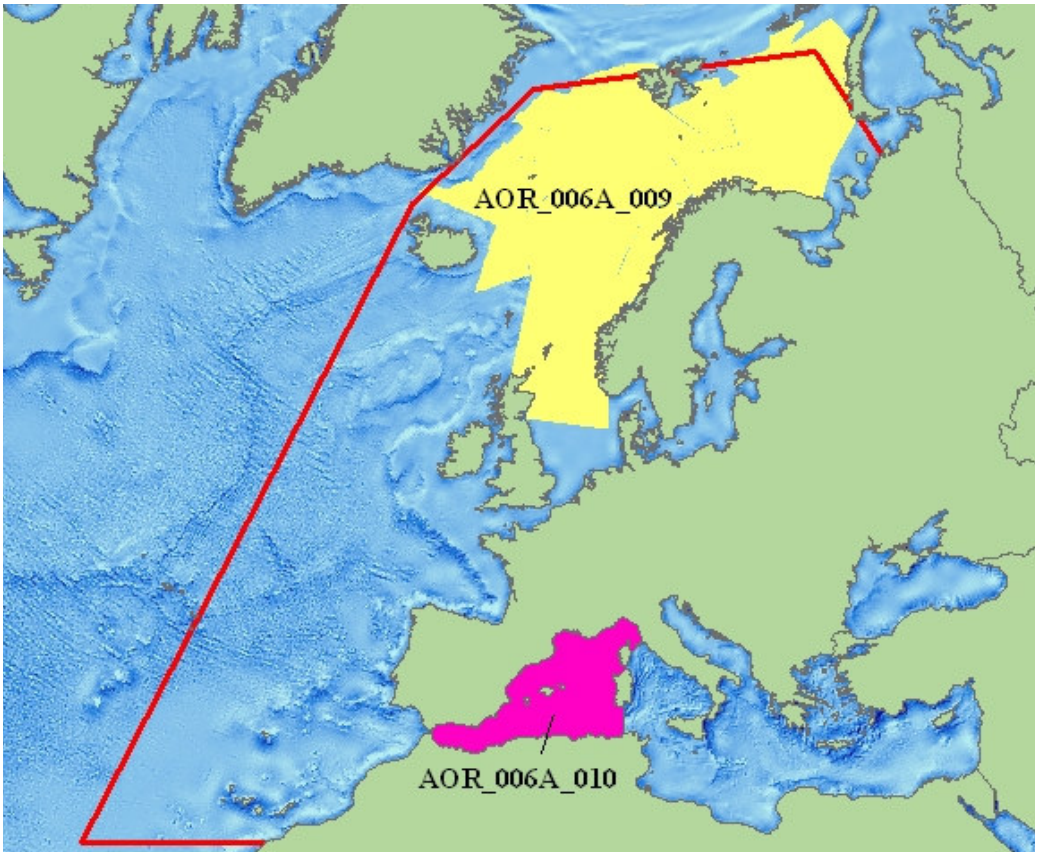

Figure 3.7.2: Large scale cetacean surveys conducted in European waters.

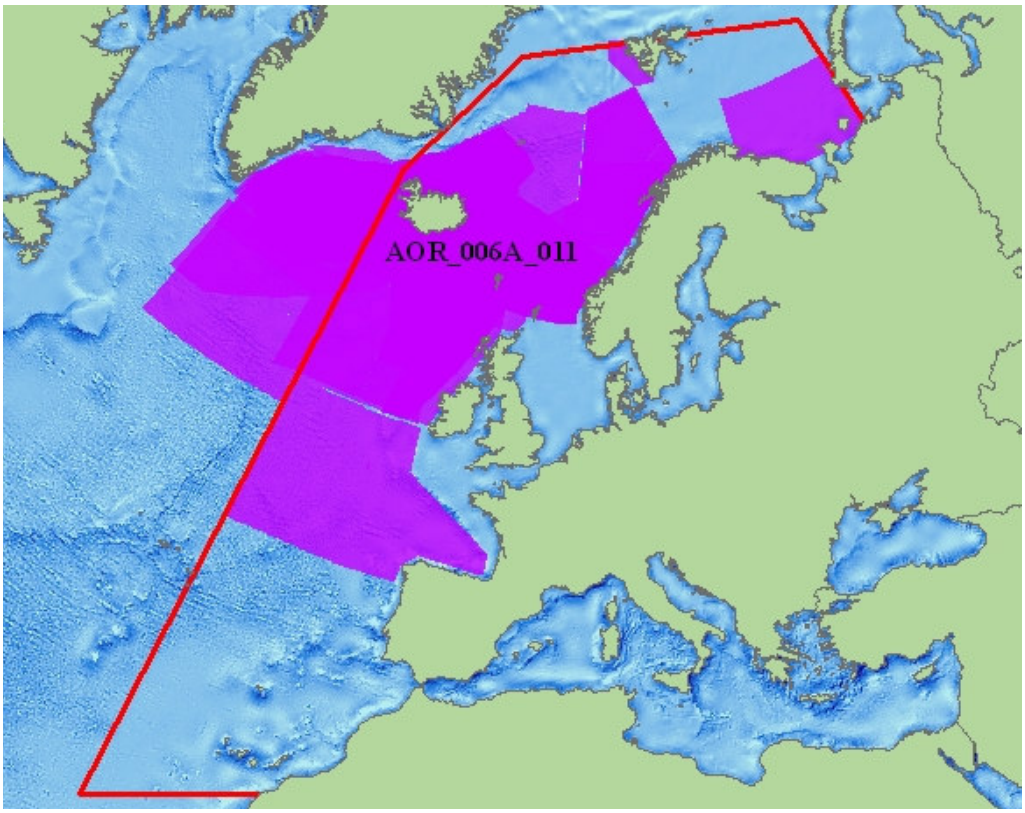

Figure 3.7.3: Survey area covered by North Atlantic Sighting Surveys.

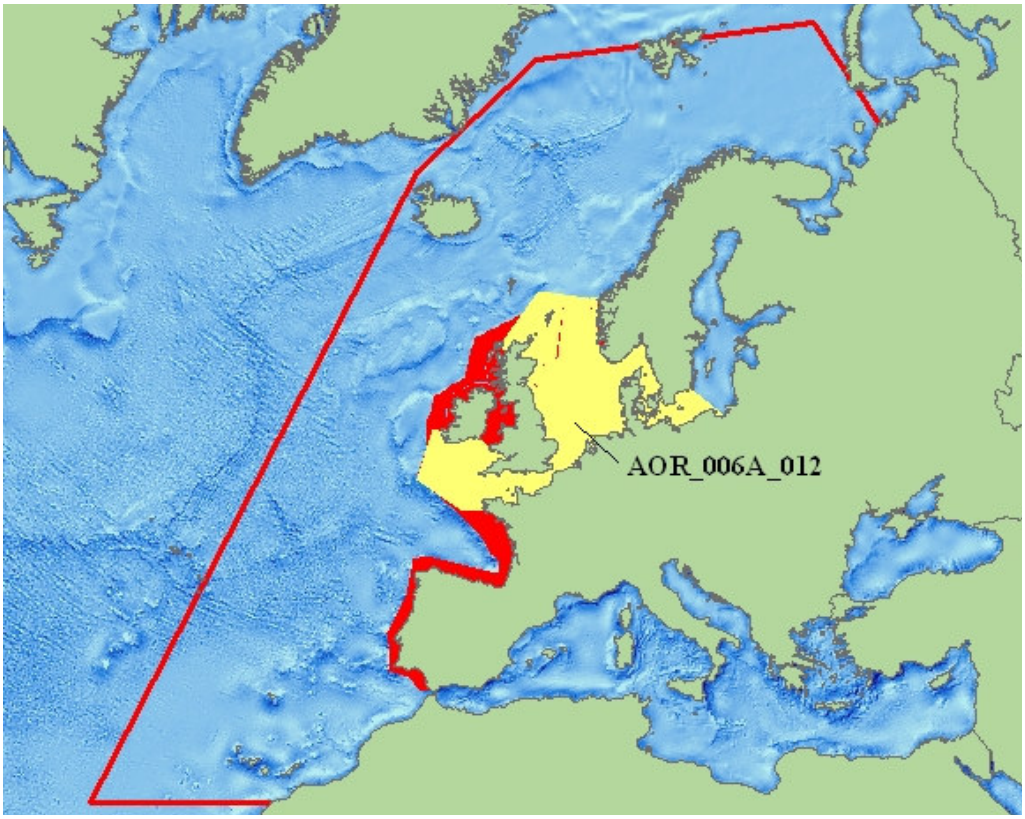

Figure 3.7.4: Area surveyed by SCANS (yellow area) and SCANS II (yellow and red areas).

3.7.1 Summary of available data for AOR 6A

AOR6A falls within the North east Atlantic Ocean, the North Sea, Barents Sea and the Mediterranean and thirty-eight cetacean species were predicted to be present within its boundaries. Of these thirty-eight species, eighteen have associated stock information (Table 3.7.1).

Abundance information exists for sixteen species in AOR6A (Table 3.7.1; Figures 3.7.1-3.7.4). However, due to data from multiple research groups, the number of repeat surveys undertaken in the same survey area is variable between species.

3.8 Area of Relevance 6B: Sakhalin, Russia

Area of relevance 6B covers Sakhalin Island and the Sea of Okhotsk.

Table 3.8.1: Summary table of the abundance data available for cetaceans in AOR_006B

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. North Pacific subspecies 2. O stock	Yes	✓	✓		AOR_006B_001	3.8.1	12
Sei whale	1. Northern hemisphere subspecies	No						
North Pacific right whale	1. Western North Pacific stock	No						
Blue whale	1. Western North Pacific stock	No						
Bowhead whale	1. Okhotsk Sea stock	No						
Fin whale	1. Northern hemisphere subspecies	No						
Baird's beaked whale	Unknown	No						
Beluga or white whale	Unknown	No						
Gray whale	1. Western North Pacific stock	No			✓	AOR_006B_002	3.8.1	60
Pacific white-sided dolphin	1. North Pacific stock	No						
Hubb's beaked whale	Unknown	No						
Humpback whale	1. Western North Pacific stock	No						
Stejneger's beaked whale	Unknown	No						
Killer whale	Unknown	No						
Dall's porpoise	Unknown	No						
Sperm whale	1. North Pacific stock	No						
Harbour porpoise	1. Western North Pacific stock	No						

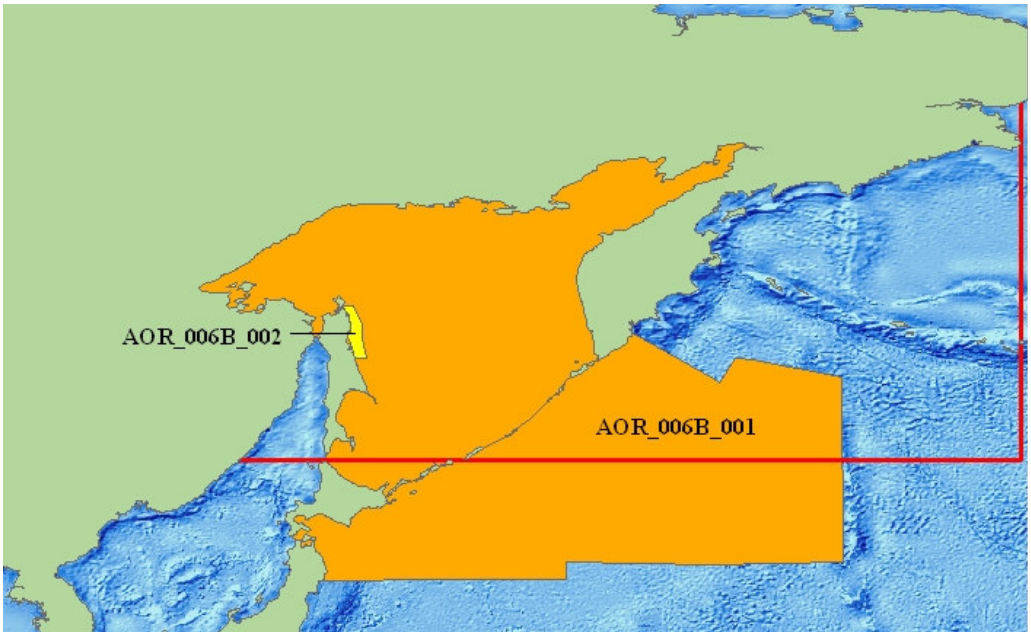

Figure 3.8.1: Cetacean surveys conducted in the Sakhalin Island region

3.8.1 Summary of available data for AOR 6B

AOR6B lies within the Sea of Okhotsk and seventeen cetacean species were predicted to be present within its boundaries. Of these seventeen species, eleven; the common Minke whale; Fin whale; Humpback whale; Blue whale, Sei whale, North Pacific Right whale, Bowhead whale, grey whale, sperm whale, Pacific white-sided dolphin and harbour porpoise have associated stock/ subspecies information (Table 3.8.1). This information is to the resolution of North Pacific subspecies or North Pacific stocks for everything but common Minke whales and Bowhead whale, where smaller stock units have been identified.
Abundance information exists for two species in AOR6B. A ship-based line transect survey was used to estimate abundance of Minke whales in the Sea of Okhotsk for 1989 and 1990 (AOR _006B_001; Buckland *et al.,* 1992a) (Figure 3.8.1). Grey whale abundance was estimated using data gathered during aerial surveys flown between 2001 and 2003. Results of surveys in all years indicated gray whales occurred in predominantly two areas, (1) adjacent to Piltun Bay and (2) offshore from Chayvo Bay, and the density of grey whales within the survey area was calculated (AOR_006B_002; Meier *et al*., 2007) (Figure 3.8.1).

The references described above contain insufficient abundance estimates to allow comprehensive trend analysis.

3.9 Area of Relevance 7: Brazil, South America

The boundary of AOR 7 was determined by the country boundary of Brazil, so the entire coastline of Brazil falls within this AOR.

Table 3.9.1: Summary table of the abundance data available for cetaceans in AOR_007

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Common minke whale	1. Dwarf subspecies	No						
Southern right whale	Unknown	No						
Antarctic minke whale	Unknown	No						
Sei whale	1. Southern hemisphere subspecies	No						
Bryde's whale	Unknown	No						
Blue whale	1. Antarctic subspecies (*B. m. intermedia*)	No						
Fin whale	1. Southern hemisphere subspecies	No						
Pygmy right whale	Unknown	No						
Long-beaked common dolphin	Unknown	No						
Short-beaked common dolphin	Unknown	No						
Pygmy killer whale	Unknown	No						
Short-finned pilot whale	Unknown	No						
Risso's dolphin	Unknown	No						
Southern bottlenose whale	Unknown	No						
Pygmy sperm whale	Unknown	No						
Dwarf sperm whale	Unknown	No						
Fraser's dolphin	Unknown	No						
Dusky dolphin	1. South American subpopulation, possibly with	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
	eastern form							
Southern right whale dolphin	Unknown	No						
Andrews' beaked whale	Unknown	No						
Blainville's beaked whale	Unknown	No						
Gervais' beaked whale	Unknown	No						
Gray's beaked whale	Unknown	No						
Hector's beaked whale	Unknown	No						
Strap-toothed whale	Unknown	No						
True's beaked whale	Unknown	No						
Humpback whale	1. Breeding stock A	Yes		✓	✓	AOR_007_001 AOR_007_003	3.9.1 3.9.1	87 52
Killer whale	Unknown	No						
Melon-headed whale	Unknown	No						
Spectacled porpoise	Unknown	No						
Sperm whale	Unknown	No						
Burmeister's porpoise	1. Atlantic subpopulation	No						
Franciscana	1. Northern stock 2. Southern (Rio Grande do Sol-Uruguay) stock	Yes			✓	AOR_007_002	3.9.1	29
False killer whale	Unknown	No						
Tucuxi	1. Freshwater form 2. Marine form	No						

Common name	Are there known to be multiple stocks in this area?	Abundance data	Abundance estimates from			Survey ID	Figure	Reference
			1980's	1990's	2000's			
Pantropical spotted dolphin	Unknown	No						
Rough-toothed dolphin	Unknown	No						
Clymene dolphin	Unknown	No						
Striped dolphin	Unknown	No						
Atlantic spotted dolphin	Unknown	No						
Spinner dolphin	1. Gray's (*S. l. longirostris*) type	No						
Bottlenose dolphin	Unknown	No						
Cuvier's beaked whale	Unknown	No						

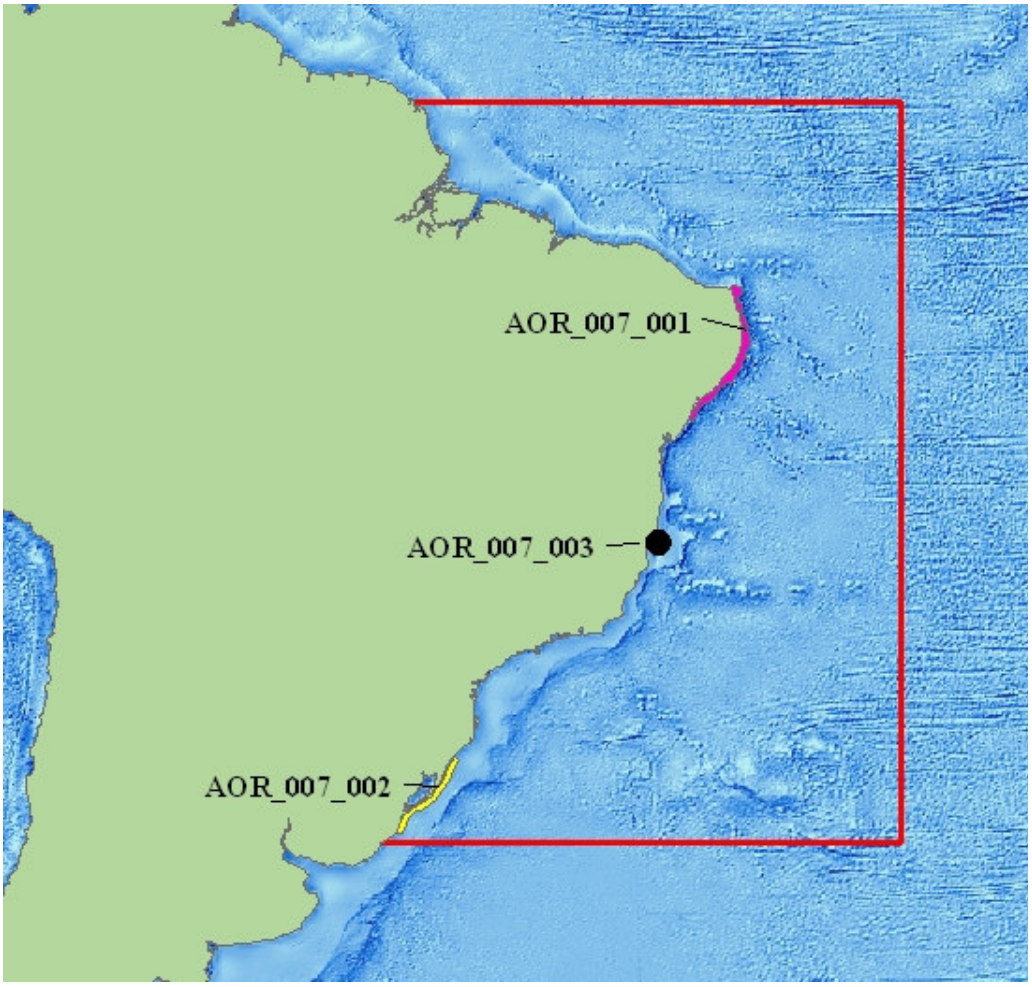

Figure 3.9.1: Cetacean surveys conducted off the coast of Brazil, South America

3.9.1 Summary of available data for AOR 7

AOR7 lies within the Western South Atlantic Ocean and forty-three cetacean species were predicted to be present within its boundaries. Of these forty-three species, ten; the common Minke whale, Fin whale, Humpback whale, Blue whale, Sei whale, Spinner dolphin, Tucuxi, Franciscana, Burmeister's porpoise and Dusky dolphin have associated stock/ subspecies information (Table 3.9.1). This information is to subspecies resolution for everything but the Humpback whale, Fransicana and Tucuxi. For the humpback whale, breeding stock information is known, as breeding stock A is shown to migrate to breed off Brazil. For the Tucuxi and Fransicana, information about forms and stocks respectively is given.

Abundance information exists for two species in AOR7. An aerial line-transect survey was used to estimate abundance of Fransicana (AOR_007_002; Danilewicz *et al*., 2006 unpublished). Humpback whale abundance was estimated from a photo-identification study in 1995 on the Abrolhos Bank, Brazil (AOR_007_003; Kinas and Bethlem, 1998) (Figure 3.9.1) and from a ship-based line transect survey during 2000 (AOR_007_001; Zerbini *et al*., 2004) (Figure 3.9.1). Neither of these study areas cover the entire breeding grounds of humpback

whales off Brazil, so the number of whales using this area is likely to be larger than indicated in these studies

It is impossible to look for population trends of humpback whales with the existing data, as estimates exist for different areas and hence there are no comparable figures for trend analysis. For the Fransicana, the data is currently unpublished and not available for analysis.

4 Overall Summary and Next Step

It is clear from this review that the level of information on cetaceans stocks and trends is highly variable between the outlined Areas of Relevance. In terms of stock information, areas 4, 5A and 5B have the best resolution of information, but for most species in other areas, stock information is unknown. In terms of abundance information, to look at how population numbers have changed over time, much variation is also evident. For areas such as 5A, 5B and 6A the highest proportion of systematic line transect surveys have been conducted. These surveys allow abundance estimates to be assigned to areas and density information to be calculated. This density information can be used to assess how species density may change over years if multiple surveys are conducted within the same areas. Other survey types such as counts from shore or photo-identification surveys can be used to look at numbers of animals passing known points or using specific coastal areas, but do not provide as robust a density estimate for a whole stock or species.

The data that forms the basis of this review has outlined that there is some potential for analysis of trend data for stocks within multiple AORs, but how robust these estimates are and if they can be used statistically to detect changes in trends cannot be determined until the completion of task 2. Additionally, this review has used primary peer-reviewed literature focussing on large-scale line transect surveys. This has enabled the review to outline the main data sources that exist within each AOR, but it should be noted that other sources of information, especially smaller scale and in the grey literature will exist. Incorporation of this data may provide further potential for trend analysis, but this will be dependent on the resolution required in task 2.

5 Further analysis

To enable the most appropriate analysis in task 2, guidance from the JIP is needed as to which species and AORs are to be taken forward for consideration in task 2. To enable task 2 to run on schedule, this guidance must be given within 2 weeks of submission of this report.

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6.2 Survey data annotated bibliography

1. Bannister, J. (2001) Status of southern right whales (*Eubalaena australis*) off Australia. Journal of Cetacean Research and Management (Special Issue) **2**, 103-110

The author reports the results of counts of southern right whales from aerial surveys conducted off the south coast of Australia. The aerial surveys began in 1976, but in 1983 the transect lines were extended further to the east and these results, from 1983 to 1997, are given. At least three flights were attempted each year, and timed to coincide with the peak in abundance of southern right whales on their breeding grounds. Data gathered in two of the years are not comparable with other years, and abundance has not been estimated for any of the years.

2. Barlow, J. (2003) Preliminary estimates of the abundance of cetaceans along the U.S. West Coast: 1991 – 2001. La Jolla, CA, USA, Southwest Fisheries Science Center, National Marine Fisheries Service: 31

This paper reports the results of a ship-based line transect survey conducted by the National Marine Fisheries Service to estimate cetacean abundance on the west coast of the US. The study area consisted of two strata; a Oregon-Washington stratum and a California stratum. The Oregon-Washington area was surveyed in 1996 and 2001 and abundance estimates were made for each of those years. The California stratum was surveyed in 1991, 1993, 1996 and 2001. The amount of effort was low in 1993 so data from 1991 and 1993 were pooled for analysis. An estimate of g(0) was taken from other studies using similar survey methods. An indirect calibration method, based on calibration from aerial photos, was used to modify group size estimates. Multiple covariates were tried in Generalised Additive Models to investigate the detection function, and the program DISTANCE was used to estimate abundance using line-transect methods.

3. Barlow, J. and B. L. Taylor. (2005) Estimates of sperm whale abundance in the northeastern temperate Pacific from a combined acoustic and visual survey. Marine Mammal Science **21**(3), 429-445

In spring 1997 a ship-based dedicated visual and acoustic survey was conducted to estimate sperm whale abundance during the breeding season in the eastern temperate North Pacific. Both the acoustic and visual data were analysed using line-transect methods and abundance estimates resulting from the two survey methods were not significantly different. g(0) was assumed to be 1 for the acoustic surveys and g(0)=0.87 was used for visual data analysis based on a previous sperm whale study.

4. Best, P. B., Rademeyer, R. A., Burton, C., Ljungblad, D., Sekigughi, K., Shimada, H., Thiele, D., Reeb, D. and D. S. Butterworth. (2003) The abundance of blue whales on the Madagascan Plateau, December 1996. Journal of Cetacean Research and Management **5**(3), 253-260

A ship-based line-transect survey, using two research vessels, survey for blue whales over 53,5833 km^2 of the Madagascan Plateau, in December 1996. The study area comprised northern and southern strata, that when combined give an abundance estimate of blue whales (mostly pygmy blue whales), for 1996 of 425 individuals. The probability of detecting whales on the trackline (g(0)) was assumed to be 1 and, after consideration of dive times and surfacing behaviour, this assumption was thought to be reasonable and would not cause bias in the abundance estimate. Using catch histories to derive the range of this species it was estimated that the study area covered approximately 1/3 of the range of this population.

5. Best, P. B., Sekiguchi, K. Rakotonirina, B. and A. Rossouw. (1996) The distribution and abundance of humpback whales off southern Madagascar, August-September 1994. Rep Int Whal Commn **46**, 323-331

A ship-based line transect survey for humpback whales, was conducted over a month in 1994, covering 26,893 km^2 on the south and east coast of Madagascar. The survey ran from the coast to the 182 metre isobath. The probability of detecting whales on the trackline was assumed to be 1, so the abundance estimates are likely underestimates of the true abundance. There is thought to be little mixing between the humpback whale concentrations off Mozambique and Madagascar despite their proximity (Best *et al*., 1994). This reference gives an abundance estimate for humpback whales from 1994

6. Birkun, A., Glazov, D., Krivokhizhin, S. and L. Mukhametov. (2003) Distribution and abundance estimates of cetaceans in the Azov Sea, Kerch Strait and northeastern shelf area of the Black Sea: results of aerial surveys in July 2001 and August 2002. (SC/55/SM15) International Whaling Commission - Scientific Committee Meeting, Berlin, Germany, (unpublished).

Aerial surveys were completed in July 2001 and August 2002. In 2001 the entire Azov Sea and Kerch Strait were covered by the survey, while in 2002 Kerch Strait, the southern portion of the Azov Sea and part of the Black Sea shelf area was surveyed. The abundance of bottlenose dolphins and harbour porpoise was estimated and no corrections for visibility or perception bias were made.

7. Blaylock, R.A., Hain, J. W., Hansen, L. J., Palka, D. L. and G. T. Waring. (1995) U.S.Atlantic and Gulf of Mexico stock assessments, U.S. Department of Commerce**:** 211.

This document contains the stock assessment reports for the U.S. Atlantic and Gulf of Mexico marine mammal stocks under NMFS jurisdiction. Marine mammal species which are under the management jurisdiction of the USFWS are not included in this report. Each stock assessment includes a stock definition and extent of geographical range, a minimum population estimate, current population trends, current status of stock, current and maximum net productivity rates, optimum sustainable population levels and allowable removal levels, estimates of annual human-caused mortality

and serious injury through interactions with commercial fisheries and subsistence hunters, and key references.

8. Borchers, D. L, McCracken, M., Gunnlaugsson, Th and M. L. Burt. (1997) Estimates of minke whale abundance from the 1987 and 1995 NASS aerial surveys (SC/5/AE/2). Tromsø and Norway, NAMMCO SC North Atlantic Marine Mammal Commission (NAMMCO)**:** 13.

North Atlantic Sighting Surveys (NASS) were completed by a number of national agencies in 1987, 1989, 1995 and 2001 in order to estimate the abundance of a number of cetacean species. Data from the 1987 and 1995 aerial NASS surveys were used by the authors to investigate the abundance and distribution of minke whales in the North Atlantic. The probability of detecting minke whales on the trackline could not be estimated, so the abundance estimates were not corrected for those animals that were missed.

9. Buckland, S. T., Bloch, D., Cattanach, K. L., Gunnlaugsson, Th., Hoydal, K., Lens, S. and J. Sigurjónsson. (1993) Distribution and abundance of long-finned pilot whales in the North Atlantic, estimated from NASS-87 and NASS-89 data. Rep. Int. Whal. Commn (Special Issue **14**), 33-49

North Atlantic Sighting Surveys (NASS) were completed by a number of national agencies in 1987, 1989, 1995 and 2001 in order to estimate the abundance of a number of cetacean species. Data from the 1987 and 1989 NASS surveys were used by the authors to investigate the abundance and distribution of pilot whales in the North Atlantic. The 1989 survey covered a wider area, extending further to the south, than the 1987 survey. The abundance estimates found during this study could be biased for a number of regions. There were inconsistencies with how group size was recorded and no correction was made for pilot whales that were diving as the vessel passed and weren't available for detection. The tendency for pilot whales to form large, loose aggregations may have resulted in pilot whales being recorded as closer to the trackline than the centre of the aggregation, resulting in abundance estimates being positively biased. Also these large aggregations were more likely to be detected than small groups, causing further positive bias.

10. Buckland, S. T. and J. M. Breiwick. (2002) Estimated trends in abundance of eastern Pacific gray whales from shore counts (1967/68 to 1995/96). Journal of Cetacean Research and Management **4**(1): 41-48

Shore counts of migrating gray whales were made during their southbound migration down the east coast of America between 1967/68 and 1995/96 to estimate trends in abundance. From 1986 the counts were made from two independent observation platforms to allow the proportion of whales detected by a single observer to be estimated, and the abundance estimate to be corrected. Between 1967/68 and 1995/96 the population is estimated to have increased at a rate of 2.5% per annum.

11. Buckland, S. T., Cattanach, K. L. and Th. Gunnlaugsson. (1992) Fin whale abundance in the North Atlantic, estimated from Icelandic and Faroese NASS-87 and NASS-89 data (SC/F91/F2). Rep. Int. Whal. Commn. **42**, 645-651

North Atlantic Sighting Surveys (NASS) were completed by a number of national agencies in 1987, 1989, 1995 and 2001. Data gathered by Icelandic and Faroese survey vessels during the 1987 and 1989 NASS surveys were used to estimate abundance of fin whales. The majority of the effort fell within the area of the East Greenland/Iceland management stock of fin whales, but some fell east of the stock boundary. Data collected from within and outside the stock boundary were analysed separately, although the data from outside the stock boundary represents only a fraction of the eastern stock. Data were stratified by group size to avoid bias caused by large groups having a higher probability of detection.

12. Buckland, S. T., Cattanach, K. L. and T. Miyashita. (1992a) Minke whale abundance in the northwest Pacific and the Okhotsk Sea, estimated from 1989 and 1990 Sighting surveys. Rep. Int. Whal. Commn **42**, 387-382

Shipboard line transect surveys were conducted in the Okhotsk Sea and northwest Pacific in the summers of 1989 and 1990 to estimate minke whale abundance in the region. The authors acknowledge that the abundance estimates of 19,209 for the Okhotsk Sea and 5,841 for the northwest Pacific are likely to be negatively biased as the probability of detecting minke whales on the trackline was assumed to be one.

13. Buckland, S. T., Cattanach, K. L. and R. C. Hobbs. (1993) Abundance estimates of Pacific White-Sided Dolphin, Northern Right Whale Dolphin, Dall's Porpoise and Northern Fur Seal in the North Pacific, 1987-1990. International North Pacific Fisheries Commission Bulletin **53**(3), 387-407

The authors report on a ship-based, line transect survey extending from Japan and southeast China to the northwest coast of America and from 70° N to 20°N. The survey area covered 13,638,245 km^2 and data were gathered primarily over the summer months between 1987 and 1990. Distance sampling was used for analysis and the probability of detecting animals on the trackline was assumed to be 1. Abundance estimates were corrected for bias resulting from size-biased sampling of schools. Movement of the animals prior to detection was thought to be problem resulting in overestimation, particularly for pacific white-sided dolphins which were thought to be attracted to the vessel. Abundance estimates for three species; Pacific white-sided dolphin; Northern Right Whale dolphin and Dall's porpoise from 1987-1990 are given.

14. Burt, M. L, Borchers, D. L. and F. Samarra. (2006) Abundance estimates from SCANS-II: stratified analysis. St. Andrews, UK, RUWPA, University of St Andrews**:** 18.

The authors report on data from the SCANS-II survey to estimate abundance of the harbor porpoise, bottlenose dolphin and common dolphin in the North Sea. The study region was divided into 17 stratum surveyed by seven ships and three aircraft.

The analysis of shipboard data was based on mark-recapture line-transect methods and the analysis of the aerial data was based on the method of Hiby and Lovell. Estimates were obtained for each stratum and the whole study region.

15. O'Cadhla, O, Borchers, D. Burt, L. M. and E. Rogan. (2001) Summer distribution and abundance of cetaceans in western Irish waters and the Rockall Trough. SC/53/O15. International Whaling Commission - Scientific Committee Meeting, (unpublished).

A ship-based survey was completed in the summer of 2000 to investigate the distribution and abundance of cetaceans in western Irish waters, including the Rockall Trough. Double-platform methods were used and g(0) was estimated. Fifteen identifiable species were recorded during the survey, but only common dolphins and Atlantic white-sided dolphins were encountered frequently enough to allow estimation of their abundance.

16. Calambokidis, J. and J. Barlow. (2004) Abundance of Blue and Humpback whales in the eastern North Pacific estimated by capture-recapture and line-transect methods. Marine Mammal Science **20**(1), 63-85

The authors analysed data from 1991, 1993 and 1996 from a ship-based line transect survey conducted by the National Marine Fisheries Service to estimate the abundance of humpback and blue whales in the study area. The detection function was modelled using DISTANCE and g(0) was estimated from independent observer data.

17. Calambokidis, J., Steiger, G. H., Ellifrit, D. K., Troutman, B. L. and C. E. Bowlby. (2004b) Distribution and abundance of humpback whales (*Megaptera novaeangliae*) and other marine mammals off the northern Washington coast. Fishery Bulletin **102**(4), 563–580

Ship based line transect surveys were conducted over two weeks in the summer of 1995, 1996, 1997, 1998, 2000 and 2002. Abundance was estimated for each of the six survey years for Humpback whales and Dall's porpoise using line-transect analysis. Abundance is likely to have been underestimated given that no adjustments were made for animals missed on the trackline. Gray and minke whales, Harbour porpoise, pacific white-sided dolphin, northern right-whale dolphin, risso's dolphin and killer whales were also recorded during the survey but were not seen frequently enough to allow abundance estimation.

18. Cañadas, A. and P. S. Hammond. (2006) Model-based abundance estimates for bottlenose dolphins off southern Spain: implications for conservation and management. Journal of Cetacean Research and Management **8**(1), 13-27

Non-systematic line transects were completed off the coast of southern Spain between 1992 and 2003 in the eastern part of the study area and between 2000 and 2003 in the western region of the study area and analysed using spatial modelling methods. Distance sampling methods were used to estimate the probability of

detecting bottlenose dolphins and spatial modelling methods were then used to estimate abundance as a function of physical and environmental data.

19. Carretta, J. V. (2003) Preliminary estimates of harbor porpoise abundance in California from 1997 and 1999 aerial surveys. La Jolla, California, Southwest Fisheries Science Center.

This report outlines preliminary results and the author's state reference to the results should not be made and any information should be taken from formal publications.

20. Carretta, J. V. and K. A. Forney. (2004) Preliminary estimates of harbour porpoise abundance in California from 1999 and 2002 aerial surveys. La Jolla, California, Southwest Fisheries Science Center.

This report outlines preliminary results and the author's state reference to the results should not be made and any information should be taken from formal publications.

21. Carretta, J. V., Taylor, B. L. and S. J. Chivers. (2001) Abundance and depth distribution of harbor porpoise (*Phocoena phocoena*) in northern California determined from a 1995 ship survey. Fishery Bulletin **99**(1), 29-39

Following a dedicated ship-based line-transect survey in November 1995 the abundance of the northern California harbour porpoise stock was estimated by the authors. The results differ significantly from abundance estimates made by the US National Marine Fisheries Service based on aerial survey data, possibly as a result of seasonal movements of harbour porpoise, lack of ship-based effort or underestimation of the proportion of porpoise missed on the trackline. Overestimation of mean group size resulting from the increased probability of detecting large groups further from the trackline was accounted for by using a regression-based estimate of mean group size. The program DISTANCE was used to estimate abundance.

22. Cattanach, K. L., Sigurjónsson, J., Buckland, S. T. and Th. Gunnlaugsson. (1993) Sei whale abundance in the North Atlantic, estimated from NASS-97 and NASS-89 data (SC/44/NAB10). Rep. Int. Whal. Commn. **43**, 315-321

Data gathered during the 1987 and 1989 North Atlantic Sighting Surveys were used to estimate sei whale abundance in the North Atlantic using standard line transect analysis. The abundance estimate resulting from the 1989 survey was remarkably higher than that resulting from the 1987 survey, possibly because the survey area in 1989 incorporated waters further to the south of the previous survey area. The probability of detecting sei whales on the trackline was assumed to be 1 and detection probability was proven to be independent of school size. This survey is thought to have not covered the entire stock area.

23. Chilvers, B. L. and P. J. Corkeron (2003) Abundance of Indo-Pacific bottlenose dolphins, Tursiops aduncus, off Point Lookout, Queensland, Australia. Marine Mammal Science **19**(1), 85-95

Photo-identification data were gathered on the Indo-Pacific bottlenose dolphins off North Stradbroke Island, Queensland during the winters of 1998 and 1999. The population was assumed to be closed and the program CAPTURE was used to estimate population size using mark-recapture techniques. Over the duration of the study 581 different individuals were recognised and population size was estimated to be 861 in 1998 and 895 in 1999. There was no significant difference between the abundance estimates from each year.

24. Christensen, I., Haug, T. and N. Øien. (1992) Seasonal distribution, exploitation and present abundance of stocks of large baleen whales (Mysticeti) and sperm whales (*Physeter macrocephalus*) in Norwegian and adjacent waters. ICES Journal of marine science **49**, 341-355

Large-scale ship-based line transect surveys were completed by the Institute of Marine Research, Norway in the North Atlantic in 1989, 1995, 1996, 1997, 1998, 1999, 2000 and 2001. The target species during these surveys was the minke whale, but abundance was estimated for other species encountered too. The authors used data gathered during the 1989 survey to estimate abundance of fin and humpback whales. During analysis, it was assumed that all cetaceans on the trackline were seen (i.e. that g(0)=1), resulting in an estimate of 2,245 fin whales in the survey area. At the time, the IWC considered there to be two stocks of fin whales present in the survey area. The survey data was stratified to correspond with the presumed stock boundary as closely as possible, resulting in an estimate of 1,900 fin whales from the North Norway Stock and 340 fin whales from the West Norway- Faroe Islands stock. Approximately 1000 humpback whales were estimated to be present in the Norwegian and Barents Seas.

25. Clapham, P., Barlow, J., Bessinger, M., Cole, T., Mattila, D., Pace, R., Palka, D., Robbins, J. and R. Seton. (2003) Abundance and demographic parameters of humpback whales from the Gulf of Maine, and stock definition relative to the Scotian Shelf. Journal of Cetacean Research and Management **5**(1), 13-22

Aerial and ship-based line transect surveys were conducted in waters from the Gulf of Maine and Georges Bank to the mouth of the Gulf of St Lawrence in the summer of 1999. During the ship-based survey there were two independent teams of observers, which allowed estimation of the probability of detecting whales on the trackline. Aerial data were not corrected for animals missed on the trackline, resulting in a possible negative bias in abundance estimates based on these data.

26. Claridge, D. E, Balcomb, K. C. and J. W. Durban. (2000) Preliminary report of the occurrence, distribution and abundance of Ziphiid and Balaenopterid whales in Northwest providence Channel, the northern Bahamas (unpublished draft report), Bahamas Marine Mammal Survey**:** 21.

Photo-identification data gathered between 1997 and 1999 from boat-based surveys were used to estimate abundance of Blainville's beaked whale in the northern Bahamas. Photos of the dorsal fin of each individual were taken for identification purposes, while photos of the head and thoracic region allowed age and sex classes to be distinguished.

27. Davis, R. W., Evans, W. E. and B. Wursig. (2000). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, Abundance and habitat associations. Volume II Technical Report. U.S. Department of the Interior, Geological Survey, Biological Resources Division, USGS/BRD/CR-1999-0006 and Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 2000-003 346pp.

This set of surveys was designed to provide seasonal information on the distribution of cetaceans in the shelf and slope waters of the northeastern Gulf of Mexico. Aerial surveys were conducted in the summers of 1996 and 1997, and in the winters of 1997 and 1998. Ship based surveys were completed in the summers of 1996 and 1997.

28. Davis, R. W. and Fargion, G. S. (Eds) 1996. Distribution and abundance of cetaceans in the north-central and western Gulf of Mexico: final report, volume II: technical report. OCS Study MMS 96-0027. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana. 357 pp.

The authors report on a 3.75 year project (1 October 1991 through 15 July 1995), in a study area bounded by the Florida-Alabama border, the Texas-Mexico border, and the 100- and 2,000-m isobaths. Seasonal, line transect aerial surveys, visual surveys from ships and acoustic surveys using a linear hydrophone array towed behind a visual survey ship were carried out to determine the distribution and abundance of cetaceans in the study area. A total of 351 cetacean groups representing 17 species were sighted on-effort during eight aerial surveys and a total of 683 shipboard marine mammal sightings of at least 19 species were made during the eleven ship surveys.

29. Danilewicz *et al*. (2006) Unpublished.

30. Findlay, K., Meyer, M., Elwen, S., Kotze, D., Johnson, R., Truter, P., Uamusse, C., Sitoe, S., Wilke, C., Kerwath, S., Swanson, S., Staverees, L. and J. Van der Westhuizen. (2004) Distribution and abundance of humpback whales, *Megaptera novaeangliae*, off the coast of Mozambique, 2003. Report to the International Whaling Commission Scientific Committee SC/56/SH12

This ship-based line-transect survey for humpback whales was conducted over 13 days in 2003. The survey trackline ran between the 20 and 200 metre isobaths and covered a survey area of 86,128 km^2. The probability of detecting whales on the trackline was assumed to be 1, therefore the abundance estimate is likely to be underestimated. This survey occurred during the winter breeding season for

humpback whales, so this 2003 estimate of 5811 humpback whales will only apply during certain months.

31. Foote, A. D., Víkingsson, G., Øien, N., Bloch, D.Davis, C. G., Dunn, T. E., Harvey, P., Mandleberg, L., Whooley, P. and P. M. Thompson. (2007) Distribution and abundance of killer whales in the North East Atlantic. Paper SC/59/SM5 presented to the IWC Scientific Committee, June 2007, Anchorage, Alaska, USA. 10pp.

This paper presents a review and comparison of killer whale sightings from across the NE Atlantic from 1970-2007 with whaling catch data from 1938-1967. The authors estimate abundance using data from the NASS surveys from 1987, 1989, 1995 and 2001. The authors show high inter-annual variability in the estimates and that some populations of killer whales in the NE Atlantic do not follow the Norwegian Spring spawning herring migration.

32. Forcada, J., Aguilar, A., Hammond, P. S., Pastor, X. and R. Aguilar. (1994) Distribution and numbers of striped dolphins in the western Mediterranean Sea after the 1990 epizootic outbreak. Marine Mammal Science **10**(2), 137-150

Following an outbreak of the morbillivirus epizootic in 1990, the western Mediterranean was surveyed by boat to estimate the number of surviving striped dolphins. The survey lasted one month in 1991 and data were analysed using distance sampling techniques. The effect of school size on detection probability was accounted for by using the expected mean school size during analysis. The effect of movement of striped dolphins prior to detection could not be accounted for, but large detection distances suggested it wasn't a problem.

33. Forcada, J., Gazo, M., Aguilar, A., Gonzalvo, J. and M. Fernández-Contreras. (2004) Bottlenose dolphin abundance in the NW Mediterranean: addressing heterogeneity in distribution. Marine Ecology Progress Series **275**, 275-287

Aerial line transect surveys were flown over the Balearic Sea and Islands in the NE Mediterranean Sea to assess the abundance of coastal bottlenose dolphins. The study area included a putative subpopulation of bottlenose dolphins in the Balearic Islands, although the results suggested that the islands contain critical habitat for the species rather than supporting a subpopulation. Aerial surveys took place in 2001 and 2002. Small boat surveys were conducted in 2003 to gather data on dive duration of the bottlenose dolphins; this was used to estimate the availability of dolphins for detection at the surface during the aerial survey and the abundance estimate was modified accordingly.

34. Forcada, J., Notarbartolo di Sciara, G., and F. Fabbri. (1995) Abundance of fin whales and striped dolphins summering in the Corso-Ligurian Basin. Mammalia **59**(1), 127-140

A ship-based line transect survey was conducted in the western Ligurian Sea and in the offshore waters off western Corsica in August 1992. Absolute abundance of

striped dolphins and fin whales were estimated using standard line transect analysis methods.

35. Forney, K. A. (1999) The abundance of California harbor porpoise estimated from 1993–97 aerial line-transect surveys. La Jolla, California, Southwest Fisheries Science Center.

Aerial surveys were flown between 1993 and 1997, and data from these surveys were combined and used to estimate harbour porpoise abundance. The probability of detecting harbour porpoise on the trackline was taken from another study and used to adjust the abundance estimate accordingly.

36. Forney, K. A., Barlow, J. and J. V. Carretta. (1995) The abundance of cetaceans in California waters. Part II: Aerial surveys in winter and spring of 1991 and 1992. Fishery Bulletin **93**, 15-26

Aerial surveys were conducted in winter and spring of 1991 and 1992 and the data combined to estimate cetacean abundance during cold water conditions using standard line-transect methods. An independent observer within the plane reported sightings missed by the primary observers, allowing g(0) to be estimated (for the categories 'small cetaceans' and 'medium and large cetaceans') and the abundance estimates to be corrected for animals missed on the trackline. Availability bias was not corrected for so the abundance estimates are possibly negatively biased. Small and large school sizes were analysed separately in an attempt to reduce the upward bias caused by large groups having higher detection probability than small groups. It is not possible to identify common dolphins to species level during aerial surveys, so the two species were grouped together for the abundance estimate here.

37. Fulling, G. L., K. D. Mullin, et al. (2003). "Abundance and distribution of cetaceans in outer continental shelf waters of the U.S. Gulf of Mexico." Fishery Bulletin **101**: 923-932.

The authors report on cetacean surveys conducted during the fall ichthyoplankton surveys from 1998 to 2001. Two stratum, east and west were surveyed and from these surveys, the abundance and distribution of cetaceans in Outer Continental Shelf waters (20–200 m deep) of the U.S. Gulf Of Mexico are given. A combination of line transect and strip-transect methods were used to estimate abundance. Line-transect methods were used for sightings detected with 25 (binoculars, which constituted the majority of sightings (129/140). Strip-transect methods were used for the 11 sightings that were made without the 25× binoculars (naked-eye sightings). Abundance estimates are given for three species; Bottlenose dolphins (*Tursiops truncates)*; Atlantic spotted dolphins (*Stenella frontalis*) and rough-toothed dolphins (*Steno bredanensis*).

38. Gannier, A., Drouot, V. and J. C. Goold. (2002) Distribution and relative abundance of sperm whales in the Mediterranean Sea. Marine Ecology Progress Series **243**, 281-293

Field surveys were conducted in the Mediterranean for four consecutive summers (June to August), using visual and acoustic ship based methods to estimate sperm

whale abundance. For the analysis the Mediterranean Sea was divided into 6 regions, and relative abundances of sperm whales in each region were calculated.

39. Garrison, L.P., Swartz, S. L., Martinez, A., Burks, C. and J. Stamates. (2002) A marine mammal assessment survey of the Southeast US Continental Shelf: February - April 2002, U.S. Department of Commerce.

A boat-based line transect survey was undertaken to assess the winter abundance and distribution of cetaceans in shelf and inner slope waters in 2002. The survey was jointly conducted by NOAA Fisheries (National Oceanic and Atmospheric Administration) and the US Navy. An independent observer onboard the survey vessel had too few independent sightings to allow visibility bias to be accounted for, therefore the probability of detecting animals on the trackline was assumed to be 1.

40. George, J. C., Zeh, J., Suydam, R. and C. Clark. (2004) Abundance and population trend (1978-2001) of western Arctic bowhead whales surveyed near Barrow, Alaska. Marine Mammal Science **20**(4), 755-773

In 2001 a visual survey was conducted from Barrow, Alaska during the spring migration of the bowhead whale. Concurrently, the area was surveyed using passive acoustic equipment, and the resultant data were used to calculate the detection probability during the visual survey. The abundance estimate was then corrected accordingly. These data are not currently contained within the database as there's no area associated with the abundance estimate.

41. Gerrodette, T. and J. Forcada. (2002a) Estimates of abundance of northeastern offshore spotted, coastal spotted, and eastern spinner dolphins in the eastern tropical Pacific Ocean. La Jolla, California, Southwest Fisheries Science Center**:** 41.

The authors report on abundance estimates from stratified large-scale line-transect surveys carried out with oceanographic research vessels in the eastern tropical Pacific Ocean in 12 different years between 1979 and 2000. The surveys were designed to estimate the abundance of northeastern offshore spotted (*Stenella attenuata*) and eastern spinner (S*. longirostris orientalis*) dolphins. In the most recent surveys in 1998-2000, estimates of the coastal subspecies of spotted dolphins (*S. attenuata graffmani*) were also possible. Estimates of dolphin abundance for each stock were based on modified line-transect methods, using covariates to model the detection process and group size. For the 21-year period, the low and high total estimates of abundance are given Weighted linear and quadratic regressions of abundance against time were not statistically significant for either northeastern offshore spotted or eastern spinner dolphins.

42. Gerrodette, T. and J. Forcada. (2002b) Estimates of abundance of western/southern spotted, whitebelly spinner, striped and common dolphins, and pilot, sperm and Bryde's whales in the Eastern Tropical Pacific Ocean. La Jolla, California, Southwest Fisheries Science Center**:** 24.

The authors report on abundance estimates from large-scale line-transect surveys carried out with oceanographic research vessels in 8 different years between 1986 and 2000. For the 15-year period, the low and high total estimates of abundance for the western/southern stock of offshore spotted dolphins (*Stenella attenuata*), the whitebelly stock of spinner dolphins (*S. longirostris*), striped dolphins (*S. coeruleoalba*), three stocks of short-beaked common dolphins (*Delphinus delphis*), short-finned pilot whales (*Globicephala macrorhynchus*), sperm whales (*Physeter macrocephalus*) and Bryde's whales (*Balaenoptera edeni*) in the eastern tropical Pacific Ocean are given. Most species/stocks did not show significant changes between the 1986-1990 and 1998-2000 periods. Pilot and Bryde's whales were significantly higher in the later period, but interpretation of changes in abundance are confounded by several factors, including movement in and out of the study area and a larger survey area in the later period.

43. Gerrodette, T., Watters, G. and J. Forcada. (2005) Preliminary estimates of 2003 dolphin abundance in the Eastern Tropical Pacific. La Jolla, California, Southwest Fisheries Science Center**:** 28.

This report outlines preliminary results and the author's state reference to the results should not be made and any information should be taken from formal publications.

44. Gowans, S., Whitehead, H. Arch, J. K. and S. K. Hooker. (2000) Population size and residency patterns of northern bottlenose whales (*Hyperoodon ampullatus*) using the Gully, Nova Scotia. Journal of Cetacean Research and Management **2**(3), 201-210

Photographs of the dorsal fin and flanks of northern bottlenose whales were gathered from ship-based surveys conducted between 1988 and 1999. Effort was highly variable each year, ranging from a few days in some summers to a few months in others. Photos from all years were combined and used to estimate population size.

45. Griffin, R. B. and N. J. Griffin. (2003) Distribution, Habitat partitioning, and abundance of Atlantic spotted dolphins, Bottlenose dolphins, and Loggerhead sea turtles on the eastern Gulf of Mexico continental shelf. Gulf of Mexico Science **21**(1), 23-34

Line transects extending from the coast to the 180m isobath between Tampa Bay and Charlotte Harbour, Florida were surveyed by ship between 1998 and 2000 and the abundance of Atlantic spotted dolphins and bottlenose dolphins was estimated. A number of assumptions of distance sampling methodology were not met during this study.

46. Gunnlaugsson, T. and J. Sigurjónsson. (1990) NASS-87: Estimation of whale abundance based on observations made onboard Icelandic and Faroese survey vessels. Reports of the International Whaling Commission **40**, 571-580

Data from Faroese and Icelandic survey vessels participating in the 1987 North Atlantic Sighting Surveys (NASS) were used to estimate fin, sperm, killer, blue, northern bottlenose and humpback whale abundance. The estimate of abundance of sei whales was considered to be a minimum estimate as the survey was conducted too early to cover the mid-summer distribution of this species. It's thought that a large number of minke whales were missed during the survey and, as a result, the abundance estimate for minke whales is thought to be significantly biased despite efforts to correct for diving minke whales. The abundance estimate for minke whales based on Icelandic survey vessel data refers to the Central North Atlantic stock, while the abundance estimate based on Faroese vessel data refers to the Northeastern Atlantic stock. Abundance estimates of sperm and northern bottlenose whales were not corrected for animals that were submerged, potentially resulting in significant bias.

47. Hammond, P.S., Berggren, P., Benke, H., Borchers, D. L., Collet, A., Heide-Jórgensen, Heimlich, S., Hiby, A. R., Leopold, M. F. and N. Øien. (2002) Abundance of harbor porpoise and other cetaceans in the North Sea and adjacent waters. Journal of Applied Ecology **39**, 361-376

The authors report on data collected during the SCANS survey conducted in summer 1994. The SCANS survey used shipboard and aerial line transect surveys over a number of stratum to estimate abundance of cetaceans in the North Sea and adjacent waters. Abundance estimates are given for harbour porpoise, minke whale, white-beaked dolphin, short-beaked common dolphin and Atlantic whitesided dolphin.

48. Hansen, L. J., Mullin, K. D. and C. L. Roden. (1995) Estimates of cetacean abundance in the northern Gulf of Mexico from vessel surveys. Miami, USA, SEFSC, Miami Laboratory**:** 9.

The authors report on a 3.75 year project (1 October 1991 through 15 July 1995), in a study area bounded by the Florida-Alabama border, the Texas-Mexico border, and the 100- and 2,000-m isobaths. Seasonal visual surveys from ships were carried out to determine the distribution and abundance of cetaceans in the study area. A total of 683 shipboard marine mammal sightings of at least 19 species were made during the eleven ship surveys.

49. Harwood, L. A., Innes, S., Norton, P. and M. C. S. Kingsley. (1996) Distribution and abundance of beluga whales in the Mackenzie estuary, southeast Beaufort Sea, and west Amundsen Gulf during late July 1992. Can J Fish Aquat Sci **53**, 2262-2273

Three aircraft were used to carry out an aerial line transect survey of the southeast Beaufort Sea, Mackenzie estuary and west Amundsen Gulf over three days in July 1992. The survey area covered was 81,686 km^2 and the abundance of Beluga whales was estimated. Efforts were made to correct the abundance estimates presented here for perception bias but not for availability bias and, therefore represent an estimate of the number of belugas visible at the surface during the 1992 survey.

50. Hill, P. S., DeMaster, D. P. and R. J. Small. (1997) Alaska Marine Mammal Stock Assessments, 1996. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-78, 150 p.

The authors report on stock assessments for all Alaska marine mammals. Each stock assessment includes a description of the stock's geographical range, a minimum population estimate, current population trends, current and maximum net productivity rates, optimum sustainable population levels and allowable removal levels, and estimates of annual human-caused mortality and serious injury through interactions with commercial fisheries and subsistence hunters.

51. Hobbs, R. C., Rugh, D. J. and D. P. DeMaster. (2000) Abundance of Belugas, *Delphinapterus leucas*, in Cook Inlet, Alaska, 1994-2000. Marine Fisheries Review **62**(3), 37-45

Aerial surveys were flown over Cook Inlet 2-3 times per year and the daily abundance of whales was averaged for each strata of the survey. The total survey area of all strata was 20,363 km^2. The averaged abundances were then summed to calculate total annual abundance. This estimate was corrected for perception bias using data from independent observers within the aircraft. Annual summed total abundance estimates of Beluga whales are given for 1994, 1995, 1996, 1997, 1998, 1999 and 2000.

52. Kinas, P. G. and C. B. P. Bethlem. (1998) Empirical Bayes abundance estimation of a closed population using mark-recapture data, with application to humpback whales, Megaptera novaeangliae, in Abrolhos, Brazil (SC/49/SH29). Rep. Int. Whal. Commn **48**, 447-450

Photo-identification data of humpback whales were collected in 1995 during the breeding season on the Abrolhos Bank, Brazil. An empirical Bayes estimation procedure was then used on the mark-recapture data to estimate abundance of this population. These data are not currently contained within the database as there is no area associated with the abundance estimate.

53. Kingsley, M. C. S. and R. R. Reeves. (1998) Aerial surveys of cetaceans in the Gulf of St. Lawrence in 1995 and 1996. Canadian Journal of Zoology **76**, 1529-1550

A dedicated aerial survey was conducted to estimate cetacean abundance in the Gulf of Lawrence in 1995 and 1996. The timing of the surveys varied between years, as did the survey area with the majority of the Gulf being surveyed in 1995 but only the northern shelf being covered in 1996. Abundance estimates were not corrected for visibility bias. Abundance and density was estimated for the three stratums that made up the survey area in 1995 and the one stratum surveyed in 1996.

54. Laake, J. L., Calamokidis, J. C., Osmek, S. D. and D. J. Rugh. (1997) Probability of detecting harbour porpoise from aerial surveys: estimating g(0). Journal of wildlife management **61**, 63-75

This study tracked porpoises from land to estimate the proportion of time harbour porpoises spent at the surface and the probability that aerial observers detected groups within 200m of the transect line. Simultaneous land-based tracking and aerial surveys were carried out and estimates of g(0) are given for both methodologies and for experienced and inexperienced observers. 30.5% of land-based sightings were observed from the aircraft and experienced observers were more successful than inexperienced observers.

55. Lowry, L. F., DeMaster, D. P. and K. J. Frost. (1999a) Alaska Beluga Whale Committee Surveys of Beluga Whales in the eastern Bering Sea, 1992-1995. Report from the Alaska Beluga Whale Committee submitted to the Small Cetacean Subcommittee of the International Whaling Commission SC/51/SM34

The authors report on an aerial, strip and line transect survey carried out in May, June and September of 1992 and June of 1993-1995. The area surveyed was approximately 52,852 km^2 but the actual area surveyed varied between years. Strip transect methods were used to estimate beluga whale density following the 1992 surveys, while line-transect methods were used with the remaining data. Efforts were made to correct for availability bias of adults, using beluga dive data from Arctic Canada, and perception bias of neonates which are smaller and harder to see from an aircraft. Because of variation in the size of the area surveyed each year, the abundance estimates are not comparable. However, estimated densities of beluga whales for each year can be compared. Abundance estimates are given for Beluga whales in 1992, 1993, 1994, early 1995 and late 1995.

56. Lowry, L. F., DeMaster, D. P., Frost, K. J. and W. Perryman. (1999b) Alaska Beluga Whale Committee Surveys of Beluga Whales in the eastern Chukchi Sea, 1996-1998. Report from the Alaska Beluga Whale Committee submitted to the Small Cetacean Subcommittee of the International Whaling Commission SC/51/SM33

The authors report on an aerial line-transect survey, covering 54,604 km^2, for Beluga whales in the eastern Chukchi Sea. The best current estimate of abundance for the eastern Chukchi Sea stock of Beluga whales is thought to be 3,710. This figure is based on a minimum abundance estimate based on aerial surveys conducted between 1989 and 1991 and has been corrected for availability bias and the increased probability of missing young animals during the survey because of small body size and dark colouration.

57. Lowry, L. F. and K. J. Frost. (1999) Alaska Beluga Whale Committee Surveys of Beluga Whales in Bristol Bay, 1993-1994. Report from the Alaska Beluga Whale Committee submitted to the Small Cetacean Subcommittee of the International Whaling Commission SC/51/SM32

The authors report on aerial line-transect surveys for beluga whales in Bristol Bay in June-July 1993 and 1994. The area surveyed was 23,561 km^2. The belugas in Bristol Bay are considered a separate stock. Efforts were made to correct for availability

bias of adults, using beluga dive data, and perception bias of neonates which are smaller and harder to see from an aircraft. Abundance estimates are given for 1993 and 1994.

58. MacLeod, K. (2004) Abundance of Atlantic white-sided dolphin (*Lagenorhynchus acutus*) during summer off northwest Scotland. Journal of Cetacean Research Management **6**(1), 33-40

The author reports the first abundance estimates for this species to the northwest of Scotland. A shipboard cetacean survey was conducted in July/August 1998 within the Atlantic Frontier and standard line-transect and DISTANCE sampling methodology was used. This paper presents abundance estimates corrected for $g(0) < 1$ using a direct duplicate method and the value of $g(0)$ was estimated to be 0.61 (CV = 0.09). Abundance was estimated in two strata; the west of the Outer Hebrides (21,371 (CV = 0.54)) and the Faroe Shetland Channel (74,626 (CV = 0.72)). These results, together with opportunistic sightings data collected during other surveys, suggest that the waters to the west of Scotland are an important habitat for the Atlantic white-sided dolphin

59. Macleod, K., Simmonds, M. P. and E. Murray. (2006) Abundance of fin (*Balaenoptera physalus*) and sei whales (*Balaenoptera borealis*) amid oil exploration and development off northwest Scotland. Journal of Cetacean Research and Management **8**(3), 247-254

A shipboard line-transect cetacean survey was conducted in July/August 1998 in the waters off northwest Scotland. Two strata, the west of the Outer Hebrides and the Faroe Shetland Channel were surveyed. Neither fin nor sei whales were recorded to the west of the Outer Hebrides whereas relatively high densities of both were recorded further north in the Faroe-Shetland Channel. Abundance was estimated as 933 (CV=0.38) fin whales, and 1011 (CV=0.35) sei whales. The high density of whales in the Faroe-Shetland Channel supports the idea that it is an important summer feeding ground for both species.

60. Meier, S. K., Yazvenko, S. B., Blokhin, S. A., Wainwright, P., Maminov, M. K., Yakovlev, Y. M. and M. W. Newcomer. (2007) Distribution and abundance of western gray whales off northeastern Sakhalin Island, Russia, 2001-2003. Environ. Monit. Assess. **134**,107-136

This study reports on >60,000 km of aerial surveys and 7,700 km of vessel surveys conducted during June to November in 2001–2003,. Results of surveys in all years indicated gray whales occurred in predominantly two areas, (1) adjacent to Piltun Bay, and (2) offshore from Chayvo Bay During all years, the distribution and abundance of whales changed in both areas, and both north–south and inshore–offshore movements were documented within and between feeding seasons. Seasonal shifts in the distribution and abundance of gray whales between and within both the areas are thought, in part, to be a response to seasonal changes in the distribution and abundance of prey.

61. Moore, S. E., Waite, J. M., Friday, N. A. and T. Honkalehto. (2002) Cetacean distribution and relative abundance on the central-eastern and the southeastern Bering Sea shelf with reference to oceanographic domains. Progress in Oceanography **55**, 249-261

The authors report on ship-based, visual, line-transect survey carried out in the central-eastern Bering Sea from the 5 July to 5 August in 1999 and the southeastern Bering Sea from 10 June to 3 July 2000, in association with a Pollock stock assessment survey. The survey area covered was 197,412 km^2 in 1999 and 156,654 km^2 in 2000. Abundance was estimated for 1999 and 2000 for 4 species encountered during the survey; fin whales, minke whales, Dall's porpoise and harbour porpoise. An estimate for humpback whales for 2000 is also given The abundance estimates were not corrected for perception or availability bias so are likely to be negatively biased. This is not a big problem for larger species because g(0) probably approaches 1, but is a major issue for smaller species (particularly harbour porpoise).

62. Moore, S. E., Waite, J. M., Mazzuca, L. L. and R. C. Hobbs. (2000) Mysticete whale abundance and observations of prey associations on the central Bering Sea shelf. J Cetacean Res Manage **2**(3), 227-234

The authors report on ship-based, visual, line-transect survey carried out in the central-eastern Bering Sea from the 5 July to 5 August in 1999. The survey area covered was 322,263km^2 and abundance was estimated for 3 mysticete species; fin whales, minke whales and humpback whales. The abundance estimates were not corrected for perception or availability bias so are likely to be negatively biased. This is not a big problem for larger species because g(0) probably approaches 1.

63. Moretti, D., DiMarzio, N., Morrissey, R., Ward, J. amd S. Jarvis. (2006) Estimating the density of Blainville's beaked whale (*Mesoplodon densirostris*) in the Tongue of the Ocean (TOTO) using passive acoustics (060328-10). Oceans 06 MTS / IEEE, Boston, unpublished.

An array of 82 bottom-mounted hydrophones was used to gather acoustic data on the distribution of Blainville's beaked whales, and these data were verified by visual observers. Six days worth of data collected in April 2005 formed the basis of the study.

64. Mullin, K. D. and G. L. Fulling (2003) Abundance of cetaceans in the southern U.S. North Atlantic Ocean during summer 1998. Fishery Bulletin **101**(3), 603-613

A ship-based line transect survey was completed between Maryland and central Florida from the 10m isolbath to the boundary of the U.S. Exclusive Economic Zone (200nmi from land) in summer 1998. Independent observers were used during the survey in an attempt to estimate the probability of detecting cetaceans, but there were insufficient independent sightings and g(0) could not be estimated. Because of a high degree of variability in the sighting rate of each species, the precision of abundance estimates resulting from this survey is poor.

65. Mullin, K. D. and G. L. Fulling (2004) Abundance of cetaceans in the oceanic northern Gulf of Mexico, 1996-2001. Marine Mammal Science **20**(4), 787-807

The authors report on ship-based line-transect surveys conducted in oceanic waters of the northern Gulf of Mexico during spring from 1996-1997 and from 1999-2001. Data were pooled from five surveys and minimum abundance estimates of 19 species are given. The program DISTANCE was used for the analysis, but g(0) was not estimated and hence assumed to be 1 for each abundance estimate. The authors state abundance estimates will be negatively biased as g(0) was <1 as observers would have missed animals on the trackline.

66. Noad, M.J., Paton, D. and Cato, D.H. (2006) Absolute and relative abundance estimates of Australian east coast humpback whales (*Megaptera novaeangliae*). Paper SC/A06/HW27 presented to the Workshop on the Comprehensive Assessment of Southern Hemisphere humpback whales, 4-7 April 2006, Hobart, Tasmania.

In 2004 a land-based survey was conducted of migratory humpback whales on the east coast of Australia from Point Lookout on North Stradbroke Island. Two different, isolated lookout points were used during the survey allowing a correction factor for missed groups to be developed. Absolute abundance of the population was then calculated. Relative abundance data was given for 1981, 1982, 1986, 1987, 1991, 1993, 1996 and 2000 as well as 2004. The whales were considered to be part of the Group E breeding stock from the Area V population.

67. O'Corry-Crowe, G. M., Suydam, R. S., Rosenberg, A., Frost, K. J., and A. E. Dizon. (1997) Phylogeography, population structure and dispersal patterns of the beluga whale *Delphinapterus leucas* in the western nearctic revealed by mitochondrial DNA. Molecular Ecology **6**, 955-970

This study reports on the analysis of mitochondrial DNA control region sequence variation of 324 beluga whales from 32 locations representing five summer concentration areas in Alaska and north-west Canada. Phylogenetic relationships among haplotypes were inferred from parsimonious networks, and genetic subdivision was examined using haplotypic frequency-based indices and an analysis of variance method modified for use with interhaplotypic distance data. Overall, results showed that the patterns of mtDNA variation in beluga whales indicate that the summering concentrations are demographically, if not phyletically distinct. Population structure appears to be maintained primarily by natal homing behaviour, while asymmetries in dispersal may be associated with the type of mating system.

68. Palka, D (1995) Abundance estimate of the Gulf of Maine harbor porpoise. Biology of the phocoenids - Reports of the International Whaling Commission (Special issue 16)**,** 27-50

A ship-based line transect survey was conducted in 1991 to estimate the abundance of harbour porpoise in the Gulf of Maine. The probability of detecting harbour porpoise on the trackline was estimated and used to correct the abundance estimates for each of the 4 survey strata.

69. Palka, D. L. (2006) Summer abundance estimates of cetaceans in US North Atlantic Navy operating areas. Northeast Fisheries Science Center Reference Document 06-03

Ship-based and aerial line transect data were gathered in the summers of 1998, 1999, 2002 and 2004 by the Northeast Fisheries Science Centre. The study area comprises multiple strata, some of which were surveyed by plane while others were surveyed by ship. Not all strata were surveyed in all 4 summers.

70. Parra, G. J., Corkeron, P. J. and H. Marsh. (2006) Population sizes, site fidelity and residence patterns of Australian snubfin and Indo-Pacific humpback dolphins: implications for conservation. Biological Conservation **129**, 167-180

Photo-identification data were gathered between 1999 and 2002 in northeast Queensland, Australia to estimate the abundance of snubfin and humpback dolphins. As this study spanned four years, the population was not assumed to be closed and instead the Jolly-Seber open population model was used to estimate the number of dolphins frequenting the study area. The population size of each species was estimated to be less than 100 individuals.

71. Paton, D. A., Brooks, L., Burns, D., Franklin, T., Franklin, W., Harrison, P. and P. Baverstock. (2006) First abundance estimate of east coast Australian humpback whales (*Megaptera novaeangliae*) utilising mark-recapture analysis and multi-point sampling. Paper SC/A06/HW32 presented to the Workshop on the Comprehensive Assessment of Southern Hemisphere humpback whales, 4-7 April 2006, Hobart, Tasmania.

Boat-based surveys were undertaken in three different locations on the east coast of Australia in 2005 to gather photo-identification data of humpback whales during their migration to the breeding grounds. The whales were considered to be part of the Group E breeding stock from the Area V population. The population was assumed to be closed during the survey period, and eight models were fitted to the data. The model that seemed to best describe the data resulted in an abundance estimate of 7,024 individuals.

72. Rosenbaum, H. C., Strindberg, S. and P. J. Ersts. (2004) Initial estimates abundance and distribution of humpback whales on their wintering grounds in the coastal waters of Gabon (southeastern Atlantic Ocean, Area B) based on aerial surveys. Report to the International Whaling Commission Scientific Committee SC/56/SH2

This survey lasted 5 days in 2002 and covered 62,635 km^2 in the region out to the 1000 metre depth contour. The study area included the entire coastline of Gabon and part of the Congolese coastline. The survey area consisted of two strata and abundance of humpback whales was estimated for each. Detection was assumed certain within 450 metres from the trackline and the data were left truncated at this distance, so the abundance estimates are likely negatively biased. Attempts were made to correct for those animals missed on the trackline (i.e. at 450 metres). The assumption that each point in the study area has an equal coverage probability was

not met and perception bias was not accounted for. The study area was small relative to the migratory and breeding grounds of humpback whales off West Africa and the number of whales using this area is likely to be much larger than indicated by this study.

73. Scheidat, M, Gilles, A., Kick, K. H. and U. Siebert. (2006) Harbour porpoise (*Phocoena phocoena*) abundance in German waters (July 2004 and May 2005) (SC/58/SM19). International Whaling Commission - Scientific Committee Meeting, St Kitts and Nevis, unpublished.

Aerial line transect surveys were conducted of German waters of the North Sea and Baltic in July 2004 and May 2005 to estimate abundance of harbour porpoise. Abundance was estimated for each stratum (North Sea and the Baltic) for each year. The probability of detecting porpoises on the trackline during both good and moderate conditions was estimated by doubling back and re-searching segments of the trackline, a method known as the "racetrack" design.

74. Schweder, T., et al. (1997) Abundance of northeastern Atlantic minke whales, estimates for 1989 and 1995. Reports of the International Whaling Commission. G. P. Donovan, IWC. **47:** 453-483.

Large-scale ship-based line transect surveys were completed by the Institute of Marine Research, Norway in the North Atlantic in 1989, 1995, 1996, 1997, 1998, 1999, 2000 and 2001. Data resulting from the surveys completed in 1989 and 1995 were analysed and the abundance of minke whales in the Norwegian Sea and adjacent waters estimated.

75. Siebert, U., Gilles, A., Lucke, K., Ludwig, M., Benke, H., Kock, K. H. and M. Scheidat. (2006) A decade of harbour porpoise occurrence in German waters – Analyses of aerial surveys, incidental sightings and strandings. Journal of Sea Research **56**, 65-80

Aerial line transect surveys of the German North Sea and the German Baltic sea were completed in 1995 and 1996. Harbour porpoise density was found to be higher in the eastern North Sea than the western Baltic. The authors also comment on incidental sightings, strandings and by-catch of porpoises in the same areas.

76. Sigurjónsson, J., Gunnlaugsson, T. and M. Payne. (1989) NASS-87: Shipboard sightings surveys in Icelandic and adjacent waters June-July 1987. Reports of the International Whaling Commission. Cambridge, UK, IWC. **39,** 395-409

North Atlantic Sighting Surveys (NASS) were completed by a number of national agencies in 1987, 1989, 1995 and 2001 and the abundance of several cetacean species estimated. This paper gives an account of the Icelandic ship-based surveys conducted in 1989. As well as recording sightings, dive time data for minke whales were gathered to allow correction for submerged minke whales missed during the sightings survey.

77. Sigurjónsson, J. and G. A. Víkingsson. (1997) Season abundance of and estimated food consumption by cetaceans in Icelandic and adjacent waters. J. Northw. Atl. Fish. Sci **22**, 271-287

The authors use data on cetacean abundance estimates from the NASS 87, 89 and 95 sightings surveys and seasonal variation in abundance estimated from sightings and/or catch data from whaling vessels to assess consumption rates by cetaceans in Icelandic and adjacent waters. They calculate the possible magnitude of consumption as 6.3 million tons in a smaller area defined as Icelandic and adjacent waters, and 8.8 million tons in the larger area north of 60°N. They note many assumptions are made for these calculations, but give final estimates naming fin whales (*Balaenoptera physalus*) and minke whales (*B. acutorostrata*) as the largest consumers in the area, followed by long-finned pilot whales (*Globicephala melas*) and northern bottlenose whales (*Hyperoodon ampullatus*).

78. Skaug, H. J., Øien, N., Schweder, T. and G. Bøthun. (2004) Abundance of minke whales (*Balaenoptera acutorostrata*) in the Northeast Atlantic: variability in time and space. Can. J. Fish. Aquat. Sci. **61**, 870-886

Large-scale ship-based line transect surveys were completed by the Institute of Marine Research, Norway in the North Atlantic in 1989, 1995, 1996, 1997, 1998, 1999, 2000 and 2001. Data gathered from the ship-based line transect surveys carried out in the Norwegian Sea and adjacent waters between 1996 and 2001 were used to estimate minke whale abundance for each of those years. Different areas were surveyed in different years, with the eastern Norwegian Sea being surveyed in 1996, the western Norwegian Sea in 1997, the North Sea in 1998, the Greenland Sea in 1999 and the Barents Sea in 2000 and 2001.

79. Smith, T. D., Allen, J., Clapham, P. J., Hammond, P. S., Katona, S., Larsen, F., Lien, J., Mattila, D., Palsbøll, P. J., Sigurjónsson, J., Stevick, P. T. and N. Øien. (1999) An ocean-basin-wide mark-recapture study of the North Atlantic humpback whale (*Megaptera novaeangliae*). Marine Mammal Science **15**(1), 1-32

A two year ocean-basin-wide photo-identification survey of humpback whales was conducted in the North Atlantic in 1992 and 1993. Data collected from the western and eastern Atlantic were used to estimate that 10,600 humpback whales populate the North Atlantic. This study is not currently contained in the database as it doesn't have an area associated with it.

80. Stensland, E., Carlén, I., Särnblad, A., Bignert, A. and P. Berggren. (2006) Population size, distribution and behaviour of Indo-Pacific bottlenose dolphin (*Tursiops aduncus*) and Humpback (*Sousa chinensis*) dolphins off the south coast of Zanzibar. Marine Mammal Science **22**(3), 667-682

The authors used mark-recapture methods to estimate abundance from photo-identification data gathered between January and March, 1999 to 2002 in a study

area of 84km^2. The study was designed to investigate population sizes, distribution and habitat preferences of two species to monitor impacts of anthropogenic activities such as tourism and by-catch. For each species, a large number of individuals were resighted between years suggesting that most of the population was photographed and that these species are at least seasonally resident in these waters. Abundance estimates for humpback dolphins for 1999 (58) 2001 (65) and 2002 (63) and for Indo-Pacific bottlenose dolphins from 1999 (150), 2000 (153), 2001 (179) and 2002 (136) are reported.

81. Thomsen, F., Laczny, M. and W. Piper. (2007) The harbour porpoise (Phocoena phocoena) in the central German Bight: phenology, abundance and distribution in 2002-2004. Helgol. Mar. Res. **61**, 283-289

The authors report the results of aerial line-transect surveys undertaken in the central German Bight between 2002 and 2004 in order to estimate harbour porpoise abundance. The use of multiple independent observers and knowledge of dive times meant that the probability of detecting harbour porpoise on the trackline could be estimated, and used to modify abundance estimates. Absolute abundance was calculated for each of the survey days using DISTANCE. The authors proposed that the study area is used as a transit route between areas of high porpoise density.

82. Wade, P.R. and T. Gerrodette. (1993) Estimates of cetacean abundance and distribution in the eastern tropical Pacific. Rep Intl Whal Commn **43**, 477-493

The authors report on large-scale research vessel surveys conducted annually from 1986 through 1990 by NMFS. Stratified line transect surveys were used to monitor the abundance of dolphin populations in the eastern tropical Pacific Ocean (ETP). Data from five surveys are pooled to give single estimates of abundance for 24 stocks representing 19 species.

83. Waite, J. M., Friday, N. A. and S. E. Moore. (2002) Killer whale (*Orcinus orca*) distribution and abundance in the central and southeastern Bering Sea, July 1999 and June 2000. Marine Mammal Science **18**(3), 779-786

The authors report on ship-based, visual, line-transect survey carried out in the central-eastern Bering Sea from the 5 July to 5 August in 1999 and the southeastern Bering Sea from 10 June to 3 July 2000, in association with a Pollock stock assessment survey. The survey area covered was 197,412 km^2 in 1999 and 156,654 km^2 in 2000. Abundance was estimated for 2000 for killer whales. Low sightings in 1999 meant abundance estimation was not possible in this year. The abundance estimate was not corrected for perception or availability bias so are likely to be negatively biased.

84. Waring, G. T., Pace, R. M., Quintal, J. M., Fairfield, C. P. and K. Maze-Foley. (2003) U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2003, U.S. Department of Commerce**:** 300.

This document contains the stock assessment reports for the U.S. Atlantic and Gulf of Mexico marine mammal stocks under NMFS jurisdiction. Marine mammal species which are under the management jurisdiction of the USFWS are not included in this report. Each stock assessment includes a stock definition and extent of geographical range, a minimum population estimate, current population trends, current status of stock, current and maximum net productivity rates, optimum sustainable population levels and allowable removal levels, estimates of annual human-caused mortality and serious injury through interactions with commercial fisheries and subsistence hunters, and key references.

85. Williams, R. and L. Thomas. (2006) Distribution and abundance of marine mammals in the coastal waters of British Columbia. Report to the International Whaling Commission Scientific Committee SC/58/019

The authors report on systematic ship-based line transect surveys carried out in the summers of 2004 (covering the entire study region) and 2005 (providing partial coverage of the study region) in the coastal waters of British Columbia, Canada. The survey area consisted of a number of strata that combined cover 83,072 km^2. The estimates provide the first abundance estimates for 6 cetacean species in that area; Harbour porpoise, Dall's porpoise, Pacific white-sided dolphin, Fin whale, Humpback whale, Minke whale and Northern Resident Killer whales. The probability of detecting cetaceans on the trackline was assumed to be 1. Conventional distance sampling methods were used to estimate the probability of detecting cetaceans that weren't on the trackline, and an unbiased estimate of mean school size was obtained using the program DISTANCE. This information was then used to estimate density and abundance.

86. Zeh, J. E., George, J. C., Raftery, A. E. amd G. M. Carroll (1991) Rate of increase, 1978-1988, of bowhead whales, Balaena mysticetus, estimated from ice-based census data. Marine Mammal Science **7**(2): 105-122

Between 1978 and 1988 an ice-based census of bowhead whales on their spring migration has been undertaken from Point Barrow in Alaska. The migrating whales comprise the Bering-Chukchi-Beaufort Seas stock. Two observation points were used for a number of years in order to correct the abundance estimates for whales missed by a single observer. In addition to this, at various times, aerial and acoustic surveys were completed to estimate the number of whales passing offshore and out of range of the visual observers and the number of whales in range but missed by observers respectively. Trends in the number of bowhead whales seen each year suggest that the population is increasing by 3.1% per year.

87. Zerbini, A. N., Andriolo, A., Da Rocha, J. M., Simões-Lopes, P. C., Siciliano, S., Pizzorno, J. L., Waite, J. M., DeMaster, D. P. and G. R. VanBlaricom. (2004). Winter distribution and abundance of humpback whales (*Megaptera novaeangliae*) off northeastern Brazil. Journal of Cetacean Research and Management **6**(1), 101-107

The authors report on shipboard, line transect, sighting surveys off the northeastern coast of Brazil, to evaluate humpback whale distribution and density in 1999 and 2000. Data from the 2000 survey were used to estimate abundance over the continental shelf from 5 to 12°S. Humpback whales were distributed from near-shore to the 800m isobath, but 93.5% of sightings were recorded shoreward of the 300m isobath. The relatively high density off northeastern Brazil suggests that the species is reoccupying historical areas of distribution and the presence of newborn individuals indicates that calving and nursing occur in the area. This estimate probably corresponds to only a portion of the breeding population.

88. Zerbini, A. N., Waite, J. M., Durban, J. W., LeDuc, R., Dahlheim, M. E. and P. R. Wade. (2006a) Estimating abundance of killer whales in the nearshore waters of the Gulf of Alaska and Aleutian Islands using line-transect sampling. Marine Biology 13

The authors report on ship-based line transect surveys, conducted in the summers of 2001-2003 in coastal waters (usually within the 1000m isobath) of western Alaska and the Aleutian Islands. Conventional and multiple covariate distance sampling methods were used to estimate the abundance of different killer whale ecotypes, distinguished based upon morphological and genetic data post-survey, in some of the 16 survey blocks. Fourteen blocks were surveyed in 2001, then two more were added and all 16 blocks were surveyed in 2002 and 2003. The area of all strata combined was 210,989 km^2. The survey design was such that abundance could be estimated for individual blocks, or for all blocks combined. The probability of detecting whales on the trackline was assumed to be 1, potentially resulting in the abundance estimates being negatively biased. Abundance estimates of resident killer whales (991 [95% CI = 379–2,585] and 1,587 [95% CI = 608–4,140]) were at least four times greater than those of the transient killer whales.

89. Zerbini, A. N., Waite, J. M., Laake, J. L. and P. R. Wade. (2006) Abundance, trends and distribution of baleen whales off western Alaska and the central Aleutian Islands. Deep-Sea Research I **53**, 1772-1790

The authors report on ship-based line transect surveys conducted in the summers of 2001-2003 in coastal waters (usually within the 1000m isobath) of western Alaska and the Aleutian Islands. Conventional and multiple covariate distance sampling methods were used to analyse the data gathered and abundance was estimated for fin, minke and humpback whales in some or all of the 16 survey blocks. Fourteen blocks were surveyed in 2001, then two more were added and all 16 blocks were surveyed in 2002 and 2003. The area of all strata combined was 210,989 km^2. The survey design was such that abundance could be estimated for individual blocks, or for all blocks combined. The probability of detecting whales on the trackline was assumed to be 1, potentially resulting in the abundance estimates being negatively biased. Annual rates of increase for fin and humpback whales were also estimated in this study.

7 Appendix 1: database structure

For this review an access database was created to hold the data for all the AORs. The database will be updated as each task is completed and it is envisaged that the information in this database will be provided to the JIP on completion of the project.

This database has a structure consisting of the following tables:

1. **Abundance**: This table holds all information for each reference that has associated abundance information
2. **Reference**: This table lists all references used to date
3. **Survey_ID**: This table holds all necessary spatial information about each survey block used within the project.

Report

Project Name:	JIP Cetacean Stock Assessment
Reference:	RA1007OGP
Project Manager:	Nicola Quick

Draft report drafted by:	Nicola Quick, Kristin Kaschner, Rodrigo Wiff & Len Thomas	
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Contents

Task 2 Deliverable:
Cetacean stock assessment in relation to Exploration and Production Industry sound

Detecting trends in global cetacean stocks within the Areas of Relevance

Summary

Task 2 delivers a report outlining which covariates may be important when assessing changes in density over time for 34 cetacean species combined globally, and for seven individual species within specific areas. Results from the global model showed that the taxonomic attributes of species and family were by far the most important covariates. For the individual models, temporal and spatial covariates generally explained the highest proportion of the total deviance across both species and area. Survey-related covariates, such as method and agency, explained little of the deviance. Power analysis showed that small changes (few tens of percent) may be detectable in some rare cases, but in most of the studies analysed, the population change would need to be at least double, before it could be detected with high power. In some cases, even an order of magnitude of population change could easily be missed. The species for which population trends can be analysed will be carried forward to Task 3 when factors influencing cetacean stocks will be investigated.

1. Introduction

In order to assess any potential relationships between Exploration and Production Industry (E&P) sound and trends in cetacean stocks within JIP areas of relevance, it is necessary to first determine the possibility of detecting trends in cetacean population using published estimates. Detecting trends in populations or stocks of marine mammals is a primary focus of much research. The ability to detect how numbers of animals may change over time can help inform many management decisions including those relating to conservation, and mitigation of anthropogenic impacts. Research into population numbers has been carried out on cetacean species across the globe; but determining changes in population trends is not straight forward. Many cetacean species are wide-ranging and not easily observed at sea. Additionally, many undertake migrations that lead to seasonal differences in distribution. These differences in distribution may be at a population level (e.g. Humpback whales migrating between feeding and breeding areas; see Clapham, 2000 for review) or may only relate to individuals of a certain sex within a population (e.g. sperm whales, where females remain in low latitudes and males migrate to feed in higher latitudes, Whitehead & Weilgart, 2000). To deal with these difficulties, researchers have developed different methods for monitoring populations and analysing data. These methods include line-transect surveys for all species present in an area (e.g. SCANS surveys; SCANS I: and SCANS II: http://biology.st-andrews.ac.uk/scans2/; SCANS-II 2008); individual species accounts of stock structures and movements (http://www.nmfs.noaa.gov/pr/sars/species.htm); photo-identification surveys of species that assess numbers within restricted geographic boundaries (e.g. Parra *et al*., 2006 and Smith *et al*., 1999) and counts of animals passing geographic points (Buckland & Breiwick, 2002 and Zeh *et al*., 1991). In addition, new methods are also being developed that try to estimate abundance of highly vocal species using acoustic monitoring techniques (Barlow & Taylor, 2005).

Although all these methods can provide a good means of monitoring populations, a further problem of how robust and comparable the data are across years to detect changes needs to be addressed.

Variation across population estimates may be caused by taxonomic, spatial, temporal and methodological differences of varying scales. Taxonomic differences will be evident when comparing estimates across species. Spatial differences will occur if survey areas cover completely different parts of a species' range, are across a range of suitable habitat, or if survey areas vary between different years. Temporal differences will occur if surveys of a population are conducted in different seasons or years. Methodological differences may be evident if several survey or analysis techniques are used. Differences between methodologies such as mark-recapture versus line-transect, where the former may give higher abundance estimates if sampling focuses only on known hotspots, and density from these hotspots is then extrapolated to a wider area. Or differences in how animals may response in terms of movement away from a ship, compared to an aerial surveying vessel. Additionally, some surveys may include correction for animals missed on the track line in line transect surveys (g(0) estimation), and differences in the application of the same methodologies will be present between different agencies.

The aim of Task 2 was to use available published data that were collected and reviewed during Task 1, to indicate robustness of trend estimates. The data collected during Task 1 used The Environmental Risk Management Capability Project (ERMC) database as a foundation. This database contains more than 1800 regional abundance estimates from over 350 surveys and covered a total of 70 marine mammal species. The data compiled in the database are estimated to equate to roughly 90% or more of all surveys conducted globally over the past 30 years. Within this database comprehensive details about relevant associated information for each survey are held. These survey details include; the geographic location, the time period and duration, the size of the survey area, survey methodology and any potential biases. By investigating the importance of these factors as covariates of the abundance estimates, it may be possible to assess whether or not the currently available data on species abundance over time are of sufficient quality to reliably detect trends in cetacean populations, at both a global scale and at an individual species or population level. As such the Task was split into two parts: a "global" analysis and, a set of "species-specific" analyses. Initially, a total of 34 cetacean species were selected from the database, which were (a) covered by ERMC, thus ensuring a very comprehensive coverage of all available information and/or (b) associated with >nine abundance estimates in the database (Table 1). Selecting all available data on a global scale for these species resulted in a subset of 1035 abundance estimates that were used in the subsequent analysis. Using these data, the principal aim of the global analysis was to determine overall which covariates had the most influence on the density estimates, both singly and in combination. The secondary aim of the global analysis was to determine how precisely temporal trends could be estimated after all other important covariates were included.

The species-specific analyses focussed on individual populations of specific data-rich species within some of the areas of relevance (identified in Task 1) for which multiple estimates exist[1]. The final step was to produce a power analysis to make a preliminary determination of what level of population trend would be observable with reasonable certainty, given the levels of variability about the trend estimates observed.

[1] Areas and species to take forward in the Task 2 analyses were agreed via email on 1st June 2008 by Russell Tait. The decision was based on the areas and species for which most data existed, as outlined in the Task 1 review. Four areas were outlined as possible for further analysis (Area 4 (Alaska); Area 5A (West coast of US); Area 5B (East coast of US); Area 6A (Europe). Within each of these areas, a few candidate species were explored for potential analysis. The final number of species and areas could not be determined until the end of the task 2 analysis. It should be noted that not all species within these areas will be considered due to lack of data and, as such, not all of the four areas may be taken forward to the end of the project.

Table 1. All species names, associated codes and abundance estimates used in the global analysis.

Common name	Scientific name	SpecID	Family	Species Group	Number of Abundance Records
Sei whale	Balaenoptera borealis	Babor	Balaenopteridae	Baleen_whales	28
Blue whale	Balaenoptera musculus	Bamus	Balaenopteridae	Baleen_whales	16
Fin whale	Balaenoptera physalus	Baphy	Balaenopteridae	Baleen_whales	95
Minke whale[2]	Balaenoptera acutorostrata Balaenoptera Bonaerensis	Baacu	Balaenopteridae	Baleen_whales	113
Humpback whale	Megaptera novaeangliae	Menov	Balaenopteridae	Baleen_whales	93
Northern right whale dolphin	Lissodelphis borealis	Libor	Delphinidae	Delphinidae	20
Short-beaked common dolphin	Delphinus delphis	Dedel	Delphinidae	Delphinidae	32
Pygmy killer whale	Feresa attenuata	Featt	Delphinidae	Delphinidae	6
Short-finned pilot whale	Globicephala macrorhynchus	Glmac	Delphinidae	Delphinidae	10
Long-finned pilot whale	Globicephala melas	Glmel	Delphinidae	Delphinidae	34
Risso's dolphin	Grampus griseus	Grgri	Delphinidae	Delphinidae	36
Atlantic white-sided dolphin	Lagenorhynchus acutus	Laacu	Delphinidae	Delphinidae	21
Pacific white-sided dolphin	Lagenorhynchus obliquidens	Laobl	Delphinidae	Delphinidae	34
False killer whale	Pseudorca crassidens	Pscra	Delphinidae	Delphinidae	11
Killer whale	Orcinus orca	Ororc	Delphinidae	Delphinidae	34
Common bottlenose dolphin	Tursiops truncatus	Tutru	Delphinidae	Delphinidae	68
Spinner dolphin	Stenella longirostris	Stlon	Delphinidae	Delphinidae	18
Melon-headed whale	Peponocephala electra	Peele	Delphinidae	Delphinidae	8
Atlantic spotted dolphin	Stenella frontalis	Stfro	Delphinidae	Delphinidae	17
Striped dolphin	Stenella coeruleoalba	Stcoe	Delphinidae	Delphinidae	36
Rough-toothed dolphin	Steno bredanensis	Stbre	Delphinidae	Delphinidae	10
Pantropical spotted dolphin	Stenella attenuata	Statt	Delphinidae	Delphinidae	27
White-beaked dolphin	Lagenorhynchus albirostris	Laalb	Delphinidae	Delphinidae	10
Beluga or white whale	Delphinapterus leucas	Deleu	Monodontidae	Toothed_whales	58
Harbour porpoise	Phocoena phocoena	Phpho	Phocoenidae	Phocoenidae	76
Finless porpoise	Neophocaena phocaenoides	Nepho	Phocoenidae	Phocoenidae	5
Dall's porpoise	Phocoenoides dalli	Phdal	Phocoenidae	Phocoenidae	84

[2] For the global analysis Minke whale refers to estimates for both the Dwarf Minke whale (*Balaenoptera acutorostrata)* and also the Antarctic Minke whale (*Balaenoptera Bonaerensis*). For the individual species analysis only estimates for the Dwarf Minke whale were used.

Sperm whale	Physeter macrocephalus	Phmac	Physeteridae	Toothed_whales	53
Southern bottlenose whale	Hyperoodon planifrons	Hypla	Ziphiidae	Toothed_whales	2
Longman's beaked whale	Indopacetus pacificus	Inpac	Ziphiidae	Toothed_whales	1
Baird's beaked whale	Berardius bairdii	Bebai	Ziphiidae	Toothed_whales	8
Cuvier's beaked whale	Ziphius cavirostris	Zicav	Ziphiidae	Toothed_whales	12
Blainville's beaked whale	Mesoplodon densirostris	Meden	Ziphiidae	Toothed_whales	6
Northern bottlenose whale	Hyperoodon ampullatus	Hyamp	Ziphiidae	Toothed_whales	12

2. Methodology

2.1. Analysis steps

The primary tool used for examining trends in cetacean populations by combining multiple surveys from the literature was generalized linear and generalized additive modelling (GLM/GAM). Since abundance estimates covering different sized areas are clearly not comparable, we used density as the response variable in these models[3]. There were many variables that could potentially be used as explanatory covariates for differences in density estimates which can be classified broadly into the following groups: taxonomic (e.g., species, family), spatial (e.g., ocean, Food and Agriculture Organization of the United Nations (FAO area), temporal (e.g., year, decade) and survey-related (e.g., survey method, agency).

In total, five analysis steps were required: Data Exploration; Determination of Weights; Global Dataset Models; Individual Area and Species Models; Power Analysis. We treat each step in turn below.

2.2 Data Exploration

The first step of data exploration was to convert reported observed abundance estimates to density estimates for each species within each survey based on the survey area. All surveys were digitized in ArcGIS 9.1, and the survey area calculated based on the produced shapefiles. Where different strata were present within one survey each stratum was entered with its associated estimates. Once density estimates had been calculated for all species within surveys, potential explanatory covariates were chosen. Graphical techniques were used to explore both potential patterns in density related to each covariate, and the number of estimates available using different covariate combinations. It was clear that many covariates had a large number of levels e.g., species, survey agency and FAO area. These levels were a direct result of the amount of data held within the ERMC database. The level of detail precluded the fitting of models for combinations of covariates, so parsimonious groupings were investigated to enable data exploration for each of the outlined covariate groups: taxonomic, spatial, temporal and survey-related. Table 2 contains a list of the potential covariates considered.

The outcome of this first phase was a set of covariates that could be used in the subsequent modelling exercises, as well as information about which ones could potentially be modelled in combination.

[3] An alternative, which we did not pursue, would be to use abundance with area as an offset in the model.

Table 2. Potential covariates considered for inclusion during exploratory data analysis. Abbreviations are those used in subsequent tables. See text for details of factor levels.

Covariate	Abbreviation	Covariate group	Type
Species	Species	Taxonomic	Factor, 87 levels
Family	Family	Taxonomic	Factor, 6 levels
Species group	SppGroup	Taxonomic	Factor, 4 levels
FAO area	FAO	Spatial	Factor, 18 levels
Ocean basin	Ocean	Spatial	Factor, 6 levels
Mean latitude	Lat	Spatial	Continuous
Maximum latitude	MaxLat	Spatial	Continuous
Minimum latitude	MinLat	Spatial	Continuous
Year	Year	Temporal	Factor (rounded mean year for multi-year estimates, 22 levels) or continuous
Decade	Decade	Temporal	Factor, 3 levels
Season	Season	Temporal	Factor, 3 levels
Survey methodology	Method	Survey-related	Factor, 6 levels
Agency	Agency	Survey-related	Factor, 33 levels
Ocean basin and grouped survey agency	OceanAgency	Spatial and survey-related	Factor, 11 levels

2.2.1 Taxonomic covariates

In total information on 63 cetacean species is held within the database, 34 of which fulfilled the criteria set for the global analysis described above. However, even within this selected group, information on abundance is variable across all species. For the global analysis the consideration of each species individually would most likely involve too many factors. Therefore, species were grouped into families (six groupings; Balaenopteridae; Delphinidae; Phocoenidae; Monodontidae; Physeteridae and Ziphiidae), containing between 1 and 20 species. An alternative taxonomic grouping combined the last three groups (two of which contained only a single species) into a single "other_toothed_whales" group (now consisting of 22 species). For the individual based analysis, species were considered to have sufficient data if the database contained more than 30 specific estimates roughly equally distributed between the different geographic subareas (unless they were not found in all three areas). Note that although information on stocks of cetaceans was outlined in Task 1, the data are considered here on a species basis[4].

2.2.2 Spatial covariates

Spatial coverage of surveys varied both within and across the areas of relevance. Surveys areas could be calculated as Km^2 and hence density estimates could theoretically be reported in the same resolution. However, since species distribution is generally unlikely to be homogeneous across entire survey areas, a spatial resolution below the size of survey areas would be misleading. Unfortunately, survey areas within the same region also tend to vary considerably between different years, thus precluding the use of survey areas as the minimum spatial unit of analysis. Therefore the surveys were allocated to specific statistical reporting areas that are used by the Food and Agricultural

[4] It is not possible to look at trends of cetaceans on a stock level for the global analysis because abundance estimates reported in the literature do not consistently provide stock information. Certain stock assessment reports do provide this level of detail and, as such, stock information can be addressed during the species and area specific section of the analysis.

Organization (FAO) (Figure 1). However, even at this relatively large spatial scale, several surveys took place over multiple FAO areas and for these surveys, datasets with one identical record for each FAO area were created, e.g. if one overall density estimate was available it would be entered for each FAO region that the survey covered. This applied to 70 surveys, turning the 219 species-specific density estimates for these surveys into 519 records, and therefore making the total number of records for the FAO-level analysis to 1334. This is a clear case of pseudoreplication, but is defensible in the context of an exploratory analysis (see discussion). FAO areas could be allocated to ocean basins reducing the number of geographic units further, to six ocean basins (Pacific, Atlantic, Indian Ocean, Mediterranean, Arctic and Antarctic). Since no survey covered more than one ocean basin the original dataset with a column for Ocean basin could be used in the model (eliminating any pseudoreplication as seen in the FAO-level approach). The result was two datasets; one including the pseudoreplication for FAO area and one with the groupings for Ocean basin. Both of these were taken forward from the exploratory analysis to the first stage of modelling. A further spatial level that was considered was average (coarse measure of shape of survey, i.e. if it is tear-shaped, or symmetrical), and minimum and maximum latitude values for the survey blocks. This approach enabled consideration of multiple abundance estimates for single species that may consist of geographically different populations.

2.2.3 Temporal covariates

Two levels of temporal information exist in the database. The first is the annual level – i.e., the year of the survey. For survey estimates that cover multiple years, the midyear was used. In subsequent modelling, when year was used as a covariate, this was rounded to the nearest integer, but non-integer values were left when year was used as a continuous covariate. Year was also used to assign surveys to decades. The second level of temporal information was season. This was used to assign surveys to one of three levels; summer (surveys conducted during the months June-November), non-summer (surveys conducted during the months December-May), and year-round (any survey longer than 6 months).

2.2.4 Survey related covariates

A number of different survey related covariates are recorded in the database. To account for different survey methodologies and analysis methods used for abundance estimation, it was necessary to produce covariates that grouped surveys into a manageable number of categories. Six groups were considered, as shown in Table 3.

Table 3. Groupings of the survey-related variables used in the analysis. G(0) refers to animals missed on the trackline during line-transect surveys, resulting in an underestimation of species abundance.

Variable code	Variable description
A	All non line transect surveys (Photo-id, counts from land, acoustic)
B	Aerial surveys not corrected for g(0)
C	Aerial surveys corrected for g(0)
D	Shipboard surveys not corrected for g(0)
E	Shipboard surveys corrected for g(0)
F	Combined Shipboard and aerial surveys not corrected for g(0)

Information about the agency conducting the survey was also a potential covariate. In total, 33 different survey agencies were represented in the database. Because most agencies operated in only

one ocean, and because of the large number of factors that would need consideration if each agency was considered separately, the agencies were grouped into a small number of categories and combined with ocean basin for the analysis (Table 4). This created a combined survey and geographic covariate.

Table 4. Groupings for ocean agency used in the analysis. NOAA: US National Oceanic and Atmospheric Administration, includes all National Marine Fisheries Science Centres

Survey-Ocean	Survey Agency Grouping
Antarctic	Other Agencies
Arctic	NOAA
Arctic	Other Agencies
Atlantic	NASS
Atlantic	NOAA
Atlantic	Other Agencies
Atlantic	SCANS
Indian ocean	Other Agencies
Mediterranean	Other Agencies
Pacific	NOAA
Pacific	Other Agencies

2.3 Determination of Weights

There are two reasons to consider weighting the density estimates when combining them in a GLM/GAM analysis. Firstly, they cover different amounts of area. Intuitively, within a defined region (such as an ocean basin) a survey that makes inferences about only a small part of that region, and hence potentially a small part of the population in it, should have a smaller influence on the analysis than one that makes inferences about a large part of that region, and hence potentially a larger part of the population. Secondly, they have different levels of precision, partly reflecting the spatial variability in the animals over the area for which the estimate is made, but largely reflecting the amount of effort devoted to the survey. It makes sense to weight more precise estimates more heavily than less precise ones. Within these principles, there are many potential ways to calculate area and precision weightings, and to combine them, some of these were investigated as described below.

2.3.1 Calculating area and precision weightings

Since we have limited information about the distribution of each species within each survey block, we cannot calculate the area of the species' range covered by each survey. As a proxy, we calculated the area weighting simply as the area of each survey, taken directly from the database.

For the precision weighting, the natural weight would be $1/SE^2$, which shows strong variability among surveys. For this weighting, the coefficient of variation (CV, defined here as standard error of the estimated density divided by estimated density) was used. For this, and all subsequent analyses, we excluded all estimates of zero abundance or those that were associated with a CV > 200%. For surveys where standard errors were reported, the CV was calculated directly. However, some records only had 95% two-sided confidence intervals (CI) recorded as a measure of precision. For these records the standard error was calculated by assuming the CIs came from a lognormal distribution. This was a reasonable assumption given the asymmetry found on the interval endpoints

with respect to the estimates. For each abundance estimate x and its associated confidence limits [a, b] two statistics C_1 and C_2 were calculated using

$$C_1 = \frac{x}{a} \text{ and } C_2 = \frac{b}{x}. \quad (1)$$

Under the assumption of lognormal distribution, these statistics should be the same, and where they were this value was used as *C* in the formulae that follow. However, for some records C_1 and C_2 were slightly different, indicating they were estimated using some method other than assuming a lognormal distribution on density (this most likely results from a resampling method). For these cases, a compromise value of *C* was calculated, as the mean of C_1 and C_2. Given a value of *C*, the estimated variance of the abundance estimate on the natural log scale is given by

$$\mathrm{var}[\log(x)] = \left(\frac{\log(C)}{Z_\alpha}\right)^2 \quad (2)$$

where Z_α is the upper α-level quartile of the standard Gaussian distribution (approximately 1.96).

This value was then back-transformed to a variance estimate using

$$\mathrm{var}(x) = (\exp[\mathrm{var}(\log(x))] - 1)x^2 \quad (3)$$

From this the standard error was calculated by taking the square root of $\mathrm{var}(x)$.

A few records (0.8% of the total data) had no reported precision of any kind. For those records we arbitrarily assigned the value of the upper 90th percentile of the distribution of CVs calculated from the estimates where some measurement of precision was reported. Our reasoning was that surveys that do not report precision are likely poorly performed, and hence imprecise; we therefore assigned a low precision (high CV) to them, in order to give them low weight in subsequent analyses.

2.3.2. Combining weightings

In theory, the optimal area weighting would be untransformed area, and the best precision weighting would be the inverse of the variance (or inverse CV^2). A sensible way to combine them would be to first normalize the weightings to put them on the same scale (so that both have a maximum value of 1, for example), and then multiply them together. However, a straight combination of the two weightings would have the undesirable effect of placing almost all the weight on very few surveys. This is because there is enormous variation in both the area covered by the surveys, and in the inverse variance. However, there is less variation in the CV^2, and as a dimensionless quantity this was the preferred value. Nevertheless, these survey CV^{2}s are still highly variable, so it may be more insightful to temper the weights to some extent. This could be achieved for example, by square root or log-transforming them, before scaling and multiplying them together. This still has the effect of weighting larger and more precise surveys more heavily in the analysis, but not to the extent that few estimates dominate, obscuring any pattern.

One way to investigate the potential effect of a proposed weighting method is to calculate the effective sample size (ESS) using that weighting, by

$$ESS = \frac{n}{1+[CV(w)]^2} \quad (4)$$

where *n* is the number of estimates available, and *CV(w)* is the coefficient of variation of the weights. If all weights are the same, then *ESS=n*, and *ESS* declines as variability in weights increases.

We used graphical methods to investigate the distribution of area and CV^2, as well as square root and logged values of these quantities and their scaled combinations to calculate the resulting ESSs. The final decision on the best weighting method was highly subjective, aiming to get as close as possible to the ideal of (normalised) area divided by (normalised) CV^2, while at the same time maintaining a reasonably high ESS.

2.4 Global dataset models

For the global analysis we used GLM/GAM modelling on the global dataset of species estimates for all surveys, in conjunction with a supervised forward selection procedure for introducing covariates into the models. The response variable, weighted estimated density was assumed to follow a gamma distribution, with an inverse link function (McCullagh & Nelder, 1989). The models were fitted using the mgcv package within the R statistical software (R Development Core Team, 2008).

We began by fitting each single covariate within each covariate group (taxonomic, spatial, temporal and survey-related; (Table 2) in turn. Continuous covariates were fitted as smooth functions, using thin-plate regression splines with the degree of smoothness determined using cross-validation (Wood, 2006; 2008). In some cases, the number of knots (related to maximum "wiggliness" of the function) had to be restricted relative to the default used by mgcv, to allow model convergence. We judged the importance of each covariate using the percentage deviance explained by its inclusion, as well as the generalized cross-validation (GCV) score. A high percentage deviance means that the covariate explains much of the variability in density estimates; however factor covariates with many levels or continuous highly wiggly functions can be expected to explain a great deal of variability simply by using many parameters. Generalized cross validation is one of several potential measures that aims to determine which model is the "best" in terms of balancing good fit (high deviance explained) against parsimony (few parameters). It determines this by estimating the ability of the model to provide predictions of future data points. Lower value GCV scores indicate better estimated out-of-dataset prediction.

Various multiple covariate models were also fitted using promising covariates from the single-factor analysis (i.e., those with low GCV and high deviance explained). Covariates from the same group (e.g., year and decade) were not fitted together, unless this was biologically reasonable (e.g., year and season could potentially be included together). Both main effects and interaction models were tried, although many main effects models failed to fit because of lack of data for some combinations of the covariate values. For interaction models, this did not present a problem, as combinations that were not represented in the dataset were not fitted by the model. However, lack of convergence was a feature of many multiple covariate models, and, in practice, only a few could be fitted successfully. Interactions between continuous covariates were fitted using tensor product smooths (Wood, 2006); interactions between continuous covariates and factors were fitted by specifying a different smooth on the continuous covariate for each level of the factor covariate.

To determine the precision of trend estimates, a particular emphasis was placed on fitting multiple covariate models that included temporal covariates.

2.5 Individual area and species models

Task 1 (Jewell *et al.*, 2008) highlighted on a global scale for which areas, the majority of data on cetacean abundance exists. These areas were then cross-referenced to the JIP areas of relevance (AOR) to determine if it was possible to assess cetacean trends within any AORs of interest to the JIP. In terms of stock information, AOR 4 (Alaska); 5A (West coast of US); 5B (East coast of US); 6A (Europe) were identified to have the best resolution of data required to investigate possible changes in abundance over time. Density estimates contained in the ERMC database were encoded based on FAO areas (Figure 1). For the individual species analysis, data were grouped into different FAO areas which overlapped and /or fully encompassed the most data-rich AORs identified during Task 1. The three areas of interest in terms of FAO areas were: (*i*) Alaska and NE Pacific (FAO area 67) in combination with West coast of Canada USA and Mexico (FAO area 77) equivalent to AOR 4 & 5A; (*ii*) NE coast of USA (FAO area 21) in combination with Central East coast of USA, Mexico and Caribbean (FAO area 31) or AOR 5B; and (*iii*) North east Atlantic (FAO area 27) or AOR 6.

This part of the analysis was carried out for specific areas and species by pairwise combinations of seven species and the three main areas. The seven species of interest were the Harbour porpoise (Phpho), Dwarf Minke whale (Baacu), Fin whale (Baphy), Humpback whale (Menov), Sperm whale (Phmac), Long-finned pilot whale (Glmel) and Striped dolphin (Stcoe). Five of these species (Harbour porpoise, Dwarf Minke whale, Fin whale, Humpback whale and Sperm whale) were considered for all three of the areas. For the remaining two species, there were insufficient or no data in some of the areas. Therefore Long-finned pilot whale was only considered in area iii, and Striped dolphin was only considered in areas i and ii. The FAO-area data set including pseudo-replicates was used for this analysis.

For all species models, GAMs were applied in the same way to the global models, i.e. by assuming a gamma distribution of the response variable with an inverse link. The covariates used in the models had temporal, spatial and survey related attributes. Temporal covariates were year and season code (non-summer, summer and year-round surveys). Spatial covariates were survey latitudes (minimum, maximum and average), and survey related attributes were agency and method. Covariates were considered both singly and by combining temporal trends as a smooth and factor variables as a main effect. The construction of smooth functions with year and latitude as covariates was also explored using tensor product basis functions. For most of the models, the degree of smoothing was estimated by default using the mgcv package; however, as with the global analysis, the maximum number of knots had to be restricted in some cases to allow convergence.

As with the global dataset models, the percentage of explained deviance and the GCV score were used to evaluate model performance.

2.6 Power Analysis

The aim of this analysis was to make a preliminary determination of what level of population trend would be observable with reasonable certainty, given the levels of variability about the trend estimates. However it was first necessary to define what is meant by “trend”, since there is no objective definition (Thomas *et al.,* 2004). A common definition used in power analysis studies (e.g., Gerrodette, 1987) is the slope of a log-linear regression, since this provides a convenient one-number summary of the pattern, is relatively easy to obtain through linear regression, and

corresponds to the growth rate coefficient in an exponential model of population growth. However, for longer time series, assuming that population trend is linear (on some scale) is often unrealistic. There are many alternatives: see Thomas *et al*., (2004) for a review. In previous sections we used generalized additive modelling to fit flexible smooth functions to the annual estimates, we also now assume that these will be used as the basis for a test for non-zero trend. One simple way to achieve this is to compare estimated smoothed population levels from recent years with those from earlier time periods. For example, one could consider comparing the mean smoothed density estimate from the past five (or some other number of) years with that from an earlier set of years to look for quantitative evidence of recent declines. This kind of approach was, to our knowledge, first proposed by James *et al*., (1990). A similar method is currently in use by the British Trust for Ornithology to assess patterns of change in UK land bird populations. For the purposes of this report, we use the following metric of population change to assess trend:

$$\Delta = \frac{\overline{D}_{2001:2005}}{\overline{D}_{1991:1995}} \qquad (5)$$

where $\overline{D}_{x:y}$ is the mean of the smoothed estimates of density for the years *x* to *y* inclusive. A value of 2, for example, indicates a population doubling over that period, while a value of 0.5 indicates a population halving. Other tests, involving different time periods or averaging over different numbers of years, could easily be constructed. Single years could potentially be used for the comparison, e.g., the first and last year of the survey, but these tend to be estimated quite imprecisely.

Since Δ is the ratio of two zero-bounded random variables, we expect its distribution to be approximately log-normal. Hence, a simple test for trend is a one-sample, two-sided z-test of the null hypothesis that the natural log of Δ is zero (i.e., that Δ is 1). Given an estimate of the variance in log (Δ) and the α-level (here assumed to be 0.05) then it is straightforward to calculate the power of the test for various levels of Δ that are considered biologically relevant (the relevant formulae are given in Steidl and Thomas (2001). Here, for the purposes of illustration, we calculate power to reject the null hypothesis of no trend over a range of Δ from an order of magnitude decrease to an order of magnitude increase.

To obtain estimates for the variance of log(Δ) that apply to the current study, we estimated the coefficient of variation (CV) of Δ from all of the single species datasets from the previous section, using the model that had the lowest GCV score while at the same time containing a smooth temporal trend term. Care should be taken when estimating the required CV, since the quantities; $\overline{D}_{1991:1995}$ and $\overline{D}_{2001:2005}$ are not independent of one another (they are both derived from the same smooth). To account for this dependency, we used a parametric bootstrap approach to obtain the variance (Wood, 2006), as follows. For each required variance, we simulated 100,000 bootstrap replicate datasets from the fitted model (i.e., by generating pseudo-random deviates with distribution given by the parameter estimates), calculated $\overline{D}_{1991:1995}$ and $\overline{D}_{2001:2005}$, and hence Δ for each of these samples. We took the variance in these 100,000 simulated values of Δ as an estimate of the required variance. Given values of CV(Δ), we calculated var(log(Δ)) using the relationship

$$\mathrm{var}(\log(\Delta)) = \log\left[1 + CV(\Delta)^2\right]. \qquad (6)$$

3. Results

3.2 Data Exploration

In total 1035 independent abundance estimates for 34 cetacean species collected in 342 different survey areas or survey blocks during 485 distinct surveys (N.B. survey areas could be covered multiple times) were used in this analysis (Figure 1). It is clear that the surveys were not equally spaced on a global scale and that many of the areas of relevance identified in Task 1 (Jewell *et al*., 2008) do not have any associated survey information (Figure 1).

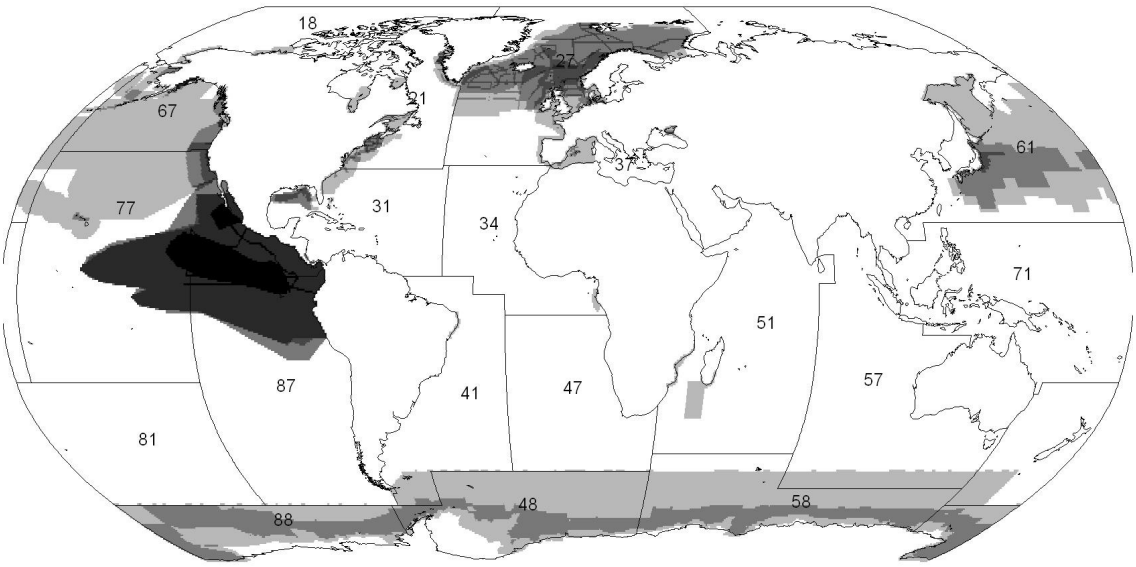

Figure 1. Global map of all FAO areas showing survey coverage and frequency for all species. Shading shows frequency of coverage; light grey: cells covered 1-2 times; light-mid grey: cells covered 3-5 times; dark-mid grey: cells covered 6-9 times; dark grey: cells covered 10-19 times; black: cells covered 20-29 times.

3.2.1 Taxonomic covariates

In total, sufficient numbers of estimates for 34 different species exist. The number of records and amount of density data per species and the number of species within each species group is variable (Figure 2). The most data rich species i.e. those that have more than 40 associated records are spread across all groups; three within the baleen whale group, one in the delphind group, two in the porpoise group and two in the toothed whale group (Figure 2). Information on all 34 species was used within the global models.

3.2.2 Spatial covariates

There was not an equal spread of surveys across either the six ocean basins or the 18 FAO areas (Figures 3 and 4). Only a small minority of all surveys were conducted in the Antarctic, Arctic, Mediterranean and Indian oceans, but the majority of all survey effort to date has been in the Atlantic or Pacific Ocean basins (Figure 3). These oceans were thus selected for closer investigation

in the single species analysis. Similarly, the five FAO areas chosen to be taken forward in the analysis had the majority of density data within them (Figure 4). However some survey data was present in every FAO area (Figure 4).

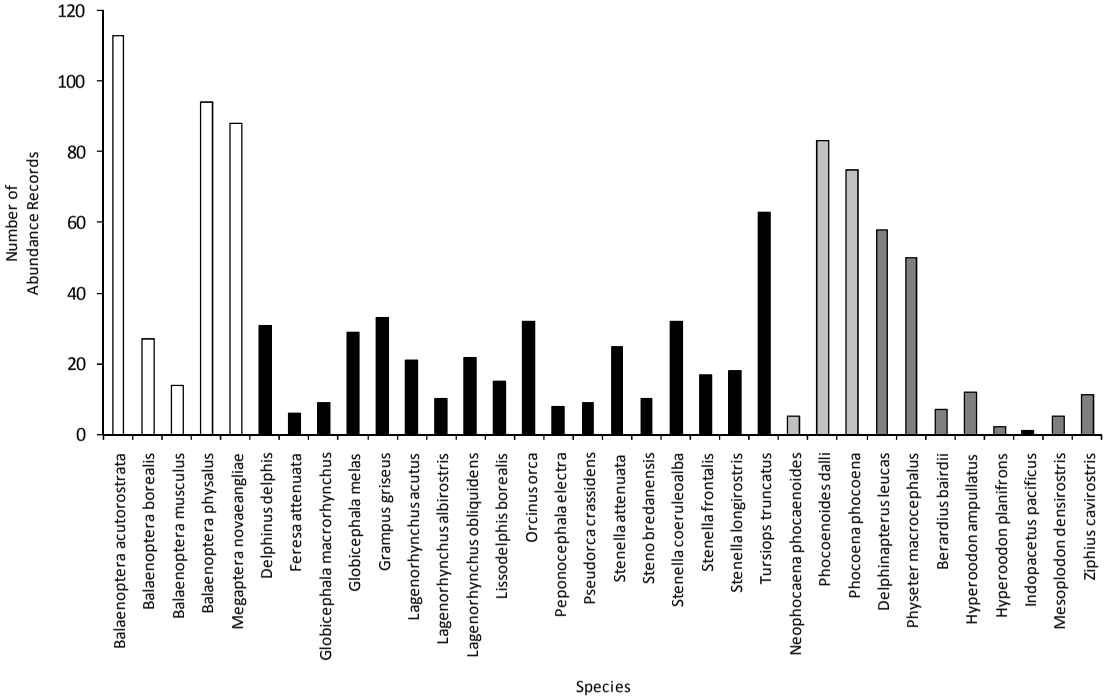

Figure 2: Number of abundance records for the 34 species used in the analysis. Different colours represent different species groups: Baleen whales (white bars), delphinds (black bars), porpoises (light grey bars) and toothed whales (dark grey bars).

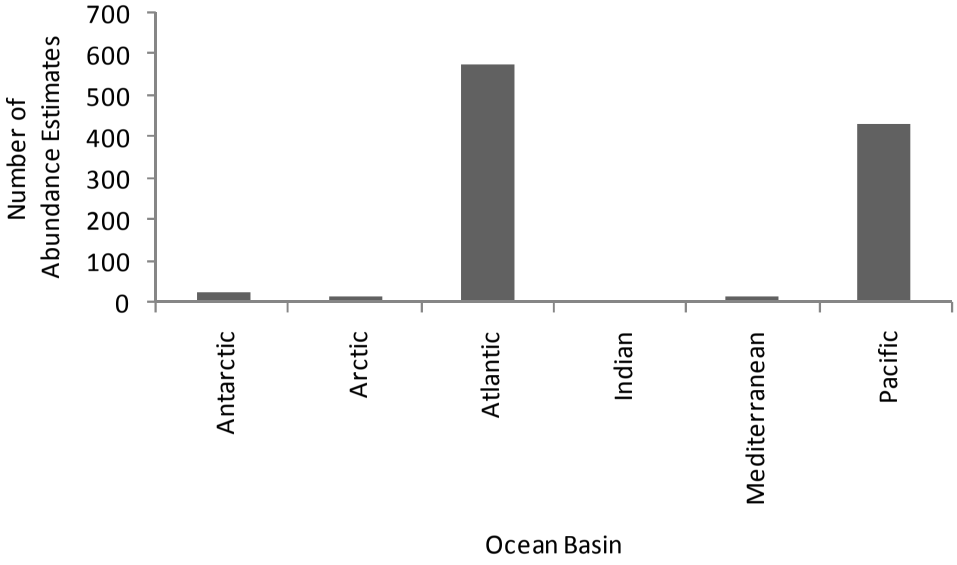

Figure 3. Number of abundance estimates per Ocean basin.

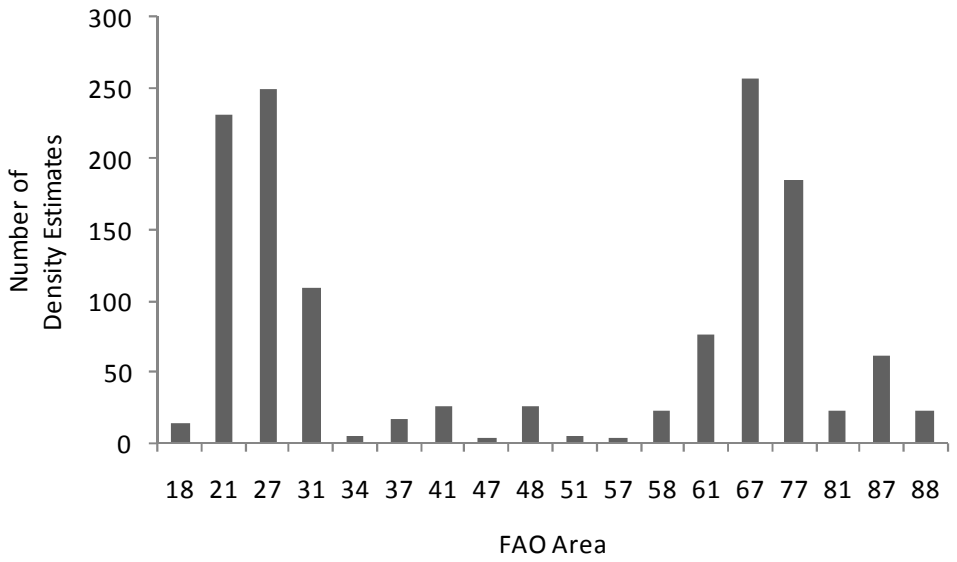

Figure 4. Number of density estimates per FAO area.

3.2.3 Temporal covariates

There was a more equal spread of surveys across decades, with a peak in the 1990's (Figure 5). For the majority of surveys (almost 350 surveys) data were collected within a single year (Figure 6). The longest survey period used to produce a distinct abundance estimate was almost 13 years. The spread of density records with mid-year showed no obvious pattern (Figure 7). The majority of surveys were undertaken in the generic summer (i.e. June-November in the Northern hemisphere and December–May in the Southern hemisphere) (Figure 8).

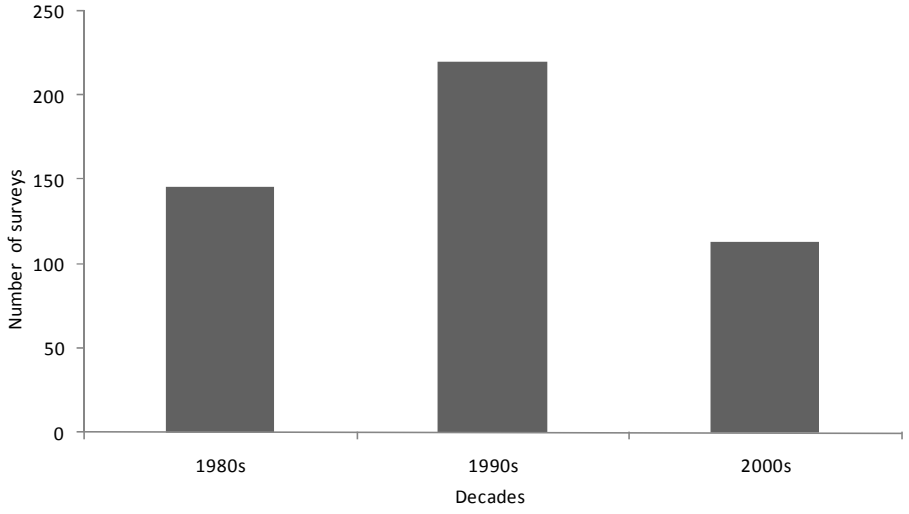

Figure 5. Number of surveys per decade.

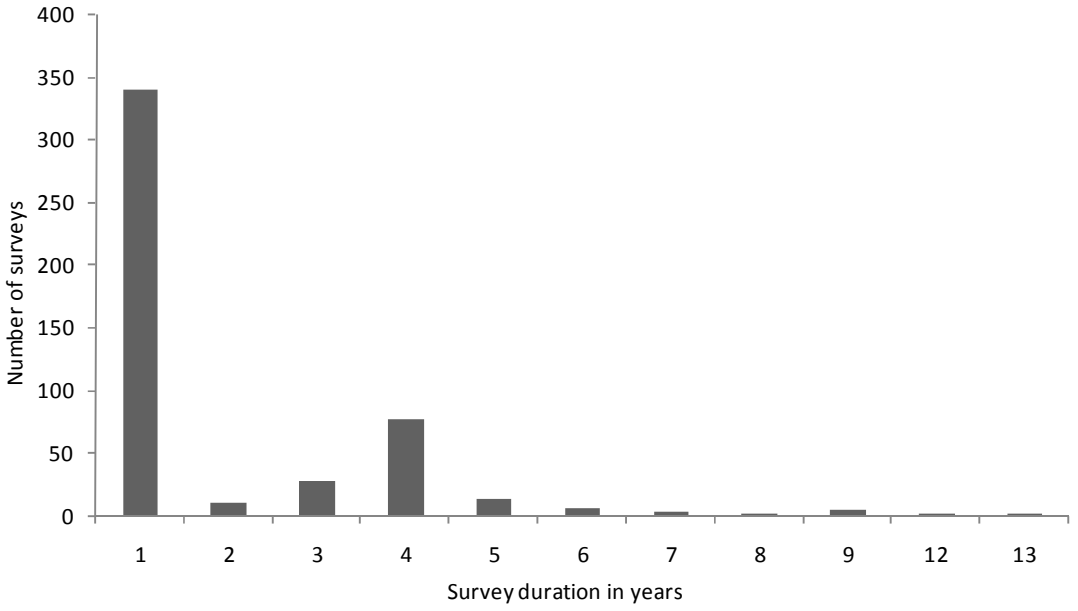

Figure 6. The variation in survey durations for all surveys (N.B. survey duration has been rounded up to the next whole year).

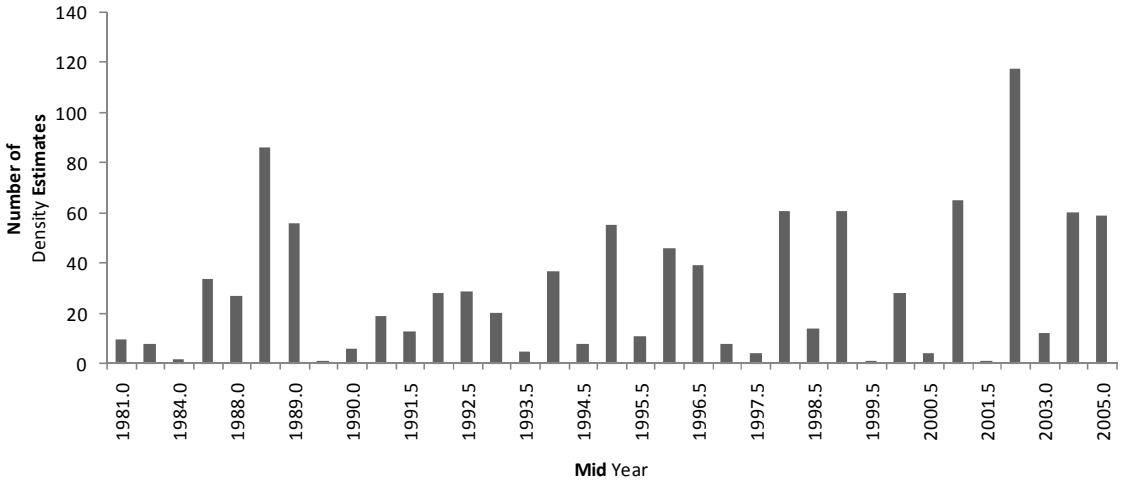

Figure 7. Density estimates with year.

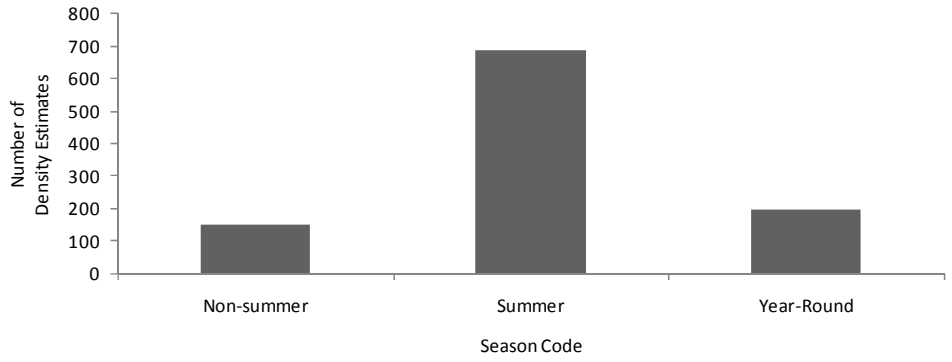

Figure 8. Number of density records with season. Summer refers to a generic summer that corresponds to the months between June-November in the Northern hemisphere and December-May in the Southern hemisphere. Likewise generic non-summer refers to remaining months in the respective hemispheres. Year-round surveys are surveys longer than 6 months in duration.

3.2.4 Survey related covariates

There is clear variation in the six survey method variables. Shipboards surveys not corrected for g(0) make up the majority of survey methodologies, and non-line transect methodologies make up the least (Figure 9). Within each Ocean Basin, there is a predominant Ocean Agency that conducts the majority of surveys. For the Antarctic, Indian Ocean and Mediterranean there were not enough surveys for one Agency to make its own category, and so all agencies were combined to make an 'other agencies' category (Figures 10 and 11). For the Arctic and the Atlantic, the other agencies category is also the predominant category. However, for the Pacific, the majority of surveys and abundance estimates are from the agency NOAA (Figures 10 and 11).

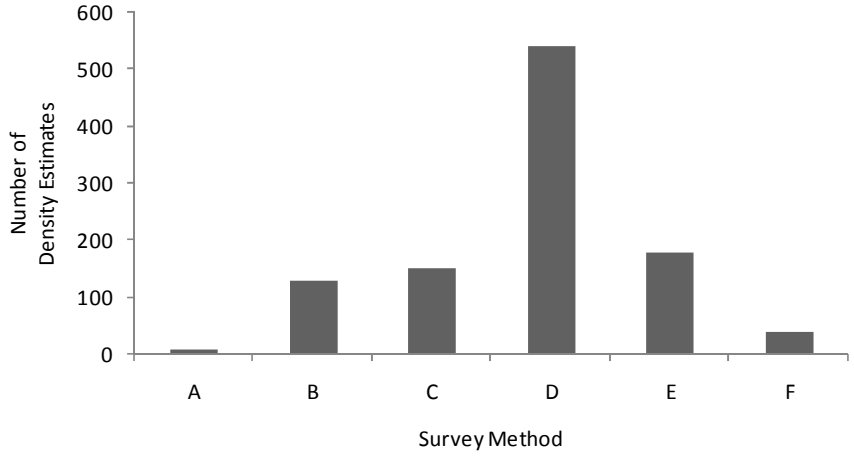

Figure 9. Number of density estimates collected by the six survey methodologies. (A=All non line transect surveys (Photo-id, counts from land, acoustic); B=Aerial surveys not corrected for g(0); C= Aerial surveys corrected for g(0); D= Shipboard surveys not corrected for g(0); E= Shipboard surveys corrected for g(0); F= Combined Shipboard and aerial surveys not corrected for g(0)).

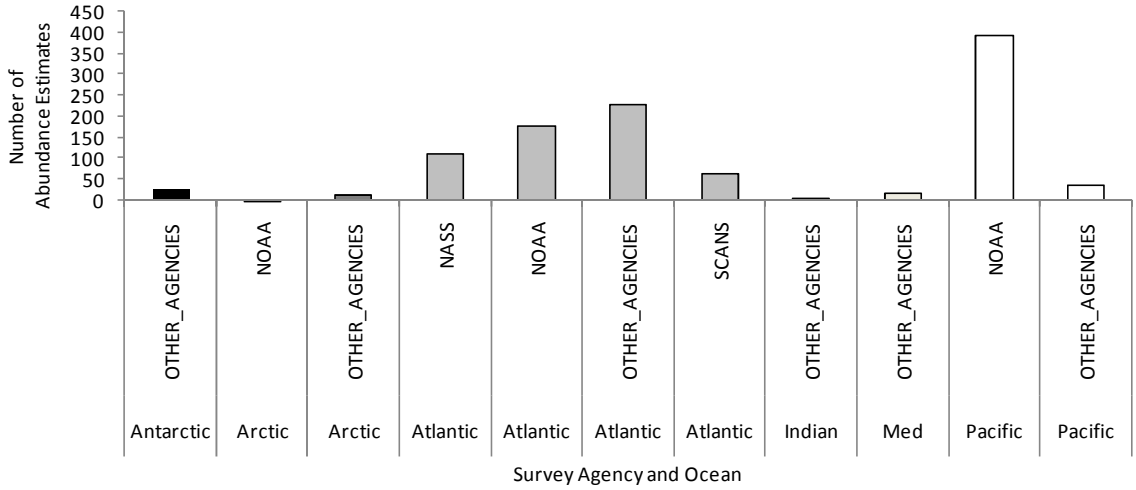

Figure10. Number of abundance estimates per agency groups across the six ocean basin categories.

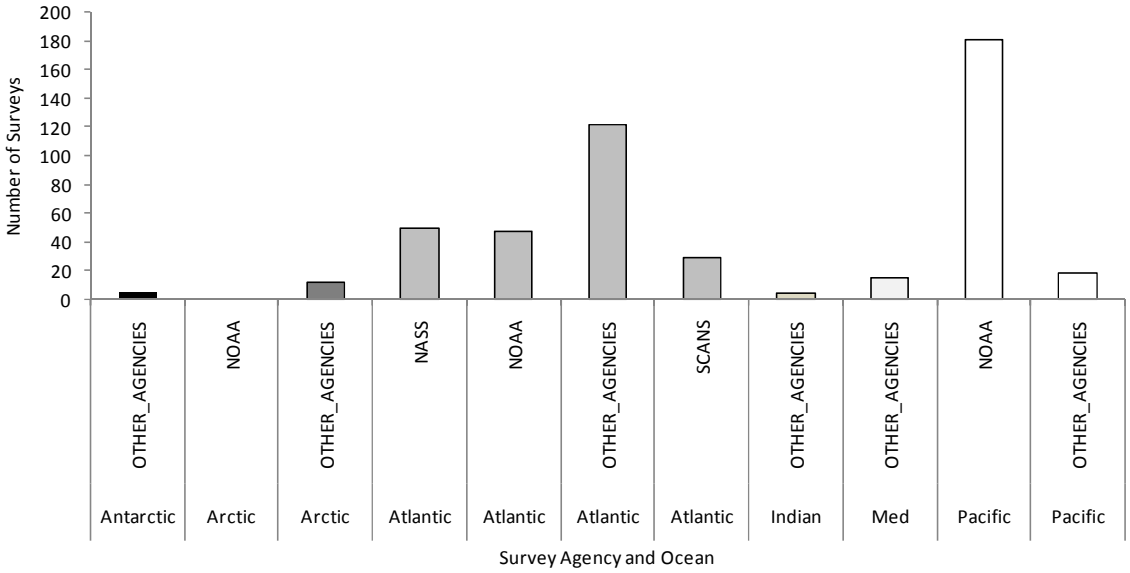

Figure 11. Number of surveys per agency groups across the six ocean basin categories.

3.3 Determination of Weights

Figure 12 gives the empirical distribution and effective sample sizes (ESSs) of various weight measures, calculated on the non pseudoreplicated global data. Note that each weighting is scaled so that its maximum value is 1.0. Distributions for the FAO-pseudoreplicated data were very similar and are not shown here. Both the natural area and precision ($1/CV^2$) weight lead to very low ESS (Figure 12, w1 and w4), and the combination of both (w7) has an ESS of only 34.2 – around $1/10^{th}$ of

the original sample size. Taking the square root of each weight has some ameliorating effect (w2 and w5), but the combined sample size is still only 158 (w8). On the other hand, taking the log of the weights (w3 and w6) has perhaps too little effect, with the combined ESS of 896 (w9) being little less than the number of data points (1035). A compromise that preserves more of the precision weighting, using 1/CV, while still using the logged area (w10) gives an ESS of 590. This compromise weighting was the one chosen for the analyses that follow. The formula is

$$w10 = \frac{\log(\text{Area})/\max(\log(\text{Area}))}{\text{CV}/\max(\text{CV})} \tag{7}$$

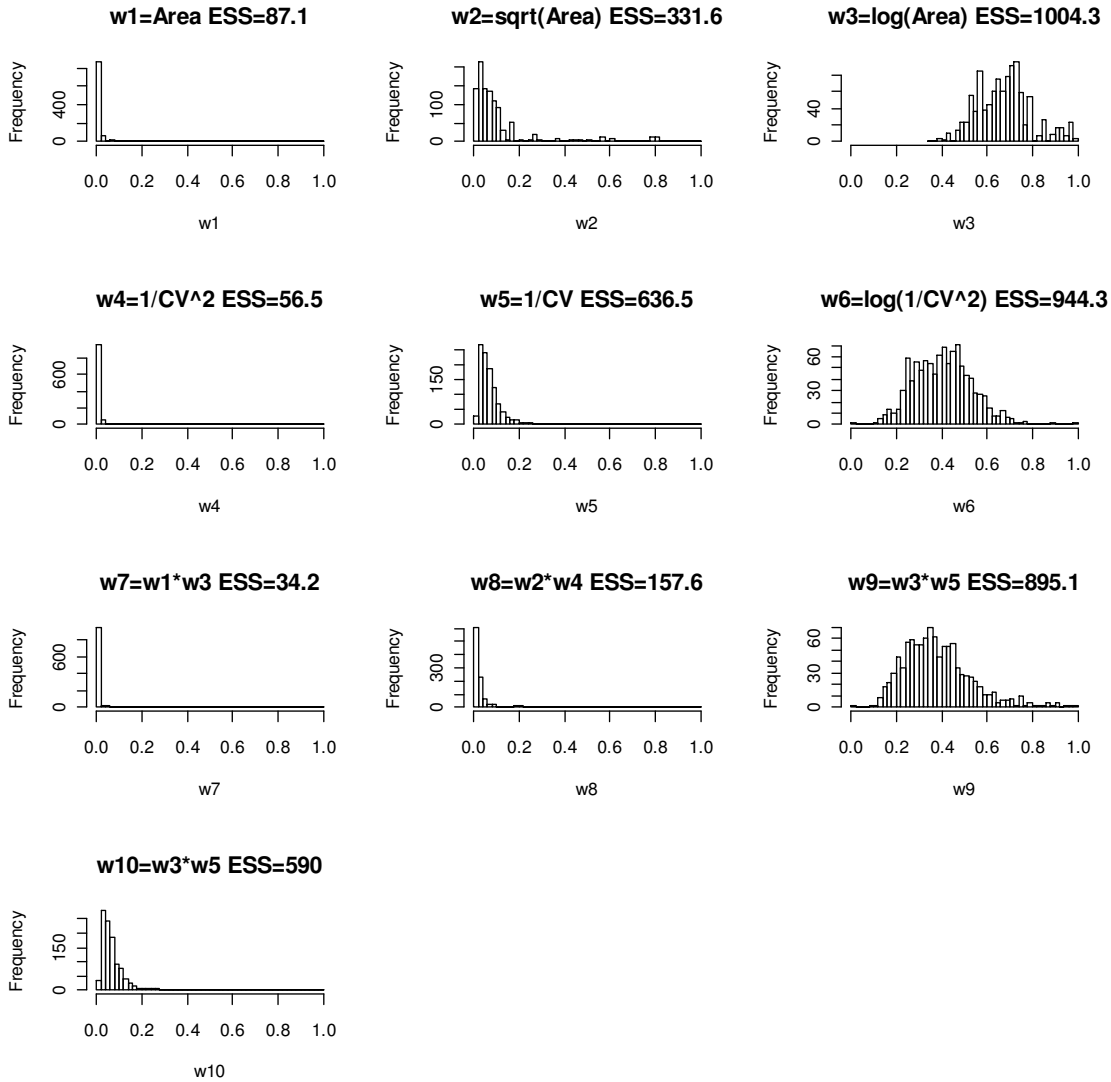

Figure 12. The distributions of the various weight measures. First line shows area weights, second line precision weights, third line combination weights, and forth line the chosen weight measure. Note that all measures have been normalized so that the maximum weight is 1.0. ESS indicates effective sample size in each case; un-weighted sample size is 1035.

3.4 Global dataset models

Single covariate analysis on the global data (Table 5) showed that taxonomic attributes (species, family and species group) are the most important in term of GCV score and explained deviance, with species as by far the most important covariate (despite requiring estimation of 34 parameters). Survey agency had the next lowest GCV score, again despite having 33 factor levels; geographic covariates were the next most important, and the combined covariate ocean agency had a GCV score close to that for survey agency, but with a more moderate 11 parameters. Temporal covariates were the least important.

Table 5. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for Global dataset (1035 observations), using single covariates. GCV is the generalized cross validation score, and s() refers to a covariate entering the model as a smooth. The best model is highlighted in bold.

Model	# parameters	GCV	% deviance explained
Single covariate analysis; global dataset (1035 obs).			
Species	**34**	**0.104**	**57.5**
Family	6	0.152	34.7
SppGroup	4	0.174	24.9
Ocean	6	0.219	5.9
s(Lat)	7.1	0.212	9.0
s(MaxLat)	6.8	0.214	8.0
s(MinLat)	8.1	0.207	11.0
Year	22	0.220	8.3
s(Year)	7.9	0.225	3.3
Decade	3	0.228	1.3
Season	3	0.230	0.3
Method	6	0.216	6.8
Agency	33	0.203	17.1
OceanAgency	11	0.208	11.1

The same models were fit to the pseudoreplicated FAO data. The results (Table 6) were broadly similar. FAO area provided a better fit (in terms of GCV) than Ocean, but was marginally worse than OceanAgency. Because of the difficulties caused by pseduoreplicating surveys to produce the FAO covariate and the lack of a strong effect of FAO area, only the non-pseudorepliacated global data were used in the multiple covariate analyses.

Table 6. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for pseudoreplicated FAO dataset (1334 observations), using single covariates. GCV is the generalized cross validation score, and s() refers to a covariate entering the model as a smooth, with a maximum of k knots (if k is shown). The best model is highlighted in bold.

Model	# parameters	GCV	% deviance explained
Single covariate analysis; FAO dataset (1334 obs).			
Species	**34**	**0.114**	**60.5**
Family	6	0.179	35.3
SppGroup	4	0.214	22.5
FAO	18	0.238	15.4
Ocean	6	0.242	12.6
s(Lat)	7.7	0.236	15.0
s(MaxLat, k=4)	3.5	0.24	11.3
s(MinLat)	8.2	0.230	17.0
Year	22	0.255	10.0
s(Year)	9.7	0.260	6.5
Decade	3	0.270	2.2
Season	3	0.276	0.1
Method	6	0.260	6.3
Agency	33	0.232	19.6
OceanAgency	11	0.235	16.0

Given that the aim of this project was to determine the precision of temporal trends, we tried fitting temporal attributes in combination with other covariates. Using a smoothed function of the mean year variable in combination with each other covariate produced almost identical results as the single covariate analysis (Table 7 part I), with the shape of the smooth being estimated by GCV to be a straight line (i.e., using 1 parameter). When treating mid-year as a factor most model combination with other covariates could not be fitted, including the combination year + species (i.e. the taxonomic attribute that produced the best model in the single covariate analysis). However, the combination of year + family explained slightly more deviance, and had a better fit in terms of GCV, than family as a single covariate. Similarly, the incorporation of temporal information slightly improved the fit of the model that included survey agency information (Table 7, part II). Most interaction models that used smoothed mean year models in combination with other covariates could not be fitted either, but the second taxonomic level attribute (i.e. family) produced a slightly better fit in terms of GCV and explained deviance (Table 7, part III).

Table 7. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for Global dataset (1035 observations), using multiple covariates. GCV is the generalized cross validation score, and s() refers to a covariate entering the model as a smooth. The best models are highlighted in bold.

Model	# parameters	GCV	% deviance explained
Combining temporal trend (smooth) with other covariates – main effects models			
s(Year) + Species	**35**	**0.104**	**57.5**
s(Year) + Family	8.1	0.150	35.6
s(Year) + SppGroup	6.3	0.171	26.0
s(Year) + Ocean	7	0.218	6.3
s(Year) + s(Lat)	8.13	0.212	9.0
s(Year) + s(MaxLat)	13.5	0.212	9.9
s(Year) + s(MinLat)	13.5	0.206	12.4
s(Year) + Season	9.8	0.225	3.7
s(Year) + Method	14.8	0.212	9.9
s(Year) + Agency	34.5	0.203	17.2
s(Year, k=7) + OceanAgency	12	0.209	11.1
Combining temporal trend (factor) with other covariates – main effects models			
Year + Species	NA	NA	NA
Year + Family	**27**	**0.148**	**38.7**
Year + SppGroup	25	0.172	28.8
Year + Ocean	27	0.205	15.1
Year + s(Lat)	NA	NA	NA
Year + s(MaxLat)	NA	NA	NA
Year + s(MinLat)	NA	NA	NA
Year + Season	24	0.220	8.5
Year + Method	NA	NA	NA
Year + Agency	53	0.198	22.1
Year + OceanAgency	NA	NA	NA
Combining temporal trend (smooth) with other covariates – interaction models			
s(Year) * Species	NA	NA	NA
s(Year, k=7) * Family	**17.18**	**0.144**	**39.2**
s(Year, k=7) * SppGroup	13.39	0.158	33.1
s(Year) * Ocean	NA	NA	NA

s(Year, Lat)	NA	NA	NA
s(Year, MaxLat, k=4)	12.7	0.213	9.6
s(Year, MinLat, k=4)	12.4	0.212	9.9
s(Year, k=7) + Season	15.56	0.213	9.8
s(Year) + Method	NA	NA	NA
s(Year) + Agency	NA	NA	NA
s(Year) + OceanAgency	NA	NA	NA
Combining >2 covariates – interaction models			
s(Year) + Species * OceanAgency	117.0	0.083	71.2
s(Year) + Family * OceanAgency	39.7	0.131	47.4
s(Year) + SppGroup * OceanAgency	32.7	0.14	41.9
s(Year) * Family + Species	NA	NA	NA
s(Year) * Family + Species * OceanAgency	**124.5**	**0.082**	**72.2**
s(Year) * SppGroup + Species * OceanAgency	121.3	0.083	71.9
s(Year) *Family + Species * Method	NA	NA	NA
s(Year) *SppGroup + Species * Method	115.0	0.092	68.3
s(Year) * SppGroup + Species * OceanAgency * Method	NA	NA	NA

The best model (using minimum GCV score) was based on a combination of main effects and interactions of four covariates; s(Year)*Family + Species*OceanAgency (Table 7, part IV). This model combined an interaction between smoothed mean year and family with an interaction between species and ocean agencies (i.e. a separate smooth trend fit to each family with different intercepts for each species-ocean-agency combination). The percentage deviance explained by this model is 72%, and it incorporates aspects of all the four main pre-selected covariates; taxonomic, temporal, spatial and survey-related. (Note, however, that all multi-covariate models were forced to include temporal covariates).

For the second level taxonomic attribute of family, the approximate significance of the smooth terms show that the only significant trend is in the Monodontidae family, which only contained a single species in our analysis. The plots of the smooths indicate a modest increase in the function over time for this family (Figure 13). This corresponds to an estimated decrease in density as these covariates were fit using an inverse link function.

Table 8. Approximate p-values for the statistical significance of the smooth terms (edf = estimated degrees of freedom; F = F statistic).

	edf	F	p-value
s(Year):FamilyBalaenopteridae	1.350	0.345	0.691177
s(Year):FamilyDelphinidae	1.000	0.760	0.433221
s(Year):FamilyMonodontidae	2.078	6.936	0.000307 ***
s(Year):FamilyPhocoenidae	1.000	0.164	0.785838
s(Year):FamilyPhyseteridae	1.000	0.262	0.704558
s(Year):FamilyZiphiidae	2.068	1.113	0.338158

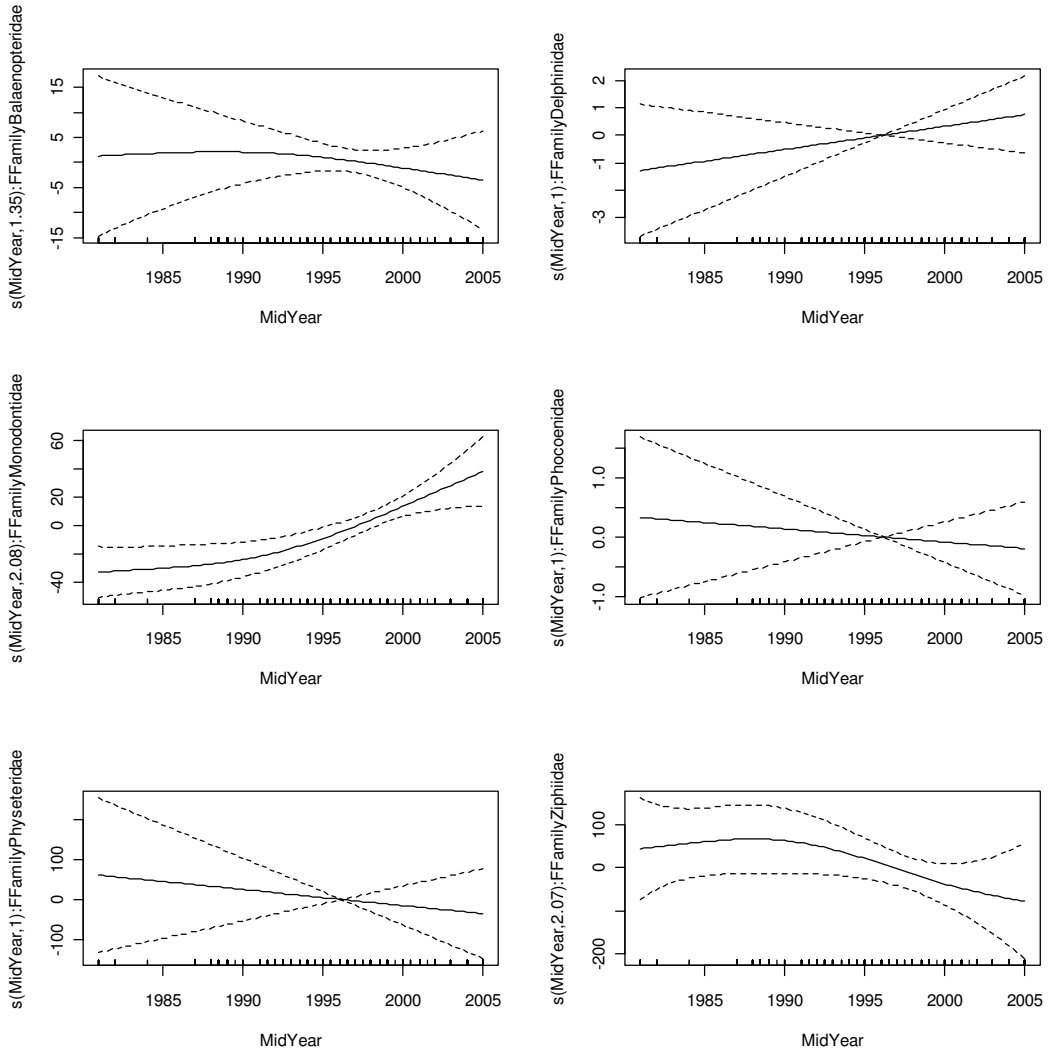

Figure 13. The estimated smooth for each family (solid lines), with +/- 2 standard errors (dashed lines). Note that this is on the scale of the linear predictor, and that an inverse link function was used so a decline on this scale indicates an increase in density over time. As an approximation for families where a horizontal line could be drawn entirely within the dashed lines, the smooth is not statistically significant (p>0.05).

Of the remaining model parameters, 16 of the 33 species-specific intercept parameters were statistically significant (i.e. $p<0.05$) indicating, as expected, differences in density between species within family, even after temporal trends have been incorporated. In addition 23 of the 84 species-ocean-agency interaction terms were statistically significant, suggesting that some differences in density between oceans and survey agencies remain even after the other factors have been accounted for.

3.5 Individual Area and Species Models

3.5.1 Frequency distributions of important covariates

In most cases, the data for individual species consisted of a fairly representative subset of the global data set with respect to most of the covariates. This is not surprising as the focus of these analyses were the most data-rich species, and areas which represented more than one third of the FAO-based data set (511 out of 1334 records). For analysis of individual species in all of the described FAO areas (67, 77, 21, 31 and 27) the most common survey method was D, which refers to Shipboard surveys not corrected for g(0). Harbour porpoise was the only species where method C was more common (aerial surveys corrected for g(0); see Figure 14). The most common survey agency was variable across species. NOAA was the most common agency for five of the species, NASS for one species and the 'other agencies' category for one species (Figure 15). For all species, the season when most of the surveys were conducted was summer (Figure 16). The distribution of surveys over years was skewed to the 1990's and 2000's for all species, except the Long-finned pilot whale (Figure 17).

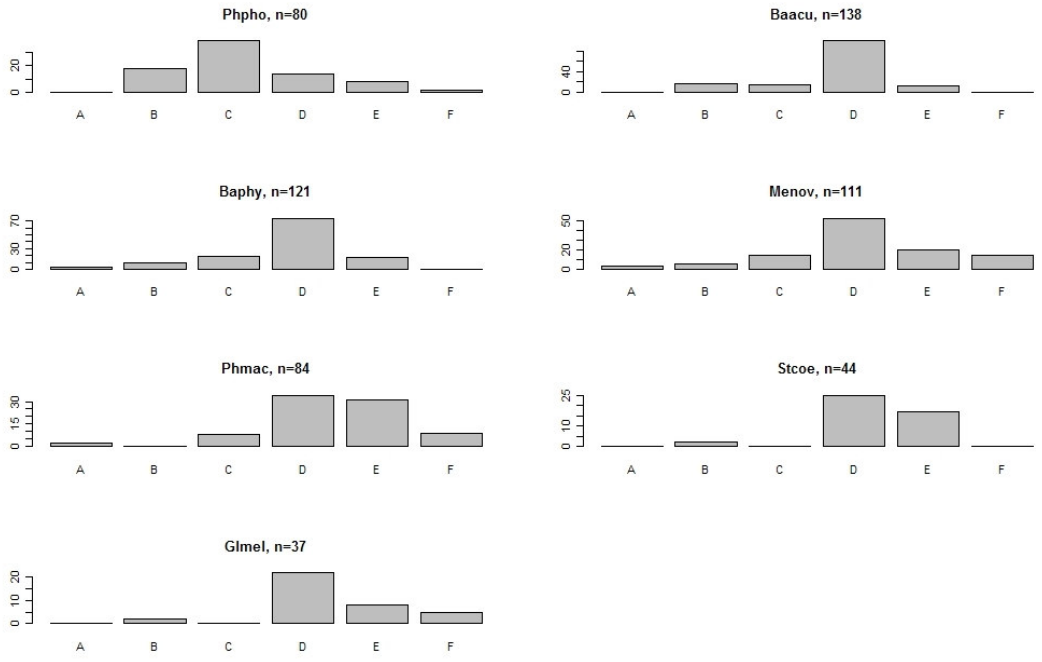

Figure 14. Frequency of surveys with survey methodologies for each species in all FAO areas (67, 77, 21, 31 and 27). Harbour porpoise (Phpho), Dwarf Minke whale (Baacu), Fin whale (Baphy), Humpback whale (Menov), sperm whale (Phmac), Long-finned pilot whale (Glmel).

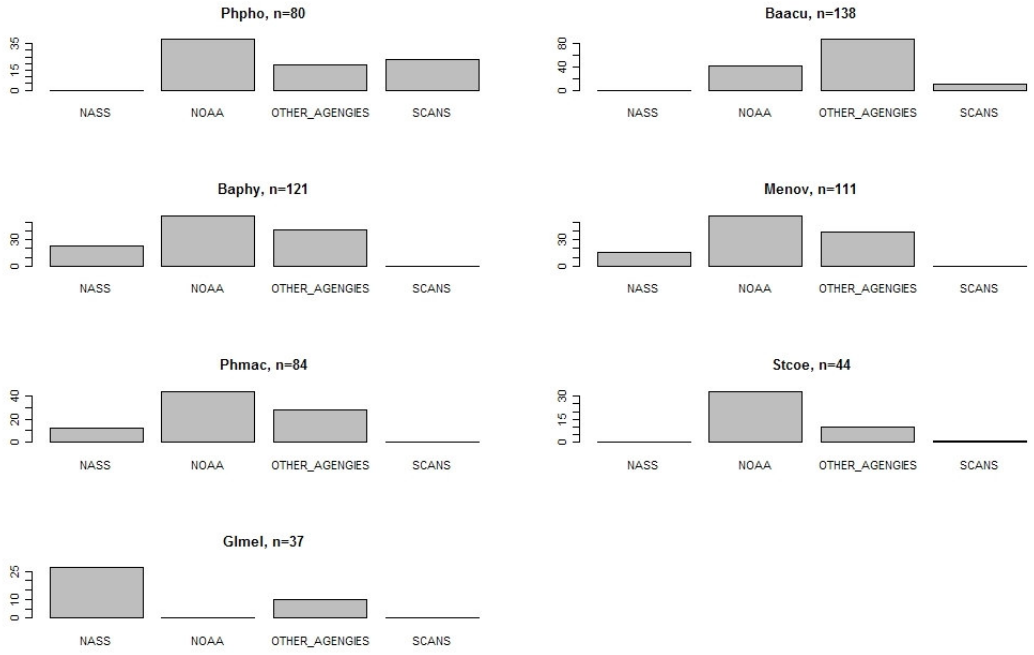

Figure 15. Frequency of surveys with agency for each species in all FAO areas (67, 77, 21, 31 and 27). Harbour porpoise (Phpho), Dwarf Minke whale (Baacu), Fin whale (Baphy), Humpback whale (Menov), Sperm whale (Phmac), Long-finned pilot whale (Glmel).

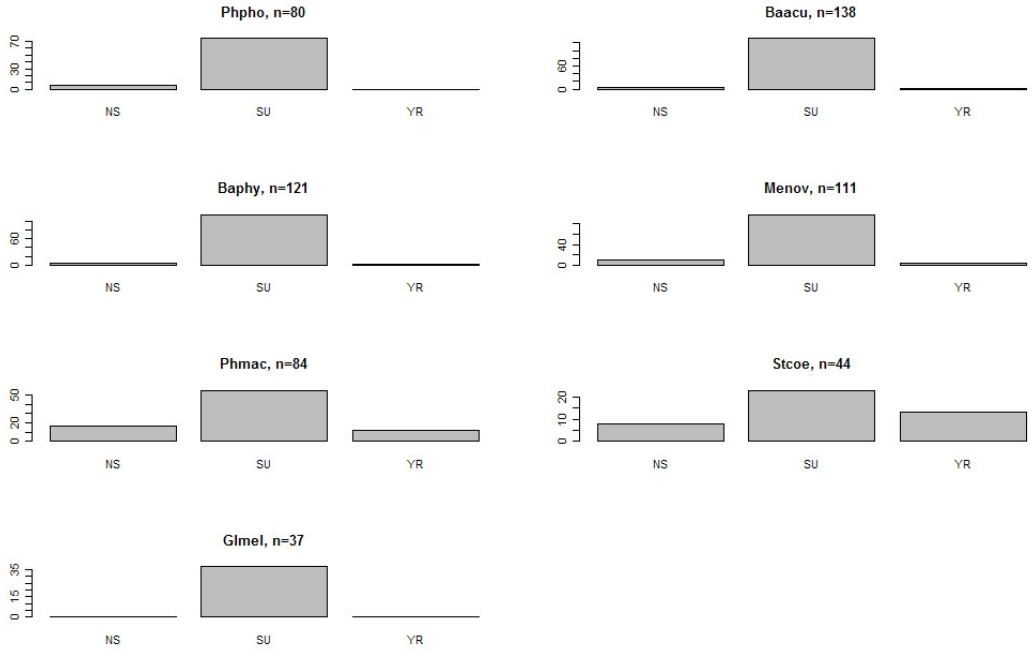

Figure 16. Number of density records with season. Summer (SU) refers to a generic summer that corresponds to the months between June-November in the Northern hemisphere and December – May in the Southern hemisphere. Likewise generic non-summer (NS) refers to remaining months in the respective hemispheres. Year-round surveys (YR) are surveys longer than 6 months in duration. Harbour porpoise (Phpho), Dwarf Minke whale (Baacu), Fin whale (Baphy), Humpback whale (Menov), Sperm whale (Phmac), Long-finned pilot whale (Glmel).

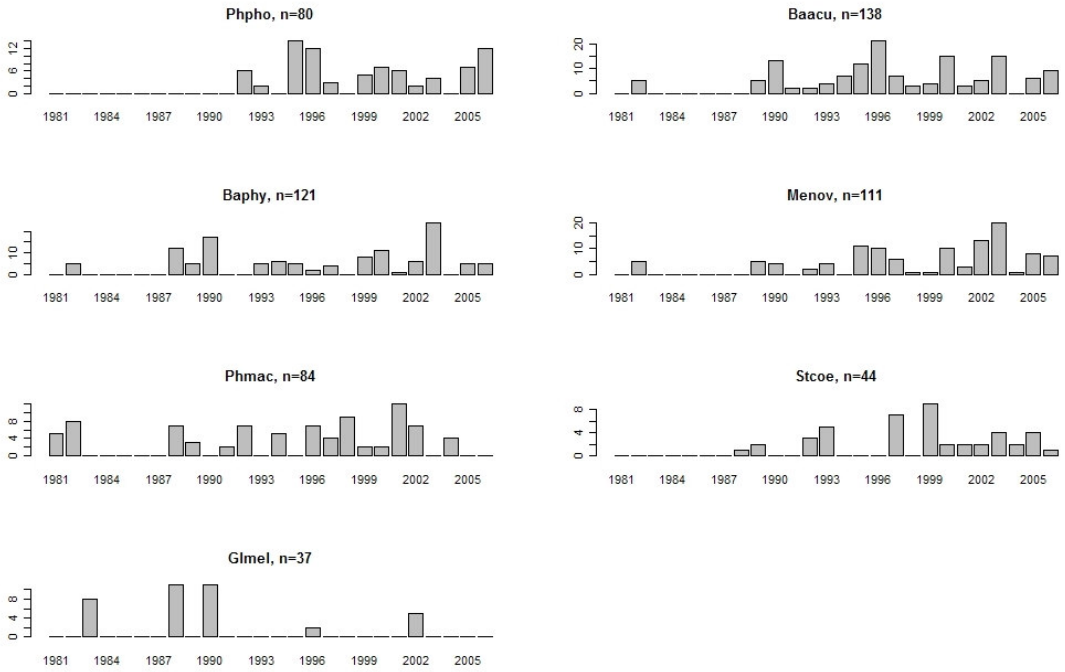

Figure 17. Frequency of surveys within year for each species in all FAO areas (67, 77 , 21 ,31 and 27). Harbour porpoise (Phpho), Dwarf Minke whale (Baacu), Fin whale (Baphy), Humpback whale (Menov), Sperm whale (Phmac), Long-finned pilot whale (Glmel).

3.5.2 Individual Species Results

3.5.2.1 Harbour porpoise

In all three focal areas, harbour porpoise has been surveyed fairly regularly over the past 10-15 years (Figure 18). Survey frequency, in terms of regular annual coverage, was highest in the north-eastern Pacific (FAO areas 67 & 77). Spatially, the most detailed information (in terms of number of abundance estimates provided for a specific survey area or block) was available for the north-eastern Atlantic (FAO area 27). However, temporal coverage in this area was very incomplete as comprehensive large scale surveys were only conducted twice in different decades (Figure 18). Geographic coverage was most extensive in the north-eastern Pacific; where surveys ranged from 30 to 55 degrees North. In contrast, latitudinal coverage in the other two areas was restricted to a narrow 10 degree latitudinal band, ranging between 40 – 48 degrees North along the east coast of the US and 48 – 58 degrees North in the north-western Atlantic. Within each of these areas, highest densities appear to have been observed consistently over time at approximately similar latitudinal ranges. Although, while highest densities were observed around 40 degrees North in the eastern Pacific, harbour porpoises in the Atlantic were most abundant between 44 – 46 degrees along the east coast of the US, and around 54 degrees North in the north-eastern Atlantic.

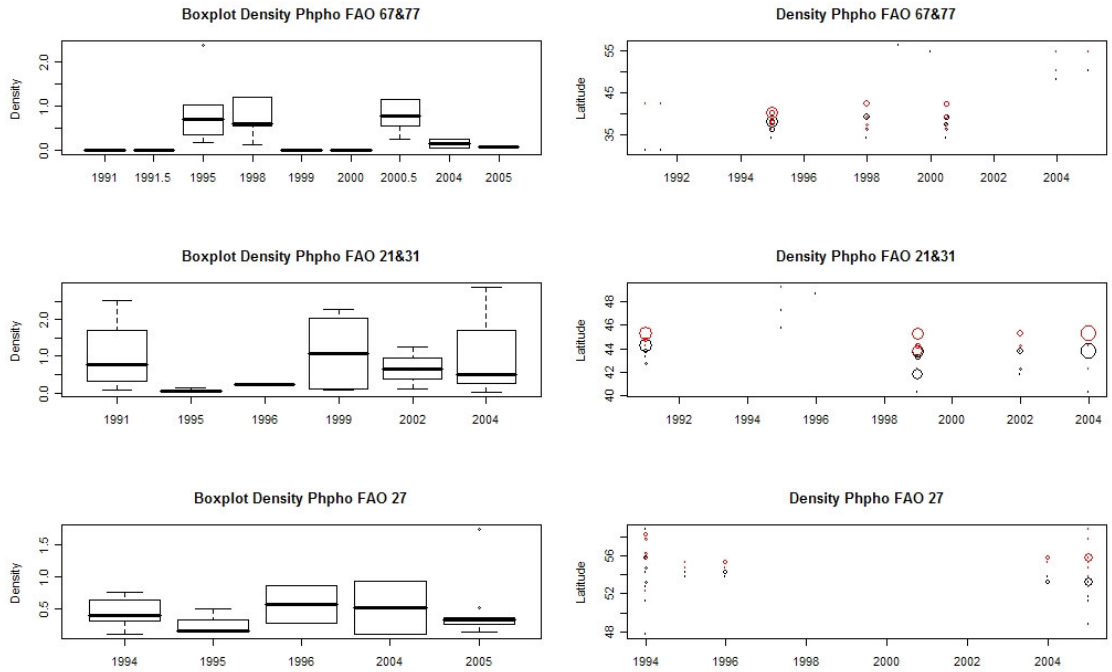

Figure 18. Harbour porpoise density through years and latitude for three areas described. Symbols on the right-hand side plots are proportional to the density. Pairs of circles show latitudinal survey extent with red representing the minimum latitude of each survey area and black the maximum.

Available survey estimates for this species were distributed relatively equally between the three focal areas. Application of the combined area and precision weighting did not reduce the sample sizes too much; all effective sample sizes were greater than 13 in each area (Table 9). In the single covariate analysis, the observed variation in harbour porpoise densities was best explained by a smoothed temporal covariate in the north-eastern Pacific [S(Year)]. While a smoothed geographic covariate explained the most deviance in the north- western Atlantic [S(Lat)]. A different temporal covariate, i.e. season performed best in the north-eastern Atlantic (Table 9, Part I). In the first two areas, slightly lower amounts of deviance could be explained by temporal trends, i.e. smoothed year of all available survey estimates or year as a factor, but GCVs were substantially higher in these cases (Table 9, Part I). However, in the north-eastern Atlantic, temporal trends cannot explain the observed variance very well at all. In all three areas, survey agency or method used to conduct the survey could only explain relatively small percentages of the overall deviance (Table 9, Part I).

In the multi-covariates analysis, combining smoothed minimum latitude with a smoothed temporal trend for the eastern Pacific could not explain more of the deviance. In addition the GCV value was slightly higher than in the single covariate analysis (Table 9, Part II). Similarly, in the north-western Atlantic, GCVs were consistently slightly higher when combining geographic covariates with smoothed or rounded temporal covariates than they were using geographic covariates alone. At the same time the percent deviance explained only increased marginally in most cases (Table 9, Part II). In the north-eastern Atlantic the proportion of deviance explained could be increased slightly with only a negligible increase in GCV by combining the best performing individual covariate (season) with a smoothed temporal covariate (Table 9, Part II).

Where modelling interactions between temporal and spatial covariates could be fit it did not produce better results in the single area, i.e. the north-eastern Pacific (Table 9, Part III).

In general, given our selected weighting of data points, the selected covariates performed best in the north-eastern Pacific and worst in the north-eastern Atlantic.

Table 9. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for Harbour porpoise. GCV is the generalized cross validation score, and s() refers to a covariate entering the model as a smooth, with a maximum of k knots (if k is shown) and NA means that the variable could not be fitted. Best models are highlighted in bold.

PHPHO	**67&77 (north-eastern Pacific) N=25 ESS=22.4**		**21&31 (north-western Atlantic) N=19 ESS=13.7**		**27 (north-eastern Atlantic) N=32 ESS=28.8**	
Variable	GCV	Deviance	GCV	Deviance	GCV	Deviance
S(Year)	**0.28234**	**88.3**	0.82161	37.2	0.26109	0.135
	k=9	**k=9**	k=6	k=6	k=5	k=5
Year	0.31054	89.2	0.92516	41.3	0.31007	3.93
Season	0.68029	51.1	NA	NA	**0.22381**	**14.4**
Agency	1.3404	3.61	0.6325	31.3	0.23739	9.2
Method	1.2104	20.4	0.7828	33.8	0.27509	1.68
S(MinLat)	NA	NA	0.69338	36.2	0.24346	12.5
S(Lat)	0.82893	49.5	**0.57764**	**47.3**	0.25072	9.1
S(MaxLat)	1.0636	31.3	0.55978	47.8	0.25345	7.62
			k=8	k=8		
Combining Temporal trends factor with other variables-main effect models						
S(Year)+Season	0.67632	55.5	NA	NA	**0.23198**	**17.1**
	k=9	k=9			**k=5**	**k=5**
Year+Season	0.31054	89.2	NA	NA	0.26401	24.2
Year+Agency	0.31054	89.2	0.92516	41.3	0.26401	24.2
Year+Method	0.34691	89.4	1.0637	51.7	0.32846	5.64
Year+S(MinLat)	0.089573	98.3	0.73753	60.1	0.30982	17
Year+S(Lat)	0.087182	98.5	0.67108	63.7	0.32001	13.3
Year+S(MaxLat)	0.088197	98.5	0.75052	59.4	0.32234	12.4
			k(lat)=8	k(lat)=8		
S(Year)+S(MinLat)	NA	NA	0.73852	39.8	0.25925	13.1
			k(year)=6	k(year)=6	k(year)=5	k(year)=5
S(Year)+S(Lat)	0.081676	98.4	**0.62409**	**49.6**	0.26638	10.1
	k(year)=9	k(year)=9	**k(year)=6**	**k(year)=6**	k(year)=5	k(year)=5
S(Year)+S(MaxLat)	**0.078131**	**98.3**	0.6241	48.6	0.27015	8.19

	k(year)=9	**k(year)=9**	k(year)=6 k(lat)=8	k(year)=6 k(lat)=8	k(year)=5	k(year)=5
Tensor product Functions- Temporal and spatial smooth variables.						
te(Year, Lat)	0.15391	97.7	NA	NA	0.27275	19.6
te(Year, MinLat)	0.31654	89.3	NA	NA	**0.26671**	**21.7**
te(Year, MaxLat)	**0.112**	**98.5**			0.2816	15.5

3.5.2.2 Dwarf Minke whale

In all three focal areas, Minke whales have been surveyed fairly regularly over the past 10-15 years (Figure 19). Survey frequency in terms of regular annual coverage was very consistent in all three areas. Latitudinal coverage was broadest in the north-western Atlantic, where surveys ranged between 30 degrees North to beyond 80 degrees North. However, spatial coverage was probably most comprehensive over time in the north-eastern Atlantic between 50 to 80 degrees North (Figure 19). In all three areas, highest densities appear to have been observed consistently over time north of 60 degrees North.

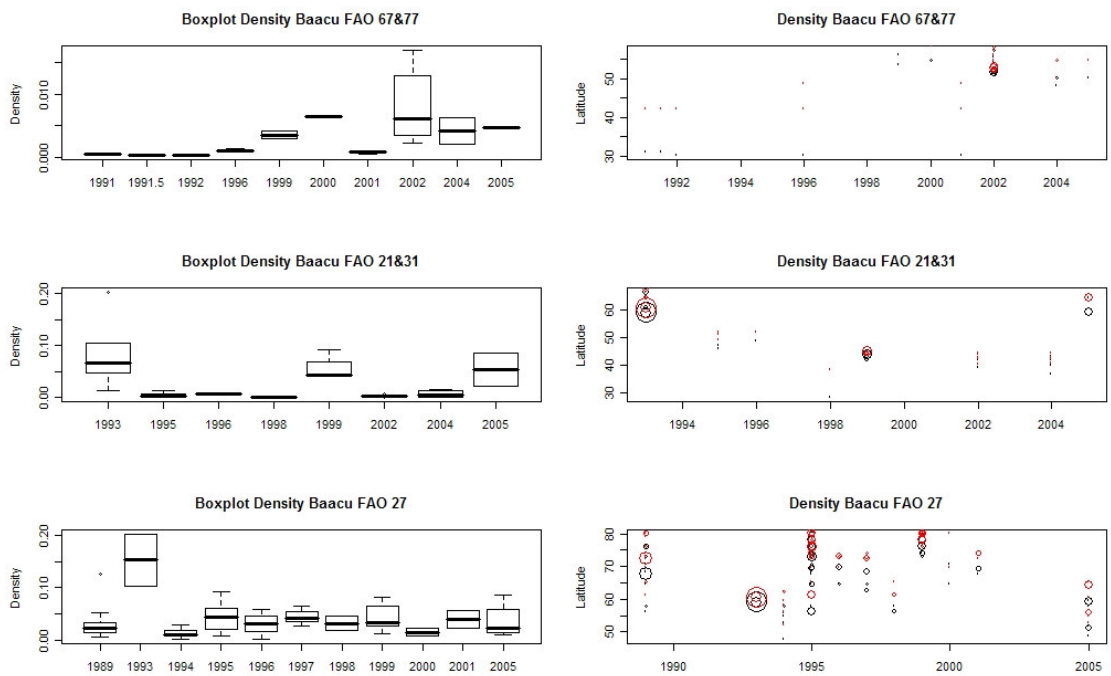

Figure 19. Dwarf Minke whale density through years and latitude for three areas described. Symbols on the right-hand side plots are proportional to the density (latitude minimum in black and latitude maximum in red). Pairs of circles show latitudinal survey extent with red representing the minimum latitude of each survey area and black the maximum.

Although the vast majority of survey estimates for this species stem from the north-eastern Atlantic, we had a minimum of 25 estimates from each area. However, the application of weighting factors reduced some of these sample sizes by almost 30%; though all ESS were > 17 (Table 10). In the single covariate analysis, the observed variation in Dwarf Minke whale densities was best explained by a smoothed temporal trend in the north-eastern Pacific [S(Year)], while Year as a covariate performed

best in the north-western Atlantic. In the north-eastern Atlantic, Year as a covariate also explained most of the deviance, but GCV was lowest for Method (Table 10, Part I). In the north-eastern Pacific, smoothed geographic covariates also explained relatively large proportions of the deviance, but those models were all associated with higher GCVs. (Table 10, Part I).

In the multi-covariate analysis, combining smoothed maximum latitude with a rounded Year covariate resulted in the lowest overall GCV and explained 97.3% of the deviance, thus representing the best model for this species in FAO areas 67 & 77. Combining the best single covariate (Year) in the north-western Atlantic with a smoothed geographical covariate [S(MinLat) improved GCV and explained deviance substantially. This represented the best model to explain variation in reported Minke whale density estimates in this area. In the north-eastern Atlantic the model with the lowest GCV did not explain as much of the deviance as the best single covariate model (Table 10, Part II).

Modelling interactions between smoothed continuous temporal and spatial covariates did not produce better results in any of the three areas (Table 10, Part III).

In general, the selected covariates, given our selected weighting of data points, performed best in the north-eastern Pacific and worst in the north-eastern Atlantic.

Table 10. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for Dwarf Minke Whale. GCV is the generalized cross validation score, *S* refers to a smooth factor, te is tensor product function, k means number of knots fitted and NA means that the variable could not be fit. Best models are highlighted in bold.

BAACU	**67&77 (north-eastern Pacific) N=26 ESS=21.6**		**21&31 (north-western Atlantic) N=25 ESS=17.2**		**27 (north-eastern Atlantic) N=66 ESS=52.6**	
Variable	GCV	Deviance	GCV	Deviance	GCV	Deviance
S(Year)	**0.13939**	**90.1**	0.86716	3.43	0.21875	1.12
			k=8	K=8		
Year	0.15761	90.4	**0.38443**	**76.6**	0.22339	24.9
Season	0.47383	40.1	NA	NA	NA	NA
Agency	0.71517	1.6	0.89062	0.821	**0.1942**	**11.6**
Method	0.16422	79.2	0.80094	25.6	0.21661	1.36
S(MinLat)	0.16222	83.4	0.82528	8.1	0.19765	9.99
S(Lat)	0.16392	83.2	0.8347	7.05	0.2045	6.87
S(MaxLat)	0.19107	78.5	0.84401	6.01	0.20965	4.53
	k=5	k=5				
Combining Temporal trends factor with other variables-main effect models						
S(Year)+Season	0.12287	88.7	NA	NA	NA	NA
Year+Season	0.15761	90.4	NA	NA	NA	NA
Year+Agency	0.15761	90.4	0.38443	76.6	0.23146	25

Year+Method	0.15761	90.4	0.38443	76.6	0.23146	25
Year+S(MinLat)	0.15374	91.7	**0.30117**	**92.2**	0.219	29
round(Year)+S(Lat)	0.15033	91.9	0.34346	83.7	0.22456	28.6
round(Year)+S(MaxLat)	**0.093149**	**97.3**	0.34706	83.3	0.23	25.4
S(Year)+S(MinLat)	NA	NA	0.50868	65.5	0.20322	11
			k(year)=8	k(year)=8		
S(Year)+S(Lat)	NA	NA	0.60969	59.3	**0.21021**	**7.24**
			k(year)=8	k(year)=8		
S(Year)+S(MaxLat)	NA	NA	0.54069	63.2	0.21526	5.54
			k(year)=8	k(year)=8		
Tensor product Functions- Temporal and spatial smooth variables.						
Te(Year, Lat)	NA	NA	**0.41086**	**79.9**	0.21659	8.27
te(Year, MinLat)	**0.10249**	**96.7**	0.41119	79.9	**0.2097**	**11.2**
te(Year, MaxLat)	NA	NA	0.434	78.8	0.22086	6.83

3.5.2.3 Fin whale

Over the past 10-15 years, Fin whales have been surveyed the most in the north-eastern Pacific (FAO 67&77); with density estimates every year since 1991 (Figure 20). In the north-western Atlantic (FAO 21&31), there are consistent surveys from 1993 until 2005, giving good temporal coverage over this time frame. In the north-eastern Atlantic (FAO 27) larger time periods between density estimates means there is less complete temporal coverage (Figure 20).

Geographic coverage was most extensive in the north-eastern Pacific where surveys ranged from 20 to 60 degrees North. In the north-western Atlantic coverage was over a 35 degree band and in the north-eastern Atlantic was over a 25 degree latitudinal band, ranging between 50 to 75 degrees North. In the north-eastern Pacific, the highest estimated densities were between 50 and 60 degrees. In the north-western Atlantic, in the band from 40-50 degrees, highest densities seem consistent over years, but within the north-eastern Atlantic, highest densities fell between 60 and 70 degrees.

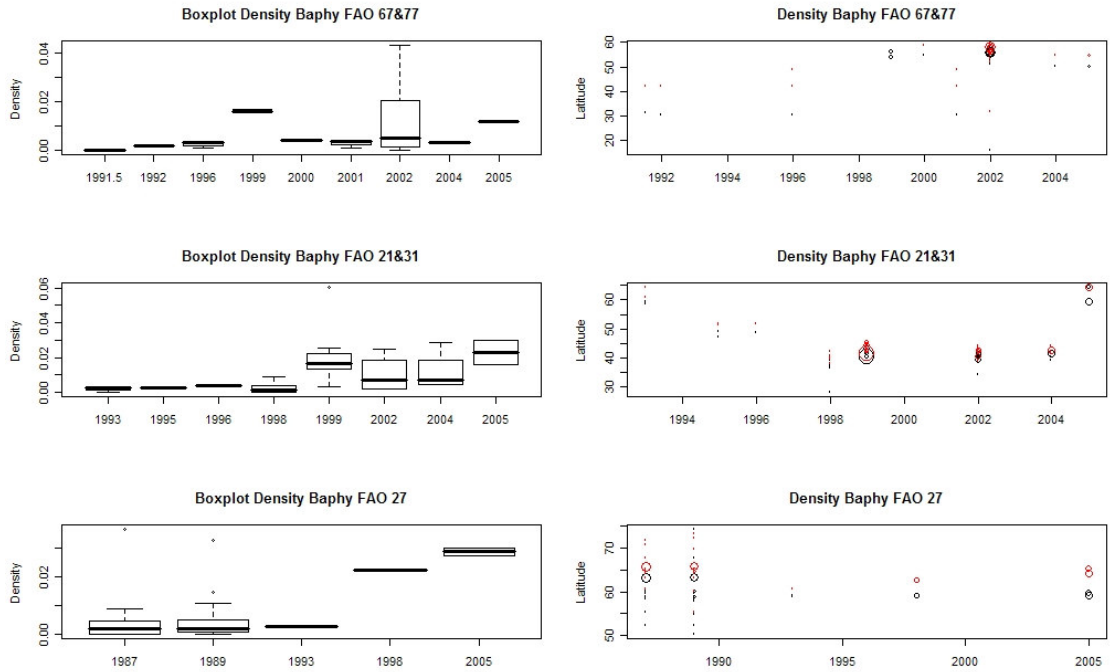

Figure 20. Fin whale density through years and latitude for three areas described. Symbols on the right-hand side plots are proportional to the density. Pairs of circles show latitudinal survey extent with red representing the minimum latitude of each survey area and black the maximum.

Available survey estimates for this species were distributed between the three focal areas, with similar numbers in the north-eastern and north-western Atlantic and least in the north-eastern Pacific. Application of the combined area and precision weighting did not reduce the sample sizes overly much, with all effective sample sizes still being greater than 23 samples in each area (Table 11). In the single covariate analysis, the observed variation in Fin whale densities was best explained by S(MaxLat) in the north-eastern Pacific, S(MinLat) in the north-eastern Atlantic, and a S(Year) in the north-western Atlantic (Table 11, Part I). Within the north-east Atlantic, the amount of deviance explained by S(Year) was very small, and no other models explained significantly more of the deviance.

In the north-eastern Pacific combining two covariates explained less of the deviance than the single temporal covariate of S(MaxLat). This can happen due to changes in the smoothness estimated for the functions. However, in the north-western Atlantic, combining S(Year) with the single covariate S(MinLat) explained more deviance; and hence gave a better model. In the north-eastern Atlantic, adding Agency to Year did increase the percentage deviance, but the final value was still only 18%. For this area no other models performed any better.

Modelling interactions between temporal and spatial factors only worked (i.e. the models converged to produce valid estimates) in the north-eastern Pacific, but they explained no more of the deviance than the best model combining temporal trends. (Table 11 part III).

In general, the selected covariates, given our selected weighting of data points, performed best in the north-eastern Pacific and worst in the north-western Atlantic.

Table 11. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for Fin whale. GCV is the generalized cross validation score, *S* refers to a smooth factor, te is tensor product function, k means number of knots fitted and NA means that the variable could not be fit. Best models are highlighted in bold.

BAPHY	**67&77 (north-eastern Pacific) N=28 ESS=23.2**		**21&31 (north-western Atlantic) N=36 ESS=28.7**		**27 (north-eastern Atlantic) N=34 ESS=27.8**	
Variable	GCV	Deviance	GCV	Deviance	GCV	Deviance
S(Year)	NA	NA	0.41193	38.9	**0.90042**	**6.36**
			k=8	k=8	**k=5**	**k=5**
Year	0.90927	51.5	0.4678	42.8	1.0779	7.94
Season	0.87512	19.1	0.49493	10.7	NA	NA
Agency	0.9991	0.134	0.54261	2.09	0.96088	0.075
Method	0.5321	50.8	0.48812	22	0.95305	0.889
S(MinLat)	0.56135	43.9	**0.39539**	**50.8**	NA	NA
S(Lat)	0.56338	45.1	0.43782	41.7	NA	NA
S(MaxLat)	**0.37158**	**78.2**	0.55417	0.00129	NA	NA
Combining Temporal trends factor with other variables-main effect models						
S(Year)+Season	NA	NA	0.36503	49.8	NA	NA
			k=8	k=8		
Year+Season	0.90927	51.5	0.41364	52.9	NA	NA
Year+Agency	0.90927	51.5	0.4678	42.8	**1.0208**	**18.7**
Year+Method	0.65841	68.5	0.38738	59.1	1.0779	7.94
Year+S(MinLat)	NA	NA	0.48536	44.8	NA	NA
Year+S(Lat)	NA	NA	0.49933	43.2	NA	NA
Year+S(MaxLat)	NA	NA	0.50303	42.8	NA	NA
S(Year)+S(MinLat)	0.60368	44.2	**0.30011**	**65.8**	NA	NA
	k(year)=9	k(year)=9	**k(year)=8**	**k(year)=8**		
S(Year)+S(Lat)	**0.57497**	**46.9**	0.33519	62.9	NA	NA
	k(year)=9	**k(year)=9**	k(year)=8	k(year)=8		
S(Year)+S(MaxLat)	NA	NA	0.43498	46.7	NA	NA
			k(year)=8	k(year)=8		
Tensor product Functions- Temporal and spatial smooth variables.						
Te(Year, Lat)	**0.62298**	**46.9**	NA	NA	NA	NA
te(Year, MinLat)	0.64555	45	NA	NA	NA	NA
te(Year, MaxLat)	0.62608	46.7	NA	NA	NA	NA

3.5.2.4 Humpback whale

Over the past 10-15 years, the Humpback whale has been surveyed the most in the north-eastern Pacific (FAO 67&77) with density estimates every year since 1991 (Figure 21). In the north-western Atlantic (FAO 21&31), there were consistent surveys from 1995 until 2005, giving good temporal coverage over this time period. In the north-eastern Atlantic (FAO 27), there was less complete temporal coverage with larger time frames between density estimates (Figure 21).

Geographic coverage was reasonably equal across areas covering from 25 to 30 degrees of latitude in each area. In the north-eastern Pacific, highest densities appear to be between 45 and 60 degrees from 2002 onwards. Prior to 2002, density across the latitude range was low. In the north-western Atlantic, the highest density was in the lower latitudes up until 2004. Post 2004, density estimates were only from higher latitudes. In the north-eastern Atlantic, the highest density appeared to be consistent across latitudes over years.

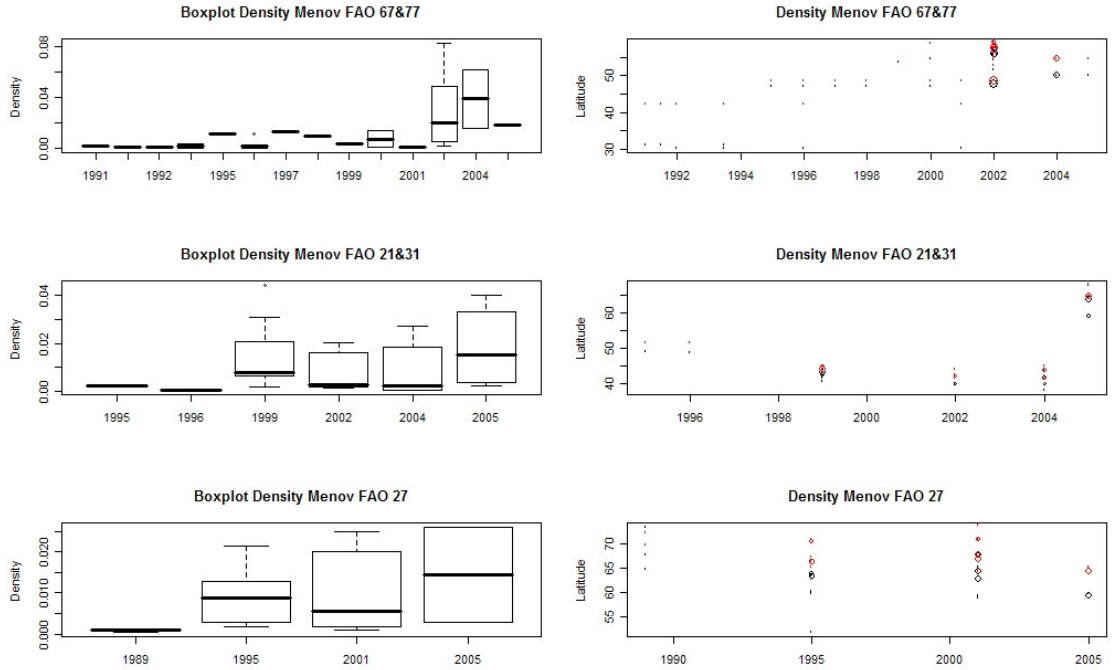

Figure 21. Humpback whale density through years and latitude for three areas described. Symbols on the right-hand side plots are proportional to the density. Pairs of circles show latitudinal survey extent with red representing the minimum latitude of each survey area and black the maximum.

Available survey estimates for this species were distributed between the three focal areas, with the majority in the north-eastern Pacific. Application of the combined area and precision weighting did not reduce the sample sizes overly much, with all effective sample sizes still being greater than 16 samples in each area (Table 12). In the single covariate analysis, the observed variation in Humpback whale densities was best explained by a smoothed geographic covariate, in the north-eastern Pacific,

[S(MinLat)], and the north-eastern Atlantic, [S(Lat)]. However, a survey related covariate (Method) performed best in the north-western Atlantic (Table 12, Part I).

In the north-eastern Pacific, incorporation of temporal trends by combining covariates explained less of the deviance than the single covariate[S(MinLat)] (Table 12 part II). In the north-western Atlantic, combining Year and Agency explained more of the deviance than the single covariate analysis. Method could not be combined in the combined temporal trend analysis. Combining covariates in this area consistently explained more of the deviance than in the single covariates models, with many of the GCV values being only slightly higher. It is worth noting that some of the models combining temporal and geographical covariates had only marginally higher GCV values than the best model. In the north-eastern Atlantic, combining a temporal with a geographic covariate explained the most deviance, with two models containing Year and a latitude covariate having similar GCV scores and deviance explained. In the north-eastern Atlantic it was also clear that combining covariates produced models that consistently explained more deviance, and generally had lower GCV values, than the single covariate models (Table 12 part II).

Modelling interactions between temporal and spatial covariates only produced better results in the north-western Atlantic, as the models could not be fitted in the other areas (Table 12, Part III).

In general, the selected covariates, given our selected weighting of data points, performed best in the north-eastern Pacific and worst in the north-western Atlantic.

Table 12. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for Humpback whale. GCV is the generalized cross validation score, *S* refers to a smooth factor, te is tensor product function, k means number of knots fitted and NA means that the variable could not be fit. Best models are highlighted in bold.

MENOV	**67&77 (north-eastern Pacific) N=40 ESS=34.7**		**21&31 (north-western Atlantic) N=25 ESS=19.8**		**27 (north-eastern Atlantic) N=21 ESS=17.4**	
Variable	GCV	Deviance	GCV	Deviance	GCV	Deviance
S(Year)	NA	NA	0.59955	27.8	0.56792	41.2
			k=6	k=6	k=4	k=4
Year	1.1411	60.4	0.66088	34.7	0.57741	41.9
Season	1.1286	21.5	NA	NA	NA	NA
Agency	1.3558	0.477	0.69075	0.0209	0.65041	18.3
Method	0.62369	56.6	**0.58836**	**29**	0.65041	18.3
S(MinLat)	**0.59759**	**71.9**	0.67828	1.82	0.72125	21
S(Lat)	0.68998	64.6	0.68086	1.45	**0.55014**	**40.8**
S(MaxLat)	0.82361	48.6	0.68397	1	0.58348	36.6
Combining Temporal trends factor with other variables-main effect models						
S(Year)+Season	NA	NA	NA	NA	NA	NA
Year+Season	1.1411	60.8	NA	NA	NA	NA

Year+Agency	1.1411	60.8	**0.66088**	**34.7**	0.57741	41.9
Year+Method	**1.0903**	**65.4**	NA	NA	0.57741	41.9
Year+S(MinLat)	1.2151	61.4	0.73636	34.7	0.51931	63.8
Year+S(Lat)	1.2083	61.6	0.7292	35.4	NA	NA
Year+S(MaxLat)	1.2028	61.8	0.68906	38.9	0.55205	57.7
S(Year)+S(MinLat)	NA	NA	0.74128	1.83	0.51444	63.1
			k(year)=6	k(year)=6	k(year)=4	k=4
S(Year)+S(Lat)	NA	NA	0.74393	1.48	**0.46466**	**63.5**
			k(year)=6	k(year)=6	**k(year)=4**	**k(year)=4**
S(Year)+S(MaxLat)	NA	NA	0.7469	1.09	0.54151	56.9
			k(year)=6	k(year)=6	k(year)=4	k(year)=4
Tensor product Functions- Temporal and spatial smooth variables.						
te(Year, Lat)	NA	NA	0.66964	28	NA	NA
te(Year, MinLat)	NA	NA	0.69078	31.3	NA	NA
te(Year, MaxLat)	NA	NA	**0.62608**	**46.7**	NA	NA

3.5.2.5 Sperm whale

Over the past 10-15 years, Sperm whales have been surveyed the most in the north-eastern Pacific (FAO 67&77), with density estimates every year since 1988 (Figure 22). In the north-western Atlantic (FAO 21&31), there are consistent surveys between 1992 and 2004, giving good temporal coverage. In the north-eastern Atlantic (FAO 27), there are only two years of survey data, this giving poor temporal coverage (Figure 22).

Geographic coverage was most extensive in the north-eastern Pacific where surveys ranged from -20 to 40 degrees north. In the north-western Atlantic coverage was over a 30 degree band, but in contrast, latitudinal coverage in the north-eastern Atlantic was restricted to a 15 degree latitudinal band, ranging between 55 and 70 degrees North. In the north-eastern Pacific densities appear uniform across the broad latitudinal range, but in the north-western Atlantic, highest densities are found between 35 and 45 degrees. There is no clear density pattern from the north-eastern Atlantic.

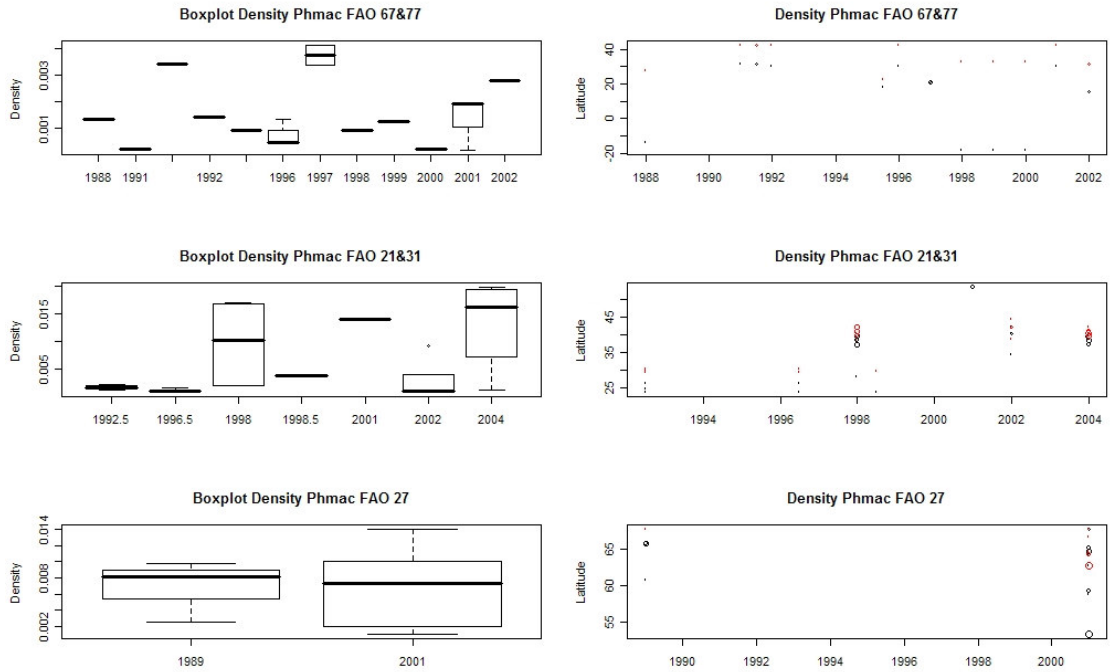

Figure 22. Sperm whale density through years and latitude for three areas described. Symbols on the right-hand side plots are proportional to the density. Pairs of circles show latitudinal survey extent with red representing the minimum latitude of each survey area and black the maximum.

Available survey estimates for this species were distributed between the three focal areas, with similar numbers in the north-eastern Pacific and north-western Atlantic, and the least in the north-eastern Atlantic. Application of the combined area and precision weighting reduced the sample sizes to between 9 and 15 (Table 12). In the single covariate analysis, the observed variation in Sperm whale densities was best explained by Year in the north-eastern Pacific, season in the north-eastern Atlantic, and a S(MinLat) in the north-western Atlantic (Table 13, Part I).

In the north-eastern Pacific, combining two temporal covariates did not explain any more of the deviance than the single temporal covariate of Year. However, in the north-western Atlantic, combining Year and Method explained more of the deviance than season on its own. Combining Year with geographic latitude covariates also explained a large amount of deviance, but all had slightly higher GCV values. Combining covariates in the north-eastern Atlantic gave mixed results. The model with the lowest GCV value explained only 0.091 % of the deviance, so it is worth considering the three models of Year + S (Latitude variables) which have higher GCV values but explain much more deviance.

Modelling interactions between temporal and spatial factors did not work in any of the areas (Table 13 part III).

In general the selected covariates, given our selected weighting of data points, performed best in the north-eastern Pacific and worst in the north-western Atlantic.

Table 13. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for Sperm whale. GCV is the generalized cross validation score, *S* refers to a smooth factor, te is tensor product function, k means number of knots fitted and NA means that the variable could not be fit. Best models are highlighted in bold.

PHMAC	**67&77 (north-eastern Pacific) N=22 ESS=15**		**21&31 (north-western Atlantic) N=23 ESS=15.3**		**27 (north-eastern Atlantic) N=11 ESS=9.7**	
Variable	GCV	Deviance	GCV	Deviance	GCV	Deviance
S(Year)	0.24452	1.04	0.21482	68.7	NA	NA
			k=7	k=7		
Year	**0.16697**	**83.1**	0.21888	62.5	0.36887	0.091
Season	0.21753	20.5	**0.14402**	**68.4**	NA	NA
Agency	NA	NA	0.24951	45.3	NA	NA
Method	0.19753	35.2	0.15861	68.6	0.36887	0.091
S(MinLat)	0.208	23.6	NA	NA	**0.32932**	**26.5**
	k=8	k=8				
S(Lat)	0.1705	45.5	NA	NA	9.3021	64.7
	k=9	k=9				
S(MaxLat)	0.18112	26.7	0.14896	78.4	7.5351	72.5
	k=6	k=6			k(lat)=9	k(lat)=9
Combining Temporal trends factor with other variables-main effect models						
S(Year)+Season	0.2304	31	0.15813	69.7	NA	NA
			k=7	k=7		
Year+Season	**0.16697**	**83.1**	0.21636	73.3	NA	NA
Year+Agency	NA	NA	0.24327	70	NA	NA
Year+Method	0.20317	83.3	**0.10637**	**88.6**	**0.36887**	**0.091**
Year+S(MinLat)	NA	NA	0.1345	85.6	0.42192	25.6
Year+S(Lat)	NA	NA	0.13587	85.7	0.38917	55.6
Year+S(MaxLat)	NA	NA	0.15176	88.8	0.45327	3.02
					k(lat)=9	k(lat)=9
S(Year)+S(MinLat)	0.23109	23.6	NA	NA	NA	NA
	k(lat)=8	k(lat)=8				
S(Year)+S(Lat)	0.18973	46	NA	NA	NA	NA
	k(lat)=9	k(lat)=9				
S(Year)+S(MaxLat)	0.20038	26.8	0.16836	77.8	NA	NA
	k(lat)=6	k(lat)=6	k(year)=7	k(year)=7		

Tensor product Functions- Temporal and spatial smooth variables.						
te(Year, Lat)	NA	NA	NA	NA	NA	NA
te(Year, MinLat)	NA	NA	NA	NA	NA	NA
te(Year, MaxLat)	NA	NA	NA	NA	NA	NA

3.5.2.6 Striped dolphin & Long-finned pilot whale

Over the past 10-15 years Striped dolphins have been surveyed the most in the north-eastern Pacific (FAO 67&77); with density estimates every 1-4 years since 1988 (Figure 23). In the north-western Atlantic (FAO 21&31) there are consistent surveys between 1992 and 2004, giving good temporal coverage over this time frame. There are no density estimates for this species from the north-eastern Atlantic, as most surveys in this region have been conducted either outside or along the northern fringes of this species known distribution (see e.g. Reid et al, 2003). Geographic coverage was most extensive in the north-eastern Pacific where surveys ranged from -20 to 40 degrees North. In contrast, latitudinal coverage in the north-western Atlantic was restricted to a 15 degree latitudinal band, ranging between 25 to 40 degrees North. In the north-eastern Pacific highest densities appear to have been observed consistently over time at approximately similar latitudinal ranges, with minimum density estimates between -10 and 20 degrees North. In the north-western Atlantic highest densities were consistently between 30 and 40 degrees North.

Specific density estimates for Long-finned pilot whales were only available from the north-eastern Atlantic. In the north-western Atlantic, its distribution starts to strongly overlap with the closely related, and difficult to distinguish, short-finned pilot whale along the coast of the US .Thus, NOAA surveys in the western Atlantic, in general, only report estimates for generic *Globicephalus spp*. Long-finned pilot whales are not known to occur in the northern Pacific. Temporal coverage in the north-eastern Atlantic was sporadic, with estimates from only 3 years (Figure 23). Geographic coverage was restricted to a 14 degree latitude band with the highest densities spread evenly across this range.

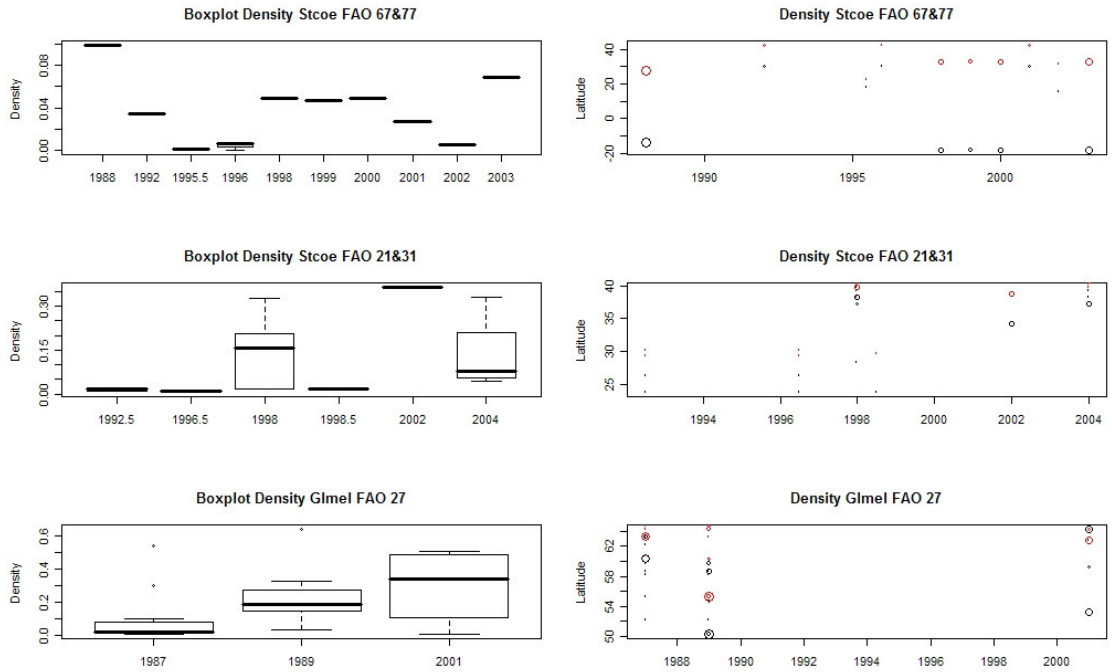

Figure 23. Long-finned pilot whale (Glmel) and Striped dolphin (Stcoe) density through years and latitude for three areas described. Symbols on the right-hand side plots are proportional to the density. Pairs of circles show latitudinal survey extent with red representing the minimum latitude of each survey area and black the maximum.

Available survey estimates for Striped dolphin were distributed evenly between the north-eastern Pacific and north-western Atlantic, with only slightly more in the north-western Atlantic. Application of the combined area and precision weighting reduced the sample sizes, with the effective sample sizes being eight and five respectively (Table 14). In the single covariate analysis, the observed variation in Striped dolphin densities was best explained by Method in the north-eastern Pacific (Table 14 Part I). However, Year explained 87% of the deviance but the GCV score was 0.15 higher than the Method model. A smoothed geographic covariate [S(MinLat)] was the best model in the north-western Atlantic. In the north-eastern Pacific, incorporation of temporal trends by combining covariates gave two models with equal GCV and deviance scores. These were Year + Season and Year + Method (Table 14 part II). In the north-western Atlantic, S(Year) + S(Lat) gave the best model. Combining a temporal with a geographic covariate in both the north-eastern Pacific and the north-western Atlantic for Striped dolphins consistently explained more of the deviance than in the single covariate models.

Survey estimates for Long-finned pilot whales were only available in the north-eastern Atlantic. Application of the combined area and precision weighting gave an effective sample size of 24 (Table 14). In the single covariate analysis, the observed variation in Long-finned pilot whale densities was poorly explained by all the covariates. A smoothed geographic covariate [S(LatMax)] was the best model, but only explained 9% of the deviance (Table 14 Part I). Incorporation of temporal trends by combining covariates did increase the deviance explained, but only by a small percentage (Table 14 part II). The best combined model was S(Year) + S(MaxLat). None of the model combinations explained more than 17% of the deviance.

Modelling interactions between temporal and spatial covariates did not work for either species in any of the areas (Table 14 part III).

In general, the selected covariates, given our selected weighting of data points, performed best in the north-western Atlantic for Striped dolphins, but performed poorly for Long-finned pilot whales in the north-eastern Atlantic.

Table 14. Weighted Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) for Long-finned pilot whale (Glmel) and Striped dolphin (Stcoe). GCV is the generalized cross validation score, *S* refers to a smooth factor, te is tensor product function, k means number of knots fitted and NA means that the variable could not be fit. Best models are highlighted in bold.

	STCOE				**Glmel**	
	67&77 (north-eastern Pacific) N=14 ESS=8.4		**21&31 (north-western Atlantic) N=18 ESS=5.2**		**27 (north-eastern Atlantic) N=26 ESS=24.3**	
Variable	GCV	Deviance	GCV	Deviance	GCV	Deviance
S(Year)	0.27934	23.6	0.12869	48.9	0.97543	6.99
			k=6	k=6	k=3	k=3
Year	0.36678	87.5	0.08009	77.5	1.0037	9.74
Season	0.22025	43.2	0.16948	25.6	NA	NA
Agency	NA	NA	0.08545	57.3	NA	NA
Method	**0.20645**	**46.8**	0.14479	27.7	0.99254	2.81
S(MinLat)	0.22612	30.7	**0.025036**	**90.5**	0.98971	3.09
	k=3	k=3	**k=8**	**k=8**		
S(Lat)	0.20976	35.7	0.028411	90.9	0.96251	5.75
	k=6	k=6				
S(MaxLat)	0.23722	27.3	0.060719	74.8	**0.9252**	**9.41**
	k=6	k=6	k=9	k=9		
Combining Temporal trends factor with other variables-main effect models						
S(Year)+Season	0.21668	53.9	0.078333	77.1	NA	NA
			k=6	k=6		
Year+Season	**0.36678**	**87.5**	0.08009	77.5	NA	NA
Year+Agency	NA	NA	0.08009	77.5	NA	NA
Year+Method	**0.36678**	**87.5**	0.045104	89.4	1.0037	9.74
Year+S(MinLat)	NA	NA	0.035606	93.3	1.076	11.5
			k=8	k=8		
Year+S(Lat)	NA	NA	0.039294	92.8	1.0498	13.6

Year+S(MaxLat)	NA	NA	0.057996 k=9	93.3 k=9	0.9975	17.9
S(Year)+S(MinLat)	0.21082 k(year)=3	46.5 k(year)=3	NA	NA	1.0207	8.21
S(Year)+S(Lat)	NA	NA	**0.030682** **k(year)=6**	**91.6** **k(year)=6**	0.9834 k(year)=3	11.6 k(year)=3
S(Year)+S(MaxLat)	NA	NA	0.062951 k(year)=6 k(lat)=9	77.1 k(year)=6 k(lat)=9	**0.92045** **k(year)=3**	**17.2** **k(year)=3**
Tensor product Functions- Temporal and spatial smooth variables.						
te(Year, Lat)	NA	NA	NA	NA	NA	NA
te(Year, MinLat)	NA	NA	NA	NA	NA	NA
te(Year, MaxLat)	NA	NA	NA	NA	NA	NA

3.6 Power Analysis

The estimated CVs for the population change index Δ varied widely from 0.12 to 1.79, with a mean of 0.85 (Table 15). Surprisingly, there was no positive relationship between the fit of the model used (in terms of the percentage deviance explained) and the CV (Δ). For example the model used for harbour porpoise in the eastern Pacific explained 98% of the deviance in the (weighted) data, but the CV(Δ) was 1.43. The model used for the same species in the north-eastern Atlantic explained only 13% of the deviance, but had an estimated CV(Δ) of 0.29. This lack of a positive relationship warrants further investigation.

The relationship between population change Δ, CV(Δ) and power is shown in Figure 24. Given the large values estimated for CV(Δ) in most cases, power is low to detect anything but the largest population changes. For example, at the mean CV(Δ) of 0.85, a population change of approximately 0.2 or 5 (i.e. a decline or increase in population size of half an order of magnitude) would be detectable with a power of 0.8 (a common benchmark for acceptable level of power). At the lowest estimated CV(Δ) of 0.12, very small population changes of the order of 0.95 or 1.05 (i.e. a 5% increase or decline) would be observable with high power; conversely at the highest estimated CV(Δ) of 1.79, even an order of magnitude change in population size would only be detectable with a power of around 0.3.

Table 15. The estimated CVs for the population change index Δ

Species	FAO Area	Model	CV(Δ)
Harbour porpoise	67 & 77	s(Year) + s(MinLat)	1.43
	21 & 31	s(Year) + s(Lat)	1.30
	27	s(Year) + s(MinLat)	0.29
Minke whale	67 & 77	s(Year) + s(Lat)	0.67
	21 & 31	s(Year) + s(Lat)	1.74
	27	s(Year) + s(MinLat)	0.24
Fin whale	67 & 77	s(Year) + s(Lat)	0.53
	21 & 31	s(Year) + s(MinLat)	0.98
	27	s(Year)	0.98
Humpback whale	67 & 77	s(Year, MinLat)	1.79
	21 & 31	s(Year)	1.58
	27	s(Year)	0.81
Sperm whale	67 & 77	s(Year) + s(Lat)	0.63
	21 & 31	s(Year)	1.14
	27	NA	NA
Striped dolphin	67 & 77	s(Year) + Season	0.18
	21 & 31	s(Year)	0.12
Long-finned pilot whale	27	s(Year) + s(MaxLat)	0.17

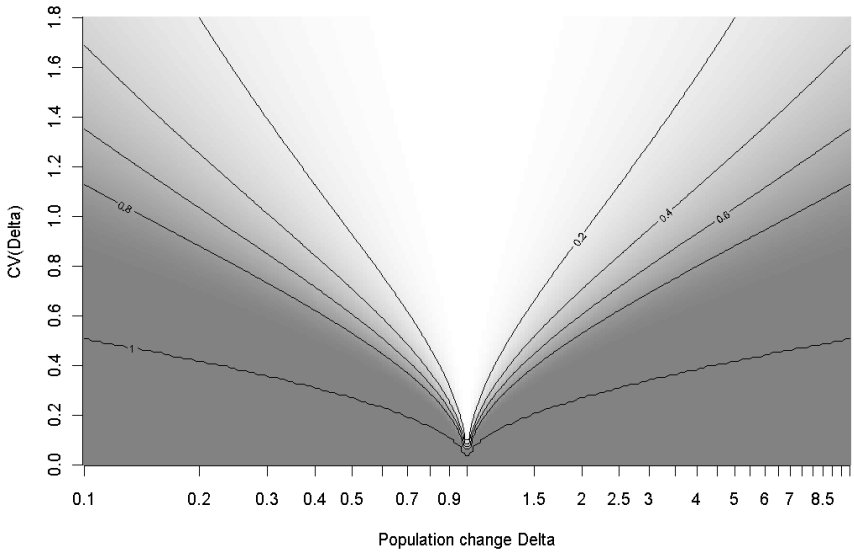

Figure 24. Power to detect population changes ranging from 0.1 to 8.5 given a range of CVs on the population change estimate.

4. Discussion

The ability to detect population trends in species of cetacean is necessary when assessing how different factors may affect a population. The primary aim of Task 2 was to use available data on abundance and density of cetacean populations to determine how robust trend estimation was on both a global scale and for individual species. It was clear during the data exploration stage, that many covariates, falling into four covariate groups (taxonomic, spatial, temporal, survey-related), may impact our ability to actually detect a trend in density. The exploratory stage of the analysis showed that data were highly variable across species. Therefore, in order to account for all the possible covariates, different levels of data exploration were required. For the FAO data it was necessary to have a specified level of pseudoreplication which, after exploration of the models, was finally combined into a more parsimonious grouping of ocean basin. These pseudo-replicated data were also used in the individual analysis, but were not a covariate in the models as the analysis was constrained to known FAO areas.

Results from the global model showed that taxonomic attributes were by far the most important covariates. For both datasets (the global dataset with no pseudoreplication and the FAO area dataset with pseudoreplication) species was the single most important factor (Tables 5 and 6). This is despite this factor having the largest number of parameters for estimation. This is not a surprising result as different species of cetacean are not found in equal numbers within ocean basins or globally. It is current practise to report species level abundance estimates in the literature, although instances of grouping species and reporting family level estimates do occur. However, even when we considered species using our second level of taxonomic attribute (family, where the 34 species were condensed into six groupings) this grouping was the second best model for both datasets (Tables 5 and 6).

Adding the temporal covariate of year in a combined model with species or family did not explain any more of the deviance than species or family on its own (Table 7). These models all had an assumption that species have the same trend in density (at least on the 1/density scale used here), but that the intercept varies by species. The fact that combining the covariates did not improve the fit, but the models with an interaction between trend and taxon did improve the fit, suggests that all species/families/groups are not showing the same trend, (e.g. S(Year)+Family has GCV 0.150; S(Year)*Family has GCV 0.144; Table 7). This is not surprising as it is unlikely that each species will have the same trend in density either across families, or within them. This confirms that the taxonomic covariates are more of a determinant of density than the temporal covariates on a global scale. However, temporal attributes might not have shown up clearly due to masking. It may be that the categories used for seasons were too broad, as during a preliminary analysis, when comparing surveys conducted only in June-August to all other surveys, an effect was noticed. This is also true for decades as the categories don't distinguish between surveys conducted within one year and those collecting data over a 10 year period (which may have started in the previous decade as the definition was based on the survey mid-year). Refining these categories and looking at different ways of grouping data may be needed for any future analyses.

When more than two categories were combined the best model was a combination of main effects and interactions of four covariates; S(Year)*Family + Species*OceanAgency (Table 7, part IV). The percentage deviance explained by this model was 72%. Although this model incorporates aspects of all four main pre-selected covariates (taxonomic, temporal, spatial and survey-related) the extra percentage deviance explained by adding in aspects of the three other covariate groups is 15%. Interestingly, survey method is not an important determinant in the model compared to other factors. Standardising methodologies and correcting for g(0) are common practice in many abundance studies, but this global analysis shows that this is not an important factor on this scale for

determining changes in density. This, in fact, is not surprising because you don't need to estimate a bias if it can be assumed to be more or less constant and/or not confounded with year. The use of g(0) is only really needed if absolute abundance is required for conservation or management purposes. Similarly, the agency conducting the survey is not an important single covariate but, OceanAgency may need consideration. However, it is impossible to tell whether ocean basin or agency alone are important determinants, as the statistical modelling suggest that some differences remain in density between Ocean Basin and Survey Agency even after the other factors have been accounted for. This is primarily due to Agency and Ocean Basin being almost completely confounded, as almost no agencies surveyed in multiple oceans. It is therefore impossible to determine whether the agency or the ocean is causing the effect. To account for this the OceanAgency covariate was used.

The estimated smooth for each family showed that it was only possible to detect a significant trend for the family Monodontidae. This may be because this family only contains one species, the Beluga whale which is restricted to a geographical area, and was exclusively surveyed using aerial line-transect surveys. In contrast, the family Physeteridae also contained only one species, the Sperm whale, but this species is present globally and surveyed by a greater number of methodologies. Because in the global analysis, the best model has no interaction between the temporal trend term S(Year)*Family and the OceanAgency term, it assumes the same trend for all OceanAgencies and species. Therefore, when you isolate some of the FAO areas (as would be the case with the geographically restricted Beluga whale) you get a significant trend, but when you lump them together over all oceans (as for the Sperm whale) the trend is not significant.

Given that the aim of this project was to determine the precision of temporal trends in stocks of cetaceans, it was clear from the global analysis that in order to try and look at changes over time it was necessary to consider species individually.

The individual models show varying results across species. For the seven species and three areas analysed, survey-related covariates (such as Method and Agency) explained little of the deviance in comparison with temporal and spatial covariates, except in a couple of cases (Table 9-14). For Dwarf Minke whales in the north-eastern Atlantic the single covariate model of Agency had the lowest GCV and explained the most deviance, although this was only 11.6% (Table 10). Similarly for Humpback whales in the north-western Atlantic and Striped dolphins in the north-eastern Pacific the single covariate model of Method had the lowest GCV scores, but other models explained more of the deviance (Tables 12 and 14). In three other cases, Year combined with either Agency or Method provided the best model for combined temporal trends (Fin whales in the north-eastern Atlantic, Sperm whales in the north-western Atlantic and Striped dolphins in the north-eastern Pacific). It is clear for these specific cases that survey related covariates are important determinants of changes in density for species within a specified area. This may be due to one survey methodology or agency being predominant in one of the areas in comparison to the other areas. It would be interesting to explore these survey related variables further, although they seem less important that the temporal and spatial covariates.

In comparison, temporal and spatial covariates generally explained the highest proportion of the total deviance across species and areas (Tables 9-14). For the Harbour porpoise in the north-western Atlantic and the north-eastern Atlantic, single covariate models of S(Lat) and Season had the lowest GCV scores, but the most deviance was explained by multi-covariate models (with Year incorporated). Similarly, in the north-eastern Pacific, a combination model of Year and MaxLat (Table 9) explained 98.3% of the deviance and had the lowest GCV score. This suggests that changes in density over time for the harbour porpoise in all three areas can be partially explained when year is considered. However, a measure of latitude should also be considered in the larger areas of the

north-eastern Pacific and the north-western Atlantic. This may be due to Harbour porpoise having a more localised distribution. Therefore, when considering density estimates over larger geographic areas a latitudinal covariate may need to be included to account for changes in density caused by movement patterns rather than a change in numbers. Similarly, for Dwarf Minke whales, Fin whales and Striped dolphins in the north-western Atlantic, Humpback whales and Long-finned Pilot whales in the north-eastern Atlantic, and the Dwarf Minke whales in the north-eastern Pacific, models incorporating year and latitude explained the most deviance. Again this may be due to non-uniform distribution of species and/or surveys across the area. However, this does suggest that for these populations there is some ability to look at changes in density over time. In only one case, Sperm whales in the north-eastern Pacific, Year alone explained the most deviance and had the lowest GCV score (Table 13). Interestingly in this case, adding season to the model had the exact same GCV score and percentage of deviance explained. Thus suggesting that season did not alter either the fit of the model or improve the amount of deviance explained.

The construction of smooth functions incorporating year and latitude using a tensor product function could not be fitted for all species in all areas. This is primarily due to the inability to accurately estimate a complex 2-d smooth function due to small samples sizes. In only seven instances did the tensor product function work (Tables 9-14). In addition, for only two of these (Harbour porpoise in the north-eastern Pacific and Harbour porpoise in the north-eastern Atlantic) did the models explain more of the deviance than either the single covariate models or combining temporal trends (Table 9). However, in both of these cases, the GCV scores were higher than the combining temporal trends models, and hence are not as good a fit.

It is clear from the individual models year is an important determinant of density, but in only one case was this covariate the singular most important in terms of explaining the most deviance. As such, spatial covariates, and in a few cases survey related covariates, may also be important in explaining changes in density.

Our intention in the power analysis was not to provide a detailed power analysis for each species and area analysed, but to obtain an initial view of what levels of population trend could be detected with high confidence. Our conclusion is that small changes (few tens of percent) may be detectable in some rare cases, but for most of the studies analysed the population change would have to be a doubling or more before it could be detected with high power. In some cases, even an order of magnitude population change could easily be missed.

The lack of relationship between goodness of fit (i.e. percent of deviance explained) and the power to detect a trend is unclear and warrants further investigation, for example, CV Δ for the Long-finned pilot whale in the north-eastern Atlantic was very small (0.17, Table 15) indicating that it would be possible to detect small trends, but our best model only explained around 17 % of the deviance. In a recent paper, Taylor *et al*., (2007) assessed the ability to detect precipitous declines in marine mammal stocks based on recent levels of survey effort. They showed that the likelihood of not detecting a precipitous decline is quite high for all the categories of marine mammals that they proposed, except for pinnipeds that haul out on land. Additionally, for a high precision species (i.e. the CV for abundance estimates is relatively low) to enable detection of a precipitous decline 80% of the time annual surveys would be advantageous, but surveys could be less frequent if the decision criterion for significance is changed. However, for a low precision species (i.e. the CV for abundance estimates is relatively high) the only option to attain an 80% probability of detection is to have annual surveys and use a higher significance criterion (Taylor *et al*., 2007).

4.1 Further work and considerations.

It is clear that in order to try and assess trends in density over time considerable data are required. However, for many species of cetacean, limited data exist over extended time periods, either because the animals are rare or the areas where they are abundant are not surveyed. Additionally, species that may only be sighted infrequently during surveys will not have abundance estimates available. There is a potential bias that the reporting of trends for rare or cryptic species, or species that are difficult to identify to species level will not take place as without the required sighting levels it is not possible to produce the necessary density values. Also a potentially worse bias would present if, for some species, trends are actually confounded with lack of sufficient data (e.g. sufficient sightings at the beginning with no trend followed by fewer sightings with no abundance estimates). One answer is to pool data across years for estimating detection function with year as a covariate.

In order to determine how commonly a species was sighted without subsequent abundance estimates being reported a review of the number of papers that reported abundance estimates for all species sighted was undertaken. This allowed an assessment of how sightings of rare or hard to observe animals may be reflected in the primary literature. If an animal is rare or hard to observe, detecting changes in density trends will be impossible if rare sightings are not reported as abundance estimates. As such there may be a bias towards detecting trends for more abundant animals. In total 34 papers were reviewed, of which 29% reported species level abundance estimates for all species sighted. All recorded sightings were used in 33% of papers, but for species where sighting numbers were low, family level or unidentified whale/dolphin combined estimates were given. Finally, 38% of papers reported sightings of species, but did not use these data for any level of abundance estimate. This is important when considering the potential effect of anthropogenic activities on cetacean populations. Clearly to gain a better understanding of rare or hard to observe animals different survey procedures may need consideration. A few examples may be (i) undertake longer surveys to increase the chance of sightings; (ii) increase effort in certain finer scale places, again to increase the chance of sightings; (iii) develop analysis techniques to use a series of sightings within an area to estimate abundance (e.g. pooling data across years).

As well as further work using power analysis to investigate the lack of relationship between goodness of fit and the power to detect a trend, it is also worth considering possible other methodological constraints to this analysis. It would be possible to add abundance estimates for more species in other areas if different types of surveys were also considered. One reason why so few estimates were collected using survey method A (non line transect methods) was that these estimates cannot consistently be attributed to a specific area. As we were using the ERMC database as a foundation, one of the selection criteria, and pre-requisites to enter it into the database, was the ability to assign it to an area. However, it is still likely, that for the global analysis the importance of taxonomic covariates would still be significant. Even with extra data added it remains unlikely that each species will have the same trend in density either across or within families. However, adding more data for individual species within specific areas may have some affect on the individual level models, and is thus worth future consideration. Furthermore, future analysis should consider the use of the log link function instead of the inverse link. The advantage of the log-link function is that estimates of density are constrained to be non-negative. In comparison, the inverse link, used in the models here does not constrain to non-negatives, and in a few cases negative estimates had to be set to zero. Finally, exploring the modelling further to look at dependencies, e.g. fitting stratum within a survey as a random effect to account for possible dependencies between estimates for species within a survey may be advantageous.

This Task has outlined the possibility of modelling a range of covariates to evaluate changes in density estimates of cetacean populations. It has highlighted possible species within determined areas to be carried forward to Task 3, when other factors influencing cetacean stocks will be investigated.

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Report

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Environmental and anthropogenic factors linked to influencing or controlling cetacean population growth rates

1. Introduction

In 2003, as part of the 2002-2010 conservation action plan for the world's cetaceans, Reeves et al. (2003) undertook a preliminary assessment of the impact of various threats on cetacean populations, both direct and indirect, such as whaling, incidental mortality in fisheries, indirect effects of industrial fisheries, competition and culls, ship-strikes, live-capture for captive display and/or research, whale and dolphin–watching, habitat loss and degradation, disturbance from industrial and military operations, chemical pollution, disease and exposure to biotioxins, and finally climate change and ozone depletion. It was reported that the total impact of the various threats cannot be calculated by simply summing their effects as though they were independent, as threats will work together and have a synergistic effect (Reeves et al. 2003). However, it is very difficult to quantify the role of synergy among threats in causing population declines (Reeves et al. 2003).

Task 3 focused on:

- Collating information on environmental and anthropogenic factors linked to influencing or controlling cetacean population growth rates for each cetacean species. Factors considered were:
  - anthropogenic mortality
  - food availability
  - disease
  - contaminants
  - climate change
- Collating data on maximum population growth rates (R_{MAX}) for each cetacean species
- Reviewing information on how cetacean populations/stocks differ in rates of recovery
- Assessing factors that may have limited the recovery of certain populations/stocks and identifying major gaps in the data
- Compiling available information on the seven species outlined in task 2, including published data on current trends in abundance, maximum rate of increase and the current status of the populations/stocks

Task 4 will:

- Evaluate the effects of exploration and production noise on cetacean populations

2. Environmental and anthropogenic factors linked to influencing or controlling cetacean population growth rates

2.1 Whaling

Whaling is defined as the purposeful killing of large cetaceans to obtain economically useful products and therefore encompasses both commercial and subsistence whaling. Many species of baleen whales have been exploited by man, in some cases for almost 900 years (Best 1993). However, over the last two hundred years commercial exploitation has led to the depletion of some populations below levels at which maximum net productivity might be expected, and for others to near extinction. Initially, whaling focused on the species that were both easy to catch and the most profitable, such as right (*Eubalaena glacialis/japonica*), bowhead (*Balaena mysticetus*), sperm (*Physeter macrocephalus*), humpback (*Megaptera novaeanglia*) and gray whales (*Eschrichtius robustus*). Once steam-powered vessels and harpoon cannons became widely available, activities focused on the elusive but valuable blue (*Balaenoptera musculus*), fin (*Balaenoptera physalus*), sei (*Balaenoptera borealis*), Bryde's (*Balaenoptera edeni/brydei),* and minke (*Balaenoptera acutorostrata/bonaerensis*) whales (Reeves et al. 2003). Other species targeted by whaling include killer (*Orcinus orca*) and pilot whales (*Globicephala macrorhynchus/melas*), beaked whales, belugas (*Delphinapterus leucas*) and narwhals (*Monodon monoceros*). Several populations were reduced to such an extent that their recovery may now be hindered by demographic and genetic factors. For example, northern hemisphere right whales, western Pacific gray whales and Antarctic blue whales have yet to show signs of recovery (Clapham et al. 1999, Reeves et al. 2003). Furthermore, due to whaling an entire trophic level of the marine ecosystem was almost eliminated, particularly in the southern hemisphere (Baker & Clapham 2004), and as equilibrium conditions shifted, the full "recovery" of some populations is now unlikely (Reeves et al. 2003). Studies undertaken on these struggling populations can provide valuable information which can be used to assess the slow and variable recovery of other stocks. For example, the rates of net reproduction might decrease rather than increase at very low levels of abundance, a phenomenon referred to as depensation or the Allee effect (Baker & Clapham 2004). See section 3.2.2 for further information.

The International Whaling Commission (IWC) was set up under the International Convention for the Regulation of Whaling in 1946 (IWC 1946). The IWC imposed a pause in commercial whaling in 1986, known as the moratorium, to allow whale stocks to recover from overexploitation (http://www.iwcoffice.org/index.htm). Although the moratorium is still maintained, Norway, Japan and Iceland take whales under objection and under Special Permit (termed "scientific whaling"). Aboriginal subsistence whaling is not covered by the moratorium on commercial whaling.

2.1.1 North Atlantic Ocean

Whaling began in the 11th century, with the Basques first sustained commercial whale fishery on right whales in the Bay of Biscay (Aguilar 1986). By 1530, Basque whaling ships were operating in the Red Bay, Labrador, as well as in the general vicinity of the Strait of Belle Isle (Clapham & Link 2006). Bowhead whales were the main species targeted, and when Britain established their Arctic hunt around 1570 they also primarily hunted this species (Clapham & Link 2006). With the colonisation of the New World, directed whaling on right whales in the western North Atlantic commenced around 1730s - the American pelagic whaling era - attaining its peak in the

19th Century. Humpback and sperm whales, along with other species, were also targeted at this time (Clapham & Link 2006). Around 1854, the Norwegian coast whaling era began and by 1900 every part of the North Atlantic Ocean had been subject to whaling at some point in time, resulting in a number of depleted and extirpated populations of great whales (Clapham & Link 2006).

The Atlantic gray whale population was believed to have been hunted to extinction by the early 1700s (Mitchell & Mead 1977, Perry et al. 1999). The western right whale population, of approximately 300-350 individuals, is labelled as endangered by the IUCN (2008 listing, see ANNEX A), and the eastern Atlantic right whale population is essentially extinct. The Svaalbard-Barents Sea bowhead whale stock is listed as critically endangered on the IUCN red list of threatened species (Clapham and Link, 2006, see ANNEX A), and has been described as functionally extinct (Clapham & Baker 2002). In the eastern Arctic current estimates of population size are 6,344 (95% CI = 3,119-12,906) for Baffin Bay – Davis Strait and 1,525 (95% CI = 333-6,990) for Foxe Basin–Hudson Bay (IWC Scientific Committee Report 2008, p28). In contrast, humpback whales have been reported to be recovering - after an estimated total catch of 29,000 individuals - with a current abundance of 11,570 individuals, close to its estimated pre-exploitation abundance (Stevick 2003a). Further, fin, sei and minke whales are believed to be generally abundant (Clapham & Link 2006). Blue whales however remain endangered in the western Atlantic, with population numbers in the low hundreds. Although, the eastern Atlantic blue whale population is larger in size, no current abundance estimates are available (Sigurjonsson 1995).

2.1.2 North Pacific Ocean

Intense commercial whaling in the northern North Pacific Ocean commenced in the early 1840s. It initially focused on right and bowhead whales and populations of these species were drastically reduced within the first few years of the fishery (Springer et al. 2006). Shortly after this, the gray whale calving grounds were discovered in Baja California and whaling reduced the population from an estimated 12,000 individuals in 1846, to approximately 2,000 by the 1880s (Springer et al. 2006). Modern whaling, using catcher boats and mounted harpoon cannons, began in Japan around 1899 (Korea in 1889, British Columbia in 1905 and Alaska in 1907) focusing on fin, sei and blue whales (Springer et al. 2006). By the 1920s, the fin and blue whale populations had declined and catches fell rapidly. The blue whale population in the North Pacific was historically small, and the combined harvest of animals off Baja California and from their northern summering grounds during 1925-1930 was approximately 1600 individuals (Springer et al. 2006).

In the early 1900s, whalers focused on targeting humpback whales. By the 1930s, 18,000 humpback whales were killed in the North Pacific and the population had fallen from an estimated 15,000 to approximately 6,000 individuals (Rice 1978). Although right whales were afforded worldwide protection in 1935, illegal hunting continued up to the 1960s, which almost drove populations in this region to extinction (Springer et al. 2006). At the end of world-war II, Japanese and Soviet pelagic whaling expanded in the North Pacific and focused on sperm, humpback, blue, fin and sei whales. Catches for all species increased rapidly each year, to peak levels in the mid-to late 1960s (Springer et al. 2006). Bryde's whales were not hunted until the 1970s, following the depletion of larger whales in more northern waters (Springer et al. 2006).

Gray whales have been exploited commercially on both sides of the North Pacific for the last two centuries. The 'Asian' or western stock of gray whales survives as a remnant population of approximately 100 animals (Weller et al. 2002, Baker & Clapham 2004). The eastern stock, considered to have been commercially extinct since the end of the 19th century, has now increased to 26,300 whales (http://www.iwcoffice.org/conservation/estimate.htm, see ANNEX B) (Baker & Clapham 2004).

The initial pre-whaling population of fin whales in the North Pacific was estimated to be 42,000-45,000 individuals (Ohsumi & Wada 1974). Between 1925 and 1975, 47,645 fin whales were reported killed throughout the North Pacific (International Whaling Commission, BIWS catch data, February 2003 version, unpublished), although information on illegal Soviet catches indicates that the Soviets over-reported c.1,200 catches of fin whales (Doroshenko 2000), possibly to hide catches of other species (Angliss & Outlaw 2006a). In the early 1970s, the North Pacific population was estimated to have been reduced to 13,620-18,630 whales (Ohsumi & Wada 1974, Perry et al. 1999). Fin whales in the North Pacific were given protected status by the IWC in 1976.

The abundance of sperm whales in the North Pacific was reported to be 1,260,000 prior to exploitation, which by the late 1970s was estimated to have been reduced to 930,000 whales (Rice 1989, Angliss & Lodge 2003). Approximately 436,000 sperm whales were taken between 1800 and the end of commercial whaling for this species. It should be noted though that this number is an underestimate, as soviet vessels under-reported their catches by 60% during the period 1949-71 (Angliss & Lodge 2003), and Japanese land based whaling operations also under-reported sperm whale catches during the post-World War II era (Kasuya 1999). There has been a prohibition on taking sperm whales in the North Pacific since 1988, though large-scale pelagic whaling ceased around 1980 (Angliss & Lodge 2003).

The northeast Pacific right whale is listed as critically endangered by the IUCN (see ANNEX A). Scarff (2001) estimated that 26,500-37,000 right whales were killed during the period 1839-1909, with the majority killed in the 1840s. A reliable abundance estimate is not available for this stock. The Cook Inlet beluga whale in Alaska is also listed as critically endangered by the IUCN, with a population of approximately 350 individuals (Kingsley 2000, Reeves et al. 2003).

Japanese small cetacean drive fishery:

The Japanese drive fishery for small cetaceans in the 19th Century mostly targeted striped dolphins (*Stenella coeruleoalba*) which made up 96% of the catches in the 1960s (Kishiro & Kasuya 1993). Information on catch statistics is lacking for the period prior to 1960, although records suggest 10,000 to 22,000 dolphins (mostly striped dolphins) were caught between 1942 and 1960. The annual catch of striped dolphins declined from 11,000 to 3,000 in the early 1970s to less than 1,000 in the early 1980s (Kasuya 1985, Kishiro & Kasuya 1993). It has been suggested that one of the main factors behind the decline in supply of striped dolphin meat was a decline in abundance of the coastal population(s) of the species. Abundance estimates for offshore and inshore population(s) of striped dolphins are 497,000 (CV=0.18) and 22,500, respectively (Kasuya, 2007; Table 1). The declining supply of striped dolphin was substituted by Dall's porpoise (*Phocoenoides dalli*) meat, taken by the hand harpoon fishery in northern Japan. Fishermen also targeted killer whales, bottlenose, pantropical spotted, and Risso's dolphins

(*Tursiops* spp., *Stenella attenuata*, and *Grampus griseus*, respectively) to supply the profitable Japanese market for small cetacean meat (Kishiro & Kasuya 1993, Reeves et al. 2003).

Two subspecies of Dall's porpoise exist in Japanese waters, the *dalli*-type colour morph and the *truei*-type colour morph, both hunted by the Japanese hand-harpoon fishery. An abundance of 226,000 individuals (see Table 1) and a rate of increase of 4% have been estimated for the *dalli*-type subspecies. The annual catch of Dall's porpoises taken by the harpoon fishery between 1960s and 1970s fluctuated between 5,000 and 10,000 individuals. In the early 1980s, the Dall's porpoise fishery expanded its geographical range and its season of operation to summer. A peak catch of 45,600 Dall's porpoises was reported for 1988 (Kasuya 1992), and between 1986 and 1989 111,500 Dall's porpoises were killed (IWC 1991, Reeves et al. 2003). Following this, the Japanese government began to regulate the hand-harpoon hunt and the reported catches decreased to less than 11,500 in 1992 (IWC 1994). The most recent published catches for both stocks are 4,614 and 9,175 individuals in 2004, for the *dalli*-type and *truei*-type, respectively (Kasuya 2007).

Table 1 Calculation of quota for small cetaceans in 1993, together with comparison against earlier catches (modified from unpublished document of the Fisheries Agency dated January 1993). (Table taken from Kasuya, 2007).

Species/Stocks	Abundance	Increase rate	Safety factor	Special allocation	Quota 1993	Annual catch (1989–1992)
Dalli-type [a]	226,000	0.04			9,000	12,265–29,0484 [c]
Truei-type [a]	217,000	0.04			8,700	
Striped d.	22,500	0.03		+50	725	749–1,225 [d]
Bottlenose d.	35,100	0.03		+50	1,100	171–1,298
Spotted d.	30,100	0.03		+50	950	6–636
Risso's d.	42,000	0.03		+50	1,300	13–298
S [b], s.f. pilot w.	20,300	0.02		+50	450	149–296 [e]
N [b], s.f. pilot w.	5,000	0.02	0.5		50	10–50 [f]
False killer w.	5,000	0.02	0.5		50	30–91
Baird's bkd. w.						54 [g]

[a] One of two stocks of Dall's porpoises off Japan; [b] Southern and northern stocks of short-finned pilot whales off Japan; [c] Quota started in 1991 with a combined figure of 17,600; [d] Quota started in 1992 with 1,000; [e] Quota started in 1992 with 400; [f] After several management attempts in 1983–85, a quota of 50 was started in 1986; [g] Quota was set at 40 in 1983, 60 (in 1988) and 54 (in 1989) for the Pacific and Okhotsk Sea, then to the current quota of 62 for the Pacific, Okhotsk Sea and Sea of Japan (IWC 1992)

2.1.3 Southern Oceans

Over six decades after the opening of the Antarctic whaling grounds, the whaling industry killed approximately two million whales in the southern hemisphere, including 725,116 fin, 401,670 sperm, 360,644 blue, 203,538 sei, 208,359 humpback, and 116,568 minke whales (Clapham & Baker 2002). Smaller numbers of southern right whales (4,338) and Bryde's whales (7,757) were taken. Whaling peaked in the 1960s with approximately 70,000 killed annually over a period of several years (Ballance et al. 2006).

In southern oceans, some populations of southern right whales (IWC 2001b) and humpback whales have shown signs of recovery under protection (Reeves et al. 2003). In contrast, the

continued small numbers of southern right whales in some areas of former abundance (e.g., around New Zealand, off Peru and Chile) (IWC 2001b), and of blue and fin whales, is a cause for concern (Clapham et al. 1999). The Antarctic blue whale is listed as critically endangered by the IUCN, and throughout the world sei and fin whales are listed as endangered, and the sperm whale as vulnerable, (see ANNEX A). Some regional subpopulations or stocks were essentially extirpated by whaling, such as blue whales around South Georgia, and humpback whales off New Zealand (Laws 1977, Baker & Clapham 2004).

2.1.4 Whaling today

Japan, Norway and Iceland have issued scientific permits as part of their cetacean research programmes. Permits for the Japanese's JARPA II (Antarctic waters) programme include 850±10% Antarctic minke whales, 50 fin whales and 50 humpback whales, and for the JARPN II (western North Pacific waters) programme include 220 common minke whales, 50 Bryde's whales, 100 sei and 10 sperm whales (http://www.iwcoffice.org/conservation/permits.htm). Under scientific permits, a total of 292 fin whales and 70 sei whales were caught in Icelandic waters between 1986 and 1989, and 200 minke whales were caught between 2003 and 2007 (http://www.iwcoffice.org/conservation/table_permit.htm). A further 7 minke whales were caught in Icelandic waters under objection to the whaling moratorium in 2006 and 2007 (http://www.iwcoffice.org/conservation/table_objection.htm). A total of 289 minke whales were caught in Norwegian waters under scientific permits between 1988 and 1994 (http://www.iwcoffice.org/conservation/table_permit.htm), while a further 8,085 minke whales were harvested under objection to the whaling moratorium between 1986 and 2007 (http://www.iwcoffice.org/conservation/table_objection.htm).

Currently, there are 5 active whaling operations in the North Atlantic, consisting of shore-based commercial whaling of minke whales by Norway (see section 4.3.1), "scientific whaling" of minke whales by Iceland and three native subsistence hunts – fin and minke whale hunt off Greenland, the eastern Canadian bowhead whale hunt, and a small hunt for humpback whales in the south-eastern Caribbean. The impact of these operations ranges from insignificant to potentially damaging, in the case of the Greenland fin whale (IWC 2004).

The international whaling commission has determined catch limits for aboriginal subsistence whaling including (http://www.iwcoffice.org/conservation/catches.htm#aborig, downloaded on the 01/09/08):

1. Bering-Chukchi-Beaufort Seas stock of bowhead whales (taken by native people of Alaska and Chukotka) - A total of up to 280 bowhead whales can be landed in the period 2008 - 2012, with no more than 67 whales struck in any year (and up to 15 unused strikes may be carried over each year).

2. Eastern North Pacific gray whales (taken by native people of Chukotka and Washington State) - A total catch of 620 whales is allowed for the years 2008 - 2012 with a maximum of 140 in any one year.

3. West Greenland fin whales (taken by Greenlanders) - An annual strike limit of 19 whales is allowed for the years 2008 - 2012.

4. West Greenland common minke whales (taken by Greenlanders) - An annual strike limit of 200 whales is allowed for the years 2008 - 2012 with an annual review by the Scientific Committee.

5. West Greenland bowhead whales (taken by Greenlanders) - An annual strike limit of 2 whales is allowed for the years 2008 - 2012 with an annual review by the Scientific Committee.

6. East Greenland common minke whales (taken by Greenlanders) - An annual strike limit of 12 whales is allowed for the years 2008 - 2012.

7. Humpback whales taken by St Vincent and The Grenadines - For the years 2008-2012 the number of humpback whales to be taken shall not exceed 20.

2.2 Fisheries interactions

Direct fisheries interactions pose a serious threat to many populations of marine mammals, especially small cetaceans, due to their slow life histories and limited potential rates of increase (Reilly & Barlow 1986, Read 2008). Direct fisheries interactions occur when cetaceans come into physical contact with fishing gear, resulting in mortality or serious injury. If the animal is subsequently discarded the process is termed bycatch, or if the animal is retained for consumption or sale the process is referred to as non-target catch (Read, 2008). Fisheries considerably alter the trophic structure, species assemblages and pathways of energy flow (Pauly et al. 1998, Jackson et al. 2001, Myers & Worm 2003), resulting in ecological changes that may have adverse consequences for cetaceans (Read, 2008). These interactions are termed indirect interactions, and are fully outlined in Plaganyi and Butterworth (2005). There are two main forms of indirect interactions, including exploitative competition by fisheries (and removing cetaceans' prey) and interference competition which involves the disruption of marine mammal feeding activities as a result of disturbance. In the future, escalating pressure on shared resources is expected due to the growth of some marine mammal populations and the requirements of an increasing human population (Plaganyi & Butterworth 2005).

As small populations of cetaceans are vulnerable to stochastic processes (e.g. inbreeding, natural disasters, or disease outbreaks), which may cause them to spiral towards extirpation, small populations therefore are more vulnerable to bycatch removals (Read, 2008). Bycatch is recognized as the primary threat to severally endangered species of marine mammals (Reeves et al. 2003), including the vaquita (*Phocoena sinus*) and Hector's dolphin (*Cephalorhynchus hectori*), and contributes to the critical conservation status of the North Atlantic right whale (Read et al. 2006).

One case in particular is the vaquita, a small porpoise endemic to the Gulf of California that is listed as critically endangered on the IUCN Red List. The population has been declining since the 1940s due to incidental capture in a wide variety of gillnet fisheries throughout their range (Reeves et al. 2003). In the last 10 years, the population has been declining annually by 10%, reduced from 567 individuals in 1997 to approximately 150 in 2007 (Jaramillo-Legoretta 1999, Jaramillo-Legorreta 2007). However, although mortality in gillnets has long been recognized as the most serious and immediate threat to the vaquita's survival (see Rojas-Bracho et al. 2006), other anthropogenic activities such as the release of pesticides into the local environment, population characteristics such as inbreeding depression, and also ecological changes to the

local environment, may also have an impact on the population (Taylor & Rojas-Bracho 1999). It has been reported that unless conservation plans are implemented immediately (with a total fishing moratorium on all entangling nets, so that vaquita mortality can be immediately reduced to zero), options for conserving the vaquita will be severely reduced within two years (Jaramillo-Legorreta 2007).

In the Eastern Tropical Pacific (ETP), large-scale incidental capture of spotted, spinner and common dolphins have been reported and, to date, approximately 5-6 million dolphins have been killed by the yellowfin tuna fishery. The number of animals incidentally killed in the yellowfin tuna fishery reached a peak towards the end of the 1960s, with almost 700,000 dolphins killed in one year, but by the end of the 1970s the kill had declined to approximately 20,000 dolphins per year (Gerrodette 2002). However, in 1986 total dolphin mortality had increased to 133,000 individuals, but due to various political and economic pressures, and various mitigation measures, dolphin mortality has drastically decreased (Gerrodette 2002). During 2006, 94% of all sets made on tuna associated with dolphins were accomplished with no mortality or serious injury to the dolphins, and the total mortality of dolphins in the fishery in 2006 was reduced to less than 900 individuals (www.iattc.org). The main stocks recovering from effects of the yellowfin tuna fishery are the coastal and northeastern offshore spotted dolphins (*Stenella attenuate attenuata*) and the eastern spinner dolphin (*Stenella longirostris orientalis*); see section 3 for further information.

Table 2 Bycatch priorities based on documented species- or population-level threats (*indicates those that meet the criteria established by Reeves et al. 2005).

Species	Location	Fishing activity
Vaquita	Gulf of California	Gillnets
North Atlantic right whale	Western North Atlantic Ocean	Vertical trap lines and gillnets
North Pacific right whale	Asia	Vertical trap lines and gillnets
*Irrawaddy dolphin, marine	Philippines	*Matang quatro* crab nets
*Irrawaddy dolphin freshwater	Mekong River, Mahakam River, Songkhla Lake, and Ayeyarwady River,	Gillnets
Ganges river dolphin	India and Bangladesh	Gillnets
Finless porpoise	Inland Sea (Japan)	Gillnets
Finless porpoise	Yangtze River	Gillnets and electrofishing
*Franciscana	Argentine, Brazil, and Uruguay	Coastal gillnets
Hector's dolphin	North Island, New Zealand	Coastal gillnets
Harbour porpoise	Baltic Sea	Gillnets
*Harbor porpoise	Black Sea	Coastal gillnets
J-stock minke whale	Japan and South Korea	Trap nets
*Dusky dolphin	Peru	Drift gillnets
Indo-Pacific humpback and bottlenose dolphins	Natal (South Africa)	Anti-shark nets
*Indo-Pacific humpback and bottlenose dolphins	South coast of Zanzibar (Tanzania)	Drift and bottom-set gillnets

Throughout its range, the harbour porpoise (*Phocoena phocoena*) is frequently incidentally captured, and gillnets appear to pose a serious threat, as harbour porpoises are extremely

susceptible to entanglement. In the western North Atlantic, Caswell et al (1998) combined the rate of increase (determined using model life tables derived from other mammal species) of this population with the uncertainty of incidental mortality and population size. Results suggested that incidental mortality exceeds the critical values established by various management agencies, and therefore by-catch was a significant threat to the harbour porpoise in this region. A harbor porpoise reduction bycatch plan in USA Atlantic gillnets came into effect in 1999. The Gulf of Maine portion of the plan pertains to all fishing with sink gillnets and other gillnets, and includes time and areas closures, some of which are complete closures; others are closed to multispecies gillnet fishing unless pingers are used in the prescribed manner (Waring et al. 2007). In 1994, amendments to the Marine Mammal Protection Act established a process in which maximum allowable annual removal limits are set for each marine mammal stock based on the potential biological removal (PBR) level, and fishing activities are subject to monitoring and regulation to assure that those limits are not exceeded (Wade 1998, Read 2003). It has not been reported though that this approach has substantially improved fishery management in the United States in terms of mitigating cetacean bycatch (Warning et al. 2007).

Table 3 Bycatch priorities based on suspected species- or population-level problems (* indicates those that meet the criteria established by Reeves et al. 2005).

Species	Location	Fishing activity
Burmeister's porpoise	Peru	Coastal gillnets
Finless porpoise, marine	China and SE Asia	Coastal nets and traps
Finless porpoise	Persian Gulf	Coastal gillnets
*Irrawaddy dolphin	Chilka Lake (India)	Gillnets
*Irrawaddy dolphin	Bay of Bengal	Heavy-mesh drift gillnets for elasmobranchs
Humpback dolphins	West Africa	Coastal gillnets
Humpback dolphins	Madagascar and East Africa	Coastal gillnets
Humpback dolphins	Asia,	Coastal gillnets
Sperm whales	Mediterranean Sea	Pelagic driftnets
Bottlenose dolphins	Black Sea	Gillnets
Bottlenose dolphins	Mediterranean	Gillnets
Marine/estuarine populations of tucuxis	Western southern Atlantic Ocean	Coastal gillnets
Freshwater tucuxis	Amazonia	Gillnets
Short-beaked common dolphin	Mediterranean Sea	Gillnets and driftnets
Striped dolphin, Risso's dolphin, long-finned pilot whale, and Cuvier's beaked whale	Mediterranean Sea	Driftnets
Short-beaked common dolphin	Western European waters	Trawl nets and gillnets
Finless porpoise	Korea and Japan	Coastal gillnets and traps
*Commerson's dolphin	Argentina	Coastal gillnets and midwater trawls
*Spinner dolphins and Fraser's dolphin	Philippines	Driftnets for large pelagics and flying fish, purse seines for small pelagic
Spinner dolphin	Sri Lanka	Drift and set gillnets in combination with direct harpooning

Table 2 outlines documented species- or population-level threats, and Table 3 outlines suspected species- or population-level threats, collated by Reeves et al. (2005) in an assessment of global priorities for reduction of cetacean bycatch. Cetacean species reported to be susceptible to high bycatch rates, and low population abundance, are the North Atlantic right whale, *Eubalaena glacialis,* the La Plata or Franciscana dolphin (*Pontoporia blainvillei*) inhabitating Argentinean, Brazilian, and Uruguayan waters, and the Atlantic humpback dolphin (*Sousa teuszii*) which is endemic to West Africa (Reeves et al. 2005). Studies to date have shown that 50-70% of the Gulf of Maine humpback whales and North Atlantic right whales have been entangled at least once in their lifetime (Knowlton et al. 2003, Robbins & Mattila 2004, Read et al. 2006). The North Atlantic right whale is one of the most endangered large cetaceans, and entanglement in lobster pots (and ship strikes) is a problem for this species (IWC 2001a, Reeves et al. 2003).

2.3 Food availability

Prey availability is an important factor that influences the rate of growth of cetacean populations. However, the feeding ecology of cetaceans is difficult to study, feeding usually occurs at depth, in offshore areas and over large spatial scales, making direct observations challenging. As a result, the population consequences of variation in prey availability and foraging success are poorly understood and have only been investigated in a few species.

Every year, whales in the southern hemisphere migrate from low latitudes in the winter to the southern ocean in the summer to feed on seasonally abundant prey (Ballance et al. 2006). Nearly two million whales have been removed from the southern hemisphere by whaling (Baker & Clapham 2004) and the dramatic reduction in the density of baleen whales has led to a greater standing stock of krill. It has been reported that this may have caused increased pregnancy rates of sei, blue and fin whales, and decreased age of sexual maturity for sei and fin whales (Gambell 1968, Lockyer 1972, Gambell 1973, Lockyer 1974, Ballance et al. 2006). See section 3.2.1 for further information on density dependence in cetacean populations.

Springer et al. (2003) suggested that depletion of baleen whales by commercial hunting forced killer whales *Orcinus orca* in the North Pacific to switch to other prey, causing sequential declines in populations of pinnipeds and sea otters. Although other evidence for this hypothesis is weak, it underscores the need to consider the impact of whaling on other predator-prey relationships, and to examine trends in population declines as well as total removals (Baker & Clapham 2004, Estes 2008).

2.4 Contaminants

Marine mammals are susceptible to the effects of anthropogenic contaminants, such as persistent organic pollutants (POPs) i.e. polychlorinated biphenyls (PCBs), dichlorodiphenylethanes (e.g. DDT, a widely used pesticide), hexabromocyclododecane (HBCD), and various heavy metals such as cadmium (Cd) and mercury (Hg). Apart from heavy metals, contaminants are reported to both biomagnify (higher levels higher up the food chain) and bioaccumulate (increasing concentration with age in individuals) and certain contaminants, such as persistent organic pollutants, are lipophilic compounds that accumulate in the lipid-rich blubber layers of marine mammals. On the whole, contaminants enter the body almost

exclusively through the diet, and levels of some contaminants can be highest in bays, estuaries and coastal waters. Therefore, it has been suggested that freshwater cetaceans may be at greater risk from pollutants than marine cetaceans (Reeves et al. 2003).

Certain pollutants are toxic and higher levels can be lethal, whereas others reduce resistance to disease through immunosupression, and/or may cause reproductive dysfunction. It has been reported that Mysticetes (baleen whales) are less contaminated – often by an order of magnitude - than some odontocetes, as they feed lower down the food chain (O'Shea and Brownell 1994, Weisbrod et al. 2000, Reeves et al. 2003). However, enzyme markers in tissues of endangered North Atlantic right whales, for example, indicate significant exposure to a nonbioaccumulative, but potentially toxic dioxin-like compound, such as one of the polycyclic aromatic hydrocarbons (PAHs) (Moore, cited in Reeves et al. 2001, Reeves et al. 2003). One of the main species of concern is the killer whale, as they are one of the most contaminated cetaceans in the world due to high concentrations of PCBs and DDT reported in the blubber of Alaskan and transient killer whales in British Columbia, (Ross et al. 2000, Ylitalo et al. 2001). Ylitalo et al. (2001) suggested that higher contaminant concentrations found in transient killer whales compared to residents could be attributed to dietary differences in the two ecotypes, i.e. transient killer whales feed predominately on marine mammals, with elevated POP levels, while resident animals are primarily piscivorous.

Even though thousands of chemicals are introduced into the world's oceans, the majority of research undertaken to date on contaminant exposure has been weighted towards organohalogens (predominately organochlorines), heavy metals and to a lesser extent radionuclides and petroleum hydrocarbons (O'Hara & O'Shea 2005). High levels of polychlorinated biphenyls (PCBs) and other organochlorines are a known concern to marine mammals. They have been reported to interfere with both the hormone and immune system of mammals and high levels in excess of 100mg/kg have been associated with reproductive abnormalities and complex diseases syndromes (Reeves et al. 2003). A Σ-PCB level of 17 $\mu g\ g^{-1}$ lipid has been reported as a threshold level for effects on reproduction in the bottlenose dolphin (*Tursiops truncatus*) (Schwacke et al. 2002). However in western European waters, Pierce et al. (2008) reported that in 40% and 47% of cases, PCB values in the blubber of female common dolphins and harbour porpoises from the Atlantic coast of Europe were frequently above the threshold at which effects on reproduction could be expected. This rose to 74% for porpoises from the southern North Sea. Interestingly, Pierce et al. (2008) stated that mature female individuals with the highest PCB concentrations in their blubber, were individuals with high numbers of *corpora albicantia* (scars of ovulation and pregnancy) on their ovaries. This suggests either that due to high contaminant burdens, females may be unable to reproduce, thus continue ovulating; or females are not reproducing for some other reason, either physical or social, and therefore accumulate high contaminant levels, as they are not able to pass their burdens onto their offspring via the placenta, or through lactation (Murphy et al. 2008). If high PCB burdens are causing a detrimental effect on the reproductive output in the common dolphin, it will have an adverse effect on the Northeast Atlantic population.

Relatively low-level exposures to some chemicals at critical life stages can result in dramatic effects on individuals and/or subtle but important population-wide impacts, by affecting population growth, maintenance and/or health (O'Hara & O'Shea 2005). For example females, through mobilization of lipid-associated toxins from the blubber, excrete toxic rich compounds to their offspring during gestation (via the placenta) and during lactation (via their lipid rich-

milk). This results in a high exposure of newborns to those chemicals and a relatively large intake of contaminants by an animal of much smaller body size (O'Hara & O'Shea 2005). However, it appears that the first born offspring tends to be the most susceptible to exposure of contaminates compared to their siblings, as first time mothers have larger contaminant loads built up over many years. Wells et al. (2005) reported higher rates of first born calf mortality were correlated with higher concentrations of PCBs in blubber and plasma of primiparous females inhabiting Sarasota Bay in Florida. Subsequent calves exhibited higher survival rates (Wells et al. 2005).

For the same bottlenose dolphin population, Hall et al. (2006) investigated the effects of different PCB accumulation scenarios on potential population growth rates. This was carried out through developing an individual-based model framework that simulated the accumulation of PCBs in the population and modified first-year calf survival, based on maternal blubber PCB levels. Their predictions though were limited both by model naivety and parameter uncertainty; data were lacking on the relationship between maternal blubber PCB levels and calf survivorship, the annual accumulation of PCBs in the blubber of females, and the transfer of PCBs to the calf through the placenta and during lactation (Hall et al. 2006). As a result, the observed population increase for the Sarasota Bay population from the mid 1990's of approximately 40% (Wells, unpublished data), which equates to an annual growth rate of approximately 1.03, lies outside the 95% confidence intervals of the best potential growth rate estimated by Hall et al. (2006).

Other recent studies have focused on the effects of increased contaminant loads on overall health status in cetaceans. Jepson et al. (2005) investigated the relationships between polychlorinated biphenyl exposure and infectious disease mortality in harbour porpoises in UK waters. The infectious disease group had significantly greater ∑25CB (chlorobiphenyl congeners) values (mean, 27.6 mg/kg lipid) than the physical trauma group (mean, 13.6 mg/kg lipid; $p<0.001$), independent of age, sex, two indices of nutritional status, season, region, and year found. Results from this study are consistent with a causal (immunotoxic) relationship between PCB exposure and infectious disease mortality (Jepson et al. 2005).

As mentioned previously cetaceans, inhabiting coastal waters, estuaries and bays may be at a greater risk from pollution, due to their proximity to the outflow of industrial waste. A small isolated population of beluga whales that are highly contaminated by pollutants, mostly of industrial origin, resides in the St. Lawrence estuary, Quebec, Canada. During the first half of the twentieth century, over-hunting was suggested to be the probable cause for the population decline from several thousand animals to approximately 500 (De Guise et al. 1995). The lack of recovery for this population has been attributed to high contaminant burdens. In recent years though, concentrations of most of the bioaccumulative and toxic (PBT) chemicals examined had exponentially decreased by at least a factor of two between 1987 and 2002 while no increasing trends were observed for any of the PBTs measured (Lebeu et al. 2007). Lebeu et al. (2007) reported that decreasing trends in PBT concentrations may be due to a decline in contamination of its prey following North American and international regulations on the use and production of these compounds, or due to a change in diet or a combination of both.

2.5 Disease

Infectious diseases have the potential to play a role in the decline of threatened wildlife populations and negatively affect their long-term viability (Gaydos et al. 2004). An outbreak of an infectious disease causing an epizootic (epidemic) mass die off can cause large scale population decline, possibly of a magnitude that may reduce population density to such an extent that stochastic events, or previously unimportant ecological factors, may further reduce the population size (Harwood & Hall 1990). Furthermore, infectious diseases alter reproductive rates by decreasing fertility and causing abortions, premature parturition and neonatal mortality (see Gulland & Hall 2005, and ref. therein).

A mass die-off can be catastrophic for a species with a limited range or low abundance (Reeves et al. 2003). Documented mass die-offs have involved bottlenose dolphins in the North Atlantic and Gulf of Mexico (Duignan et al. 1996), striped dolphins in the Mediterranean Sea (Aguilar 2000), various cetacean species in the Gulf of California (Vidal & Gallo-Reynoso 1996), harbour porpoises in the Black Sea (Birkun et al. 1999), Indo-Pacific hump-backed dolphins in the Arabian (Persian) Gulf (Ross 1994), and humpback whales in a small area of the western North Atlantic (Geraci et al. 1989). These events have highlighted the susceptibility of cetaceans to epizootic diseases (e.g., morbilliviruses) and biotoxins (e.g., dinoflagellates popularly known as "red tide" organisms), as well as discharges of highly toxic substances (e.g., cyanide) into the marine environment (Reeves et al. 2003). Morbillivirus infection claimed the lives of large numbers of inshore bottlenose dolphins *(Tursiops truncatus)* off the Atlantic coast of the United States in 1982, 1987-1988, and 1993-1994 (Lipscomb et al. 1994a, Kraft et al. 1995, Duignan et al. 1996) and of more than a thousand Mediterranean striped dolphins *(Stenella coeruleoalba)* in 1990-1992 (Domingo et al. 1990, Van Bressem et al. 1991, 1993, Aguilar and Raga 1993). However, it is believed that the 1987-88 mass die off of bottlenose dolphins in the western North Atlantic was triggered by a toxin (Wade 2002a).

It has been proposed that the total population size of Mediterranean Sea striped dolphins, which is listed as vulnerable by the IUCN, has declined by 30% over three generations (ca. 60 years) (Reeves & Notarbartolo di Sciara 2006). It has been suggested that PCBs and other organochlorine pollutants with potential for immunosuppressive effects may have triggered the mass-die off event or enhanced its spread and lethality (Aguilar and Borrell, 1994). Furthermore, unusual luteinized cysts with the potential to impede ovulation have been found in the ovaries of Mediterranean striped dolphins; these cysts have been associated with the high levels of PCB exposure (Munson et al. 1998). The occurrence of the cysts and the reproductive impairment induced by PCBs may be depressing reproductive rates, hence inhibiting demographic recovery, along with decreased food availability caused by overexploitation by fisheries (Reeves & Notarbartolo di Sciara 2006).

Gaydos et al. (2004) assessed the role of infectious disease in killer whales, and they reported the marine *Brucella* species, cetacean poxvirus, cetacean morbilliviruses and herpesviruses, should be highlighted as high priority pathogens for further investigation (Gaydos et al. 2004). Marine Brucella species are Gram negative bacteria closely related to better known terrestrial pathogens in the genus Brucella (Cloeckaert et al. 2001). Infection by Brucella has been documented to cause abortion in captive bottlenose dolphins (Miller et al. 1999), and in wild Atlantic white-sided dolphins (Dagleish et al. 2007), harbour porpoises (Deaville et al. 2008), striped dolphins (Gonzalez et al. 2002) and a number of other cetacean species in the eastern

North Atlantic Ocean (Foster et al. 2007). Historically, herpes viruses were thought to cause cutaneous (Van Bressem & Van Waerebeek 1996) and mucosal (Lipscomb et al. 1996) lesions in odontocetes (Gaydos et al. 2004). Recently, they have been documented to cause severe encephalitis in harbor porpoises (Kennedy et al. 1992) in the eastern North Atlantic, and systemic disease in bottlenose dolphins (Blanchard et al. 2001) in the western North Atlantic (Gaydos et al. 2004).

2.6 Climate Change

The ICES working group on marine mammal ecology (WGMME) (2007) stated that apart from ice-dependent species, where climate change may show a disruption to breeding, feeding habitat and food availability, most other species should show fairly plastic responses, as they are long lived and are likely to show some degree of adaptation to slowly developing change. As highly derived long-lived (i.e., K-selected) species, Arctic marine mammals are ill equipped to respond quickly to rapid climate change (Moore & Huntington 2008). However, for the majority of marine mammals it is difficult to demonstrate causal relationships between changes in distribution, abundance or condition with climate change and variation, due to a lack of baseline data and a lack of relevant long-term datasets.

How marine mammals respond to environmental change depends on a species' adaptability, its natural history and the temporal and spatial scale of perturbation (Huntington & Moore 2008). As a result of climate change, important Arctic habitat, most notably sea ice (Parkinson et al. 1999), will be altered or destroyed - causing extensive redistribution of mobile species, the disappearance of non-mobile species throughout portions of their range, and possible species extinction (Thomas et al. 2004). The physical manifestations of climate change will affect Arctic marine mammals both directly and indirectly by changing their habitat and encouraging increased human presence and activities in Arctic regions (Ragen et al. 2008). The cumulative impact of independent (additive) and interacting (synergistic) risk factors will determine the overall significance of climate change for marine mammals (Moore & Huntington 2008). Apart from a loss of habitat, decreased survival of marine mammals - due to a decline in the health status of animals as they are unable to forage effectively - and declines in reproductive output will occur, if growth and condition of individuals are compromised (Ragen et al. 2008).

Tynan and De Master (1997) suggested alterations in the extent and productivity of ice-edge systems may affect the density and distribution of important ice-associated prey of marine mammals, such as Arctic cod *Boreogadus saida* and sympagic ("with ice") amphipods. Changes in sea ice extent and concentration thus have the potential to alter the distribution, range and migration patterns of marine mammals associated with ice habitats, and thus indirectly affect nutritional status, reproductive success, and ultimately the abundance and stock structure of these species (Tynan & DeMaster 1997).

However, the full reliance of ice-associated whales, narwhals, bowhead and beluga whales on sea ice-mediated ecosystems is unclear (Laidre et al. 2008), and although arctic cetaceans are highly adapted to Arctic seas, they also inhabit open waters for part of the year (Moore et al. 2000, Moore & Huntington 2008). In the case of bowhead whales, reductions in sea ice may actually enhance feeding opportunities on prey, both produced in and/or advected to their summer and autumn habitats (Moore & Laidre 2006). Specifically, the western Arctic population

has increased steadily at 3.4% (George et al. 2004) during (roughly) two decades of sea ice loss in the Alaskan Beaufort Sea (Walsh 2008). An increasing growth rate suggests that current trends in sea ice reduction are not hindering recruitment to this population as it recovers from overhunting by commercial whaling (Moore & Huntington 2008). However, an increasing growth rate, as a consequence of a reduction in sea ice, cannot be assumed for all bowhead whale populations. As such, an interpretation is confounded by the severely depleted state from which bowhead whales are recovering (Laidre et al. 2008). Similar comparisons cannot be made for narwhals or belugas, as trends in population size estimates are unavailable (Moore & Huntington 2008). However, it has been reported that narwhals are one of the most sensitive Arctic marine mammal species primarily due to its reliance on sea ice, specialized feeding, and restricted habitat range (Laidre et al. 2008).

For other whale species that seasonally inhabitat Arctic waters, such as fin, gray, minke, humpback and killer whales, their plasticity of reported behavior within Arctic waters is indicative of species that can adapt their migration habits based upon prey availability (Moore & Huntington 2008). It has been suggested that the larger mysticetes appear to be availing of resources, which may have increased as a result of the boost to pelagic community production from reductions in sea ice (see Moore & Huntington 2008, and ref. therein).

3. Population growth rates

3.1 Maximum population growth rates (R_{MAX})

R_{MAX} is the maximum rate at which a population can increase under optimal environmental conditions and is determined by both the intrinsic life history characteristics of the species and extrinsic factors such as disease, competition and predation. Cetaceans have developed life history traits that enable them to remain relatively buffered from inter-annual variability in environmental conditions (Wade 2002a). As a consequence, though, they are highly vulnerable to overexploitation.

Population growth rates can be determined by estimating a trend in abundance data. A trend is a measure by which a population grows or declines over a certain time period, and the slope of a linear regression on the natural logarithm of abundance represents the rate of increase (r) of a population (Barlow & Reeves 2002). Statistical power is a measure of the probability of detecting a significant change in a population, if that population is truly growing or declining. It has been suggested that, for cetaceans, at least 10 annual surveys with good precision (CV <20) are required to yield a high probability (>80%) of detecting a 50% change in total population size (Barlow & Reeves 2002). Population growth rates are also calculated in an indirect method using life history data, either obtained from long-term behavioural studies or from mortality data. Life history traits such as the age at which a female first starts reproducing, the inter-birth interval and how many years a female can reproduce, all determine how quickly a population can increase, as well as factors such as availability of resources (Wade 2002a).

As outlined in section 1, human exploitation of baleen whales has in many cases reduced populations to well below levels at which maximum net productivity might be expected, although the ultimate degrees of depletion varied form stock to stock (Best 1993). Table 4 presents the most recent available data collated by the Maximum Sustainable Yield Rate (MSYR) workshop in Alaska in 2007 (IWC 2008) on trends in abundance (calculated from published and

unpublished data) for the baleen whale stocks. Available data on trends is lacking for a large number of the baleen whale stocks listed in Table 4. However, the working group did highlight a number of cases where data are available for analysing trends in abundance, such as North Pacific humpback whales, and possibly the common minke whales and Bryde's whales in the western North Pacific. It should be noted that there is a general lack of information on estimates of R_{MAX} for the majority of cetacean species; therefore in most cases in this report, rates of increase are presented for cetacean stocks (and not the MSYR).

Table 4 Adapted from Table 1 in the report of the MSYR (Maximum Sustainable Yield Rate) workshop 2007 (SC/60/REP 5). The Workshop undertook a detailed review of the available published and unpublished literature that presented estimates of trends in abundance for cetacean stocks. The table below has followed Scientific Committee discussions where possible, although for some species/areas stock structure has not been examined by the Scientific Committee for many years. For southern hemisphere baleen whales, where no recent Scientific Committee discussions/agreements have occurred, it was agreed to use ocean basins. Given problems of interpreting CPUE data as a simple index of abundance, uncritical analyses of such data to give trends are not included.

Species: 'Stock'	Trend in abundance (CI)	Data available including time periods	Reference
Blue whale			
North Atlantic			
Western			
Central	5.2 (3.0-7.4)	1979-1988	(Sigurjónsson and Gunnlaugsson, 1990)
	9 (2.0-17)	1987-2001	(Pike et al. 2007)
Eastern			
North Pacific			
Western			
Eastern			
Southern Hemisphere			
	7.3 (1.4-11.6)	1968/69-2001/02 (Integrated trend estimation)	(Branch et al. 2004)
	8.2 (3.8-12.5)	1978/79-2003/04	(Branch, 2007)
Pacific			
Indian			
Atlantic			
Pygmy Blue Whale			
Fin Whale			
North Atlantic			
Newfoundland-Labrador			
Nova Scotia			
East Greenland-Iceland			
Spain/Portugal/ British Isles			
North Norway	5 (-13-+26)	1988-98	(Vikingsson et al. 2007)
North Pacific			

East China Se.0a			
Eastern	4.8 (-1.6-+11.1)	1987-2003	(Zerbini et al. 2003) CI updated by Cooke.
Western			
Southern Hemisphere			
Pacific			
Indian and Pacific	10.2 (4.8-15.6)	1995/96-2004/05	(Matsuoka et al. 2006)
Atlantic			
Sei Whale			
North Atlantic			
Western (Nova Scotia)			
Central (Iceland-Denmark Strait)			
Eastern			
North Pacific			
Western			
Central			
Eastern			
Southern Hemisphere			
Pacific			
Indian			
Atlantic			
Bryde's Whale			
North Atlantic			
Western			
Eastern			
North Pacific			
Western	4.7 (-2.2-+11.6)	1988-2002	(Shimadata et al. 2007)
East China Sea			
Eastern			
Southern Hemisphere			
South Atlantic			
South Indian Ocean			
South African inshore			
Solomon Islands			
Western South Pacific			
Eastern South Pacific			
Peruvian			
Antarctic minke Whale			
Indian			
Pacific			
Common minke whale			
North Atlantic			
North-eastern			
Central			
Western			
North Pacific			
Western ('inshore' 'offshore')			

'J' Stock			
Eastern ('remainder')			
Humpback whale			
North Atlantic			
Western North Atlantic (Caribbean)	3.1 (SE=0.5)	1979-1993	(Stevick et al. 2003)
Eastern North Atlantic (Cape Verde)			
North Pacific			
Eastern	7-8 (no CI)	1991-2003	(Calambokidis and Barlow, 2004; Calambokidis et al., 2004)
Central	7.0 (no CI)	1993-2000	Mobley et al. 2001
Western			
Southern Hemisphere			
BSA (Brazil)	7.4 (0.6-14.5)	1995-1998	(Ward et al. 2006)
BSB (West Africa)			
BSC (East Africa)	12.1 (7.1-17.1)	1988-2002	(Findlay and Best, 2006)
BSD (Western Australia)	10.1 (0.9-19.3)	1982-1994	(Bannister and Hedley 2001)
BSE (Eastern Australia)	10.6 (10.1-11.1)	1984-2002	(Noad et al. 2006)
BSF (Oceania)			
BSG (Ecudor)			
Northern Indian Ocean			
Gray Whale			
North Pacific			
Western gray whale	2.9 (90% 1.9-4.0)	1994-2006	(Cooke et al. 2007)
Eastern gray whale	1.86 (SE=0.32)	1967/68-2001/02	(Punt et al. 2004; Rugh et al. 2005)
Bowhead whale			
Bering-Chukchi-Beaufort	3.4 (1.7-5.0)	1978-2005 (census and catches 1978-2001 (census)	(Brandon and Wade, 2006; Zeh and Punt, 2005)
Spitzbergen			
Eastern Canada-western Greenland			
Sea of Okhotsk			
North Atlantic right whale			
Western			
Eastern			
North Pacific right whale			
Western			
Eastern			
Southern right whale			
SE Atlantic	7.3	1971-2003	(Best et al. 2005)

	(6.6-7.9)		
SW Atlantic	6.8 (5.8-7.8)	1971-2000	(Cooke et al. 2003)
SE Pacific			
SW Pacific	8.3 (5.1-11.4)	1983-1997	(IWC, 2001; Bannister, 2001)

* refer to rates including an allowance for hunting mortality

In general, some of the observed rates of increase (6.8 to 12.1%, Table 4) are much higher than expected for large baleen whales. As a result of whaling, many stocks are considered well below their maximum net productivity level (see section 2.1) and it can therefore be expected that they are exhibiting per capita growth rates near their maximum. However, the magnitude of the increased rates estimated for some of the more depleted stocks raises questions over whether they are in fact biologically feasible, as the maximum possible growth rate for any cetacean population is restricted by its gross annual recruitment rate (Best 1993).

Although maximum growth rates using direct methods have not been determined for harbour porpoise populations, several attempts have been made to estimate potential population growth rates using indirect methods. For harbour porpoises inhabiting the western North Atlantic, Barlow and Boveng (1991) used a re-scaled human life table and estimated the upper bound of the annual potential growth rate as 9.4%. It has been stated though that this maximum theoretical rate may not be achievable for any real population (Waring et al. 2007). Woodley and Read (1991) used a re-scaled Himalayan thar life table to estimate a likely annual growth rate of 4%. Caswell et al. (1998) also estimated a potential population growth rate for this region and incorporated the uncertainties in survivorship and reproduction using a Monte Carlo method. The median potential annual rate of increase was determined to be c. 10%, with a 90% confidence interval of 3-15%, although these values were determined with considerable uncertainly (Caswell et al. 1998). The majority of researchers use 4% as the maximum net productivity rate for this species and subsequently for other small cetaceans, as 4% was obtained by theoretical modeling showing that cetacean populations may not grow at rates much greater than 4%, given the constraints of their reproductive life history (Barlow et al. 1995, Waring et al. 2007).

As part of an assessment for the IUCN, Taylor et al. (2007) estimated the rate of increase for each cetacean species using five life history parameters; age of first reproduction, interbirth interval, oldest age of a reproducing female, calf survival rate, and annual non-calf survival rate. The results from this study are presented in Table 5. It should be noted that 18 of the 22 species of beaked whales lack sufficient data to estimate the rate of increase. These species are a cause of conservation concern due to recent stranding events linked with anthropogenic noise (Cox et al. 2006, Taylor et al. 2007).

The highest rates of increase were determined for common and Antarctic minke whales, with rates of 9% and 8%, respectively. The reason for the high rate of increase for both these species is due to the very low interbirth interval, approximately 1 year, and a low age at first reproduction of 8 years. The life history data on minke whales were obtained from populations that were recovering from human exploitation (see section 2.1), which resulted in a shorter inter-birth interval. Another baleen whale species with a relative high rate of increase of 6% was the southern right whale, which was attributed to a low age at first reproduction, and high

observed survival rates for adults and calves. For all other baleen whales outlined in Table 5, the minimum interbirth interval was two years. In general, the female baleen whale reproductive cycle is two years, with a gestation period of approximately one year, and a lactation period of six months (Lockyer 1984).

However as mentioned previously, Taylor et al. (2007) only used data from one population and within any species large-scale variations in life history parameters can occur between different geographical areas. For example, data presented in the study was obtained from harbour porpoises in the western North Atlantic, with an age at first birth of five years and interbirth interval of one year, producing a population growth rate of 11% (Gaskin et al. 1984, Read & Gaskin 1990, Caswell et al. 1998). A longer interbirth interval of greater than one year has been determined in the eastern North Atlantic (Learmonth 2004), and for Californian waters (Hohn & Brownell 1990). Hohn & Brownell (1990) suggested that the increase in the interbirth interval off California compared to the Bay of Fundy may reflect the heavy exploitation of harbour porpoises and higher prey abundance in the Bay of Fundy.

For common dolphins, a rate of increase of 2% was determined for animal's inhabiting the eastern tropical Pacific, based on an age at first reproduction of 9 years, an interbirth interval of 2.1 years, and a pregnancy rate of 47% (Danil & Chivers 2007). Other populations of common dolphins, though, have reported lower reproductive rates, such as the eastern North Atlantic with a longer interbirth interval of 3.79 years and lower pregnancy rate of 26% (Murphy et al. in press). Both populations (ETP and eastern North Atlantic) have a similar age at first reproduction and age of oldest reproducing female, however if data from the eastern North Atlantic population were used within the Taylor et al. (2007) study, a much lower population growth rate would be estimated for this species. Variations between both *D. delphis* populations in the length of the inter-birth interval may be as a result of inherent population differences; for example, the tropical environment and highly productive waters of the Costa Rica Dome in the ETP enable females to calve all year round, also facilitate females to reproduce more often and shorten the length of their resting period (Danil & Chivers 2007, Murphy et al. in press). However, the ETP population may also be exhibiting signs of density dependent compensatory responses as a result of earlier large-scale incidental captures in the yellowfin tuna fishery (see section 3.2.1), or high contaminant burdens are reducing reproductive output in the eastern North Atlantic population (Murphy et al. in press, section 2.4).

The aim of the Taylor et al. (2007) report was to produce generation length and percentage mature information for all cetacean species. For some species, such as Commerson's dolphin and Hector's dolphin, data were omitted, as they were obtained from populations known to have experienced high levels of bycatch so the oldest individuals were possibly no longer available to be sampled. Where data were unavailable for parameters, for example survival rates, data obtained from southern right whales, humpback whales and bottlenose dolphins populations were used for other species (see Taylor et al. 2007 for further information). As mentioned previously, Taylor et al. (2007) used a single population with complete data for each species, rather than averaging between populations. Further, even though the model took into account senescence in some species, it did not include a decline in reproductive output as individuals mature, which has been noted in various cetacean species such as common dolphins (Danil & Chivers 2007, Murphy et al. in press). Therefore, the estimated rates of increase calculated within the Taylor et al. (2007) study should be used with caution.

Table 5 Reproduced from Taylor et al. 2007. Where data were available for species, values for the five-parameter model together with output for growth rate (r) and estimates of generation length (T) and percent mature (P) for current conditions (at the calculated r) and pre-disturbance conditions (when r = 0). Parameters are: AFR—age of first reproduction, IBI—interbirth interval, O—oldest age of a reproducing female, Oe—estimated oldest age of a reproducing female, S_0—calf survival rate, S_A—annual non-calf survival rate, r—population growth rate given the preceding parameters. Empirical estimates are in bold.

Species	**Common name**	**AFR**	**IBI**	**O**	**Oe**	**S_O**	**S_A**	**r**
Balaena mysticetus	Bowhead whale	**20**	**3.10**	**118**	58	0.82	0.98	0.03
Balaenoptera acutorostrata	Minke whale	8	**1.00**		51	0.81	0.96	0.09
Balaenoptera bonaerensis	Antarctic minke whale	**8**	**1.20**		51	0.81	0.96	0.08
Balaenoptera borealis	Sei whale	**9**	**2.50**	**53**	54	0.81	0.96	0.04
Balaenoptera edeni	Bryde's whale	**9**	**2.50**		53	**0.84**	**0.93**	0.00
Balaenoptera musculus	Blue whale	**11**	2.50		65	0.82	0.98	0.05
Balaenoptera omurai	No common name	9	2.50		54	0.81	0.96	0.04
Balaenoptera physalus	Fin whale	**10**	**2.24**		62	0.81	0.96	0.04
Berardius arnuxii	Arnoux's beaked whale	14	2.00		49	0.80	0.95	0.02
Berardius bairdii	Baird's beaked whale	**14**	2.00	**54**	53	0.81	0.96	0.03
Cephalorhynchus commersonii	Commerson's dolphin	**7**	2.00		28	**0.67**	**0.91**	0.01
Cephalorhynchus eutropia	Chilean dolphin	7	2.00		27	0.80	0.95	0.04
Cephalorhynchus heavisidii	Heaviside's dolphin	7	2.00		28	0.80	0.95	0.04
Cephalorhynchus hectori	Hector's dolphin	**7**	**2.00**		27	**0.79**	**0.94**	0.03
Delphinapterus leucas	Beluga	**7**	**2.88**	**35**	39	0.80	0.95	0.02
Delphinus capensis	Long-beaked common dolphin	9	2.10		31	0.80	0.95	0.03
Delphinus delphis	Short-beaked common dolphin	**9**	**2.10**	**26**	31	0.80	0.95	0.02
Eschrichtius robustus	Gray whale	**10**	**2.00**		55	**0.70**	**0.95**	0.03
Eubalaena australis	Southern right whale	**8**	**3.12**		57	**0.91**	**0.99**	0.06
Eubalaena glacialis	North Atlantic right whale	**10**	**4.00**	**69**	57	**0.88**	**0.99**	0.05
Eubalaena japonica	North pacific right whale	9	4.00		57	0.90	0.99	0.05
Globicephala macrorhynchus	Short-finned pilot whale	**11**	**6.90**	**40**	43	0.83	0.99	0.01
Globicephala melas	Long-finned pilot whale	**12**	**3.30**	**40**	43	0.83	0.99	0.04
Species	**Common name**	**AFR**	**IBI**	**O**	**Oe**	**S_O**	**S_A**	**r**
Grampus griseus	Risso's dolphin	**11**	**2.40**		37	0.80	0.95	0.02

Hyperoodon ampullatus	Northern bottlenose whale	14	**2.00**	**27**	48	0.80	0.95	0.00
Hyperoodon planifrons	Southern bottlenose whale	14	2.00		46	0.80	0.95	0.02
Inia geoffrensis	Amazon river dolphin	**6**	**1.50**	**18**	31	0.80	0.95	0.05
Kogia breviceps	Pygmy sperm whale	**6**	**2.00**	**23**	36	0.80	0.95	0.04
Kogia sima	Dwarf sperm whale	**6**	**2.00**	**22**	32	0.80	0.95	0.04
Lagenodelphis hosei	Fraser's dolphin	**8**	**2.00**	**18**	33	0.80	0.95	0.01
Lagenorhynchus acutus	Atlantic white-sided dolphin	**10**	**2.50**	**27**	32	0.80	0.95	0.01
Lagenorhynchus albirostris	White-beaked dolphin	10	2.50		34	0.80	0.95	0.02
Lagenorhynchus obliquidens	Pacific white-sided dolphin	**10**	**4.70**	**46**	32	0.80	0.95	0.01
Lagenorhynchus obscurus	Dusky dolphin	**7**	**2.40**	**35**	30	0.80	0.95	0.03
Lissodelphis borealis	Northern right whale dolphin	**12**	**2.00**	**42**	31	0.80	0.95	0.02
Lissodelphis peronii	Southern right whale dolphin	12	2.00		31	0.80	0.95	0.02
Megaptera novaeangliae	Humpback whale	**6**	**2.36**		55	**0.76**	**0.96**	0.05
Monodon monoceros	Narwhal	**8**	**3.00**	**50**	39	0.81	0.96	0.03
Neophocaena phocaenoides	Finless porpoise	**8**	**2.00**	**33**	27	0.80	0.95	0.04
Orcinus orca	Killer whale	**14**	**5.02**	**41**	48	**0.91**	**0.99**	0.02
Phocoena dioptrica	Spectacled porpoise	6	1.50		30	0.80	0.95	0.07
Phocoena phocoena	Harbour porpoise	**5**	**1.00**	**24**	27	0.80	0.95	0.11
Phocoena sinus	Vaquita	**6**	**2.00**	**21**	25	0.80	0.95	0.04
Phocoena spinipinnis	Burmeister's porpoise	6	1.50		30	0.80	0.95	0.07
Phocoenoides dalli	Dall's porpoise	**5**	**1.00**	**35**	29	0.80	0.95	0.12
Physeter macrocephalus	Sperm whale	**12**	**5.00**	**59**	51	0.83	**0.97**	0.03
Pontoporia blainvillei	Franciscana	**5**	**1.50**	**24**	26	0.80	0.95	0.08
Sotalia fluviatilis	Tucuxi	8	**2.50**	**30**	30	0.80	0.95	0.02
Sotalia guianensis	Estuarine dolphin	**8**	**2.00**	**30**	30	0.80	0.95	0.04
Sousa chinensis	Indo-pacific hump-backed dolphin	**11**	**3.00**	**40**	33	0.80	0.95	0.01
Sousa teuszii	Atlantic hump-backed dolphin	11	3.00	**0**	33	0.80	0.95	0.00
Stenella attenuata	Pantropical spotted dolphin	**13**	**3.00**	**45**	33	0.80	0.95	0.00
Stenella clymene	Clymene dolphin	7	3.00		29	0.80	0.95	0.02
Stenella coeruleoalba	Striped dolphin	**11**	**3.38**	**49**	33	0.80	0.95	0.01
Stenella frontalis	Atlantic spotted dolphin	**12**	**3.00**		31	**0.76**	0.95	0.01
Stenella longirostris	Spinner dolphin	**7**	**3.00**	**26**	30	0.80	0.95	0.01
Tursiops aduncus	Indo-Pacific bottlenose dolphin	9.5	3.80		33	0.76	0.95	0.00
Tursiops truncatus	Common bottlenose dolphin	**9.5**	**3.80**	**48**	37	**0.76**	**0.95**	0.00

Reilly and Barlow (1986) also calculated theoretical rates of increase for dolphins based on vital rate data and a Leslie matrix model. The maximum value obtained for the rate of increase was 9%. There was no evidence that the highest rates of increase calculated (8-9%) can be achieved by any real dolphin population, as tradeoffs exist between survival and reproduction (Reilly & Barlow 1986). In the Leslie matrix model, the rate of increase was most sensitive to calving interval and non-calf survival, followed by age at first birth, and relatively insensitive to changes in calf survival rate (Reilly & Barlow 1986).

Population growth rates using direct methods have been determined for small cetaceans inhabitating the Eastern Tropical Pacific (ETP). Trends were estimated by fitting the log-linear model $\log(Nt) = \log(N0) + rt$, where t was time in years and the fit was weighted by the squared inverse of the coefficient of variation (Gerrodette et al. 2008). Exponential population growth was expected for the main stocks recovering from effects of the yellowfin tuna fishery; i.e. coastal and northeastern offshore spotted dolphins and eastern spinner dolphins (see section 2.2). Between 1998 and 2006, when reported dolphin bycatch was at low levels relative to population sizes, three of the officially depleted dolphin stocks (coastal and northeastern offshore spotted and eastern spinner dolphins) were estimated to be growing at rates considered to be near the 4-8% maximum possible for dolphins (Reilly & Barlow 1986) (Table 6), indicating that the stocks are recovering (Gerrodette et al. 2008). Whereas, the western/southern offshore spotted dolphins were estimated to be declining at 8% per year. However, this may be as a result of spotted dolphins migrating into waters inhabited by the northeastern offshore stock and needs to be investigated further. For bottlenose dolphins, which were rarely taken in the fishery, it was suggested that the decadal difference in abundance might be as a result of a change in habitat for the species (Gerrodette 2005).

In general, calculating rates of increase using an indirect method, i.e. using mortality and behavioural data via theoretical modelling, will only give an rough indication of the potential maximum growth rate of a species and, due to limited available representative data, these models do not take into account declines in individual reproductive output with age, changes in habitat quality, availability and competition for food resources, impacts of disease and contaminants on the population, and geographic and temporal variations in calf and non-calf survival rates.

Table 6 Reproduced from Gerrodette et al. (2008). Estimates of the exponential rate of change r, with lower and upper limits of the 95% confidence interval on the estimate, for 10 ETP dolphins stocks: for two time periods 1986-2006 and 1998-2006.

	1986 - 2006			1998 – 2006		
Species / stock	*r*	lwr95	upr95	*r*	lwr95	upr95
NE offshore spotted	0.010	-0.014	0.034	0.035	-0.002	0.071
W/S offshore spotted	-0.023	-0.058	0.013	-0.080	-0.189	0.028
Coastal spotted	0.104	0.004	0.204	0.077	-0.091	0.245
Eastern spinner	0.019	-0.013	0.051	0.092	-0.017	0.202
Whitebelly spinner	-0.005	-0.054	0.043	0.062	-0.302	0.425
Striped	-0.004	-0.028	0.020	0.012	-0.095	0.119
Rough-toothed	0.026	-0.022	0.074	0.081	-0.071	0.232
Short-beaked common	0.047	-0.012	0.107	-0.006	-0.221	0.208
Bottlenose	0.040	0.020	0.060	-0.004	-0.033	0.024
Risso's	0.011	-0.017	0.040	0.039	-0.112	0.189

3.2 Factors that may have limited the recovery of certain populations/stocks and variations in rates of recovery for different cetacean stocks

3.2.1 Density dependence

Cetacean populations are regulated through density-dependent changes in reproduction and survival, and it has been suggested that food resources are the main causative agent in the expression of density dependence (Fowler 1984, 1987, Lockyer 1990). In general, increasing population growth rates at low densities and decreasing growth rates at high densities have been reported in marine mammals (Fowler 1981, Hohn et al. 2007). Cetacean population sizes may be reduced due to various reasons such as over-harvesting, incidental capture, habitat loss, degradation and fragmentation.

In long-lived mammal species, indices such as juvenile survival, reproductive parameters (e.g. age at first reproduction and pregnancy rates or birth rates), and the mean body size (or size of body parts) have been suggested as the parameters most likely to be influenced by population level, and are most sensitive to cumulative effects of exposure to conditions over space and time (Fowler 1981, Murphy et al. in press). Observed changes in the above-mentioned biological parameters are expected if variations occur in the availability of food resources, due to changes in the environment and/or changes in population abundance. For example (using a simplistic approach) a decline in population abundance leads to an increase in available resources for the remaining individuals. Increased availability of resources should lead to an increase in an individual's reproductive output and, consequently, an increase in the population growth rate. This increase in reproductive output could be as a result of the following factors; an increase in the pregnancy rate, an increase in the proportion of females simultaneously pregnant and lactating, a shortening of the calving interval, and/or a decrease in the age at

which sexual maturity is attained. An increase in an individual's growth rate also occurs, and this may enable individuals to attain physical maturity at an earlier age.

As populations become relatively large, they tend to have lower population growth rates, and they eventually stop increasing. This form of density dependence is termed compensation, and the level at which a population stabilises is called its carrying capacity (Wade 2002a). Within increasing populations approaching carrying capacity, density dependence would first affect the rate of immature survival, then the age of sexual maturity, the birth rate, and finally adult survival rate. It is important to determine when density dependence is taking place, so it can be distinguished from human-caused effects on population growth, which may require conservation action.

Some of the main evidence of the effects of density dependence altering reproductive parameters has been observed in baleen whales. In the Antarctic, it has been suggested that an increase in food resources as a result of whaling is the main factor causing: early migration to the southern Ocean and the redistribution of sei whales; reported increases in pregnancy rates of sei, blue and fin whales; and decreased age at sexual maturity for sei and fin whales (see Ballance et al. 2006 and ref. therein). For Icelandic fin whales, changes in individual growth rates and mean age at sexual maturation were suggested to be linked to major climatic trends, such as the mid-1960s climatic reversal in the northern North Atlantic (Lockyer 1990). The pregnancy rate and the weight of foetuses were correlated with female body fat condition, which was suggested to be correlated to prey abundance, albeit with a time delay of one to two years (Lockyer 1990).

In an ideal situation, complete biological assessment studies are desirable for assessing the effects of density dependence on populations, including assessment of life history parameters, abundance and trend data, multi-species interactions, environmental factors and resource availability (Lockyer 1990). However in reality, assessing effects of density dependence on reproductive parameters will be become confounded by other extrinsic independent factors such as contaminants (see section 2.4).

3.2.2 Allee effect

Density dependence can also work in the opposite direction, where per capita population growth is slowed at very small population sizes, i.e. the Allee effect. The Allee effect can occur from inbreeding depression, lack of ability to find a mate or behavioural changes that might accompany a reduction in population numbers, such as decreased foraging success or protection from predators (Wade 2002a). On the whole, species are more vulnerable to extinction when populations are small, as they lose genetic variation (heterozygosity) making the chance of extinction due to disturbances, and to environmental fluctuations, greater.

3.2.3 Environmental variability

It has been reported that environmental variability influences net recruitment rates at all population levels, but has larger effects on populations which occupy suboptimal habitats, or are close to their carrying capacity (IWC, 2008). This is because the net recruitment rate cannot exceed R_{MAX}, however good the conditions, but it can drop far below R_{MAX} (or become negative) in poor environmental conditions through a combination of reduced recruitment and/or increased mortality as has happened, for example, to right and gray whale populations.

3.2.4 Variations between cetacean populations/stocks

Very limited data are available for assessing variations in the rates of recovery for different cetacean stocks, within a species and or within a region (see Table 4).

Northern right whale stocks, which are still severely depleted, have shown no signs of recovery or at least no substantial population growth, even though commercial whaling ceased in 1949 (Perry et al. 1999). In 1990, Kraus (1990) suggested that mortality from anthropogenic sources (ship strikes and entanglement in fishing gear) were important factors for inhibiting population growth for the western North Atlantic right whales, and it has been reported that more than half of the living right whales have experienced at least one ship strike or one entanglement (IWC 2001a, Reeves et al. 2003). Intensive long-term effort has tried to reduce human-induced mortality and injury to the western North Atlantic right whale population, but evidence of decreased survival and reproductive rates indicates that the population may be declining (Caswell et al. 1999, Reeves et al. 2003). Whereas in the Pacific Ocean, large unreported kills by Soviet whalers in the 1950s and 1960s may have ended any chance of the right whale's recovery in the eastern and central North Pacific (Brownell et al. 2001).

In contrast, Best (1993) reported rates of population increase for southern right whales ranging from 1.27 to 7.3%. Recent estimates range from 6.3 to 8.3% (Table 4), and it has been stated that if these high rates of population increase continue in the future, this will lead to a substantial recovery of at least some of the southern hemisphere populations (Reeves et al. 2003). Current abundance estimates for southern right whales are c. 7,500 individuals (ANNEX B). A major factor delaying recovery was the illegal and unreported killing of more than 3300 southern right whales by the Soviet Union between 1951/1952 and 1971/1972 (Tormosov et al. 1998, Reeves et al. 2003).

As mentioned previously (in section 2.1), gray whales were over exploited on both sides of the North Pacific, and in 1937 the convention for regulation of whaling listed gray whales as a protected species. Since then the eastern stock - considered to have been commercially extinct since the end of the 19th century - has increased in abundance to approximately 26,300 individuals (ANNEX B). It has been suggested that the eastern North Pacific gray whale population has recovered to a level close to carrying capacity (Wade 2002a), which may be a reason for exhibiting a low rate of increase of 1.86% (see Table 4). However, other studies suggest that the current population is still below 50% of carrying capacity (Wade 2002b, Baker & Clapham 2004). In contrast, the western population has an abundance of approximately 100 animals (Weller et al. 2002, Baker & Clapham 2004) and a low rate of increase (2.9%) for a depleted stock. However, this population suffers from poor reproduction, low calf survival, nutritional stress, and potential habitat exclusion as a result of oil and gas developments (Clapham et al. 1999, Weller et al. 2002). The impact of commercial whaling by the soviets in the 1960s - and in Korean waters at the same time - may be responsible for suppressing any recovery equivalent to that of the America stock (Clapham & Baker 2002).

Although no trend data are available for minke whales (Table 1), a number of populations are doing relatively well. The north-eastern North Atlantic stock has increased to c. 120,000 individuals. This stock was reduced to an estimated 40-70% of its pre-exploitation level of abundance. Estimates for the southern hemisphere Antarctic minke whales are c.761,000 individuals (1982-1989) although no current estimates are available. The total catch from

1957/1958 to 1986/1987 reported by Japan and the Soviet Union (and possibly including a few unspecified "dwarf" minke whales) was 98,202 (Horwood 1990). In contrast, low numbers have been reported for the O and J-stocks in the western North Pacific, with 25,000 minke whales estimated for the Northwest Pacific and the Okhotsk Sea (ANNEX B). The J-stock is thought to have declined by more than 50% because of intensive whaling in the past by China, Taiwan, the Republic of Korea, and Japan. The O-stock is also well below its pre-exploitation abundance but is less depleted than J-stock. Japan continues to hunt North Pacific minke whales, and individuals are also incidentally captured in South Korean waters (Kim 1999) and in set nets in Japan (Tobayama et al. 1992). It has been reported that the scale of removals from J-stock is sufficient to cause serious concern for this population's long-term survival (Baker et al. 2000, Reeves et al. 2003).

Positive trends off Iceland and California contrast with the complete absence of blue whales today off southern Japan, and their apparent rarity in the Gulf of Alaska (Reeves et al. 2003). Although some populations of blue whales in the northern hemisphere appear to have recovered at least partially from their massive over-exploitation in the early to mid-twentieth century, others have not (Clapham et al. 1999). More than 350,000 blue whales were taken by whaling fleets in the southern hemisphere from 1904 to 1967, when they were given legal protection. Thousands more were killed, but not reported, by Soviet whaling fleets in the 1960s and 1970s (Reeves et al. 2003). Numbers of blue whales in the Antarctic remain extremely low, c. 2,300 (including pygmy blue; ANNEX B). Prior to the start of Antarctic commercial exploitation in the early 1900's, there were an estimated 180,000 (range = 150,000–210,000) blue whales in the southern hemisphere (Gambell 1976). Although high rates of increase (7.3-8.2%) have been reported for blue whales in this region, it has been suggested that Antarctic minke whales may have hindered the recovery of blue whales in the southern ocean, due to interspecific competition (Fraser et al. 1992). However, no conclusions can be made about this type of competition until further behavioral and distributional information is collected (Kawamura 1994, Clapham & Brownell Jr 1996, Perry et al. 1999).

For sperm whales, low rates of population growth have been attributed to overexploitation of mature males in high latitudes resulting in insufficient large mature males that are "accessible" to adult females (Whitehead et al. 1997). In contrast, humpback whales, which were extensively overhunted worldwide and bore the brunt of illegal Soviet catches in the southern hemisphere, are showing strong rates of population growth in the North Pacific and for a large number of stocks in the southern hemisphere (Clapham & Baker 2002). A lower rate of increase was reported for the western North Atlantic population, although unlike other regions this population was not affected by illegal whaling in the mid to latter half of the last century.

In the case of small cetaceans, abundance of northeastern offshore spotted dolphins was reduced to about one-fifth and eastern spinner dolphins to about one-third of their pre-fishery level (Wade et al. 2007). Since 1993, only a very small fraction of both *Stenella* populations in the ETP have been reported incidentally caught by the yellowfin tuna fishery, and a recovery of both dolphin populations was expected (Gerrodette 2002). However, by 2002 there was no clear indication of a recovery for either population and several hypotheses have been put forward as to why there was an apparent failure to recover such as; cryptic effects of repeated chase and encirclement on survival and/or reproduction (e.g., internal injuries, stress, hyperthermia); separation of suckling calves from their mothers during the fishing process; unobserved or observed but unreported mortality; ecosystem or environmental changes; effects

due to break up of dolphin schools (e.g., increased predation, social disruption); ecological effects due to removing tuna from the tuna-dolphin association; and lags in recovery due to other interspecific effects (Gerrodette 2002, 2005). Not only have lower pregnancy rates (compared to ETP *D. delphis*) been reported for both *Stenella* populations, calf production has also been reported to be declining in both populations since 1987 (Gerrodette 2005). However the most recent study, which includes new updated (and revised) estimates for the depleted stocks now suggests that these populations are beginning to recover (Gerrodette et al. 2008).

4. Case studies for species identified for assessing trends in population abundance

As part of Task 2, three areas – eastern North Atlantic, western North Atlantic and eastern North Pacific – and seven species were selected for assessing trends in cetacean stocks within JIP areas of relevance. These included the fin whale, humpback whale, common minke whale, sperm whale, long-finned pilot whale, striped dolphin, and harbour porpoise. Five of these species (fin whale, humpback whale, common minke whale, sperm whale and harbour porpoise) were considered for all three of the areas. Due to a lack of data, the long-finned pilot whale was only assessed for the eastern North Atlantic, and the striped dolphin was only assessed for the western North Atlantic and the eastern North Pacific. This section within Task 3 will compile available information on the seven species, inhabiting the three areas, which will include published data on current trends in abundance (rate of increase), maximum rate of increase and the current status of the stocks/populations, if known.

For American waters, the majority of information was obtained from marine mammal stock assessment reports published by the National Marine Fisheries Service (NMFS: http://www.nmfs.noaa.gov/pr/sars/species.htm). Available data for European waters was compiled from various sources.

4.1 Fin whale

4.1.1 Eastern North Atlantic

Five putative stocks have been identified in the eastern North Atlantic; North Norway, West Norway-Faroe Islands, Greenland-Iceland, West Greenland, and British Isles-Spain-Portugal. A pre-exploitation population estimate of 58,000 whales has been determined for northern hemisphere waters (Evans 1987, Notarbartolo di Sciara & Evans 2008). No current abundance estimate is available for the whole North Atlantic population, but recent sightings surveys suggest >46,000 whales, which includes an estimate of 17,000 individuals for British Isles–Spain–Portugal stock (Buckland 1992, IWC 1992, Notarbartolo di Sciara & Evans 2008). The IWC abundance estimate for the central and eastern North Atlantic Ocean is 30,000 (95% CI 23 000–39 000) individuals, and 3,200 (95% CI 1,400-7,000) individuals in the West Greenland stock (IWC, see ANNEX B). The Northern Norway stock has been reported to be increasing at a rate of 5% (Table 4; data obtained between 1988 and 1998); however there are insufficient data available to determine population trends, and the net and maximum rates of increase for the other stocks.

Fin whales have been protected from commercial whaling since 1986, although small numbers are taken by the subsistence whale fishery in Greenland (Notarbartolo di Sciara & Evans 2008).

An annual strike limit of 19 whales has been approved by the IWC for the West Greenland stock, for the period 2008 to 2012 (see section 2.1).

4.1.2. Western North Atlantic

One stock exists in this region, the western North Atlantic stock. It is believed that fin whales sighted in U. S. Atlantic waters undergo migrations into Canadian waters, open-ocean areas, and possibly subtropical or tropical regions (Waring et al. 2007). Although there are no data to support the hypothesis that fin whales have distinct large-scale annual migrations within the western North Atlantic Ocean (Watkins et al. 2000). The best abundance estimate available is 2,269 (CV= 0.37) individuals, although this is a minimum estimate as only part of the stock's range was surveyed in 2006 (Waring et al. 2007). There are insufficient data available to determine population trends, and current and maximum net productivity rates. For the period 2001 to 2005, the minimum annual rate of human-caused mortality and serious injury for the western North Atlantic stock was 2.4 whales/yr, which is less than the calculated potential biological removal limit of 3.4 whales/yr (Waring et al. 2007). This stock is identified as a strategic stock by the NMFS, as the fin whale is listed as an endangered species.

Currently, the IWC scientific committee is undertaking an extensive re-assessment of North Atlantic fin whales, which includes the testing of different stock structure hypotheses.

4.1.3. Eastern North Pacific

Two stocks have been reported within the eastern North Pacific, the California-Oregon-Washington stock and the Alaskan (Northeast Pacific) stock. For the Alaskan stock, an estimate of 5,700 individuals was calculated for waters west of the Kenai Peninsula. However again, this is only a minimum estimate, as surveys only covered a small portion of the stocks range. For fin whales inhabiting coastal waters south of the Alaska Peninsula (Kodiak and Shumagin Islands), an annual increase of 4.8% (95% CI 4.1-5.4%) was determined for the period 1987 to 2003 (Zerbini et al. 2006). The mean annual mortality and serious injury rate from anthropogenic activities for fin whales in American waters is zero (Angliss & Outlaw 2006a).

The California-Oregon-Washington stock is a genetically differentiated stock (Bérubé et al. 2002) with a minimum abundance of 3,279 (CV = 0.31) individuals (Carretta et al. 2007); data were obtained by ship surveys carried out in the summer/autumn of 1996 (Barlow & Taylor 2001) and 2001 (Barlow 2003). It has been suggested that fin whales have increased in abundance in Californian coastal waters between 1979/80 and 1991 (Barlow 1994) and between 1991 and 1996 (Barlow 1997), although these trends were not significant. Total incidental mortality due to fisheries (1.0/yr) and ship strikes (0.4/yr) appears to be less than the calculated PBR (15 whales/yr) (Carretta et al. 2007).

Mizroch et al. (1984) estimated that fin whales in the entire North Pacific were less than 38% (16,625 out of 43,500) of historic carrying capacity. Fin whales in the North Pacific were given protected status by the IWC in 1976 and, as a result, the fin whale is listed as endangered under the Endangered Species Act (ESA), and is classified as depleted under the MMPA (Marine Mammal Protection Act) (Carretta et al. 2007). Both stocks in the eastern North Pacific are classified as strategic stocks by the NMFS.

4.2 Humpback whale

This species was heavily exploited by commercial whaling which reduced its worldwide abundance by more than 90%. Within the North Atlantic, the humpback whale exhibits strong individual site-fidelity for different feeding areas, including the Gulf of Maine, Gulf of St Lawrence, and waters off Newfoundland/Labrador, Greenland, Iceland and Norway (Clapham & Evans 2008). The North Atlantic population has showed signs of recovery from exploitation and, in 1992, it was estimated between 10,400 and 11,570 individuals inhabit this region (Smith 1999, Stevick 2003a), although this value is negatively biased (Waring et al. 2007). There are insufficient data to reliably determine current population trends for humpback whales in the whole North Atlantic (IWC).

4.2.1 Eastern North Atlantic

One stock has been reported in the eastern North Atlantic, and photo-identification and genetic studies have indicated that the eastern North Atlantic stock migrates primarily to the West Indies (March 2002, Stevick 2003b), although some individuals may winter near the Cape Verde Islands (Reiner 1996, Clapham & Evans 2008). Genetic analysis suggests a third, unknown, breeding area (Clapham & Evans 2008). Øien (2001) estimated 889 (CV=0.32) humpback whales in the Barents and Norwegian Seas (Waring et al. 2007). Despite strong fidelity to specific feeding grounds, whales from the whole North Atlantic mix spatially and genetically in the West Indies during the wintertime. The status of eastern North Atlantic stock is unknown (Clapham & Evans 2008).

4.2.2. Western North Atlantic

Only one stock inhabits the western North Atlantic, the Gulf of Maine stock, and the distributional range of this stock encompasses waters off the eastern coast of the United States (including the Gulf of Maine), the Gulf of St. Lawrence, Newfoundland/Labrador, and western Greenland (Katona & Beard 1990). During the winter, the majority of whales migrate to the West Indies, although a significant number of animals are also found in the mid and high-latitudes (Clapham et al. 1993, Swingle et al. 1993).

The best recent abundance estimate for the Gulf of Maine stock in US waters is 847 whales (CV=0.55), derived from a 2006 aerial survey. This estimate was not significantly different from an earlier 1999 survey, which reported 902 (CV=0.41) individuals (Waring et al. 2007). Stevick et al. (2003) calculated an average growth rate of 3.1% (SE=0.005) for the period 1979 to 1993. Barlow and Clapham (1997) applied an interbirth interval model to photo-identification mark-recapture data and estimated a higher growth rate of 6.5% (CV=0.012) for the stock; which is close to its theoretical maximum growth rate of 7.2% (Brandão et al. 2000). Clapham et al. (2003) updated Barlow and Clapham's (1997) analysis using data from the period 1992 to 2000. The updated population growth rate estimate was either 0% using a calf survival rate of 0.51, or 4.0% using a calf survival rate of 0.875. It is not known why a decline in the growth rate has occurred, although it has been suggested that the decline is an artifact resulting from a shift in distribution, as the calf survival rate has not declined since 1996 (Waring et al. 2007). For the period 1992/93, an estimate of 2,509 individuals (CV = 0.077) was determined for eastern Canadian waters, though this estimate is almost certainly negatively biased (Baird 2003).

For the period 2001 to 2005, the (minimum) annual rate of human-caused mortality and serious injury to the Gulf of Maine stock was 4.2/yr (U.S. waters - 3.8; Canadian waters - 0.4), which was

higher than the calculated PBR of 1.1/yr (see Warning et al. 2007). Consequently, this stock is listed as a strategic stock by the NMFS, and because the North Atlantic humpback whale is listed as an endangered species. For the period 2008 to 2012, a catch limit of 20 humpback whales was allocated to the aboriginal subsistence whale fishery in St Vincent and the Grenadines (IWC, see section 2.1).

4.2.3. Eastern North Pacific

In the eastern North Pacific, three separate stocks exist; eastern North Pacific, central North Pacific and western North Pacific. All stocks migrate between their respective summer/fall feeding areas to winter/spring calving and mating areas (Calambokidis et al. 1997b, Baker et al. 1998). The eastern stock winters in coastal central American and Mexican waters, and then migrates to the waters off California and southern British Columbia; the central stock winters off the Hawaiian Islands and migrates to waters off northern British Columbia, southeast Alaska and Prince William Sound; and finally the western stock winters off Japan and migrates, primarily, to waters in the Bering Sea (Angliss & Outlaw 2006b, Angliss & Outlaw 2008).

Eastern stock

Barlow (2003) estimated 1,314 (CV=0.30) humpback whales in Californian, Oregon, and Washington waters based on summer/fall ship line-transect surveys in 1996 and 2001. Calambokidis et al. (2004) estimated the abundance for this region using photo-identification data obtained between 1991 and 2003 (using Petersen mark-recapture estimates). These data show a general upward trend in abundance followed by a large (but not statistically significant) decline in the 1999/2000 and 2000/2001 estimates (Carretta et al. 2007). The 2002/2003 population estimate (1,391, CV=0.22), however, is higher than any previous estimate which may indicate that a real decline did not occur (Calambokidis et al. 2003), or that a real decline was followed by an influx of additional whales from other areas (Calambokidis et al. 2004, Carretta et al. 2007). The rate of increase for the eastern stock was 8% per year for the period 1988/90 to 1997/98 (Calambokidis et al. 1999, Carretta et al. 2007), and between 7 and 8% for the period 1991 to 2003 (IWC, Table 4). The estimated annual mortality and injury due to entanglement (1.2/yr), other anthropogenic sources (0.2/yr), and ship strikes (0.2/yr) in Californian waters is less than the U.S. PBR allocation of 2.3/yr (Carretta et al. 2007).

Central stock

The sum of the available estimates for the known feeding areas is 2,036 individuals (149 in Prince William Sound, 651 in Kodiak, 961 in Southeast, and 275 in British Columbia), which is a minimum estimate (Angliss & Outlaw 2006b, Angliss & Outlaw 2008). This value is well below the Calambokidis et al. (1997) estimate of 4,005 humpback whales based on data collected between 1991 and 1993 (Angliss & Outlaw 2006b, Angliss & Outlaw 2008). Mobley et al. (2001) conducted annual surveys of humpback whale breeding grounds in Hawaii and determined a rate of increase of 7%, for the period 1993 to 2000 (Angliss & Outlaw 2008). The number of animals in the southeast Alaskan portion of its distribution has increased, and the 2000 estimate of 961 whales (Straley et al. 2002) was substantially higher than estimates from the early and mid-1980s (Angliss & Outlaw 2008). The calculated annual mortality and serious injury rate is 5/yr, which is considered a minimum and it is not known if the level of human-caused mortality and serious injury exceeds the PBR allocation of 12.9 /yr (Angliss & Outlaw 2006b, Angliss & Outlaw 2008).

Western stock

There are no reliable abundance estimates for this stock, as surveys of known feeding grounds are incomplete, and not all feeding areas are known. A minimum abundance estimate of 394 (CV = 0.084) individuals (Calambokidis et al. 1997b) is used for calculations in determining the potential biological removal for this stock (Angliss & Outlaw 2006b, Angliss & Outlaw 2008). No information is available on trends in abundance, and the maximum rate of increase for this stock has been set at 7% (Wade & Angliss 1997). The calculated incidental annual mortality rate by U. S. commercial fisheries is 0.2/yr. However, there is a lack of available data on fishery-related mortalities in Japanese, Russian, and international waters.

The humpback whale is listed as endangered under the Endangered Species Act, and therefore designated as depleted under the MMPA. Consequently, all stocks in the western North Pacific are classified as strategic stock by the NMFS.

4.3 Common Minke whale

In the North Atlantic, minke whales have been divided into four management stocks by the International Whaling Commission (Donovan 1991) and recent genetic, tagging, and mark-recapture studies support these divisions on large geographical scales, although actual stock boundaries remain unclear (NAMMCO online document-b).

4.3.1 Eastern North Atlantic

Two minke whale stocks have been identified in this region; the central North Atlantic and Northeastern Atlantic. Genetic studies indicate that the west Greenlandic and central North Atlantic stocks are distinct from the Northeastern Atlantic stock (NAMMCO online document-b), and a separate subpopulation/stock has been proposed to inhabit the North Sea (Andersen et al. 2003). However, it has also been reported that genetic differentiation is relatively weak between all stocks in the eastern North Atlantic (Anderwald et al. 2008). The most recent abundance estimate published for the central and Northeastern Atlantic stocks is 174,000 (CI: 125,000-245,000) individuals (IWC, ANNEX B).

Minke whaling was developed in the 20th century by Norway, Iceland, Greenland and Canada, and stocks were reduced to between 45 and 70% of their pre-exploitation abundance levels (Reeves et al. 2003, Anderwald et al. 2008). In 1986, the IWC instituted a temporary moratorium on commercial whaling; however Norway is not bound by the moratorium and has continued whaling under objection, catching a total of 8,085 minke whales since 1986 (http://www.iwcoffice.org/conservation/table_objection.htm). Since 1993, quotas for the Norwegian minke whale hunt have been established using the Revised Management Procedure (RMP), developed by the IWC (NAMMCO online document-b). Aboriginal subsistence hunting is continued in Greenland, and an annual strike limit of 200 minke whales has been allocated to this fishery for the period 2008 to 2012. Iceland has hunted minke whales for scientific research, and 200 minke whales were caught between 2003 and 2007 (http://www.iwcoffice.org/conservation/table_permit.htm). It is believed though that the harvesting of minke whales in the northeastern and central stocks are well within sustainable levels and therefore do not constitute a threat to those stocks (NAMMCO online document-b). Recent surveys indicate that minke whale abundance is stable or increasing in all areas, and may be approaching pre-exploitation levels (Sigurjónsson 1995, NAMMCO 1999). However, there

are insufficient data to determine population trends and net and maximum rates of increase for stocks in this region.

4.3.2. Western North Atlantic

Only one stock has been reported in the western North Atlantic, the Canadian east coast stock, and the best available abundance estimate is 3,312 (CV=0.74) individuals, which was derived from a 2006 aerial survey (Waring et al. 2007). There are insufficient data to determine population trends and net and maximum rates of increase for this stock. Incidental capture has been reported in - or attributed to - various fisheries, including the Northeast bottom trawl fishery and the Northeast/Mid-Atlantic lobster trap/pot fishery, although not all incidences have resulted in mortalities (see Waring et al. 2007, and ref. therein). For the period 2001 to 2005, the (minimum) U.S. estimated average human-caused mortality rate was 2.6/yr, which was lower than the calculated PBR allocation of 19/yr (Waring et al. 2007). During the period 1997 to 2001, no mortalities or serious injuries were reported in Canadian waters (Waring et al. 2007).

4.3.3. Eastern North Pacific

Two separate stocks exist in the eastern North Pacific; the Alaskan and California-Oregon-Washington stocks. The number of minke whales off California, Oregon, and Washington is estimated to be 898 (CV = 0.65) individuals, based on ship line transect surveys conducted in summer and autumn of 2001 and 2005 (Barlow 2003, Forney 2007). There are insufficient data to determine population trends, and net and maximum rates of increase for this stock. As the annual mortality due to fisheries (0.0/yr) and ship strikes (0.0/yr) is less than the calculated PBR for this stock (5.4/yr), they are not considered a strategic stock under the U.S. MMPA (Carretta et al. 2007).

Because the “resident” minke whales off California, Oregon and Washington appear behaviorally distinct from migratory whales farther north, minke whales off Alaska are considered a separate stock (Angliss & Outlaw 2006c, Angliss & Outlaw 2008). For the Alaskan stock, results of the surveys carried out in 1999 and 2000 provided provisional (not corrected for animals missed on the trackline or responsive movement) abundance estimates of 810 (CV = 0.36) and 1,003 (CV = 0.26) minke whales in the central-eastern and southeastern Bering Sea, respectively (Moore et al. 2002). These estimates are only considered a partial abundance estimate. There are insufficient data to determine net and maximum rates of increase for this stock. Minke whales are not listed as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act, and the number of human-related removals for this stock is currently thought to be minimal (Angliss & Outlaw 2006c).

4.4 Sperm whales

Numbers were substantially reduced by whaling, and in the North Atlantic alone over 20,000 animals were taken since 1950. It has been reported that within the North Atlantic abundance is increasing, though at an unknown rate (Gordon & Evans 2008). No reliable abundance estimates are available for the North Atlantic Ocean, or for other populations. There are no published estimates of growth rates of any sperm whale population (Best 1993, Carretta et al. 2007).

4.4.1 Eastern North Atlantic

Only one stock exists in the eastern north Atlantic, ranging from deep waters off the continental shelf between the Iberian Peninsula and Norway, along the Mid-Atlantic Ridge and around the mid-Atlantic islands (Azores, Madeira, Canaries, Cape Verdes). Females and juvenile males have a limited range and are confined to warmer waters, generally with sea surface temperatures greater than 15° C, and between approximately 45°N and 45°S (Gordon & Evans 2008). Only large males are found in the high latitudes, sometimes close to the ice edge, but generally in deeper, more productive waters.

4.4.2. Western North Atlantic

Two stocks have been reported in this region, the northern Gulf of Mexico and western North Atlantic stocks.

Western North Atlantic stock

In 2004, a survey of the U.S. Atlantic outer continental shelf and slope waters (depths > 50m) between Florida and Maryland (27.5°N and 38°N) was conducted between June and August. Data were analyzed to correct for visibility bias (*g(0)*) and group-size bias, and an abundance of 2,197 (CV =0.47) individuals was estimated (Waring et al. 2007). Combining this value with an abundance estimate of 2,607 (CV=0.57) sperm whales determined for the waters between Maryland to the Bay of Fundy - data obtained from a line-transect sighting survey conducted between June and August 2004 (Palka 2006) produces a combined best abundance estimate of 4,804 individuals for the western North Atlantic stock (Waring et al. 2007). However, this stock abundance estimate is only based upon a small proportion of its known range. There are insufficient data to determine population trends for this stock, and current and maximum net productivity rates are unknown (Waring et al. 2007).

Gulf of Mexico stock

The estimate of abundance for sperm whales in U.S. oceanic waters, pooled from 2003 to 2004, is 1,665 (CV=0.20) individuals (Mullin 2007), which is the best available abundance estimate for this species in the northern Gulf of Mexico (Waring et al. 2007). The Gulf of Mexico is composed of waters belonging to the U.S., Mexico and Cuba, and U.S. waters comprise approximately 40% of this region. There are insufficient data to determine population trends for this stock, and current and maximum net productivity rates are unknown (Waring et al. 2007).

Both stocks in the western North Atlantic Ocean are listed as strategic stocks by the NMFS, as the sperm whale is listed as endangered under the ESA (Waring et al. 2007).

4.4.3. Eastern North Pacific

Two stocks have been reported in the eastern North Pacific, the Alaskan (North Pacific) and the California-Oregon-Washington stocks.

California-Oregon-Washington stock

Carretta et al. (2007) reported an abundance of 2,265 (CV=0.34) individuals which was calculated from data obtained by ship surveys conducted in 2001 (Barlow 2003) and 2005 (Forney 2007). This estimate was corrected for diving animals not seen during surveys. Sperm whale abundance appears to have been rather variable off California between 1979/80 and 1996 but, overall, abundance did not show any obvious trends (Carretta et al. 2007). The annual

rate of human caused mortality and serious injury (0.2/yr) is less than the calculated PBR for this stock (3.4/yr).

Alaskan stock
There is no abundance estimate available for this stock, nor any reliable information on trends in abundance. As the PBR is unknown - due to the lack of abundance data - the annual U.S. commercial fishery-related mortality cannot be considered insignificant (Angliss & Outlaw 2008).

As sperm whales are listed as endangered under the ESA, both stocks are considered as depleted and strategic stocks under the MMPA (Carretta et al. 2007).

4. 5 Long-finned pilot whale

4.5.1 Eastern North Atlantic

Only one population (stock) exists within the eastern North Atlantic Ocean, and a separate population inhabits the Mediterranean Sea. Overall, genetic studies have indicated that there is very little variability in pilot whales mitochondrial DNA throughout the North Atlantic (NAMMCO online document-a). The 1989 North Atlantic Sightings Survey (NASS) survey covered the largest area of potential pilot whale habitat, generating an abundance estimate of 778,000 (CV=0.295) individuals for the eastern North Atlantic (Buckland et al. 1993), although this survey did not extend fully into British waters. In the 20th century, pilot whales have been harvested in the Faroe, Shetland and Orkney Islands, Iceland, Greenland, the eastern U.S. and Newfoundland, Canada (Martin et al. 1990, Nelson 1996). They continue to be harvested in the Faroe Islands and Greenland, and in recent years the annual pilot whale catch by the Faroese has ranged between 228 and 2909 individuals (NAMMCO online document-a). During the period 1709 to 1999 a total of 246,434 pilot whales and 1,766 pods have been caught (NAMMCO online document-a). Pilot whales are taken in southwest Greenland on an opportunistic basis and annual catches of up to 365 animals have been reported (NAMMCO online document-a). Incidental capture has also been reported in the eastern North Atlantic, and off France an estimated 50 to 100 animals/year were killed in fishing nets (Perrin 1994).

4.6 Striped dolphin

4.6.1 Western North Atlantic

Striped dolphins inhabiting the northern Gulf of Mexico and the western North Atlantic are considered as separate stocks for management purposes although movements between both regions may exist. The best estimate of abundance for striped dolphins for US waters in the Gulf of Mexico is 3,325 (CV=0.48) individuals (Mullin 2007). The level of past or current (direct) human-caused mortality in the northern Gulf of Mexico is unknown (Waring et al. 2007). For the western North Atlantic stock, the best abundance estimate for striped dolphins is the sum of the estimates from the two 2004 U.S. Atlantic surveys, resulting in an abundance of 94,462 (CV =0.40) individuals (Warning et al. 2007). The total U.S. fishery-related mortality and serious injury rate for this stock is less than the calculated PBR (Waring et al. 2007). For both stocks, current and maximum net productivity rates are unknown (Waring et al. 2007).

4.6.2 Eastern North Pacific

Only one stock has been reported in the eastern North Pacific, the California-Oregon-Washington stock, with the majority of individuals inhabiting waters off the Californian coast (Carretta et al. 2007). Estimates of abundance from surveys conducted in 1991/93, 1996, and 2001 in California waters were 28,396 (CV = 0.31), 5,489 (CV = 0.48), and 22,316 (CV = 0.65) individuals, respectively (Barlow 2003). There is no reported trend in abundance for this stock (Carretta et al. 2007). The 1996 to 2001 weighted average abundance estimate for this stock is 13,934 (CV= 0.53) individuals (Barlow 2003). Insufficient data are available to determine the net and maximum rates of increase (Carretta et al. 2007)

4.7 Harbour porpoise

4.7.1 Eastern North Atlantic

A number of subpopulations/stocks have been proposed to inhabit the eastern North Atlantic, including the Iberian/Bay of Biscay, Ireland/Western UK, North Sea, Kattegat/Inner Danish waters, Baltic Sea, North Norway/Barents Sea, Faroe Islands, and Iceland (IWC). There are no published data on population trends, or net and maximum rates of increase in this region.

From line-transect surveys (SCANS I) conducted in July 1994, the population abundance in continental shelf waters was estimated at 341,366 (CV = 0.14; 95% CI 260 000–449 000) individuals, including *c.*250 000 in the North Sea, 33,000 in the Baltic Sea, and 36,000 in the Celtic Sea (Hammond et al. 2002, Evans et al. 2008). A follow up survey in July 2005 (SCANS II) covered a wider geographical area and produced an estimate of 386,000 (CV=0.20) individuals (Hammond 2008), and a new estimate of 335,000 harbour porpoises for the 1994 surveyed region after re-analysing the data. Overall abundance in the North Sea did not change substantially. However, results from the SCANS II survey reported that densities in the southern part of the North Sea increased, while densities in more northern regions declined, between 1994 and 2005 (Hammond 2008).

This species was hunted in drive fisheries in the Baltic Sea and Faroe Islands, and up to 3,000 harbour porpoises were taken in a single year by the Danish drive and net fishery, which lasted from the sixteenth century until the mid-twentieth century (Reeves et al. 2003). Current threats include incidental capture in a variety of fishing gear, including bottom-set gillnets for hake, cod, turbot and sole; fixed nets or traps for cod or salmon; herring weirs; trawls, drift nets, and purse seines for cod, herring or plaice (Evans et al. 2008). Independent observer schemes estimated annual bycatches in English and French bottom-set gillnet fisheries of at least 6% of the harbour porpoise population in the Celtic Sea, and the Danish bottom-set gillnet fishery killed approximately 4% of the harbour porpoises inhabiting the central and northern North Sea (see Evans et al. 2008, and ref. therein).

4.7.2. Western North Atlantic

Three stocks have been identified within this region, the Gulf of Maine-Bay of Fundy stock, the Gulf of St Lawrence stock, and the Newfoundland and Labrador stocks.

The Gulf of Maine-Bay of Fundy stock inhabits both American and Canadian waters. In the summer (July to September), harbour porpoises are concentrated in the northern Gulf of Maine and southern Bay of Fundy, and during the rest of the year porpoises are widely dispersed,

inhabiting waters from Maine to New Jersey. However, there does not appear to be a coordinated migration, or a specific migratory route, to and from the Bay of Fundy region (Waring et al. 2007). The best current abundance estimate for this stock is 89,054 (CV=0.47) individuals, based on a 2006 survey, and there is no available information on trends in abundance. Section 3 outlines estimates of R_{MAX} for this stock, and currently 4% - a value based on theoretical modeling - is used (Waring et al. 2007). The PBR allocation is 610/yr, and the total annual estimated average human-caused mortality is 734/yr (CV=0.16) in U.S. and Canadian waters (Waring et al. 2007). Consequently, this stock is listed as a strategic stock by the MMPA (Waring et al. 2007).

4.7.3 Eastern North Pacific

In the eastern North Pacific, harbour porpoises are found in coastal and inland waters, from Point Conception, California to Alaska and across to Kamchatka and Japan. They appear to have more restricted movements along the western coast of the U.S., compared to the eastern coast (Carretta et al. 2007). Eight eastern North Pacific harbour porpoise stocks have been proposed including the Morro Bay, Monterey Bay, San Francisco-Russian River, Oregon/Washington, Inland Washington, southeast Alaska, Gulf of Alaska, and Bering Sea stocks.

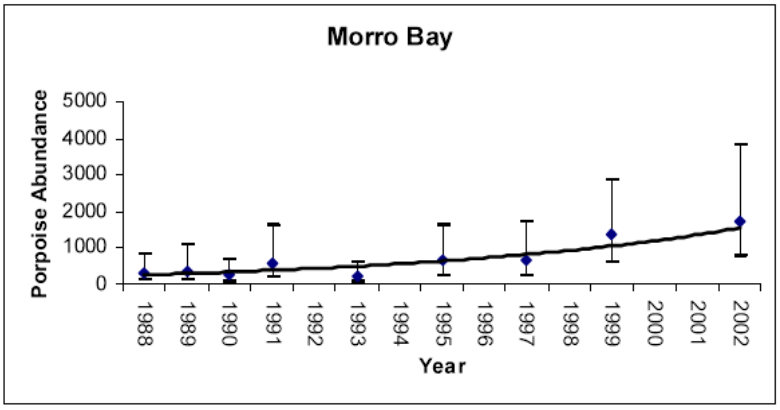

Figure 1 Aerial survey estimates of abundance for the Morro Bay stock of harbour porpoise, 1988-2002. Error bars represent lower and upper 95% confidence intervals. Solid line represents a linear regression on the natural logarithm of abundance over time. The slope of this regression is statistically significant ($p < 0.002$). Figure reproduced from Carretta et al. 2007.

The Morro Bay stock

Based on aerial surveys carried out in 1999 and 2002, an abundance of 1,656 (CV = 0.39) porpoises has been determined for this stock (Carretta & Forney 2004). There has been an increasing trend in porpoise abundance in Morro Bay since 1988, which is statistically significant ($p < 0.002$; see Figure 1). Based on 5 years of fishing effort data (1998-2002), mean annual takes of 4.5/yr is less than the PBR allocation of 10/yr, and therefore this stock has been issued a non-strategic classification (Carretta et al. 2007). As a reliable estimate of the maximum net productivity rate is not available for this stock, the default R_{MAX} value of 4% is used.

The Monterey Bay stock

Based on 1999 and 2002 aerial surveys, the estimate of abundance for this stock is 1,613 porpoises (CV = 0.42) (Carretta & Forney 2004). A linear regression of the natural logarithm of

abundance over time for the Monterey Bay stock is not statistically significant (p=0.64, Figure 2). It has been suggested that the observations of harbour porpoise movements in this region are not directly related to sea surface temperature, but rather to the more complex distribution of potential prey species in this area. As a reliable estimate of the maximum net productivity rate is not available for this stock, the default R_{MAX} value of 4% is used. The annual human-mortality in 2001 (after implementation of the emergency closure for central California gillnet fisheries) was 9.5/yr, which was less than the calculated PBR (10/yr) for this stock (Carretta et al. 2007). Therefore the Monterey Bay harbour porpoise population is not considered strategic under the MMPA. In 2003, a permanent set gillnet closure inside of 60 fathoms was implemented, effectively eliminating set gillnets from most harbour porpoise habitat (Carretta et al. 2007).

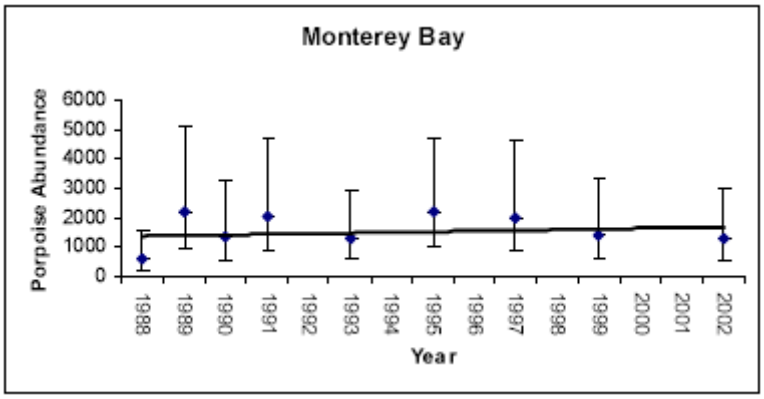

Figure 2 Aerial survey estimates of abundance for the Monterey Bay stock of harbour porpoise, 1988-2002. Error bars represent lower and upper 95% confidence intervals. Solid line represents a linear regression of the natural logarithm of abundance over time. The slope of this regression is not statistically significant (p = 0.64; taken from Carretta et al. 2007).

San Francisco-Russian River stock

Abundance of the San Francisco–Russian River harbour porpoise stock appeared to be stable or declining between 1988 and 1991 and, following this, has steadily increased since 1993 (Carretta et al. 2007). However, the slope of the linear regression on the natural logarithm of abundance over time is not statistically significant (p = 0.24, Figure 3). Based on 1999-2002 aerial surveys the estimate of abundance for this stock is 8,521 animals (CV = 0.38) (Carretta & Forney 2004). A reliable estimate of the maximum net productivity rate is not available for this stock. As the known human-caused mortality or serious injury rate (0.8/yr) is less than the PBR (63/yr), this stock is not considered a strategic stock under the MMPA.

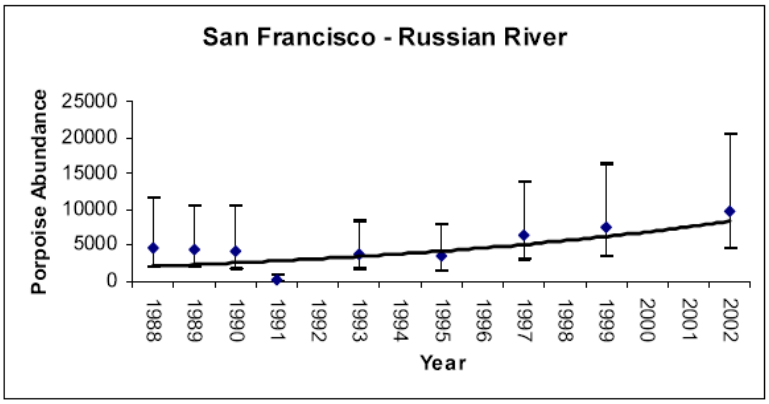

Figure 3 Aerial survey estimates of abundance for the San Francisco-Russian River stock of harbour porpoise, 1988-2002. Error bars represent lower and upper 95% confidence intervals. Solid line represents a linear regression of the natural logarithm over time. The slope of this regression line is not statistically significant (p=0.24; taken from Carretta et al. 2007).

California/Southern Oregon Stock

Forney (1999) assessed trends in relative harbour porpoise abundance in central and northern California based on aerial surveys conducted from 1989 to 1995. No significant trends were evident over this time period for the northern California stock. The 1997-99 survey results continue to show a lack of any trend in relative abundance (Figure 4; Carretta et al. 2007). Based on pooled 1997-99 aerial survey data, including data from both inshore and offshore areas, the updated estimate of abundance for the northern California/southern Oregon harbour porpoise stock is 17,763 harbour porpoise (CV=0.39). Growth rates have not been determined for this region. Because the known human-caused mortality or serious injury (≥0/yr) is less than the PBR (259/yr), this stock is not considered a strategic stock under the MMPA (Carretta et al. 2007).

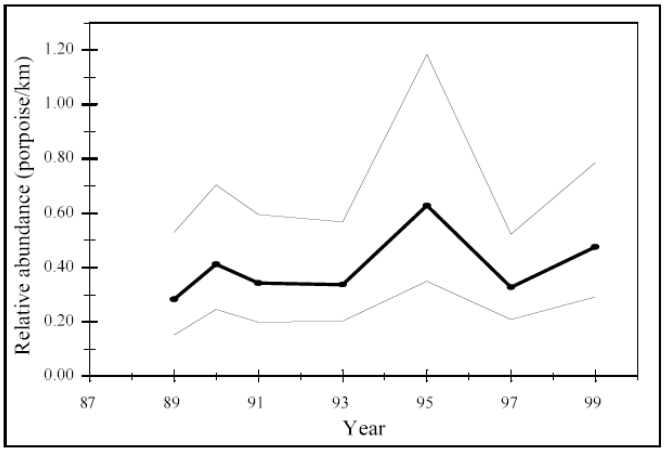

Figure 4 Relative abundance (+/- one standard error) of northern California (Russian River to CA/OR border) harbour porpoise 1989-99, adjusted for sea state and cloud cover (following methods of Forney 1995; taken from Carretta et al. 2007).

Oregon-Washington coastal stock

An estimated abundance of 37,745 (CV=0.38) harbour porpoises has been determined for coastal Oregon (north of Cape Blanco) and Washington waters. There are no reliable data on population trends of harbour porpoise in this region, however, the uncorrected estimates of abundance for the Oregon/Washington Coast stock in 1997 (11,599) and 2002 (11,036) were not significantly different (Carretta et al. 2007). Growth rates have not been determined for this stock. The level of human-caused mortality and serious injury (0.6) does not exceed the PBR (277/yr), and therefore, the Oregon/Washington Coast stock of harbour porpoise is not classified as strategic by the NMFS.

Washington Inland waters stock

The estimated abundance for the Washington Inland Waters stock of harbour porpoise is 10,682 (CV=0.38) animals (Carretta et al. 2007). There are no reliable data on long-term population trends of harbour porpoise in this region, however, the uncorrected estimate of abundance in Washington inland waters was significantly greater in 2002/2003 than in 1996 (3,123 vs. 1,025) (Calambokidis et al. 1997a, Carretta et al. 2007). The data suggest there is a decline in abundance of harbour porpoise in southern Puget Sound although reasons for the apparent decline are unknown. It has been suggested that it may be related to fishery interactions, pollutants, vessel traffic, or other factors, though recently confirmed sightings of harbour porpoise in central Puget Sound have been reported (Carretta et al. 2007). Based on available data, the total level of human-caused mortality and serious injury (15.4/yr) is less than the calculated PBR (63/yr), and the Washington Inland Waters harbour porpoise stock is not classified as strategic by the NMFS.

Bering Sea stock

The abundance of harbour porpoises in Bristol Bay was determined in 1991 and 1999, resulting in estimates of 10,946 and 66,078 individuals, respectively (Angliss & Outlaw 2008). The higher estimate in 1999 was attributed to the fact that a large numbers of porpoises were observed in 1999 in an area which was not surveyed intensely in 1991; the use of a second correction factor for the 1999 estimate confounds direct comparison (Angliss & Outlaw 2008). At present, there is no reliable information on trends in abundance for the Bering Sea stock. A reliable estimate of the maximum net productivity rate (R_{MAX}) is not currently available for this stock. The estimated level of human-caused mortality and serious injury (0.35/yr) does not exceed the PBR (545/yr) (Angliss & Outlaw 2008). However, as abundance estimates are old and information on incidental mortality in commercial fisheries is not well understood, the Bering Sea stock of is classified as a strategic stock by the NMFS.

Gulf of Alaska stock

The latest estimate of abundance (41,854; CV = 0.224) is based on surveys conducted in 1998, and is considerably higher than the previous estimate in the 1999 stock assessment (8,271; CV = 0.309; Angliss & Outlaw 2008). The large variations in abundance stems from changes in the area covered by the two surveys (Angliss & Outlaw 2008). At present, there is no reliable information on trends in abundance for the Gulf of Alaska stock, and a reliable estimate of the maximum net productivity rate (R_{MAX}) is not currently available. Estimated level of human-caused mortality and serious injury (70/yr) is not known to exceed the PBR (347/yr) (Angliss & Outlaw 2008). However, because the abundance estimate is outdated and information on incidental harbour porpoise mortality in commercial fisheries is not well understood, the Gulf of

Alaska stock of harbour porpoise is classified as a strategic stock by NMFS (Angliss & Outlaw 2008).

Southeast Alaska stock

The abundance of harbour porpoise in southeast Alaska was determined in 1993 and 1997, resulting in estimates of 10,301 and 17,076 animals, respectively. However, these estimates are not directly comparable for various reasons; for example, the area surveyed in 1997 was larger than in 1993. There is no reliable information on trends in abundance for this stock, and a reliable estimate of the maximum net productivity rate (R_{MAX}) is not currently available. The estimated level of human-caused mortality and serious injury (0/yr) is not known to exceed the PBR (137/yr). However, as the abundance estimates are old, long-term survey information suggests a decline in the southeast Alaska population and information on incidental harbour porpoise mortality in commercial fisheries is not well known, the southeast Alaska stock is classified as a strategic stock by the NMFS (Angliss & Outlaw 2008).

5. Importance of influencing factors

As can be seen within this report, various factors are linked to controlling cetacean population growth rates and the extent of their control depends on the status of the cetacean population, i.e. low population size, and the severity of the effect, i.e. prolonged large-scale whaling. For large baleen whales, it appears that the main influencing factor has been large-scale anthropogenic removal through whaling. For smaller cetaceans, anthropogenic removal, including both direct takes and incidental capture appear to be the main influencing factors, though other factors such as contaminants, disease, climate change and food availability may also have a strong influence on an already depleted population.

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ANNEX A

Cetacean update of the 2008 IUCN Red List of Threatened Species (downloaded from http://cmsdata.iucn.org/downloads/cetacean_table_for_website.pdf, on the 01.09.08)

Eubalaena glacialis	North Atlantic right whale species	Endangered (EN)
Eubalaena japonica	North Pacific right whale species	Endangered (EN)
Eubalaena japonica	Northeast Pacific right whale subpopulation	Critically Endangered (CR)
Feresa attenuata	Pygmy killer whale species	Data Deficient (DD)
Globicephala macrorhynchus	Short-finned pilot whale species	Data Deficient (DD)
Globicephala melas	Long-finned pilot whale species	Data Deficient (DD)
Grampus griseus	Risso's dolphin species	Least Concern (LC)
Hyperoodon ampullatus	Northern bottlenose whale species	Data Deficient (DD)
Hyperoodon planifrons	Sothern bottlenose whale species	Least Concern (LC)
Indopacetus pacificus	Tropical bottlenose whale species	Data Deficient (DD)
Inia geoffrensis	Boto species	Data Deficient (DD)
Kogia breviceps	Pygmy sperm whale species	Data Deficient (DD)
Kogia sima	Dwarf sperm whale species	Data Deficient (DD)
Lagenodelphis hosei	Fraser's dolphin species	Least Concern (LC)
Lagenorhynchus acutus	Atlantic white-sided dolphin species	Least Concern (LC)
Lagenorhynchus albirostris	White-beaked dolphin species	Least Concern (LC)
Lagenorhynchus australis	Peale's dolphin species	Data Deficient (DD)
Lagenorhynchus cruciger	Hourglass dolphin species	Least Concern (LC)
Lagenorhynchus obliquidens	Pacific white-sided dolphin species	Least Concern (LC)
Lagenorhynchus obscurus	Dusky dolphin species	Data Deficient (DD)
Lipotes vexillifer	Baiji species	Critically Endang. (Poss. Extinct) (CR (P
Lissodelphis borealis	Northern right whale dolphin species	Least Concern (LC)
Lissodelphis peronii	Southern right whale dolphin species	Data Deficient (DD)
Megaptera novaeangliae	Humpback whale species	Least Concern (LC)
Megaptera novaeangliae	Arabian Sea Arabian Sea humpback whale subpop	Endangered (EN)
Megaptera novaeangliae	Oceania humpback whale subpopulation	Endangered (EN)
Mesoplodon bidens	Sowerby's beaked whale species	Data Deficient (DD)

Mesoplodon bowdoini	Andrew's beaked whale species	Data Deficient (DD)
Mesoplodon carlhubbsi	Hubb's beaked whale species	Data Deficient (DD)
Mesoplodon densirostris	Blainville's beaked whale species	Data Deficient (DD)
Mesoplodon europaeus	Gervais' beaked whale species	Data Deficient (DD)
Mesoplodon ginkgodens	Ginkgo-toothed beaked whale species	Data Deficient (DD)
Mesoplodon grayi	Gray's beaked whale species	Data Deficient (DD)
Mesoplodon hectori	Hector's beaked whale species	Data Deficient (DD)
Mesoplodon layardii	Strap-toothed whale species	Data Deficient (DD)
Mesoplodon mirus	True's beaked whale species	Data Deficient (DD)
Mesoplodon perrini	Perrin's beaked whale species	Data Deficient (DD)
Mesoplodon peruvianus	Pygmy beaked whale species	Data Deficient (DD)
Mesoplodon stejnegeri	Stejneger's beaked whale species	Data Deficient (DD)
Mesoplodon traversii	Spade-toothed whale species	Data Deficient (DD)
Monodon monoceros	Narwhal species	Near Threatened (NT)
Neophocaena phocaenoides	Finless porpoise species	Vulnerable (VU)
Neophocaena phocaenoides asiaeorientalis	Yangtze finless porpoise subspecies	Endangered (EN)
Orcaella brevirostris	Irrawaddy dolphin species	Vulnerable (VU)
Orcaella brevirostris	Ayeyarwady River Irrawaddy dolphin subpopulation	Critically Endangered (CR)
Orcaella brevirostris	Mahakam River Irrawaddy dolphin subpopulation	Critically Endangered (CR)
Orcaella brevirostris	Malampaya Sound Irrawaddy dolphin subpopulation	Critically Endangered (CR)
Orcaella brevirostris	Mekong River Irrawaddy dolphin subpopulation	Critically Endangered (CR)
Orcaella brevirostris	Songkhla Lake Irrawaddy dolphin subpopulation	Critically Endangered (CR)
Orcaella heinsohni	Australian snubfin dolphin species	Near Threatened (NT)
Orcinus orca	Killer whale species	Data Deficient (DD)
Peponocephala electra	Melon-headed whale species	Least Concern (LC)
Phocoena dioptrica	Spectacled porpoise species	Data Deficient (DD)
Phocoena phocoena	Harbour porpoise species	Least Concern (LC)
Phocoena phocoena	Baltic Sea harbour porpoise subpopulation	Critically Endangered (CR)
Phocoena phocoena relicta	Black Sea harbour porpoise subspecies	Endangered (EN)
Phocoena sinus	Vaquita species	Critically Endangered (CR)
Phocoena spinipinnis	Burmeister's porpoise species	Data Deficient (DD)
Phocoenoides dalli	Dall's porpoise species	Least Concern (LC)

Physeter macrocephalus	Sperm whale species	Vulnerable (VU)
Platanista gangetica	South Asian river dolphin species	Endangered (EN)
Platanista gangetica gangetica	Ganges dolphin subspecies	Endangered (EN)
Platanista gangetica minor	Indus River dolphin subspecies	Endangered (EN)
Pontoporia blainvillei	Franciscana species	Vulnerable (VU)
Pontoporia blainvillei	Rio Grande do Sul/Uruguay franciscana subpopulation	Vulnerable (VU)
Pseudorca crassidens	False killer whale species	Data Deficient (DD)
Sotalia fluviatilis	Tucuxi species	Data Deficient (DD)
Sousa chinensis	Indo-Pacific humpback dolphin species	Near Threatened (NT)
Sousa chinensis	Eastern Taiwan Strait humpback dolphin subpopulation	Critically Endangered (CR)
Sousa teuszii	Atlantic humpback dolphin species	Vulnerable (VU)
Stenella attenuata	Pantropical spotted dolphin species	Least Concern (LC)
Stenella clymene	Clymene dolphin species	Data Deficient (DD)
Stenella coeruleoalba	Striped dolphin species	Least Concern (LC)
Stenella frontalis	Atlantic spotted dolphin species	Data Deficient (DD)
Stenella longirostris	Spinner dolphin species	Data Deficient (DD)
Stenella longirostris orientalis	Eastern spinner dolphin subspecies	Vulnerable (VU)
Steno bredanensis	Rough-toothed dolphin species	Least Concern (LC)
Tasmacetus shepherdi	Shepherd's beaked whale species	Data Deficient (DD)
Tursiops aduncus	Indo-Pacific bottlenose dolphin species	Data Deficient (DD)
Tursiops truncatus	Common bottlenose dolphin species	Least Concern (LC)
Tursiops truncatus ponticus	Black Sea bottlenose dolphin subspecies	Endangered (EN)
Ziphius cavirostris	Cuvier's beaked whale species	Least Concern (LC)

ANNEX B

IWC estimates of population abundance

(Taken from http://www.iwcoffice.org/conservation/estimate.htm, downloaded on the 01.09.08)

Estimating the abundance of animals that spend most of their time below the surface is difficult. The Scientific Committee has developed guidelines on how to best estimate abundance of whales from ships and aeroplanes for use in the RMP (Revised Management Procedure). Other methods include a combination of visual and acoustic techniques (e.g. bowhead whales off Alaska) or mark- recapture techniques using the natural marks found on some species that allow individuals to be identified (e.g. humpback whales in the North Atlantic). Because of the considerable scientific uncertainty over the numbers of whales of different species and in different geographical stocks, the International Whaling Commission decided in 1989 that it would be better not to give whale population figures except for those species/stocks which have been assessed in some detail. This does not mean that there are not other published estimates of some species or populations or areas.

At present, these are the best estimates (and associated confidence intervals) for some species and areas.

Population	Year(s) to which estimate applies	Approximate point estimate	Approximate 95% confidence limits
MINKE WHALES			
Southern Hemisphere	1982/83 - 1988/89	761,000	510,000 - 1,140,000
	Current	*The Commission is unable to to provide reliable estimates at the present time. A major review is underway by the Scientific Committee.*	
North Atlantic (Central & Northeastern)	1996-2001	174,000	125,000 - 245,000
West Greenland	2005	10,800	3,600 - 32,400
North West Pacific and Okhotsk Sea	1989-90	25,000	12,800 - 48,600
BLUE WHALES			
Southern Hemisphere (excluding pygmy blue)	1997/98	2,300	1,150 - 4,500
The estimated rate of increase is 8.2% (95% confidence interval 3.8-12.5%) per year between 1978/79 and 2003/04			

FIN WHALES			
North Atlantic (Central & Northeastern)	1996-2001	30,000	23,000 - 39,000
West Greenland	2005	3,200	1,400 - 7,200
GRAY WHALES			
Eastern North Pacific	1997/98	26,300	21,900 - 32,400
The population was increasing at a rate of 3.2% (95% confidence interval 2.4% - 4.3%) over the period 1967/68 - 1987/88 with an average annual catch of 174 whales.			
Western North Pacific	2007	121	112 - 130
BOWHEAD WHALES			
Bering-Chukchi- Beaufort Seas stock	2001	10,500	8,200 - 13,500
The net rate of increase of this population since 1978 has been estimated as about 3.2% per year (95% confidence interval 1.4% - 5.1%).			
Off West Greenland	2006	1,230	490 - 2,940
HUMPBACK WHALES			
Western North Atlantic	1992/93	11,600	10,100 - 13,200
A rate of population increase of 3.1% (SE=0.005) was obtained from the Gulf of Maine for the period 1979-1993			
Southern Hemisphere south of 60S in summer (i.e. incomplete)	1997/98	42,000	34,000 - 52,000
Rates of increase. East Australia: 1981-96 12.4% (95%CI 10.1-14.4%). West Australia: 1977-91 10.9% (7.9-13.9%)			
North Pacific	2007	at least 10,000	not yet available
Rates of increase of about 7% have been reported for the eastern North Pacific, 1990-2002.			
RIGHT WHALES			
Western North Atlantic	2001	about 300	not available
Southern Hemisphere	1997	about 7,500	not available
There is evidence of increase rates of 7-8% for populations of Argentina, Australia and South Africa			
PILOT WHALES			
Central & Eastern North Atlantic	1989	780,000	440,000 - 1,370,000

Report

Project Name:	JIP Cetacean Stock Assessment
Reference:	RA1007OGP
Project Manager:	Nicola Quick

Draft report drafted by:	Rebecca Jewell, Gordon Hastie, Len Thomas, Nicola Quick, Philip Hammond	
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Task 4 Deliverable:

Cetacean stock assessment in relation to Exploration and Production Industry sound

Relating cetacean stock trends to E&P sound production within the Areas of Relevance

Summary

The Task 4 deliverable provides a review of the behavioural and physiological responses of cetaceans to exploration and production (E&P) sound and an assessment of the relative importance of oil and gas E&P in influencing cetacean population trends. The relative importance of E&P sound was assessed using a combination of Generalised Additive Modelling (GAM) using a framework similar to that developed during Task 2 but with a number of modifications, and a weighted stepwise linear regression analysis; this was applied to the four most data-rich species contained in the global cetacean database. Factors highlighted in the Task 3 report that influence or control cetacean populations were considered during this analysis, as were the explanatory covariates reported in the Task 2 report. When modelling the density of each of the species using GAMs, latitude was the explanatory covariate most frequently retained in the final model. Environmental covariates (sea surface temperature and North Atlantic oscillation index) and the number of seismic surveys conducted in the area of interest were the covariates most frequently retained in the linear models of between-year population change. The number of seismic surveys conducted was significantly related (at the 5% probability level) to the rate of population change of two of the four cetacean species modelled. The rate of population change of fin whales in the western Atlantic was negatively influenced by the number of seismic surveys conducted (p-value-0.0487) while the number of seismic surveys conducted had a positive effect on the rate of population change of sperm whales in the same area (p-value=0.00613). In addition to this analysis, the global distribution of E&P sound in relation to the Areas of Relevance defined by the JIP was mapped.

1. Introduction

The primary objective of this project is to determine the potential relationships between trends in cetacean populations and E&P sound data within JIP areas of interest. As has been outlined previously, in order to assess any potential relationships between E&P sound and trends in cetacean stocks, it is necessary to first determine the possibility of detecting trends in cetacean populations using published estimates. Knowledge of trends in cetacean populations is vital for informing management decisions, particularly regarding conservation and mitigation measures. However, as was demonstrated during Task 2, trends are often extremely difficult to detect due to low statistical power. During Task 2 the mean estimated

coefficient of variation associated with the population change index meant that only a population change of half an order of magnitude would be detectable with a power of 0.8 (Quick *et al*., 2008). The number and precision of density estimates and the rate of change of cetacean density all effect the detectability of population trends (Gerrodette, 1987); statistical power is highest when there are many, precise density estimates and the rate of change of cetacean density is high (Gerrodette, 1987). The analysis conducted for this task attempted to reduce the problems caused by a low number of density estimates by interpolating values for years without estimated density values, from existing values, allowing the predicted rate of population change between years to be used as a response variable when investigating the effect of seismic activity on trends in cetacean populations.

Anthropogenic sound, such as E&P sound, is known to elicit physiological and behavioural responses in marine mammals (see Section 2), but whether those responses manifest in population-level effects is hard to quantify. Population level impacts could occur as a result of direct physiological effects that influence survival, or through indirect behavioural effects, such as interrupted foraging, mating and migration (Wintle, 2007). Attempts to look for impacts of anthropogenic sound at the population level have been hampered by a lack of abundance estimates for many populations and the poor precision of those abundance estimates that do exist (Gordon *et al*., 2004) and there is an urgent need for systematic research in this area (Perry, 1998). During this task, seismic covariates reflecting the number of surveys, number of kilometres surveyed and intensity of seismic survey activity were used in the models to try and quantify any relationships between trends in cetacean populations and E&P sound.

Cetacean distribution is related to environmental conditions and as a result environmental variability, unless accounted for, can hinder attempts to measure cetacean abundance and detect trends (Forney, 2000). Environmental variability can alter cetacean distribution and mean that surveys of the same area conducted at different times (within or between years) can survey varying proportions of the population of interest, increasing variability and decreasing statistical power. As a result, when trends are detected, if environmental variability has not been accounted for it cannot be known whether the trend represents a change in the abundance of the population, or reflects a change in distribution relative to the survey area. During Task 4, environmental covariates were incorporated in the models but the measures considered were constrained by the spatial resolution of the modelling – the finest spatial scale considered was one Food and Agriculture Organisation of the United Nations (FAO) area (Figure 1.1, Appendix). As a result, only environmental covariates on the scale of ocean basin (sea surface temperature in the North Atlantic and the North Atlantic Oscillation index) were considered during this task. Anthropogenic causes of mortality were considered for each of the four key species and where possible this information was incorporated into the models.

2. Review of behavioural and physiological responses of cetaceans to E&P sound

2.1 Exploration and Production Sound

Oil and gas exploration and production is associated with a range of significant sound sources. These include seismic exploration, vessel and drilling sound, and other sound sources such as pumping, pipe laying, and pile driving. A comprehensive review of the associated sound profiles has been carried out by Richardson *et al.* (1995).

2.1.1 Seismic exploration

Marine seismic surveys produce some of the most intense sounds in the oceans. They utilise arrays of airguns to produce powerful sound waves with sudden releases of pressurised air bubbles. They are designed to deliver a defined and uniform sound pulse (at approximately 200 Hz) downwards towards the seabed. However, by the nature of the pulses, significant energy is also produced up to, and beyond 22 kHz. Source levels of airguns have been measured at 240-246 dB re 1 µPa at 1m directly below the airguns and 220-230 dB re 1 µPa at 1m horizontally from the array (Caldwell and Dragoset, 2000).

2.1.2 Vessel sound

Vessels ranging from small boats to large super-tankers are used during oil and gas E&P and are a significant contributor to overall background sound levels in the ocean. Vessel-generated sound is dominated by propeller cavitation and is a combination of narrowband tonal sounds and broadband sounds with energy spread continuously over a wide range of frequencies. Broadband source levels for most small ships (support and supply ships) are around 170-180 dB re1 µPa. Most acoustic energy is below 1,000 Hz. Larger ships (super-tankers) produce source levels of between 172 and 198 dB re1 µPa (Richardson *et al*., 1995).

2.1.3 Drilling sound

Drilling generally takes place from artificial islands, platforms, or drill-ships accompanied by support vessels. Several studies of underwater sound associated with drilling have been carried out. Drilling from artificial islands produces relatively high levels (maximum= 124 dB re: 1 µPa at 1km) of broadband (10-10 000 Hz) sound underwater that principally affects frequencies between 700 and 1400 Hz (Blackwell *et al*., 2004). Most sound from drilling platforms is below 200 Hz (Malme and Mlawski, 1979). When drilling takes place, auxiliary sound is generated by activities including supply boat and helicopter movements. Drill ships produce maximum broadband source pressure level across the 10 Hz to 10 kHz band of about 190 dB re 1µPa at 1m (RMS).

2.1.4 Additional sources of sound

Additional sound generated includes borehole logging, casing, cementing, perforating, pumping, pipe laying, and pile driving.

2.2 Overview of impacts on cetaceans

A range of high source level sounds are produced by oil and gas exploration and production, which have the potential to have a number of effects on cetaceans. Effects from anthropogenic sound sources can broadly be divided into direct physical, chronic stress, perceptual, behavioural, and indirect effects, but currently limited data exists to quantify these effects with respect to seismic survey sound. However, a number of reviews provide more detailed information on the responses of marine mammals to oil and gas E&P sound (Evans and Nice, 1996; Gordon *et al*., 2004; Harwood and Wilson, 2001; Richardson *et al*., 1995; Richardson and Würsig, 1997), and individually or cumulatively, adverse effects have the potential to lead to population level effects for cetaceans through energetic deficiencies, reductions in viability, and direct injury or mortality. A very brief introduction to these four types of effects from anthropogenic sound is given below.

2.2.1 Physical effects

These may include damage to body tissues, gross damage to ears, permanent threshold shifts (PTS; i.e. permanent reduction in auditory sensitivity), temporary threshold shift (TTS; temporary reduction in auditory sensitivity with recovery), and chronic stress effects that may lead to reduced viability.

2.2.2 Perceptual effects

These include masking of potentially biologically significant sounds (echolocation, communication, and predator avoidance signals).

2.2.3 Behavioural effects

These can encompass a wide range of effects and include disruption of foraging, avoidance of areas, changes to dive and respiratory patterns, and disruption of mating. This could also lead to behaviourally mediated physical damage and stranding.

2.2.4 Indirect effects

These include effects on prey species resulting in reduced prey availability.

2.3 Impacts on relevant species

To date, there is little direct evidence of physical damage or chronic stress resulting from oil and gas sound on the relevant species. However, there are a number of studies that report behavioural and indirect effects. A summary of the speculative broad-scale sensitivities of relevant cetacean species to oil and gas E&P sound is provided in Table 2.1 (Appendix).

2.3.1 Sperm whale

Observations of behavioural responses by sperm whales to seismic surveys can be contradictory. Some unpublished evidence suggests a strong effect; Mate *et al.* (1994) reported that sperm whale density in a preferred area in the Northern Gulf of Mexico decreased to approximately 1/3 of pre-survey levels for the two days after the seismic survey started and to zero for five days after that (although the authors point out that these observations were unexpected and it would be wrong to assume that any causal

relationship was demonstrated). Indications that sperm whales in the Southern Ocean respond to seismic surveys at extreme ranges are provided by Bowles *et al.* (1994). They observed that sperm whales ceased vocalising during some, but not all, periods when a seismic survey vessel was heard firing at a range of 370 km. These were apparently not startle responses to a novel stimulus, as the seismic source had been audible intermittently over the two weeks in which they had monitored acoustically in the area, and had been surveying for some time before the start of their study. The sound source was later found to be an array of 8 x 16l Bolt air guns with an estimated source level of 263 dB re 1µPa @ 1m (back calculated from far field measurements and assuming a point source). At these ranges the seismic pulses had a duration of c. 3 seconds, ranged in frequency from 30-500 Hz and received levels of 120 dB re 1µPa were measured at a range of 1,070 km.

In a recent study, controlled exposure experiments were used to assess whether there were behavioral changes of sperm whales tagged with D-tags at known received levels from airgun sounds (Benoirt-Bird *et al*., 2008). The results showed no evidence of horizontal avoidance to acoustic exposure that ranged from 111 to 147 dB re 1 µPa (rms) (131 to 164 dBp-p re 1 µPa). However, the typical diving behavior was curtailed during full-array firing for one whale that was approached most closely. In contrast, seven other whales continued to make deep foraging dives during controlled acoustic exposure. Further results showed that the 7 foraging whales decreased the pitching movements generated by swimming motion during full-array exposure as compared to post-exposure periods. The small sample size (7 animals) was too small to provide definitive results. However, the odds-ratio of whether these data supported the conclusion of a change in behavior, conducted using Bayesian analyses indicated that there was a decrease in foraging activity of approximately 20%.

In contrast to these reports, other observations suggest that sperm whales show little response and are not excluded from habitat by seismic surveys [e.g. (Rankin and Evans, 1998)]. Swift (1998) used acoustic monitoring techniques to determine the relative abundance and distribution of sperm whales before, during, and after a three week seismic survey on the Rockall Bank west of Scotland. Observations extended over seven weeks to include one week pre-survey, three weeks during the survey, and a week of post-survey monitoring. Acoustic detection rates were actually higher during the seismic survey period than the weeks before or after the survey. It is possible that whale density increased in response to the seismic survey but more probable that changes in detection rates were the result of a seasonal change in sperm whale distribution. Swift (1998) also found no significant difference in detection rates between 'guns on' and 'guns off' periods during the seismic survey itself, suggesting a lack of short-term responses as well. However, it should be remembered that, using hydrophones, these researchers were able to detect sperm whales at ranges of c. 5 miles and this may have made changes in behavior and distributions at lesser ranges more difficult to detect. Madsen *et al.* (2002) describe how a seismic survey was shot over a 13 day period during a detailed study of sperm whale acoustic behavior and the authors were thus able to look for responses to these opportunistic exposures. Whales did not fall silent when air guns started and the authors did not detect any changes to their normal vocal behavior in response to air guns.

Behavioural responses by sperm whales to other vessel traffic can be relatively strong. In New Zealand, when vessels were close, sperm whales have been shown to decrease surface durations, reduce the number of blows per surfacing, and increase the number of dives

without fluking (Gordon *et al*., 1992). Other reactions that have been recorded include course changes and shallow dives (Gaskin, 1964; Reeves, 1992).

2.3.2 Fin, sei and common minke whales

Stone (2003) summarised reports from observers on seismic vessels operating in UK waters between 1998 and 2003, collated by the UK Joint Nature Conservation Committee. When sightings of baleen whales [minke whales (*Balaenoptera acutorostrata*), sei whales (*B. borealis*), and fin whales (*B. physalus*)] were combined, ranges to animals were higher for sightings made during surveys than those made at other times. Furthermore, they altered course more often and tended to orient away from the survey vessel. Fin/ sei whales were less likely to remain submerged while the airguns were firing, possibly because levels of sound would have been greater at depth than near the water surface.

2.3.3 Harbour porpoise

A recent series of controlled exposure experiments has revealed the risk of auditory damage to porpoises from seismic surveys. During the study a porpoise was exposed to single fatiguing sound impulses (produced by an airgun) at increasing received levels (Lucke *et al.*, 2008). Immediately after each exposure the porpoises hearing threshold was tested for any significant changes at three selected frequencies. The received levels of the airgun impulses were increased until TTS was reached at one of the frequencies. At 4 kHz the TTS-criterion was exceeded when the porpoise was exposed to a single impulse at a received sound pressure of 200 dB peak-peak re 1 µPa and a sound exposure level of 164 dB re 1 µPa2·2. Furthermore, the recovery from TTS, i.e. the return of the hearing sensitivity to pre-exposure levels took much longer in the harbour porpoise compared to the other toothed whale species that have been tested (e.g. bottlenose dolphins (*Tursiops truncatus*) and Belugas (*Delphinapterus leucas*). Modelling the impact range of multiple exposures revealed a risk for auditory effects in harbour porpoises over relatively large distances.

Gordon *et al.* (1998) reported on experimental exposures to harbour porpoises in inshore waters around Orkney, UK, using a small source (3 x 40 cu in air guns; source level approx 228 dB re: 1 µ Pa zero-to-peak @ 1 m), and on harbour porpoise detection rates made during commercial seismic surveys. In both cases, porpoise groups were detected acoustically using semi-automated detection equipment (Chappell *et al*., 1996). During experiments in inshore waters the sound source was slowly brought into a bay while a small boat conducted continuous acoustic survey lines within the bay. No changes in the rate of acoustic contact were observed during two-hour periods before, during, and after the controlled exposure. Harbour porpoises were not excluded from an area of preferred habitat by short-term exposure to this modest source. The authors caution, however, that these results can only be applied to the very precautionary experimental approach that they employed which involved using a small source and short exposure periods. Detections of porpoises were also made during full-scale seismic surveys to the north of Shetland. The same acoustic detection equipment was deployed from a guard vessel that kept station about one mile ahead of the seismic vessel. At this range, there were no significant differences in acoustic detection rates for porpoises during periods when the guns were firing and when they were off (during turns between lines for example). This might be taken

as lack of evidence for avoidance by harbour porpoises at ranges of a mile and more, though it is of course possible that avoidance could have occurred at shorter ranges than this. In contrast, Bain and Williams (2006) documented apparent avoidance responses by harbour porpoises at ranges of over 70 km from airguns. At these ranges, the received levels were less than 145 db re 1µPa RMS. Potential avoidance responses are supported by Stone (2003) who showed that harbour porpoises remained significantly further from the airguns during periods of shooting on surveys with large airgun arrays; median distance increased from around 0.5 to 1.5 km from the source vessel during seismic shooting.

Behavioural responses by harbour porpoises to other vessels have been recorded as part of a number of line transect surveys (e.g. Hammond *et al*., 2002, Palka and Hammond, 2001). Avoidance may occur up to 1-1.5 km from a ship (Barlow, 1988) and can be strong within 400m (Polacheck and Thorpe, 1990).

2.3.4 Humpback whale

McCauley *et al.* (1998) detail observations of humpback whales, migrating off Western Australia, during both full-scale seismic surveys and experimental exposures to a single air gun. Comparison of the onshore-offshore distribution of sightings made during pre-seismic aerial surveys and the distribution of sightings from the seismic survey vessel did not indicate any gross disruption of the whale's migration route. However, all pods followed by the independent tracking vessel were observed to respond to the seismic vessel. Reactions were varied and included high-speed swimming (10-15 knots) very close to the surface, course changes at ranges of 5-8 km, and erratic zigzag movements. On two occasions, animals spent an unusually high proportion of time at the surface. The authors speculated that this could be due to reduced sound levels in surface waters. Observers on the seismic vessel made proportionally more sightings within 3 km and relatively fewer at ranges of greater than 3 km during periods when guns were off. Sighting rates at ranges >3 km were approximately 3 times higher during "guns on" than "guns off" periods. This is consistent with whales avoiding the survey vessel out to ranges of >3 km. Total sightings rates were highest during 'transition periods': the periods when guns were turned on and when they were turned off. It was suggested that this could be a startle effect or curiosity, causing whales to come to the surface.

Controlled exposure experiments using an active small air gun array were conducted in an adjoining bay. With the air gun array operating, the source vessel approached groups of humpback whales while a dedicated observation vessel tracked their movements. Whales generally showed speed and course changes to avoid coming closer than 1-2 km to the air gun vessel. However, on several occasions whales were observed to approach and circle the seismic vessel at ranges within 100-400 m (expected exposures 192-177 dB re 1µPa^2 peak-to-peak). In summary, humpback whales showed avoidance behaviour at a range of 5-8 km from a full-scale array and maintained a stand-off range of 3-4 km. Typical received levels at 5 km were measured as 162 dB re 1µPa^2 peak -to- peak. During the trials with a smaller air gun, avoidance was usually evident at 2 km at which received levels were 159 dB re 1µPa^2 peak-to-peak. McCauley *et al.* (1998) suggested that different age classes of humpback whales might exhibit different levels of sensitivity.

Studies of the behavioural responses to drilling sound are limited. However, Malme *et al.* (1985) measured responses by humpback whales to playbacks of the sound generated by

drillships and production platforms. No clear avoidance responses were evident at broadband received levels up to 116 dB re 1 µPa. Studies of responses to vessels are more comprehensive and a review is provided by Richardson *et al.* (1995). Measured responses vary considerably from cases where little or no reaction was observed (e.g. Watkins, 1981) to cases where significant response was observed [e.g. a study on humpback whales reactions to the approach of boats showed that whales increased their swim speed significantly, and adopted a much more direct path after boats left (Scheidat *et al.*, 2004)].

2.3.5 Striped dolphin

Although there is no direct evidence of physical or behavioural effects on this species, there are reports of responses by other delphinids. For example, Goold (1996) reported a reduction in vocalisation rates by common dolphins and suggested that dolphins avoided the immediate vicinity of the airgun array while firing was in progress.

2.3.6 Long-finned pilot whale

Stone (2003) summarised reports from observers on seismic vessels operating in UK waters between 1998 and 2003, collated by the UK Joint Nature Conservation Committee. Results suggest that pilot whales oriented away from the survey vessel and changed their direction of travel during seismic runs with fewer movements in parallel with the ship, and more movements away from ship and across the ships path. In a recent study (Weir, 2008), a pod of 15 short-finned pilot whales (*Globicephala macrorhynchus*) was monitored before, throughout, and following a 30-min ramp-up procedure during a 2-D seismic survey off Gabon. No change in behaviour was apparent during the initial period of the ramp-up. However, 10 min into the ramp-up procedure (at airgun volume of 940 in^3), the nearest whale subgroup turned sharply away from the airguns. Subsequent behaviour included milling, tail-slapping, and a change of the animal's direction of movement directly away from the course of the source vessel. The author (Weir, 2008) speculates that this represents an avoidance response by the pilot whales to the ramp-up.

2.4 Conclusions

- A range of high source-level sounds are produced by oil and gas E&P.
- Main concerns derive from sound produced by seismic exploration. This has the clear potential to cause physical damage to all relevant cetacean species at certain distances or decibels. However, there is little direct evidence that sound produced has caused physical injury or perceptual effects.
- There is published evidence for behavioural effects of seismic, drilling, and vessel generated sound.
- Although direct evidence of effects of E&P sound in some of the relevant species is lacking, with respect to the sounds produced, there is potential for E&P sound to lead to population level effects for all relevant species.

3. Spatio-temporal distribution of E&P sound

As previously described, oil and gas exploration and production sound can arise from a number of sources, including seismic surveys, vessel and drilling sound (see Section 2.1) and the total amount of sound entering the environment is difficult to quantify. The best

available data on oil and gas E&P sound data is from the seismic surveys conducted during the exploration stage of the process so, seismic survey data were used to investigate the relationships between oil and gas E&P sound and cetacean trends in this assessment.

The following review of oil and gas E&P sound is based on the commercially available IHS seismic data set (http://energy.ihs.com/Products/Smarket/). While it is not a complete record of E&P seismic surveys, it is marketed as "the most complete in the industry", containing over 47,000 records of geophysical surveys from around the world, and it is the sole source of data used in this review. The same dataset was used during the statistical analysis to explore trends in cetacean populations in relation to oil and gas E&P sound and it was assumed that any seismic surveys missing from the IHS database were missing at random. The validity of this assumption is difficult to quantify, but a distinct lack of records of seismic surveys from the west coast of America suggests that the assumption may not hold. Alternative sources of seismic survey data were investigated and a summary of these is available in Section 7.1 (Appendix).

The IHS database was supplied in the form of a GIS file with the centre location of each seismic survey marked on a map of the world. Associated with each centre point was a record in the attribute table containing the details of the survey, for example, the start year of the survey, the seismic type (e.g. 2D, 3D, 4D) and kilometres surveyed. The attribute table associated with the shapefiles was used to generate the summary statistics below and to formulate the seismic covariates later used in the analysis

3.1 General overview of E&P sound data

Of the 47744 records in the database, 93% (n=44297) refer to geophysical surveys using seismic methods (Figure 3.1, Appendix). As our interest is in the sound produced during oil and gas exploration and production, only records of surveys utilising seismic methods (rather than gravity or magnetic methods) were investigated further. The database contained a mixture of onshore and offshore seismic surveys; only records of offshore surveys were of interest. Offshore surveys comprised 38% of the surveys that utilised seismic methods. Of the remaining records (i.e. records of offshore seismic surveys, n=16958) in the database, the majority (99%) of records contained information on the type of seismic survey used (2-dimensional [2D], 3-dimensional [3D], 2+3D or 4-dimensional [4D]) and the proportion of surveys falling into each category was examined (Figure 3.2). Two-dimensional surveys utilise one streamer containing multiple hydrophones to detect the sound waves reflected from subsurface structures of the seabed. Using a single streamer means that a 2-dimensional cross section of the seabed can be investigated. Three-dimensional surveys have a higher intensity and utilise multiple streamers (usually 6) towed in parallel to allow reflected sound waves to be used to construct a 3-dimensional image of the seabed. This seismic method is used when a higher resolution cross-section of the seabed is required. Some surveys in the database are listed as being a mixture of 2D and 3D methods. 4D surveys utilise the same methodology as 3D surveys but are repeated over time to monitor changes in hydrocarbon reservoirs due to production.

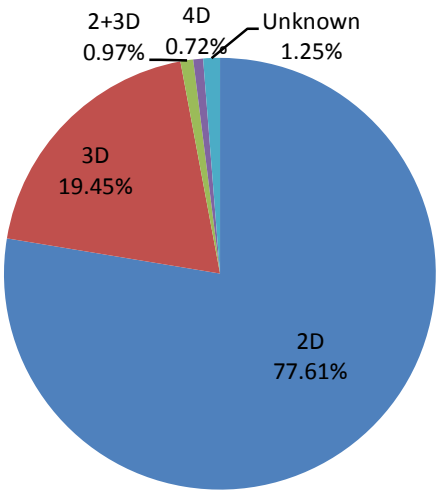


Figure 3.2: Proportion of offshore seismic surveys recorded using each seismic survey type.

3.2 Spatial distribution of E&P sound

A preliminary look at the spatial distribution of the offshore seismic surveys contained within the database revealed that the majority of surveys occurred within European waters (Figure 3.3).

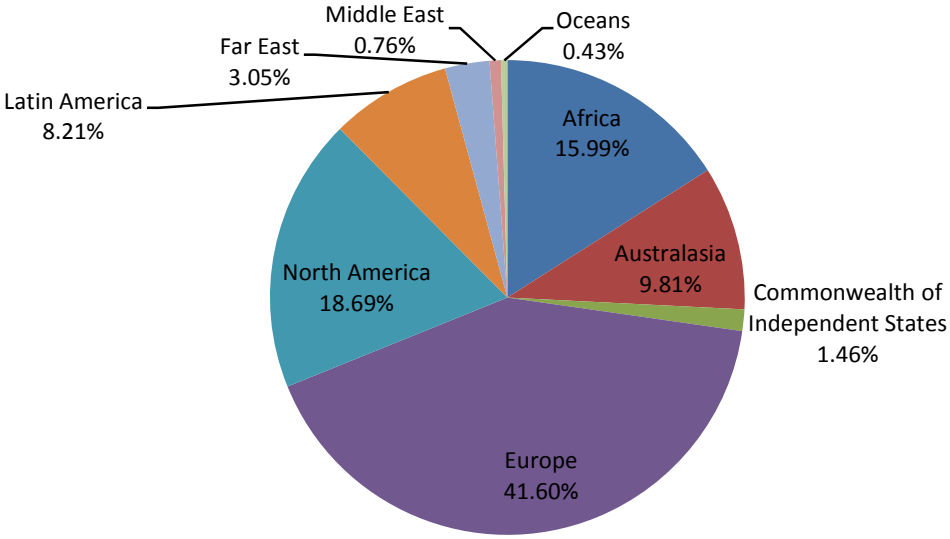


Figure 3.3: Broad scale spatial distribution of offshore seismic survey records within database. These regions do not correspond to AORs in Figure 3.4.

This pattern is also apparent when looking at a map of the global distribution of seismic surveys (Figure 3.4).

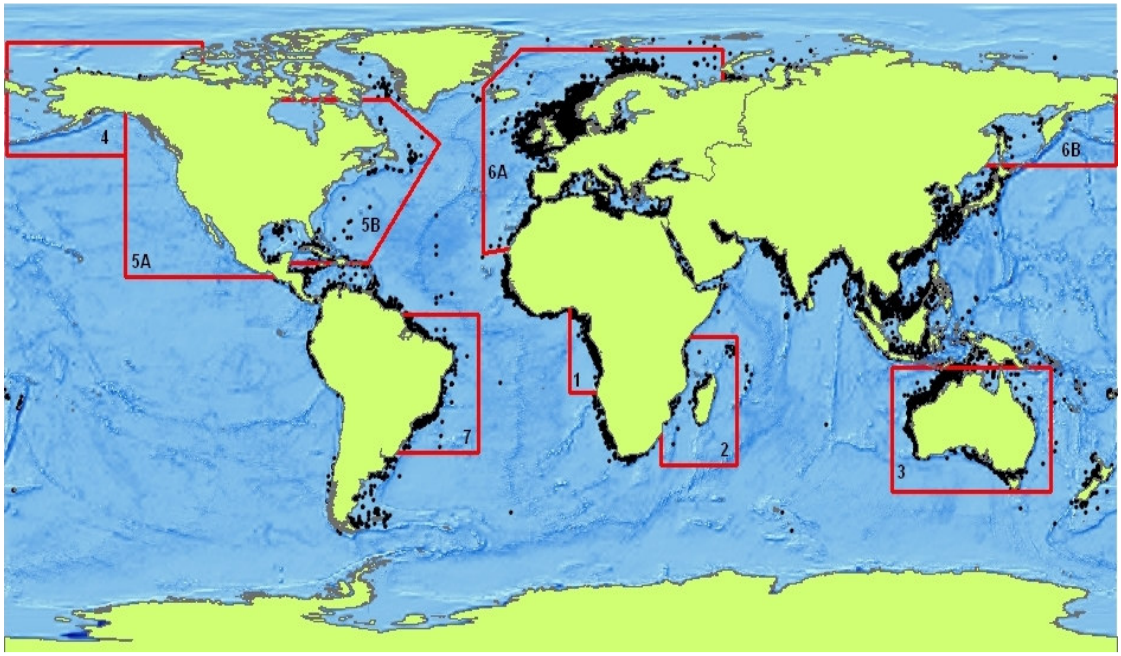


Figure 3.4: Global distribution of seismic survey records (the centre points of which are shown with a black dot) contained within the IHS database in relation to the Areas of Relevance. The regions described in Figure 3.3 do not correspond to AORs.

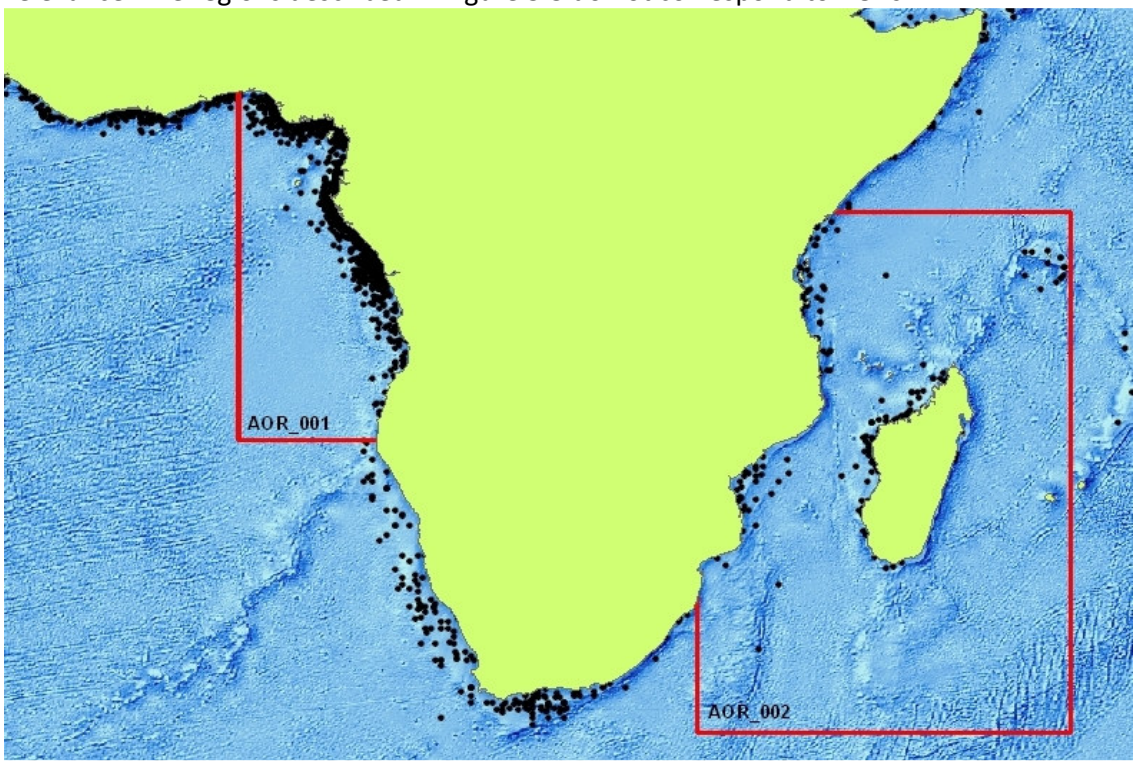


Figure 3.5: Distribution of seismic surveys in AOR_001 and AOR_002.

Seismic surveys have been conducted in both AOR_001 and AOR_002, but activity has been particularly intense off the west coast of Africa (AOR_001) (Figure 3.5).

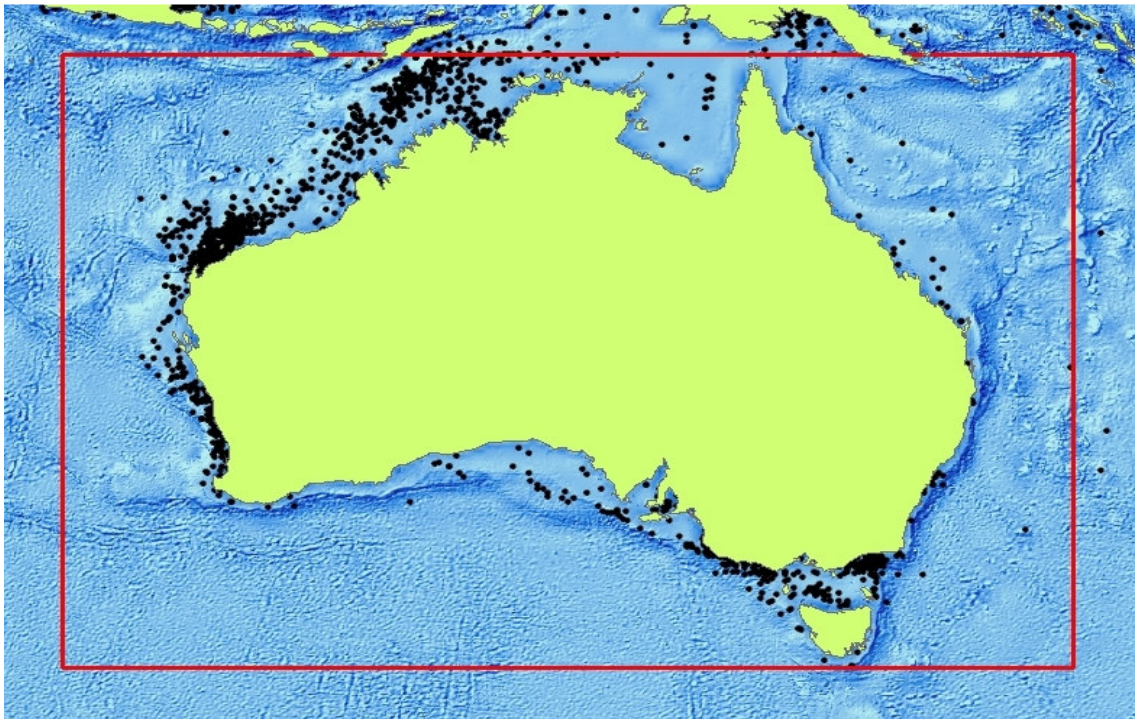


Figure 3.6: Distribution of offshore seismic surveys in AOR_003.

Seismic surveys have been conducted in all Australian offshore regions, but the largest numbers of surveys have been completed in Bass Strait and off the coast of western Australia (Figure 3.6).

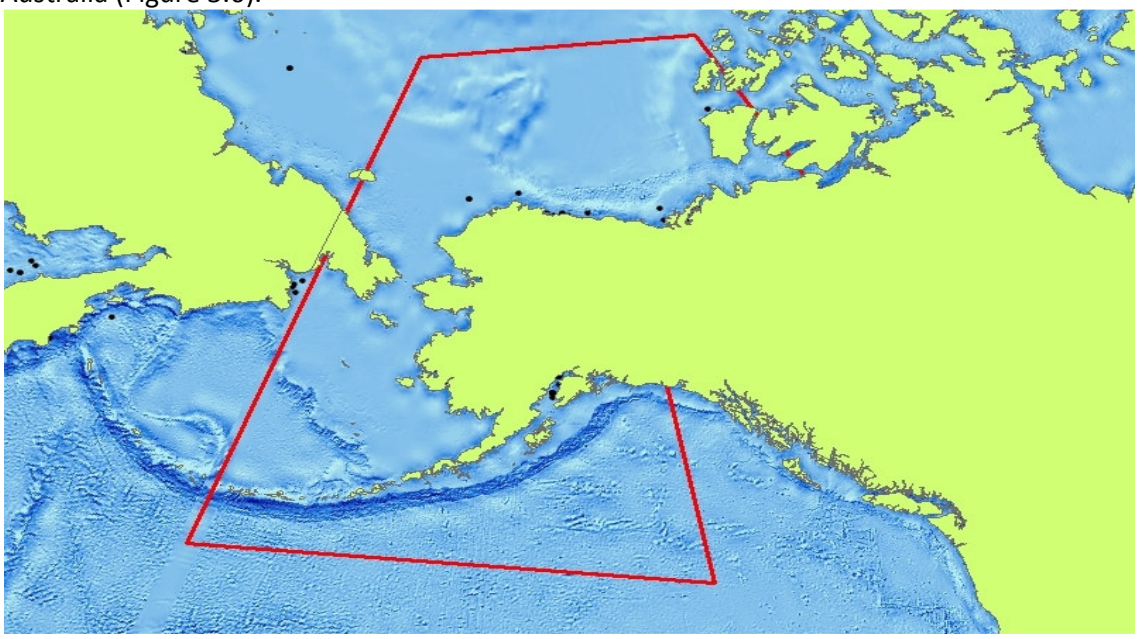


Figure 3.7: Distribution of offshore seismic surveys in AOR_004.

The IHS seismic survey database contains only a small number of records of seismic surveys conducted in Alaskan waters (Figure 3.7). A quick search of the National Archive of Marine Seismic Surveys online database (http://walrus.wr.usgs.gov/NAMSS/index.html - see section

7.1, Appendix) suggests that 87 offshore seismic surveys have been conducted in this area. The IHS seismic survey database does not seem to be representative of the seismic effort conducted in this region.

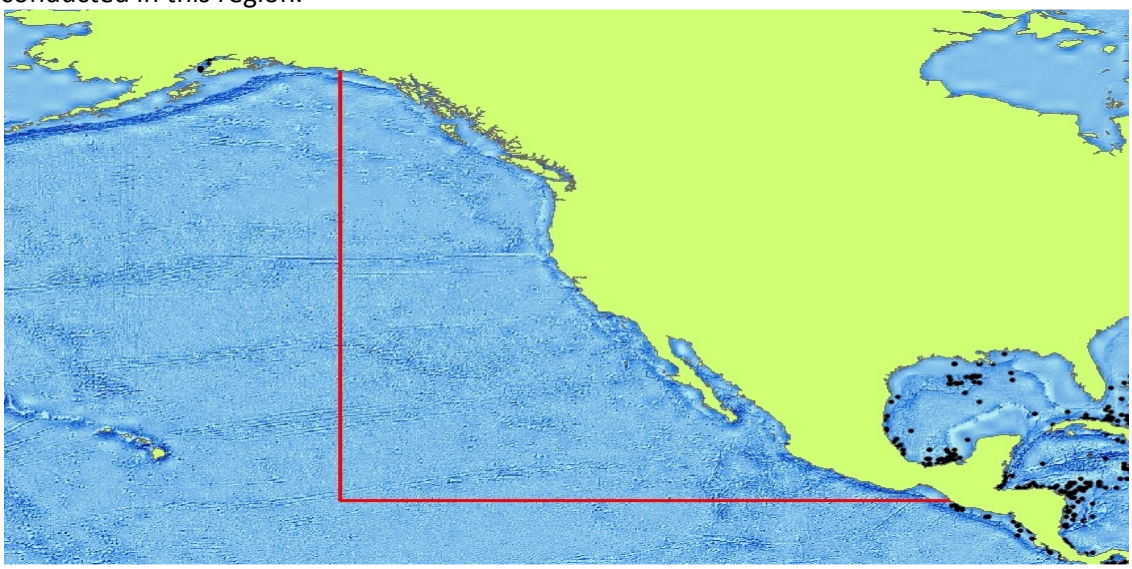


Figure 3.8: Distribution of offshore seismic surveys in AOR_005A.

The IHS seismic survey database does not contain any records of seismic surveys conducted off the west coast of Canada, the USA or Mexico (Figure 3.8). Again, the National Archive of Marine Seismic Surveys online database suggests that 72 seismic surveys have been conducted off the west coast of Canada and the USA.

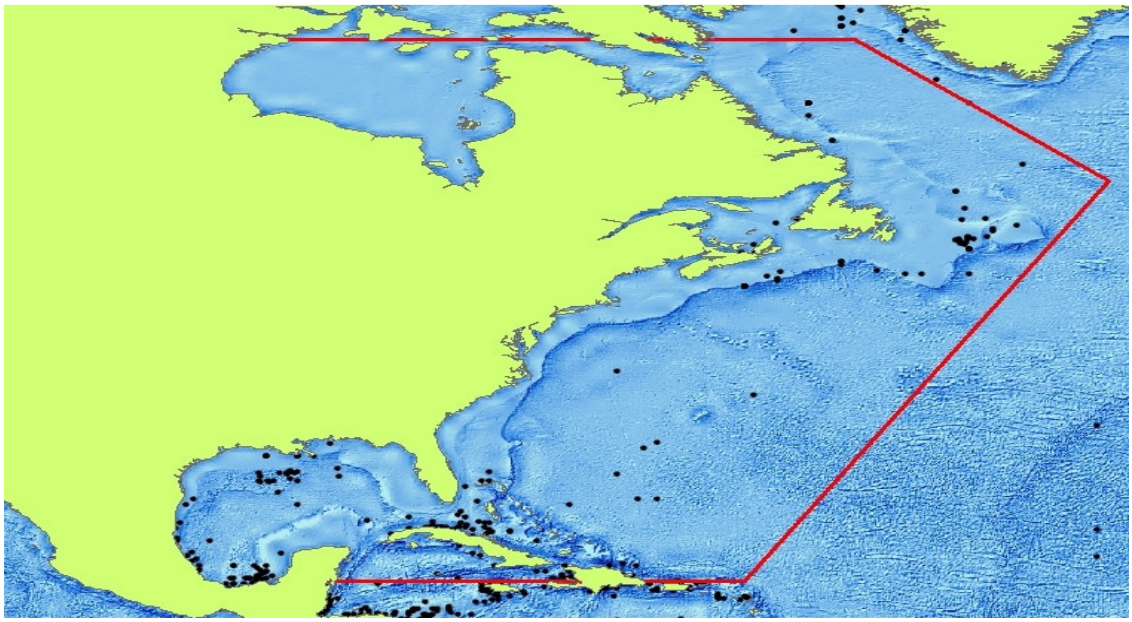


Figure 3.9: Distribution of offshore seismic surveys in AOR_005B.

Oil and gas exploration in AOR_005B seems to have been concentrated in the Gulf of Mexico and Caribbean Sea (Figure 3.9). Another cluster of seismic surveys occurred on the Grand Banks to the east of Newfoundland, Canada.

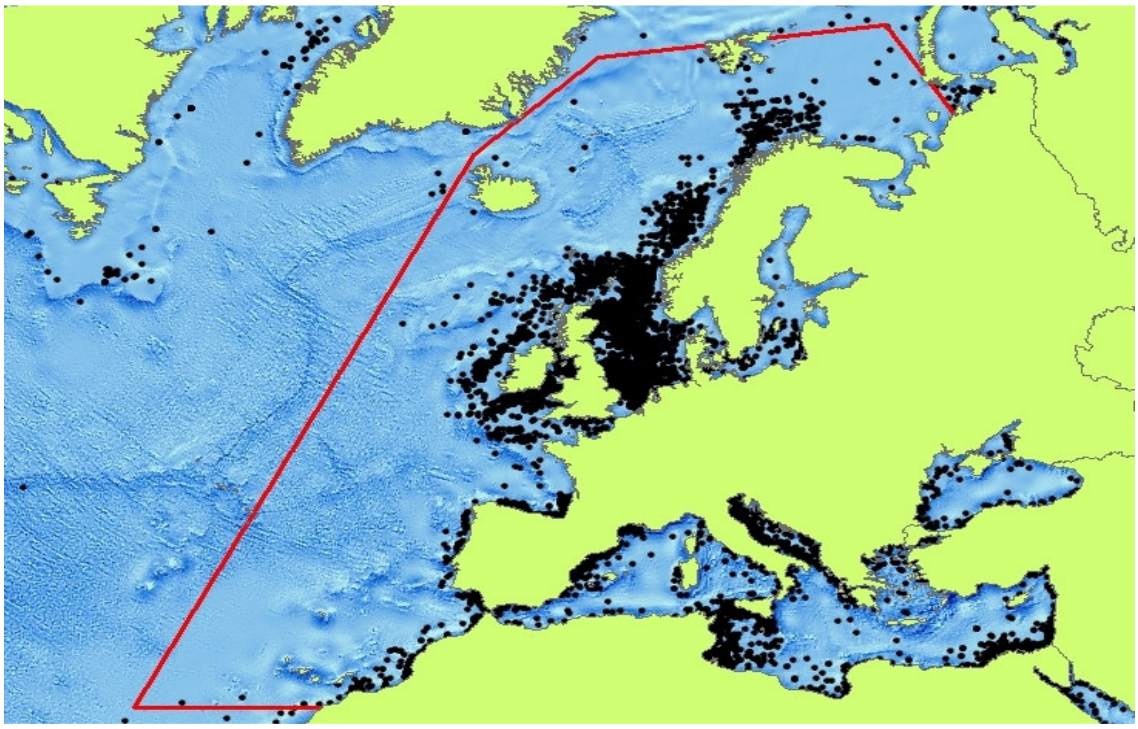


Figure 3.10: Distribution of offshore seismic surveys in AOR_006A.

Shelf waters in the NE Atlantic have been extensively surveyed for oil and gas reserves (Figure 3.10), seismic survey effort has been particularly high in the North Sea and around the coasts of the United Kingdom, Ireland and Norway. The offshore waters of Spain and Portugal have been heavily surveyed, as have regions within the Mediterranean.

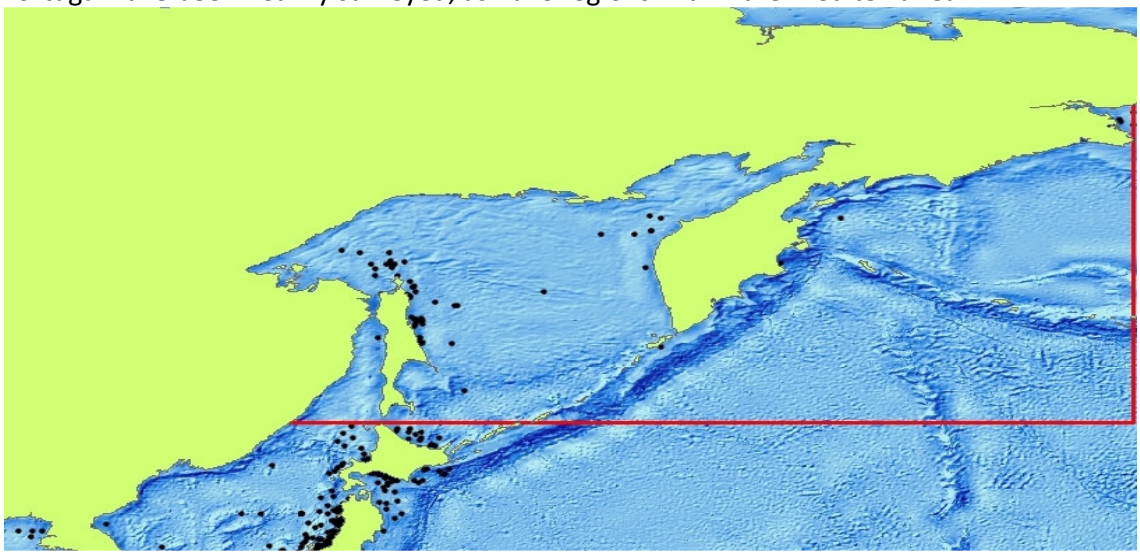


Figure 3.11: Distribution of offshore seismic surveys in AOR_006B.

Most of the seismic surveys conducted in AOR_006B have occurred close to Sakhalin Island in the Sea of Okhotsk, while a small number have been to the north of Kamchatka Peninsula (Figure 3.11).

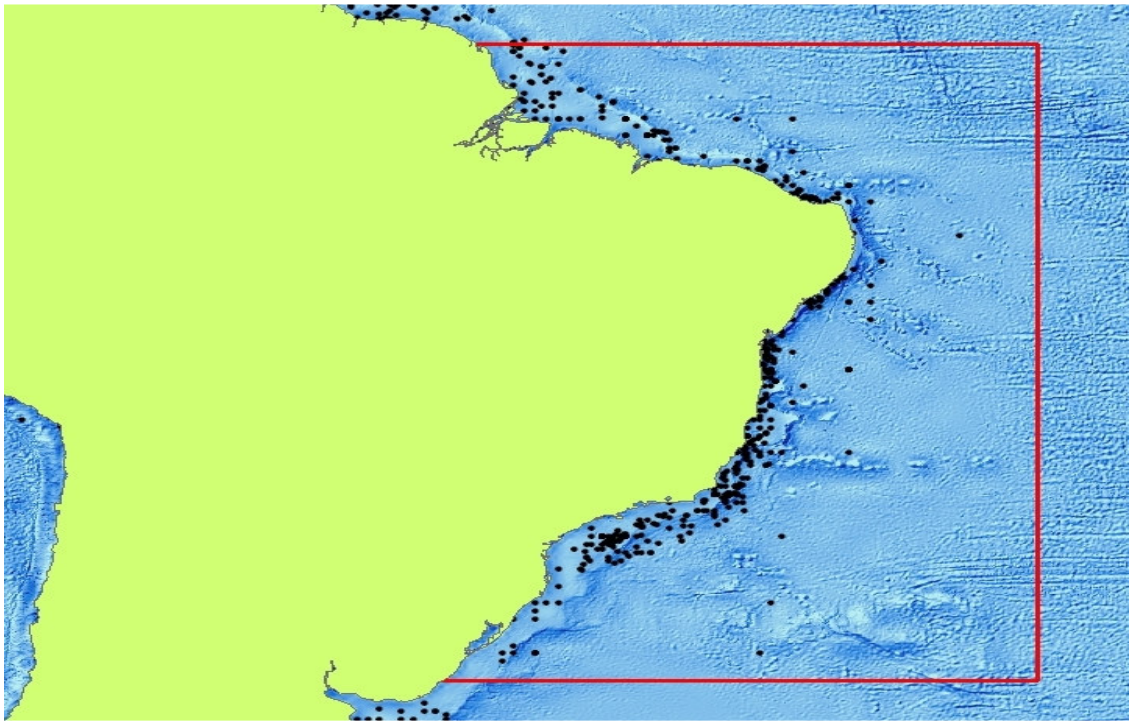


Figure 3.12: Distribution of offshore seismic surveys in AOR_007.

A large number of seismic surveys have been conducted off the coast of Brazil in South America (Figure 3.12).

Calculating the total number of shot points fired better reflects the variation in intensity of seismic activity of different seismic survey types. The number of shot points fired in the two main areas of interest (AOR_005B and AOR_006A) was calculated for each seismic survey type. The number of shot points fired during 2D and 2+3D seismic surveys was calculated by multiplying the number of kilometres surveyed (across all years) by 40, while for 3D and 4D surveys the total area surveyed (during 3D and 4D surveys) was multiplied by 800 (ASCOBANS, 2006). The totals number of shot points was obtained by summing the total from each seismic survey type. The necessary information was often missing from the IHS seismic survey database and as a result the mean number of kilometres surveyed or area surveyed calculated for each year may not be accurate; therefore the number of shot points fired can only be considered an estimate. The number of shot points fired during seismic surveys of each type was used to generate density surface maps of seismic effort in the two main areas of interest (AOR_005B and AOR_006A). Although a number of seismic surveys have been undertaken in the Caribbean Sea the total number of shot points fired during those surveys was not particularly high, as most of the surveys were 2D (Figure 3.13A). The majority of surveys conducted in the Gulf of Mexico were 3D and of a higher intensity, with more shot points fired (Figure 3.13B). Combining all seismic survey types reveals that the highest densities of shot points have been fired in the Gulf of Mexico and off Nova Scotia (Figure 3.14). The IHS seismic survey dataset did not contain any records of 4D seismic surveys conducted within FAOs 21 + 31.

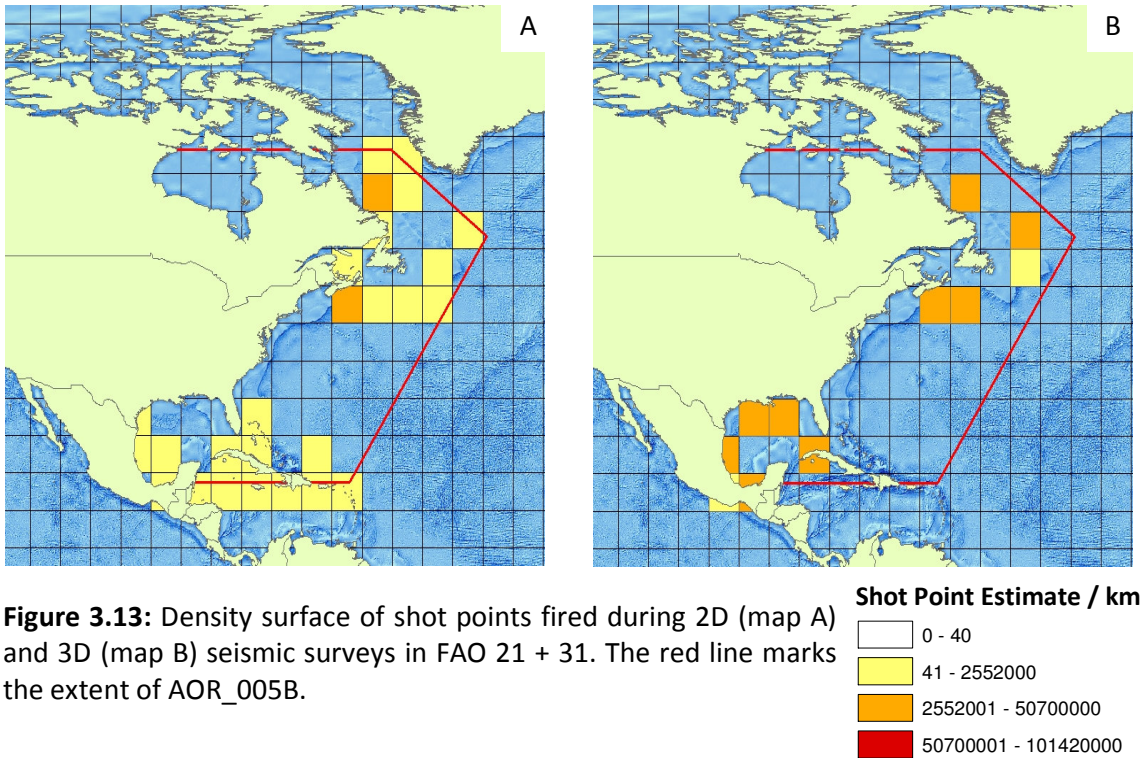


Figure 3.13: Density surface of shot points fired during 2D (map A) and 3D (map B) seismic surveys in FAO 21 + 31. The red line marks the extent of AOR_005B.

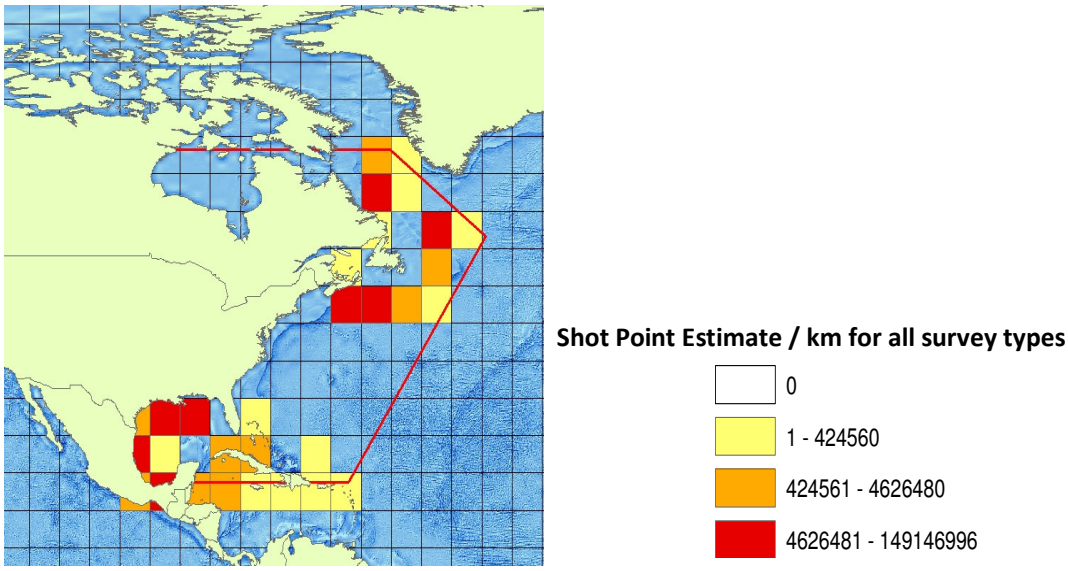


Figure 3.14: Density surface of shot points fired during all seismic survey types combined in FAO 21 + 31. The red line marks the extent of AOR_005B.

Seismic activity has clearly been intense in the North Sea, off the coast of Norway and in waters surrounding the United Kingdom, Ireland and parts of the Mediterranean including the Balearic Sea, the Libyan Sea and the Adriatic Sea (Figures 3.15 and 3.16).

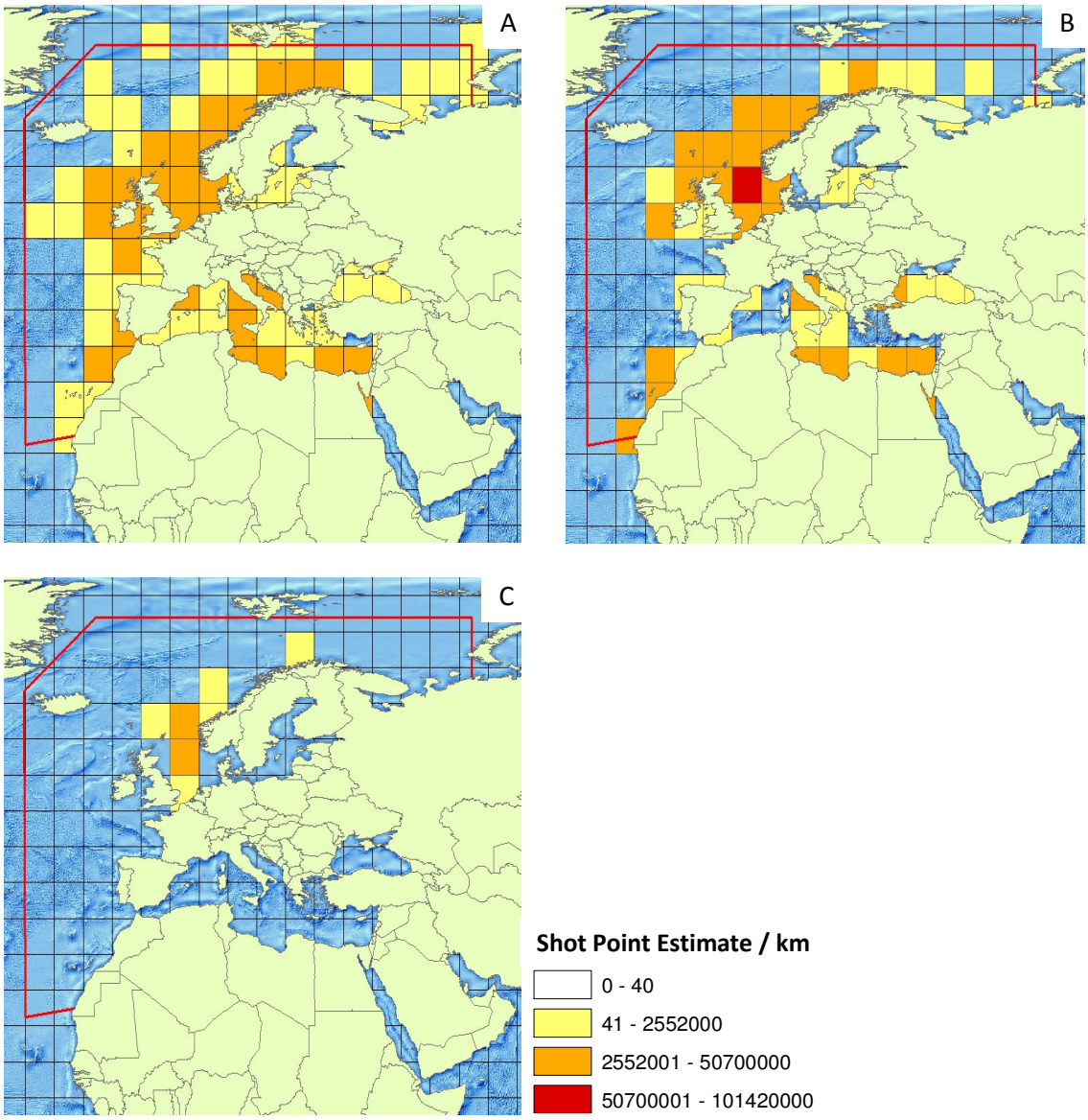


Figure 3.15: Density surface of shot points fired during 2D (map A), 3D (map B) and 4D (map C) seismic surveys in FAO 27. The red line marks the extent of AOR_006A.

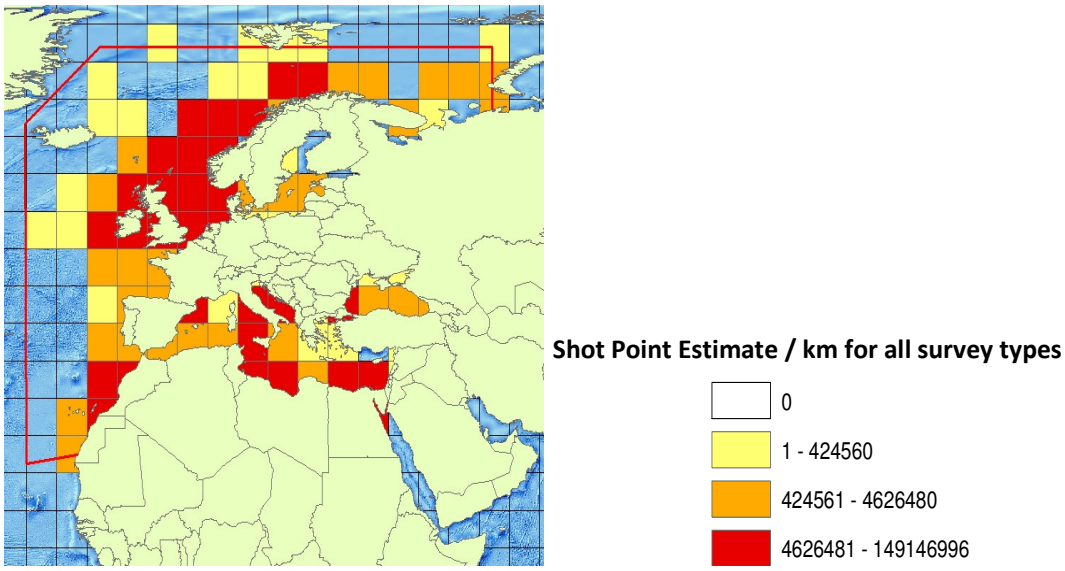


Figure 3.16: Density surface of shot points fired during all seismic survey types combined in FAO 27. The red line marks the extent of AOR_006A.

3.3 Temporal trends in E&P sound

On a global scale, the number of offshore seismic surveys detailed in the IHS database peaked in the 1980s (n=4788). The number of records of seismic surveys in the database is slightly lower for the 1990s (n=4140) and similar to the number of records for the 1970s (n=3819). The proportions of surveys of each type being conducted has varied between decades (Figure 3.17) with an increasing proportion of 3D and 4D surveys being conducted since the 1980s. Across all areas and decades, the majority of seismic surveys conducted have been 2D seismic surveys.

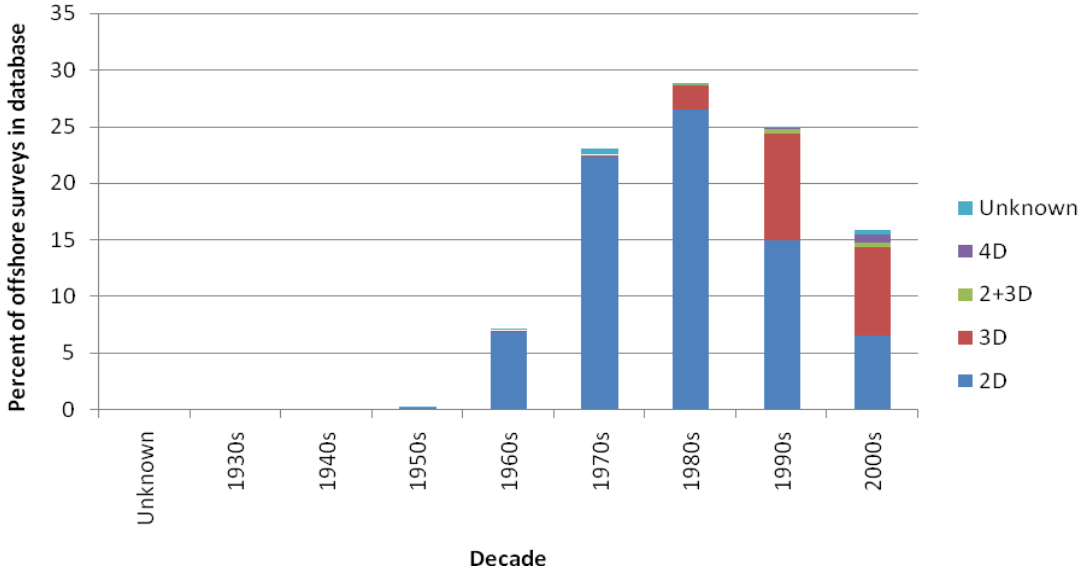


Figure 3.17: Temporal distribution of seismic survey records, with proportion of surveys made up of each seismic survey type shown.

3.3.1 East coast USA: AOR_005B/ FAO 21+31

A different trend can be seen in surveys conducted in waters offshore of the east coast of Canada, the US and the Gulf of Mexico where the number of both 2D (Figure 3.18) and 3D (Figure 3.19) surveys conducted has increased since the mid-1980s. (Note that the scales on both the primary and the secondary y axis differ between graphs). The increase in the number of all types of seismic survey conducted since the mid 1980s corresponds with an increase in the number of shot points fired over the same period (Figure 3.20).

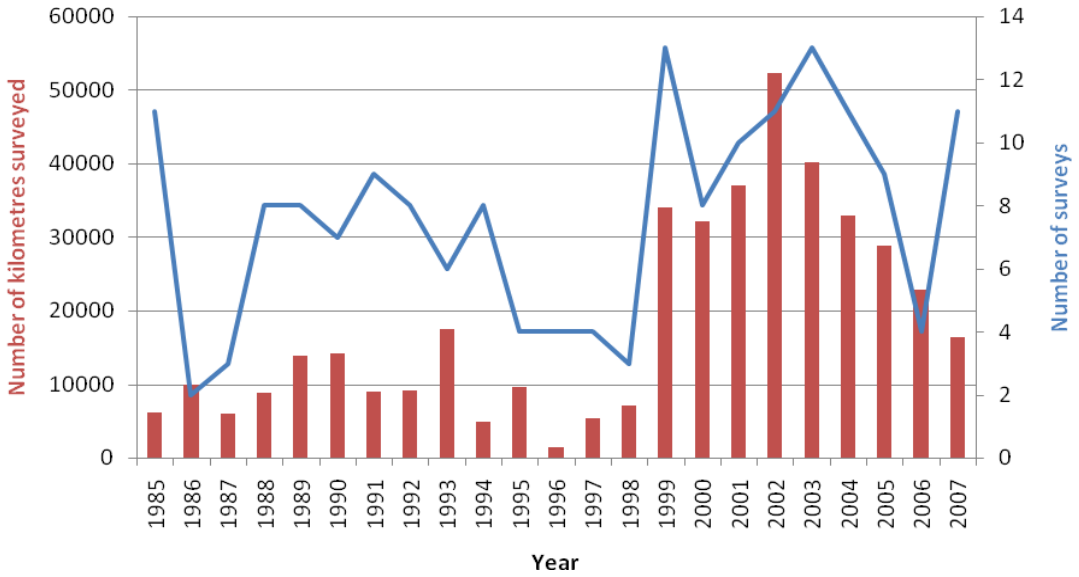


Figure 3.18: Number of 2D offshore seismic surveys conducted and kilometres surveyed in FAO 21+31 since 1985.

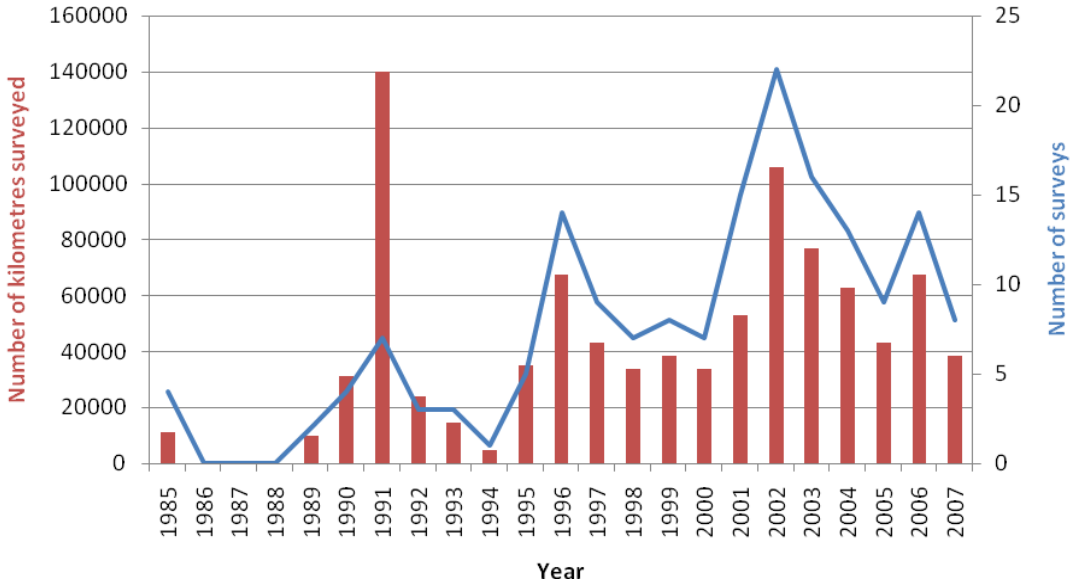


Figure 3.19: Number of 3D offshore seismic surveys conducted and kilometres surveyed in FAO 21+31 since 1985.

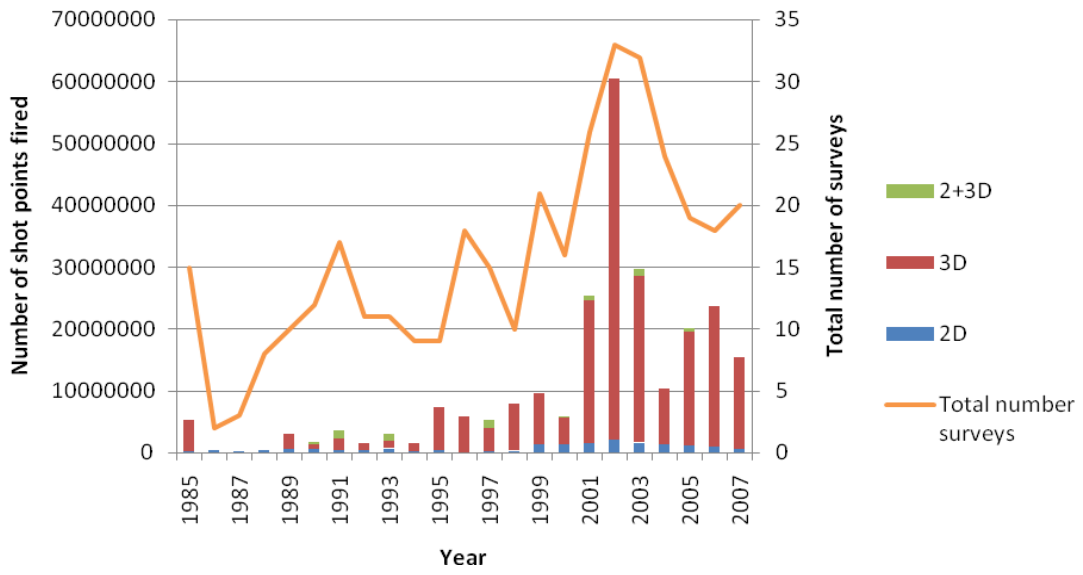


Figure 3.20: Number of shot points fired and surveys conducted in FAO 21+31 since 1985.

3.3.2 Europe: AOR_006A / FAO 27

The global trend of an overall decline in the number of seismic surveys conducted since the 1980s is apparent in FAO 27 (Figures 3.21 and 3.22), as is an increase in the relative number of 4D surveys since the late 1990s (Figure 3.23). However, the number of 3D surveys conducted and the number of kilometres surveyed was highest in the mid 1990s. Despite a decrease in the number of seismic surveys conducted over the time period of interest (1985 to 2007), the increase in the number of 3D and 4D surveys conducted in the later part of this period resulted in peaks in the number of shot points fired in the mid 1990s and mid 2000s (Figure 3.24).

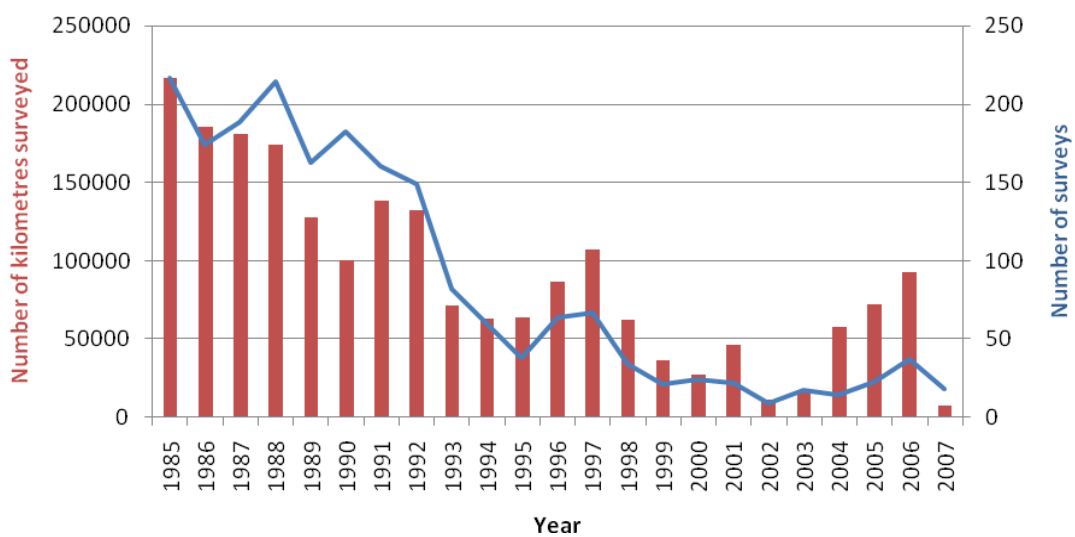


Figure 3.21: Number of 2D offshore seismic surveys conducted and kilometres surveyed in FAO 27 since 1985.

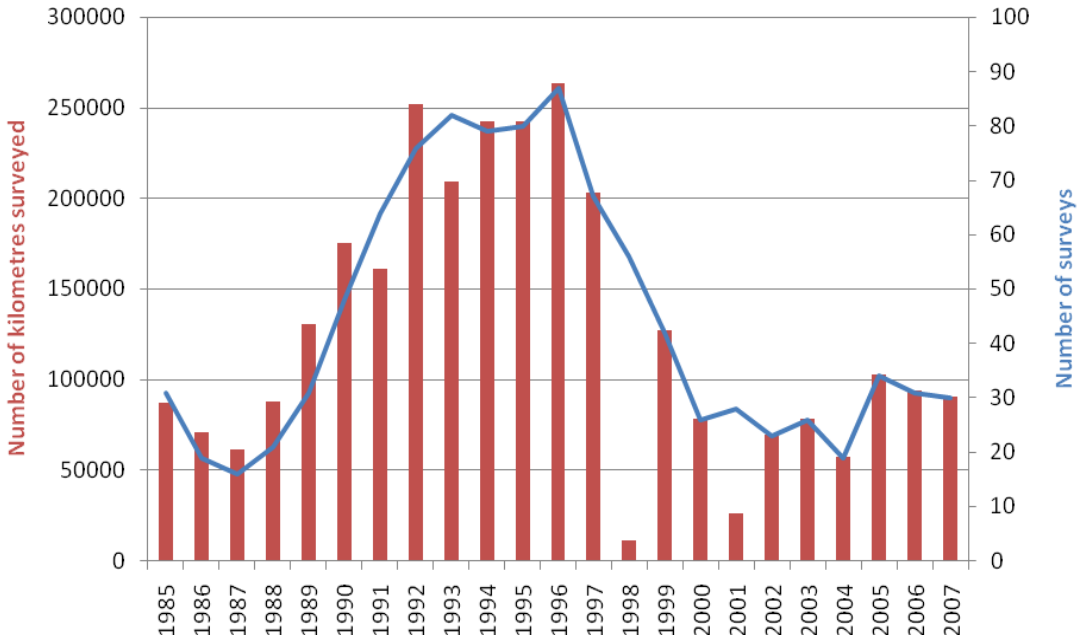


Figure 3.22: Number of 3D offshore seismic surveys conducted and kilometres surveyed in FAO 27 since 1985.

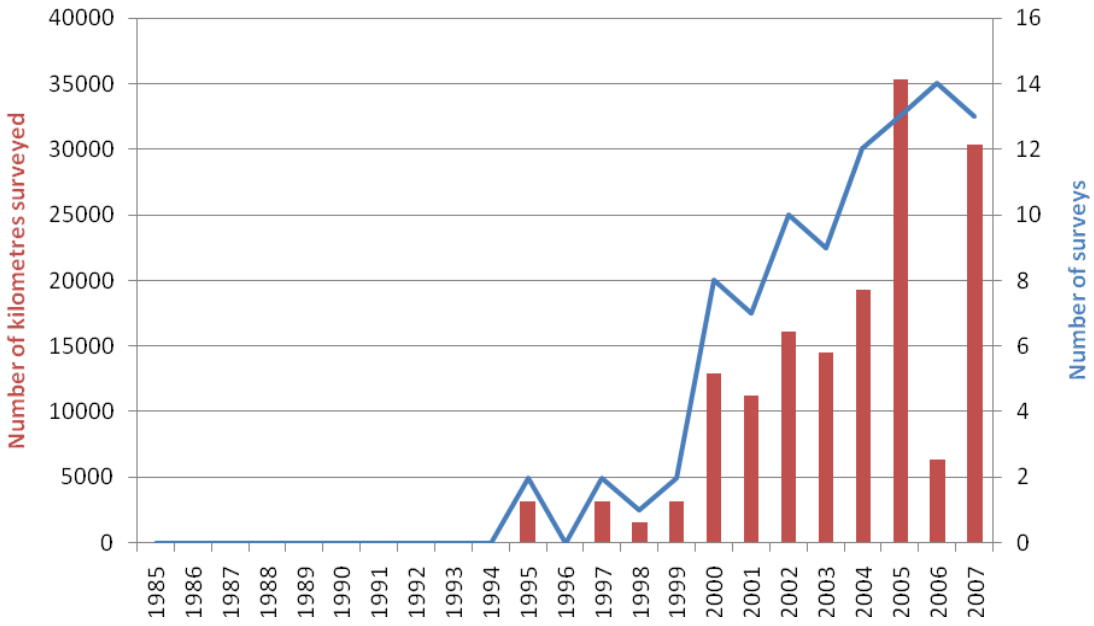


Figure 3.23: Number of 4D offshore seismic surveys conducted and kilometres surveyed in FAO 27 since 1985.

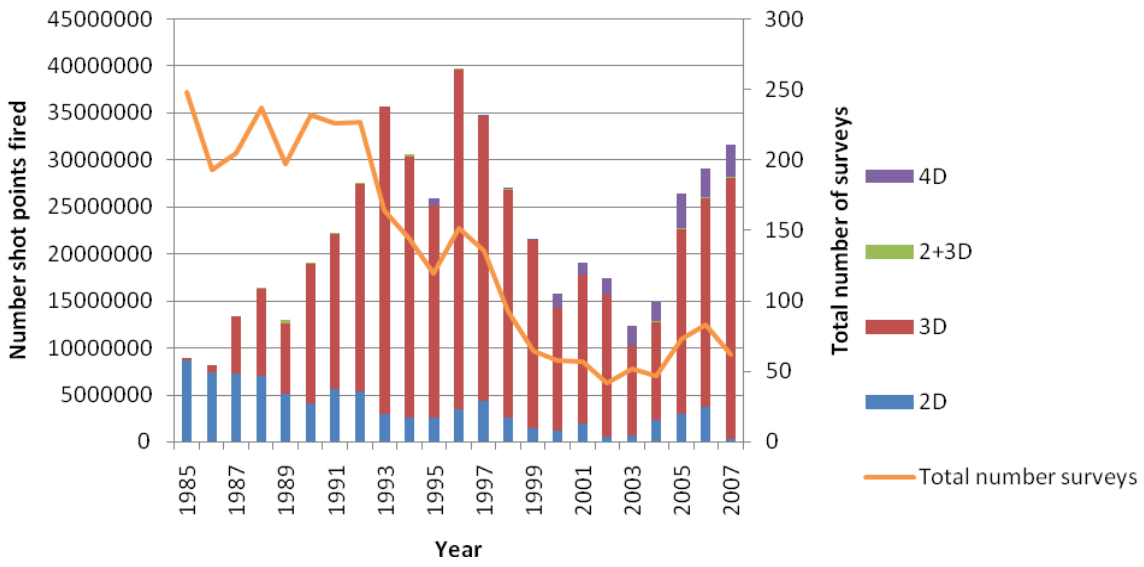


Figure 3.24: Number of shot points fired and surveys conducted in FAO 27 since 1985.

4. Statistical analysis of E&P sound data

4.1 Analysis steps

The overall goal of the analysis was to determine if there were any relationships between annual changes in density for particular species in defined geographic areas, and covariates related to environmental conditions and oil and gas E&P sound. Before this analysis could be performed, two initial steps were required to yield suitable data. Firstly, as shown in Task 2, published density estimates from surveys can be affected by a number of factors not related to true density in an area, such as the survey method used, survey agency, season, etc. Hence the first step was to correct the annual density estimates obtained from published sources for these factors, using Generalized Additive Modelling methods similar to those in Task 2 to predict the log density of that species for each year. Secondly, even for the most data-rich species, there were many years for which no survey estimates were available, making it impossible to directly calculate between-year population change. Hence the second step was to use interpolation to estimate the population densities in the missing years, producing a continuous time series of density estimates. After these two preparatory steps, annual population change could be calculated, and then a regression analysis of population change against environmental, anthropogenic mortality and oil and gas E&P-related covariates could be performed.

Effort was focused on the four species from Task 2 for which the most continuous and extensive density data were available: minke whales in FAO 27, and fin, humpback and sperm whales in FAOs 21 and 31. FAO 27 encompasses the NE Atlantic and includes the seas around Europe, with the exception of the Mediterranean which falls outside FAO 27 (Figure 1.1, Appendix). FAO 21 includes the waters off the east coast of Canada and the northern part of the USA and the waters to the west of Greenland, while FAO 31 includes the waters

off the east coast of the southern USA and the Gulf of Mexico. Data from FAOs 21 and 31 were modelled together as one area.

In total, five analysis steps were undertaken: Covariate data exploration; Generalised Additive Modelling of survey estimates; Predicting cetacean density; Interpolation of density estimates over time; Linear regression of population change. We treat each step in turn below.

4.1.1 Covariate data exploration

4.1.1.1 Environmental covariates

The spatial scale at which the density data were modelled precluded the use of fine resolution environmental covariates. Therefore only ocean-wide covariates applicable to the North Atlantic were considered. The two environmental covariates tried in the linear regression models of population change were annual measures of the North Atlantic Oscillation (NAO) index and Sea Surface Temperature (SST) data for the North Atlantic.

The NAO (http://www.cpc.noaa.gov/products/precip/CWlink/ENSO/verf/new.nao.shtml) is a measure of the deviation from a long term average of the sea level pressure between the low pressure system of Iceland and the high pressure system of the Azores. The index is associated with particular climatic conditions across the North Atlantic, including wind speed, temperature and precipitation. Mean monthly NAO index data were available, and these values were averaged to form an annual value.

Average monthly sea surface temperature data were available for the North Atlantic (from 5-20° North and 60-30° West) for each month from 1950 (http://www.cpc.noaa.gov/data/indices/sstoi.atl.indices). The geographical area the SST data originates from covers parts of the North Atlantic not included in the AORs, but it was assumed the data were representative of SST trends within the AORs. The mean SST value for each year since 1980 was calculated for use as a covariate.

4.1.1.2 External factors influencing cetacean populations

Information on environmental and anthropogenic factors that may influence or control cetacean population growth rates was presented in the Task 3 report (Murphy, 2008). In summary, whaling was the factor that most influenced cetacean population levels and growth rates for species of large whales. For smaller cetaceans, anthropogenic removal of individuals through by-catch and directed takes were the most influential factors. Information on the numbers of whales removed by whaling (both commercial and aboriginal), and the level of mortality through by-catch was sought for the 4 species of interest. The only source of anthropogenic mortality considered large enough to try in the models was the catch of minke whales in Norwegian and Icelandic waters (FAO 27) made under objection to the zero catch limit set by the International Whaling Commission in 1985/6 and under Special Permit (Table 4.1, Appendix). Of the other species, there are also aboriginal subsistence catches of fin whales off the west coast of Greenland (FAO 21) and humpback whales around St. Vincent and the Grenadines (FAO 31) but numbers of individuals caught are very low.

4.1.1.3 E & P sound data

Three covariates were extracted from the seismic database to investigate the relationships between trends in cetacean populations and E&P sound: the number of surveys conducted, the number of kilometres surveyed and the number of shot points fired. Annual values were calculated for each covariate in each area (i.e. FAO 27 and FAOs 21 + 31 combined).

The number of surveys conducted is the total number across all seismic survey types, so does not reflect the differences in sound production levels between the different seismic survey types. Nor does it reflect variation in the spatial scale of the seismic surveys. The covariate was obtained by counting the number of seismic survey records contained within the database for each year within each of the areas of interest.

The number of kilometres surveyed was calculated by finding the mean number of kilometres surveyed during each seismic survey type for each year of interest. Not all records contained the number of kilometres surveyed, so often the mean number of kilometres surveyed was calculated using considerably fewer than the total number of surveys. As a result this covariate may not accurately represent trends in the number of kilometres surveyed using each seismic survey type and should be treated with caution. The number of surveys used to calculate the mean annual number of kilometres surveyed using each seismic type is given in tables 4.4 and 4.5 (Appendix). In the instance that none of the surveys of a particular type in a given year had the number of kilometres reported, the mean number of kilometres surveyed during that type of seismic survey across all years was used to calculate the number of kilometres surveyed that year.

The third and final seismic covariate used in the models was the total number of shot points fired per year. The total number of shot points fired each year was calculated for 2D and 2+3D seismic surveys by multiplying the number of kilometres surveyed each year by 40, while for 3D and 4D surveys the total area surveyed each year was multiplied by 800 to obtain the total number of shots fired in the area of interest (ASCOBANS, 2006). The totals obtained for each seismic survey type were then summed for each year. This covariate should better reflect the variation in sound produced during the different survey types. As with the number of kilometres surveyed, there was often information missing and the mean number of kilometres surveyed or area surveyed calculated for each year may not be accurate; this is a limitation of the seismic survey data. Refer to tables 4.6 and 4.7 (Appendix) for further information.

4.1.2 Generalised Additive Modelling

The density of each of the species of interest was modelled using the statistical software R (R Development Core Team, 2008) using Generalised Additive Models and a methodology similar to that used during the Task 2 analysis (Quick *et al.*, 2008). Cetacean density was assumed to have a Gamma distribution and a log link function was used. The response variable, cetacean density was weighted according to the precision of the density estimate and the size of the area surveyed (1/CV and log of area). A forward selection scheme was used, in which the first model fit was a Year only model; then all other covariates (Table 4.8) were added in turn and the minimum AIC model selected. Each model included year as a factor variable to allow the model to be used to predict cetacean density by year. Continuous covariates were fitted as smooth functions, using thin-plate regression splines with the degree of smoothness determined using generalized cross-validation (Wood, 2006;

2008). In some cases the number of knots (related to maximum "wiggliness" of the function) had to be restricted to allow model convergence. Interaction terms were tried between variables, but models containing interaction terms could not be used to predict density because of missing values in some combinations of the covariates. Model selection was based on Akaike's Information Criterion (AIC), with some consideration also given to the generalised cross-validation (GCV) score (asymptotically both give the same model ordering, so any differences will be due to small sample size effects).

Table 4.8: Potential covariates tried in the Generalised Additive Models. Abbreviations are those used in the results tables.

Covariate	Abbreviation	Type
Year	Year	Factor, 11 levels
Mean latitude of survey	Lat	Continuous
Minimum latitude	MinLat	Continuous
Maximum latitude	MaxLat	Continuous
Survey methodology	Method	Factor, 5 levels
Survey platform	MethodPlat	Factor, 3 levels
G(0) corrected	MethodG0Corr	Factor, 2 levels
Survey agency	Agency	Factor, 5 levels
Season	Season	Factor, 3 levels

The number of levels reported for the factor covariates is the maximum number possible. The number of factor levels for the covariate Year depended on the number of density estimates so varied by species. Minke whales in FAO 27 had density estimates from the most years, with 11 years of data. The different factor levels of the covariate Method are given in Table 4.9. There were three levels to the factor covariate MethodPlat: aerial, ship or both. The density estimates used in the modelling are either corrected for animals missed on the track line or not, so the factor covariate MethodG0Corr has two levels. The maximum number of survey agencies that estimated density of the four species of interest was 5 but was usually less; for example, minke whale density in FAO 27 was estimated by 4 survey agencies, so the factor covariate Agency had 4 levels for minke whales. Season had three levels, summer surveys, non-summer surveys and year-round surveys.

Table 4.9: Different levels of the factor covariate Method.

Level code	Level description
B	Aerial surveys not corrected for g(0)
C	Aerial surveys corrected for g(0)
D	Shipboard surveys not corrected for g(0)
E	Shipboard surveys corrected for g(0)
F	Combined shipboard and aerial surveys

4.1.3 Predicting cetacean density

The model with the lowest AIC score was used to predict cetacean density, provided it did not contain interaction terms. If the best model contained covariates besides Year, a mean value of the covariate (if it was continuous) was used to generate predictions for each of the years with density estimates. If the best model contained a factor variable (besides Year) an arbitrary level of the factor variable was used when using the model to generate predictions. The level of the factor variable could be arbitrarily chosen as the predicted densities were only used to calculate the rate of population change, which would not be influenced by the choice of factor level. The predictions were checked to assess whether they were plausible. In the event that the model generated obviously unrealistic predictions of density, either the model was adjusted in an attempt to create more realistic predictions or the predictions generated by the next best model were examined. The variance of the density predictions was calculated.

4.1.4 Linear interpolation of density estimates

Linear interpolation was used to generate estimates of density for those years in which there had been no survey effort and for which there were no density estimates. In the years between known density estimates the population was assumed to have either increased or declined exponentially (i.e. population change was assumed to be loglinear). For example, if density was known for years t and $t+k$, then the log of the population change between pairs of years (e.g. t to $t+1$, $t+1$ to $t+2$, ..., $t+k-1$ to $t+k$) could be calculated using equation 1 (where D represents density) and density values for missing years could be derived.

$$\exp\left(\frac{log(D_{t+k}) - log(D_t)}{k}\right) \quad (1)$$

The log-linear interpolation resulted in a continuous time series of data within the range of the first and last surveys which allowed the between-year population change to be calculated (on the log scale) using equation 2.

$$\exp\left(\log(N_t + 1) - \log(N_t)\right) \quad (2)$$

4.1.5 Linear regression of population change

A weighted stepwise linear regression analysis was then conducted to investigate the relationships between the log of the rate of population change and the new explanatory covariates (Table 4.10). The values of log population change between one year and the next were associated with annual values of the explanatory covariates from the earlier year. The response variable, log of population change, was weighted according to the precision of the estimate of population change (i.e. 1/variance of the population change) then weights were re-scaled so that the mean weight was 1 to allow the AIC to be used for models selection. Least-squares regression was used, under the assumption of normally distributed residuals, and no temporal autocorrelation in residuals.

A stepwise model selection process was used based on the AIC score. The choice of the best model was often double checked by doing an analysis of variance (ANOVA). The R^2 value (the coefficient of determination) of the final model was used to assess the fit of the model to the data. The model output of the final model, and predictions made using the model,

were used to investigate the relationships detected between the response variable and explanatory covariates. On occasion, a Generalised Additive Model containing the same covariates as the final linear model, (and assuming a normally distributed response), was applied to the data to check for nonlinear relationships.

Table 4.10: Explanatory covariates tried in the linear regression analysis.

Covariate	Abbreviation	Covariate category	Type
Sea Surface Temperature	SST	Environmental	Continuous
North Atlantic Oscillation Index	NAO	Environmental	Continuous
Log of number of whales caught*	logWhaling	Anthropogenic	Continuous
Number seismic surveys conducted	nSurvey	E&P sound	Continuous
Number of kilometres surveyed	kmSurvey	E&P sound	Continuous
Number of shot points fired	nShots	E&P sound	Continuous

* This covariate was only used in the linear regression of minke whale population change as that was the only species for which a relatively large number of individuals were removed by whaling. The log of the number of individuals caught each year was used to put the explanatory variable onto the same scale as the response variable, log of population change.

4.2 Results

4.2.1 Data exploration of new covariates

4.2.1.1 Environmental covariates

The mean annual North Atlantic Oscillation index between 1980 and 2006 is shown in Figure 4.1. Mean sea surface temperature (Figure 4.2) and the mean annual North Atlantic Oscillation index were used as proxy measures of primary productivity and environmental variability.

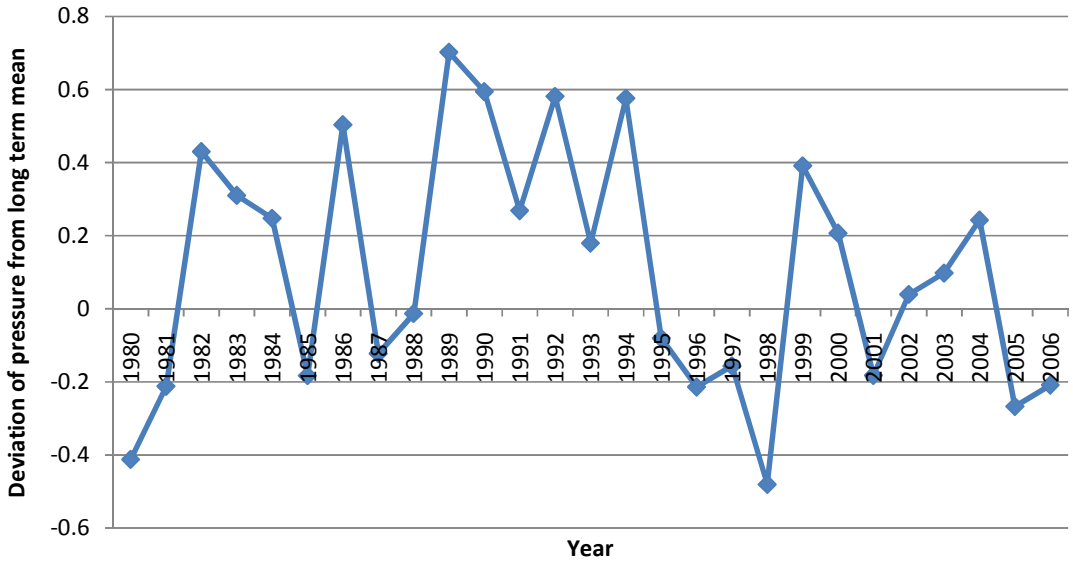


Figure 4.1: Mean annual North Atlantic Oscillation index.

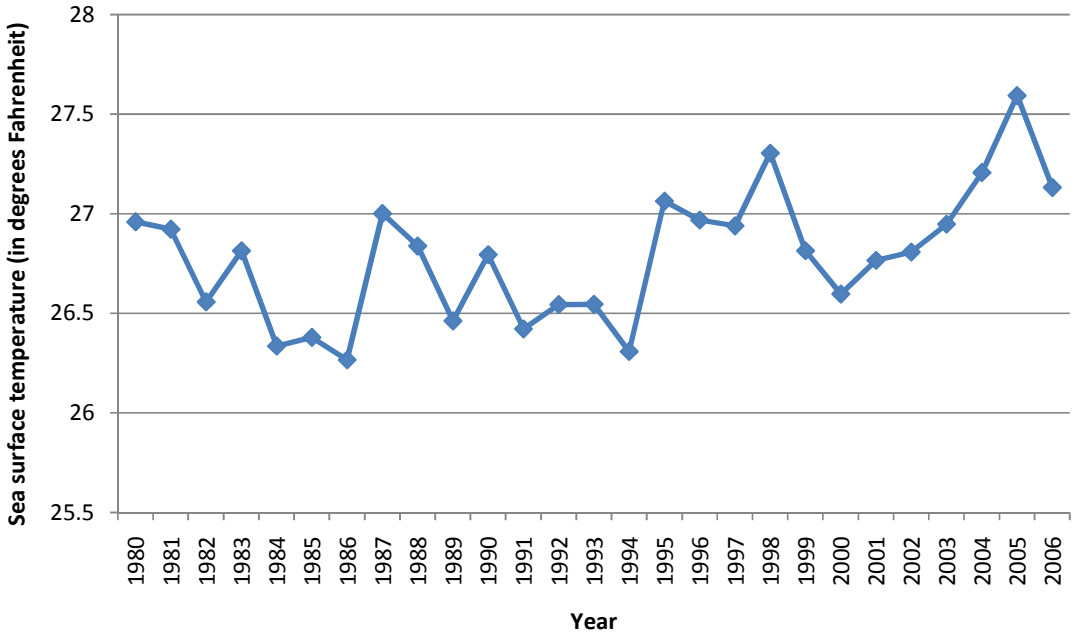


Figure 4.2: Mean annual sea surface temperature in the North Atlantic.

4.2.1.2 External factors influencing cetacean populations

The number of minke whales caught by the commercial fishery in Norwegian and Icelandic waters (Figure 4.3) was used as an explanatory covariate in the model of minke whale population change.

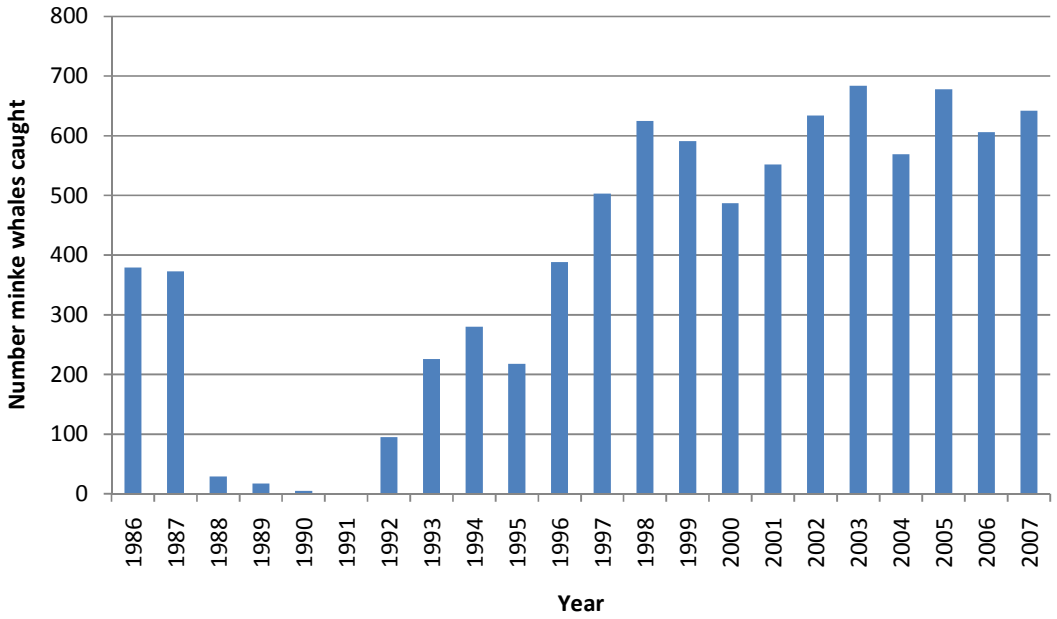


Figure 4.3: Number of minke whales caught by commercial whale fishery in Norwegian waters (FAO 27).

4.2.1.3 E & P sound data

The seismic survey data were examined in detail in section 3 of this report. In this section we specifically investigate the distribution of the three seismic survey covariates, the number of surveys conducted, the number of kilometres surveyed and the number of shot points fired, in each of the two areas of interest.

Of the 5720 records of surveys that occurred within FAO 27, 54.5% (n=3117) fell within the period of interest (1985 to 2007). The number of surveys, kilometres surveyed and number of shot points fired are illustrated in Figures 4.4 to 4.6.

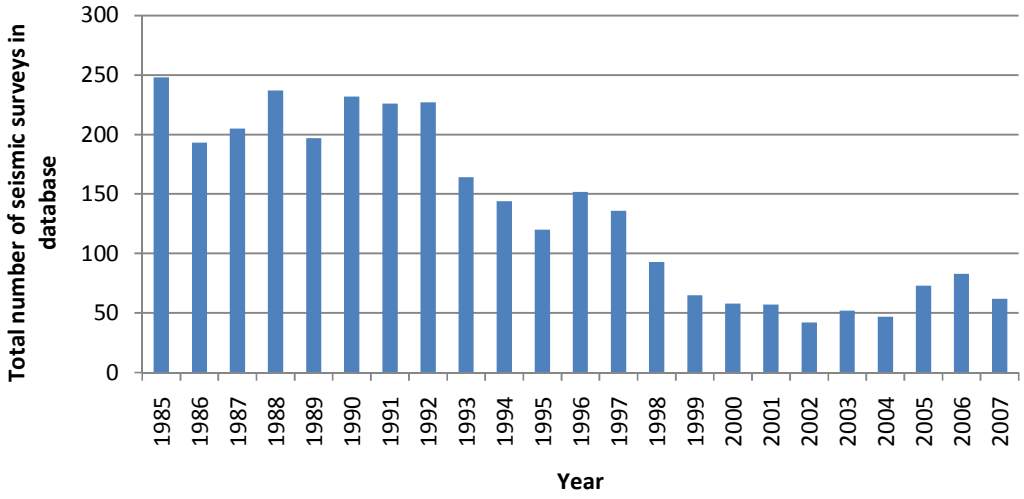


Figure 4.4: Total number of seismic surveys recorded in the IHS database in FAO 27 between 1985 and 2007.

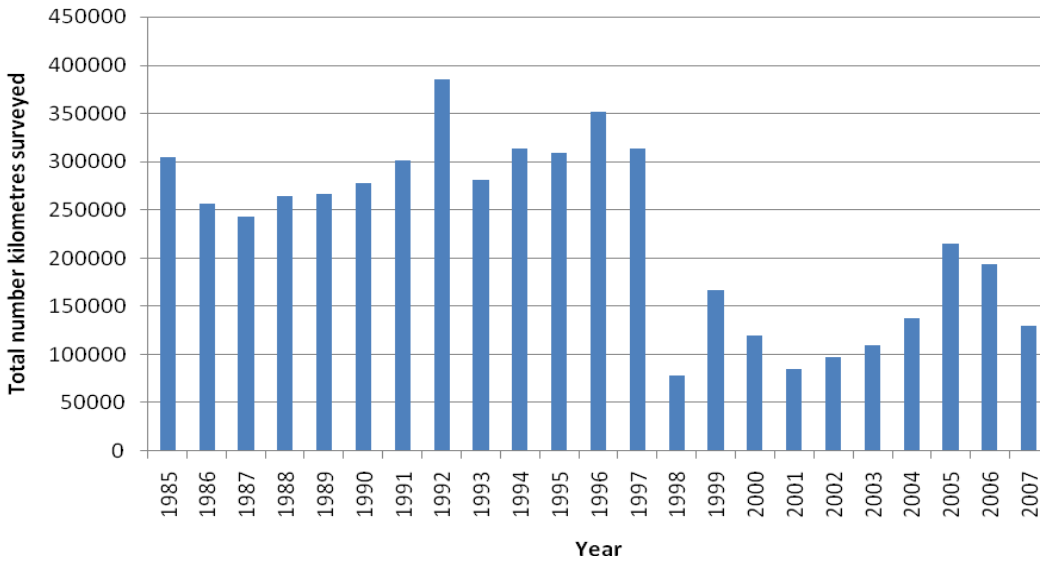


Figure 4.5: Total number of kilometres surveyed in FAO 27 between 1985 and 2007.

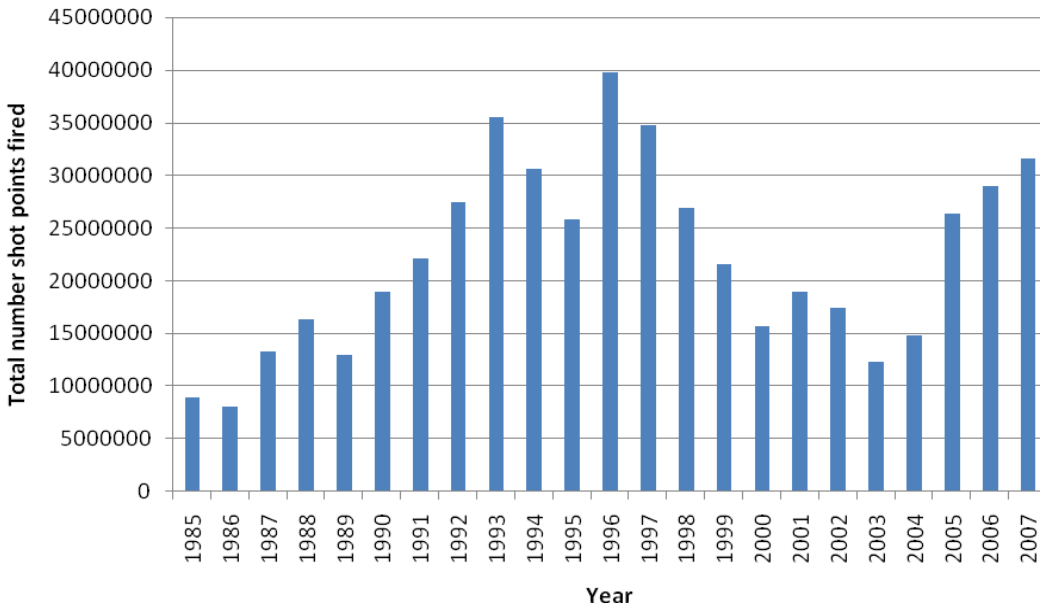


Figure 4.6: Total number of shot points fired during seismic surveys in FAO 27 between 1985 and 2007.

Only 1.8% (n=784) of all the seismic surveys in the database fell within FAOs 21+31, and 45.8% of those surveys (n=359) began between 1985 and 2007. The number of surveys, kilometres surveyed and number of shot points fired are illustrated in Figures 4.7 to 4.9.

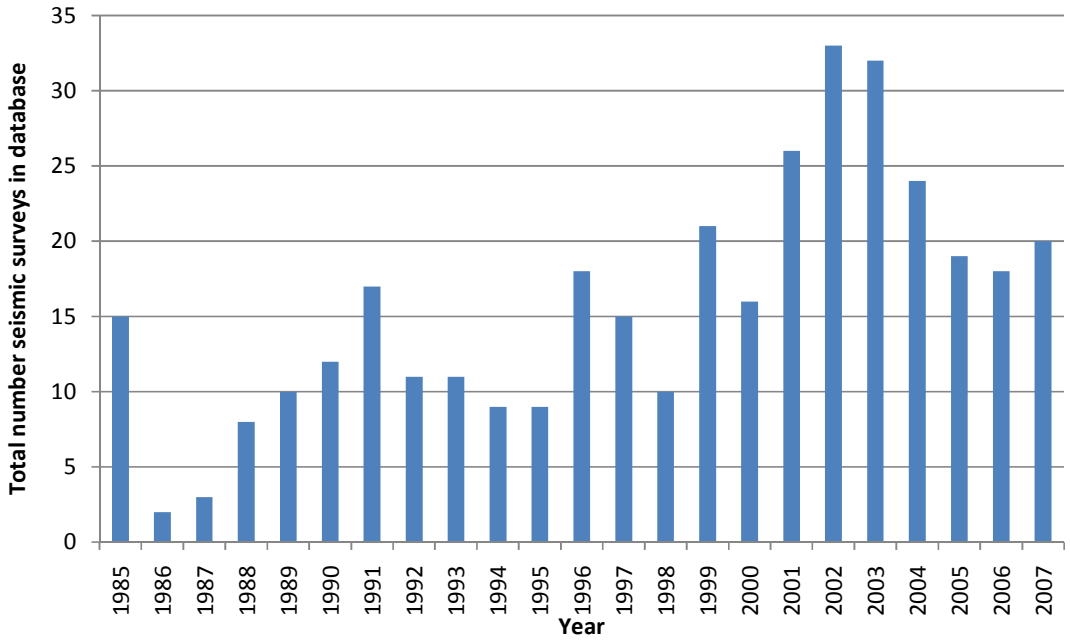


Figure 4.7: Total number of seismic surveys, recorded in the IHS database, conducted in FAOs 21+31 between 1985 and 2007.

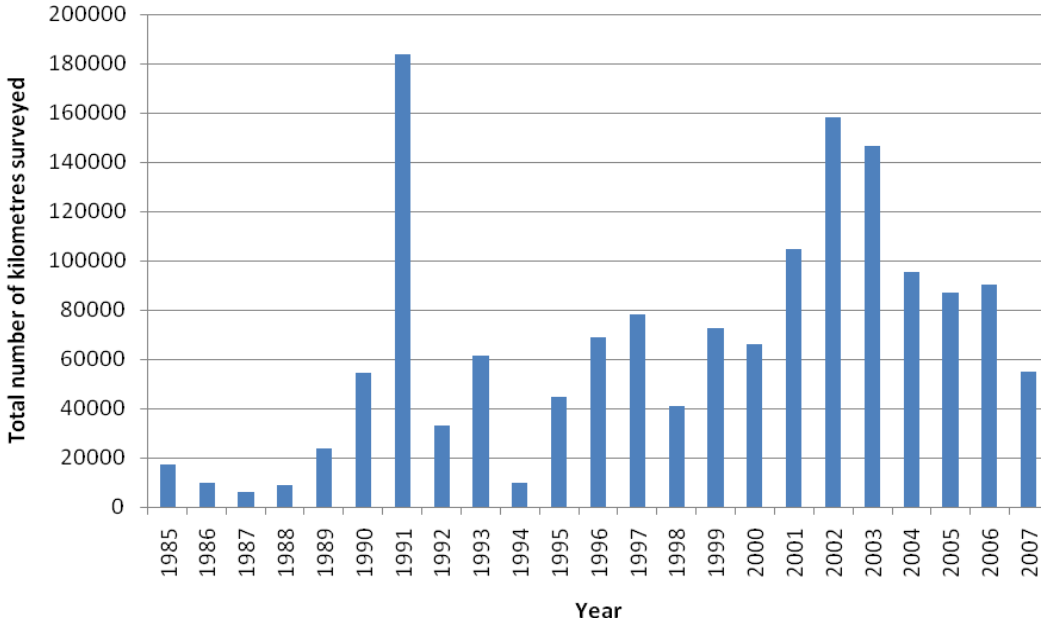


Figure 4.8: Total number of kilometres surveyed in FAOs 21+31 between 1985 and 2007.

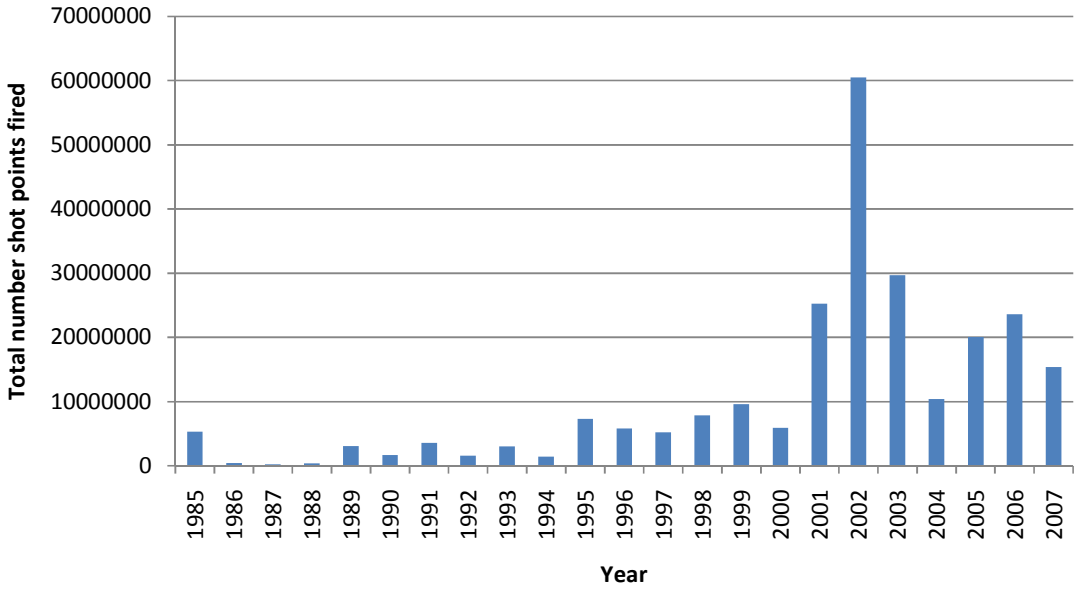


Figure 4.9: Total number of shot points fired during seismic surveys in FAOs 21+31 between 1985 and 2007.

4.2.2 Minke whales FAO 27

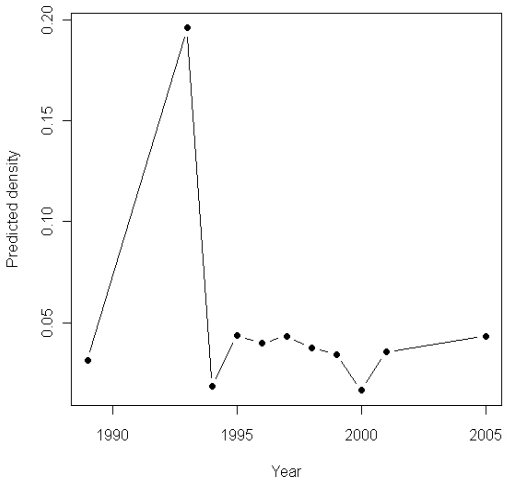
Estimates of minke whale density were available for 11 years between 1989 and 2005. Nine of the density estimates came from consecutive years. Generalised additive modelling of the minke whale density data from surveys in FAO 27 found that a model containing year as a factor (which was compulsory for generating predictions of density over years) and a smooth of the minimum latitude of the survey area best explained the patterns in the data (Table 4.11). None of the density estimates had been corrected for g(0) and all surveys were in summer, so the effect of these could not be investigated. The agency conducting the survey and the survey platform type (shipboard or aerial) did not explain as much deviance in the data as the spatial covariates (measures of latitude).

Table 4.11: The best three generalised additive models of minke whale density in FAO 27.

Model	Covariates retained	AIC	GCV	% deviance explained
1	Year + s(MinLat)	-309.1	0.518	28.9
2	Year	-307.3	0.527	24.9
3	Year + s(Lat)	-307.1	0.534	27.4

The best model (model 1) was then used to predict minke whale density for those years in which there was effort (i.e. the years the data originate from) using the mean value of

minimum latitude (Figure 4.10), and the data were interpolated to produce density estimates for the years with missing data (Figure 4.11).



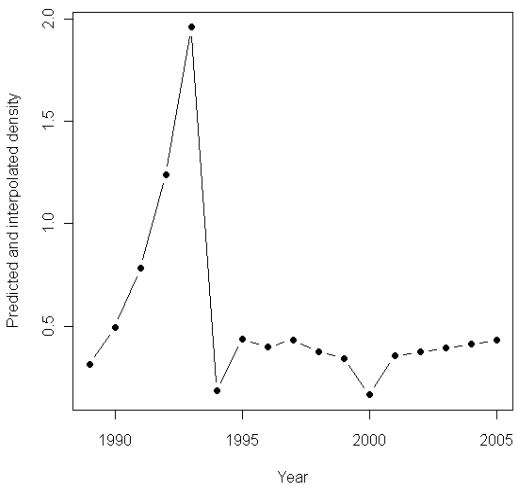


Figure 4.10 (on the left): Predicted minke whale density for years with survey effort.

Figure 4.11 (on the right): Predictions of minke whale density including interpolated values for years without survey effort.

Many of the predicted densities had a high level of variance associated with them, resulting in the confidence intervals of the differences between densities in pairs of years being large (Figure 4.12).

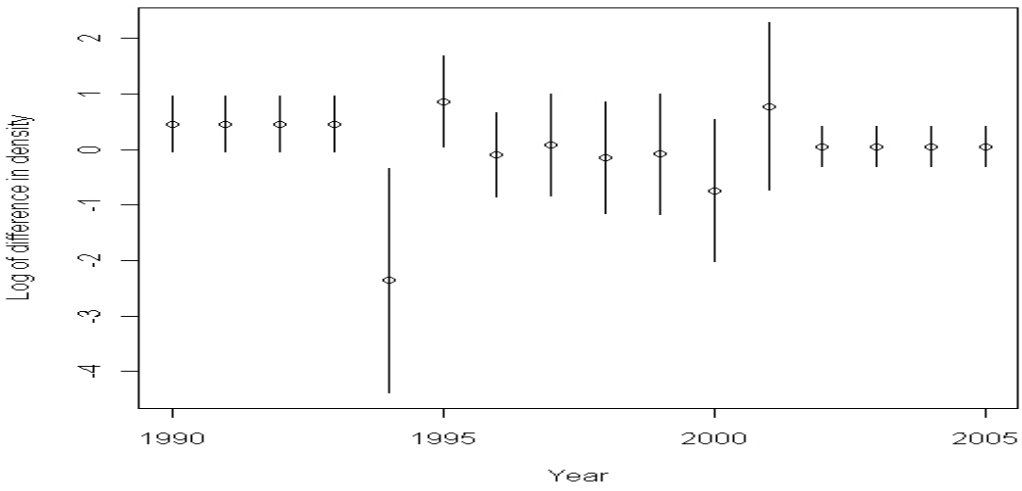


Figure 4.12: The difference in predicted log density between pairs of years, on the log scale, with 95% confidence intervals.

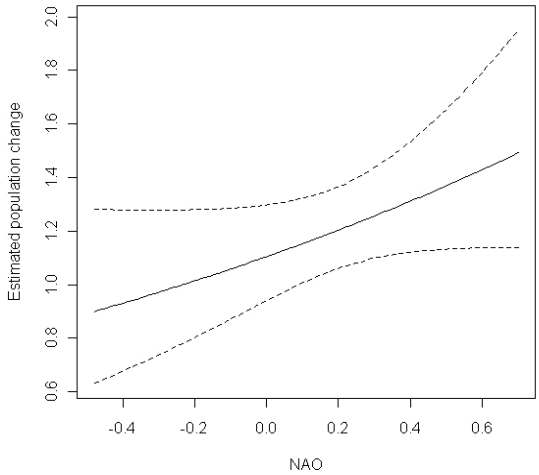
Plotting the difference in predicted density between years against each of the new covariates did not reveal any obvious trends but each of the covariates was included in a

precision-weighted linear regression. The response variable was the difference in log minke whale density between pairs of years, and a stepwise selection procedure based on the AIC value selected a final model containing the covariates sea surface temperature (SST) and North Atlantic Oscillation index (NAO) (Table 4.12).

Table 4.12: The best three linear models of population change of minke whales in FAO 27.

Model	Covariates retained	AIC
1	NAO + SST	-41.5
2	NAO	-40.8
3	nSurvey	-40.7

The R-squared value of the final model was 0.44, suggesting that 44% of the variation in the weighted data was explained by the model (Dobson, 1983). The p-value of the NAO covariate (0.098) suggested there was weak evidence that the null hypothesis that the difference in minke whale density between years does not vary with the NAO index could be rejected. The parameter estimate for NAO was positive, suggesting that the rate of population change of minke whales in FAO 27 is positively influenced by NAO (Figure 4.13). A p-value of 0.146 for the SST covariate provided no evidence against the null hypothesis at the 5% probability level and an analysis of variance showed that its inclusion only resulted in a minor improvement in the fit of the model. Removing SST from the final model increased the significance of NAO in the model, but did not alter the relationship between NAO and the difference in minke whale density between years. The parameter estimate for SST was negative suggesting that the rate of population change is perhaps weakly negatively influenced by SST (Figure 4.14).



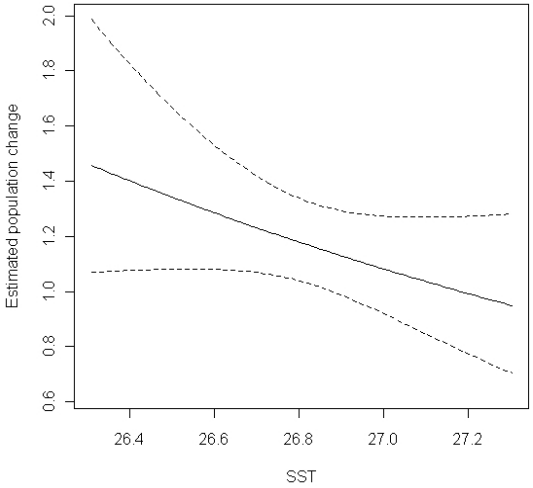


Figure 4.13 (on the left): Estimated population change for minke whales in FAO 27 over a range of NAO values (with constant SST). Dotted lines show the 95% confidence intervals of the predictions.

Figure 4.14 (on the right): Estimated population change for minke whales in FAO 27 over a range of SST values (with constant NAO).

Predictions (on the scale of the log of the difference in density between pairs of years) of the best linear model are shown in red alongside the density data (predicted by the best generalised model for years with data and linearly interpolated for years without) in black in Figure 4.15. The bars show 95% confidence intervals.

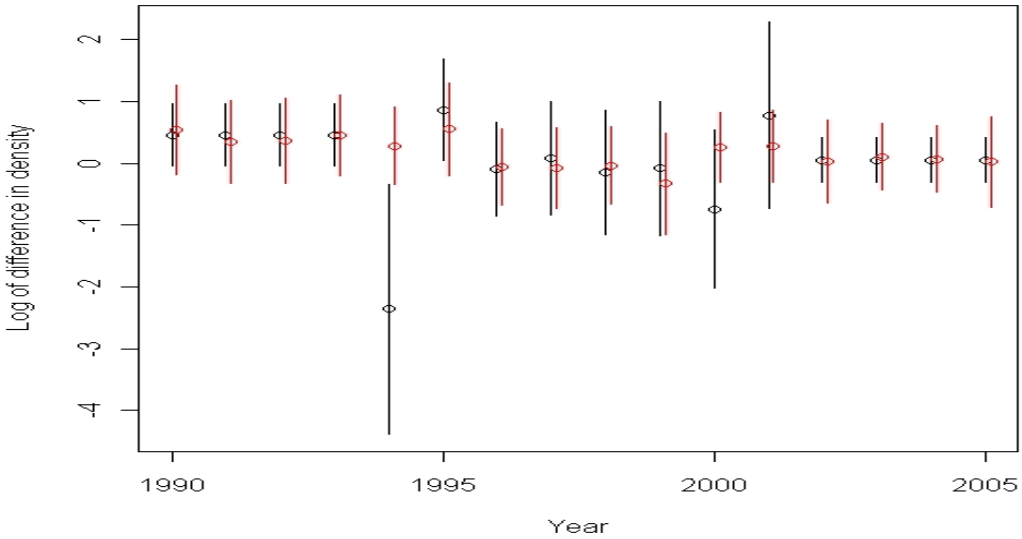


Figure 4.15: Predictions, from the best linear model, of population change (in red) alongside the log density data (predicted and interpolated) in black for minke whales in FAO 27.

A generalised additive model containing the same covariates as those retained in the final linear model was fitted to the population change data. The model output showed that, despite the greater flexibility of GAMs, a linear relationship between the covariates and the response variable had been estimated. This implies that the use of a linear model is appropriate and that relationships in the data were adequately captured during the linear regression.

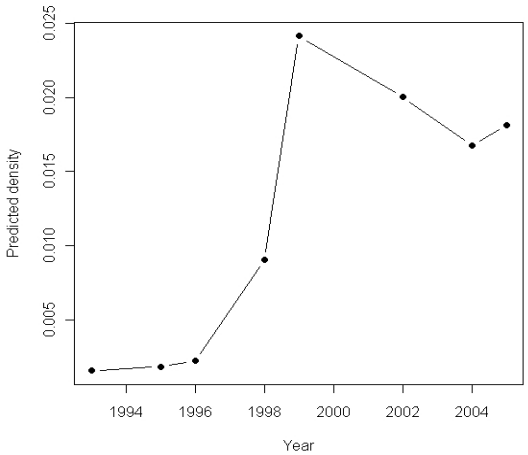
4.2.3 Fin whales FAO 21 +31

Fin whale density estimates were available for 8 years between 1993 and 2005. Data were only available for 2 pairs of consecutive years. The best generalised additive model of fin whale density from surveys in FAO 21 + 31 (Table 4.13) contained year as a factor (as was compulsory) and a smooth of the minimum latitude of the survey area, the same as was found for minke whales in FAO 27. Four different survey methods were represented in the fin whale data, including methods that corrected for animals missed on the trackline, and the third best model contained an interaction term between year and whether density estimates had been corrected for g(0) or not. The survey platform used during the survey (aerial or shipboard) did not perform well as a covariate in the models, nor did season or the agency that conducted the survey.

Table 4.13: The best three generalised additive models of fin whale density in FAO 21 + 31. ":" between two covariates represents an interaction term in the model.

Model	Covariates retained	AIC	GCV	% deviance explained
1	Year + s(MinLat)	-282.0	0.765	72.8
2	Year + s(Lat)	-280.1	0.817	72.4
3	Year:MethodG0Corrected	-279.2	0.811	69.8

Model 1 was used to predict fin whale density for those years in which there was effort using the mean value of minimum latitude (Figure 4.14). Between 1996 and 1999 an implausible 5-fold population increase was predicted, suggesting these results and the subsequent analysis based on them should be treated as exploratory. The data were interpolated to produce density estimates for the years with missing data (Figure 4.15).



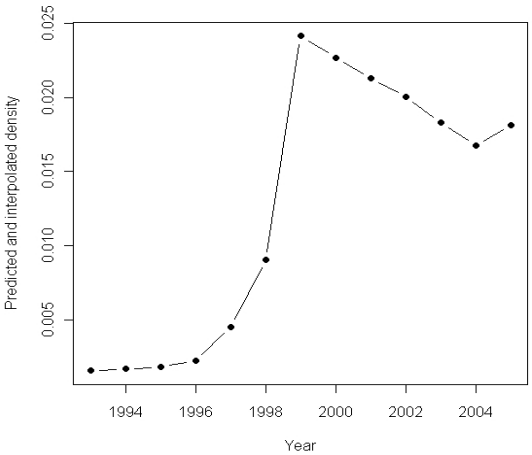


Figure 4.14 (on the left): Predicted fin whale density for years with survey effort.

Figure 4.15 (on the right): Predictions of fin whale density including interpolated values for years without survey effort.

Several of the predicted densities had a high level of variance associated with them, in particular the first and last density estimates, resulting in the confidence intervals of the differences between densities in pairs of years being large (Figure 4.16).

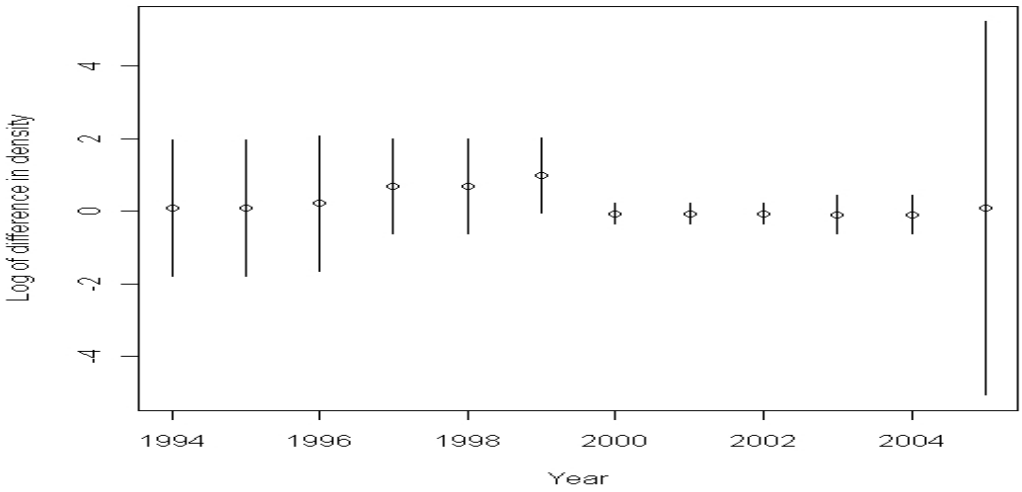


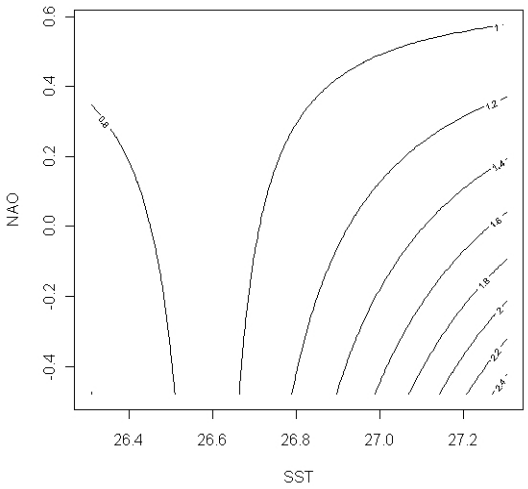
Figure 4.16: The difference in predicted log density between pairs of years, on the log scale, with 95% confidence intervals.

A precision-weighted linear regression of the new covariates against the difference in predicted log density between years resulted in a final model containing the covariates sea surface temperature (SST) and North Atlantic Oscillation index (NAO) as main effects and as an interaction term, and the number of seismic surveys conducted (Table 4.14).

Table 4.14: The best three linear models of population change of fin whales in FAO 21 + 31. ":" between two covariates represents an interaction term in the model.

Model	Covariates retained	AIC
1	SST + nSurvey + NAO + SST:NAO	-53.7
2	SST + nSurvey + NAO + nShots	-53.1
3	SST + nSurvey + NAO + SST:NAO + kmSurvey	-52.7

The R-squared value of the final model was 0.86, indicating the model was a good fit to the weighted data. There was strong evidence that the difference in density of fin whales between years varied with SST (p-value = 0.0033), and some evidence that the number of seismic surveys conducted and the NAO influenced fin whale population change (p-values of 0.0487 and 0.0317 respectively). The parameter estimates were positive for both NAO and SST, suggesting that the rate of population change of fin whales in FAO 21 + 31 is positively influenced by NAO and SST (Figure 4.17). The parameter estimate for nSurvey was negative, suggesting that the rate of population change is negatively influenced by the number of seismic surveys conducted (Figure 4.18), but the confidence intervals are wide.



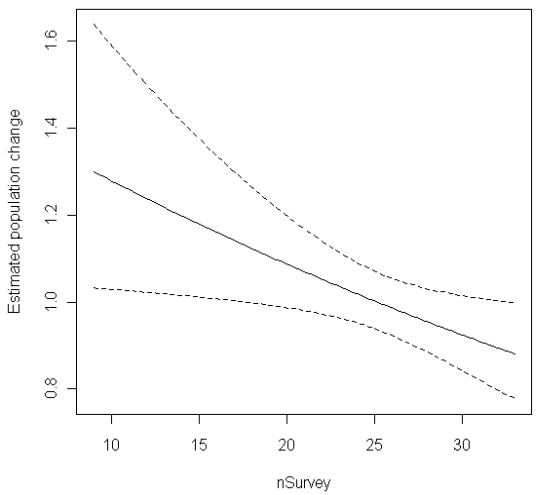


Figure 4.17 (on the left): Estimated population change for fin whales in FAO 21 + 31 over a range of SST and NAO values (with constant nSurvey).

Figure 4.18 (on the right): Estimated population change for fin whales in FAO 21 + 31 over a range of nSurvey values (with constant NAO and SST). Dotted lines show the 95% confidence intervals of the predictions.

Predictions (on the scale of the log of the difference in density between pairs of years) of the best linear model are shown in red alongside the density data (predicted by the best generalised model for years with data and linearly interpolated for years without) in black in Figure 4.19. The bars show 95% confidence intervals.

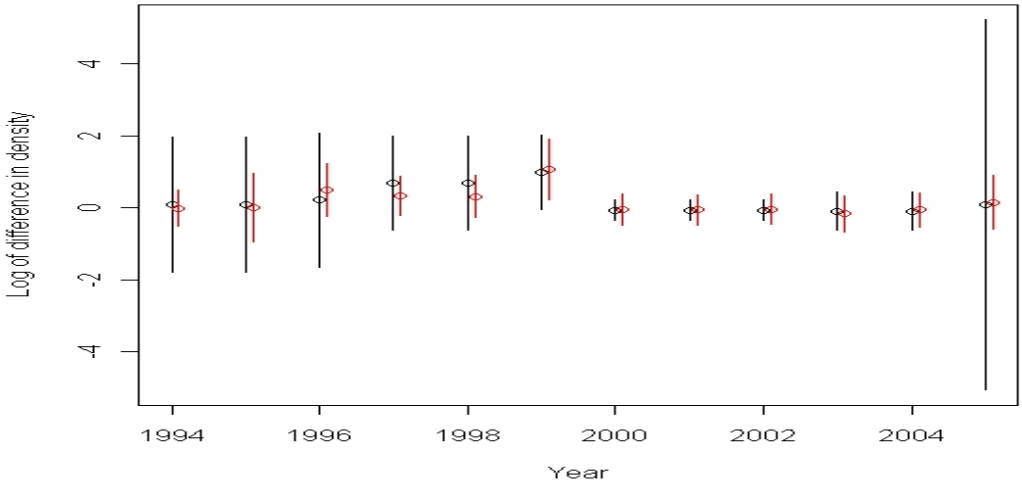


Figure 4.19: Predictions, from the best linear model, of population change (in red) alongside the log density data (predicted and interpolated) in black for fin whales in FAO 21 + 31.

A generalised additive model containing the same covariates as those retained in the final linear model was fitted to the data to check for nonlinear relationships. The model output

from the GAM was very similar to the output from the linear model suggesting existing relationships had been adequately detected.

4.2.4 Humpback whales FAO 21 +31

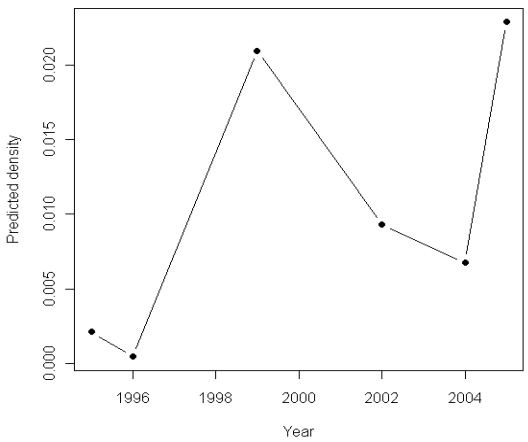
Humpback whale density had been estimated for 6 years between 1995 and 2005; only two estimates came from consecutive years. Generalised Additive modelling of the humpback whale density data from FAO 21 + 31 (Table 4.15) found that a model containing an interaction between year as a factor and the survey platform used (i.e. aerial or shipboard) had the lowest AIC score. However, often there was only one survey platform used in each year, making it difficult to choose which level of the factor variable should be selected if the model was used to generate predictions of density. The next best model (model 3) without an interaction term contained year and method, both as main effects. However, closer inspection of the data (Figure 4.20, Appendix) showed there was confounding between the year 2005 and method D (shipboard surveys not corrected for g(0)), therefore the effects of year and method could not be separated and the model should not be used to generate predictions. The next best model without an interaction term (model 5) contained the covariates year and MethodG0Correct but also generated problems. There was only one year in which a mixture of surveys corrected for g(0) and not corrected for g(0) were completed (Figure 4.21, Appendix) and the resultant parameter estimate for g(0) suggested that density estimates that had been corrected for g(0) were more than 6 times higher than those not corrected for g(0). Density estimates corrected for g(0) are rarely more than three times higher than those that are not corrected, and this is for small cetaceans that are more difficult to detect, so these model results do not seem to have accurately captured the pattern in the data. Moving down the list, model 6 contained year as a factor variable and a smooth of the maximum latitude at which the survey was conducted. This model generated highly suspicious predictions too (Figure 4.22, Appendix), possibly because a small number of surveys were conducted at a relatively high latitude. Instead the next best model, simply containing year as a factor variable, was considered to be the most suitable for generating predictions of humpback whale density for the years with data.

Table 4.15: The best seven generalised additive models of humpback whale density in FAO 21 + 31. ":" between two covariates represents an interaction term in the model.

Model	Covariates retained	AIC	GCV	% deviance explained
1	Year:MethodPlat	-171.4	1.381	61.4
2	Year:Method	-169.8	1.535	62.0
3	Year + Method	-165.9	1.684	52.9
4	Year:MethodG0Corr	-162.5	1.818	43.0
5	Year + MethodG0Corr	-162.5	1.818	43.0
6	Year + s(MaxLat)	-160.8	1.925	39.6
7	Year	-160.5	1.869	34.7

This model (model 7) was used to predict humpback whale density for those years for which there were density estimates (Figure 4.23). The data were interpolated to produce density

estimates for the years with missing data (Figure 4.24).



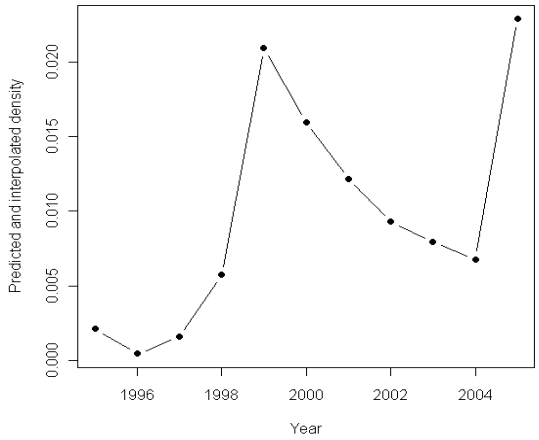


Figure 4.23 (on the left): Predicted humpback whale density for years with survey effort.

Figure 4.24 (on the right): Predictions of humpback whale density including interpolated values for years without survey effort.

The first and last predicted densities had a high level of variance associated with them resulting in the confidence intervals of the differences between densities in pairs of years being large (Figure 4.25).

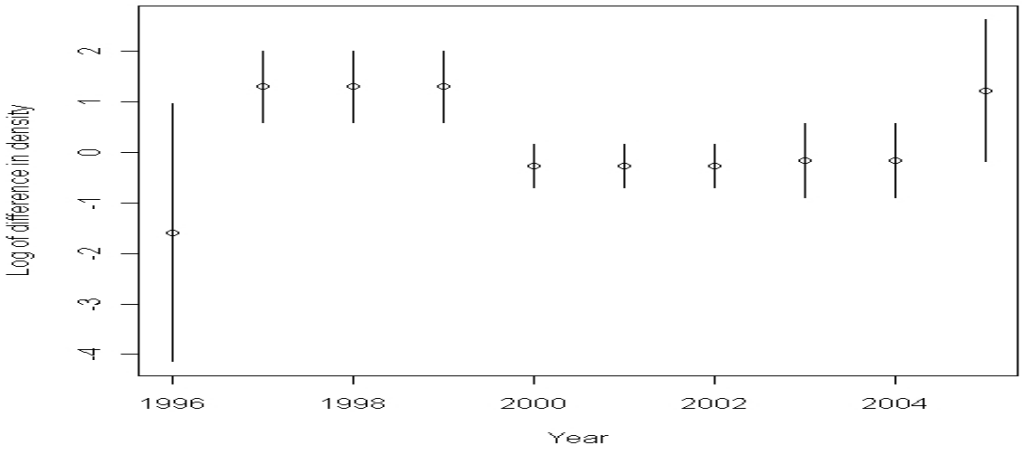


Figure 4.25: The difference in predicted log density between pairs of years, on the log scale, with 95% confidence intervals.

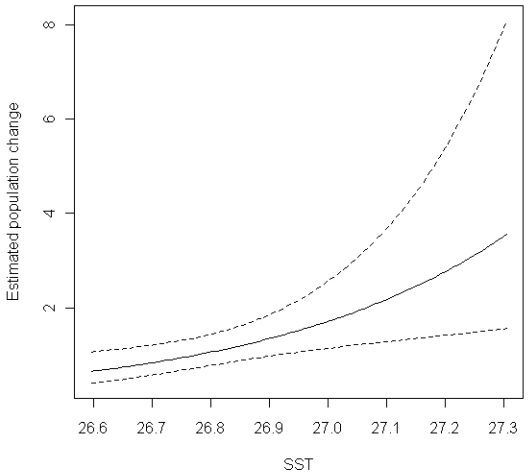
Each of the new covariates was included in a precision-weighted linear regression. The response variable was the difference in log humpback whale density between pairs of years, and a stepwise selection procedure based on the AIC value selected a final model containing

the covariates sea surface temperature (SST) and the number of seismic surveys conducted (nSurvey) as main effects (Table 4.16).

Table 4.16: The best three linear models of population change of humpback whales in FAO 21 + 31.

Model	Covariates retained	AIC
1	SST + nSurvey	-12.4
2	SST + nSurvey + kmSurvey	-12.2
3	SST + nSurvey + NAO	-11.8

The multiple R-squared value of the final model was 0.65. The p-value for SST (0.0218) showed there was relatively strong evidence that humpback whale population change is related to SST in FAOs 21 and 31. The parameter estimate was positive for SST, suggesting that the relationship between humpback whale population change and SST is a positive one (Figure 4.26). However, the p-value for nSurvey (0.1171) did not provide sufficient evidence (at the 5% probability level) to reject the null hypothesis of there being no relationship between humpback whale population change and the number of seismic surveys conducted. The parameter estimate for nSurvey was negative, suggesting that the rate of population change may be weakly negatively influenced by the number of seismic surveys conducted (Figure 4.27), but the confidence intervals are wide.



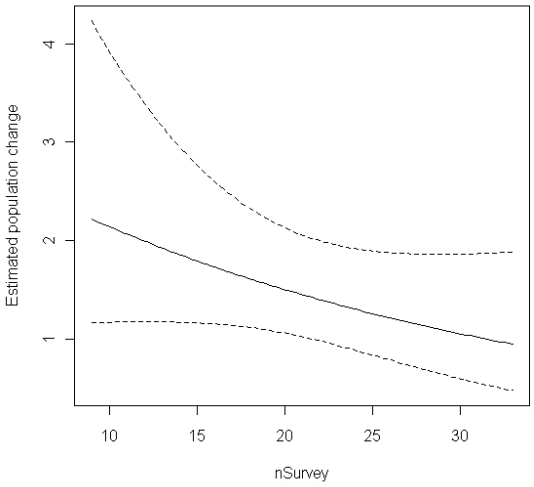


Figure 4.26 (on the left): Estimated population change for humpback whales in FAO 21 + 31 over a range of SST values (with constant nSurvey).

Figure 4.27 (on the right): Estimated population change for humpback whales in FAO 21 + 31 over a range of nSurvey values (with constant SST). Dotted lines show the 95% confidence intervals of the predictions.

Predictions (on the scale of the log of the difference in density between pairs of years) of the best linear model are shown in red alongside the density data (predicted by the best generalised model for years with data and linearly interpolated for years without) in black in Figure 4.28. The bars show 95% confidence intervals.

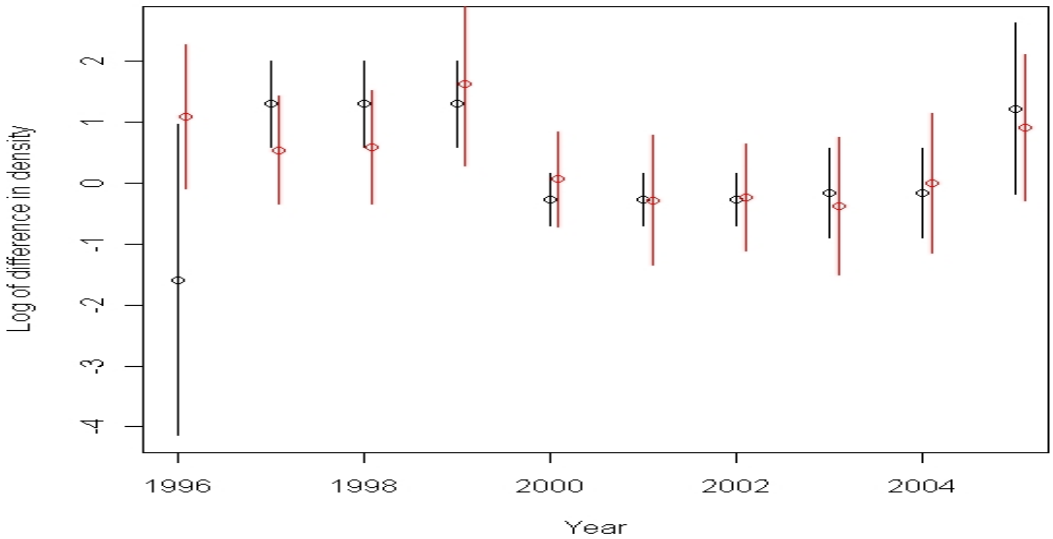


Figure 4.28: Predictions, from the best linear model, of population change (in red) alongside the log density data (predicted and interpolated) in black for humpback whales in FAO 21 + 31.

4.2.5 Sperm whales FAO 21 +31

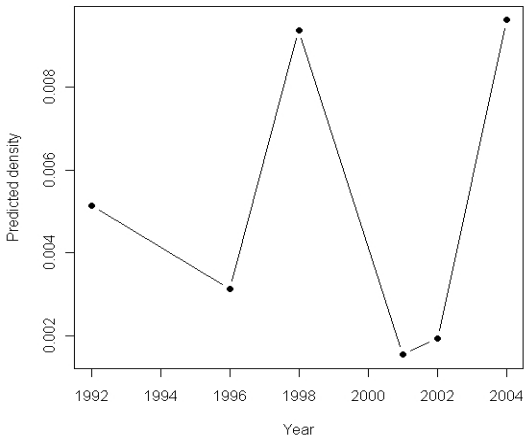
Sperm whale density in FAO 21 + 31 had been estimated in 6 years between 1992 and 2004; only two of the estimates came from consecutive years. The best generalised additive model of sperm whale density from surveys in FAO 21 + 31 (Table 4.17) contained year as a factor and a smooth of the minimum latitude of the survey area, the same as was found for minke whales in FAO 27 and fin whales in FAO 21 + 31. Four different survey methods were represented in the sperm whale data, including methods that corrected for animals missed on the trackline, and surveys were conducted in summer and non-summer. The three best models all contained a spatial covariate (a measure of latitude) in addition to year, while the two next best models contained methodological covariates in addition to year.

Table 4.17: The best three generalised additive models of sperm whale density in FAO 21 + 31. ":" between two covariates represents an interaction term in the model.

Model	Covariates retained	AIC	GCV	% deviance explained
1	Year + s(MinLat)	-214.9	0.371	91.5
2	Year + s(MaxLat)	-213.3	0.377	90.4
3	Year + s(Lat)	-211.5	0.408	89.6

However, closer inspection of model 1 revealed an extremely high parameter estimate for 2001 and when the model was used to predict sperm whale density the estimate for 2001 was approximately 80 times higher than for other years (Figure 4.29, Appendix). Reducing the flexibility of the generalised additive model (by reducing the number of knots) solved this problem; minimum latitude was then fitted in the model as a linear term and the

predictions (using a mean value of latitude) were much more realistic (Figure 4.30). This modified version of model 1 explained 80.9% of the deviance in the data and had a GCV score of 0.459. The data were interpolated to produce density estimates for the years with missing data (Figure 4.31).



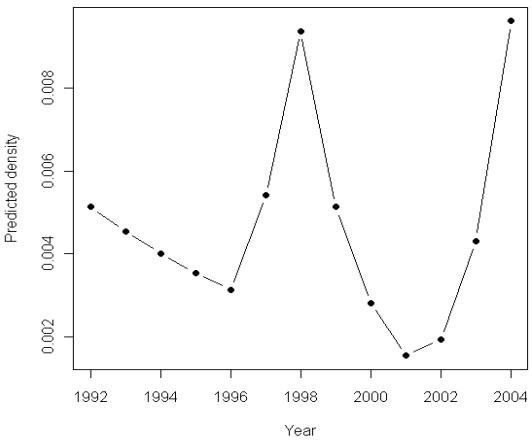


Figure 4.30 (on the left): Predicted sperm whale density for years with survey effort.

Figure 4.31 (on the right): Predictions of sperm whale density including interpolated values for years without survey effort.

Many of the predicted densities had a high level of variance associated with them resulting in the confidence intervals of the differences between density in pairs of years being large, especially in 2002 (Figure 4.32).

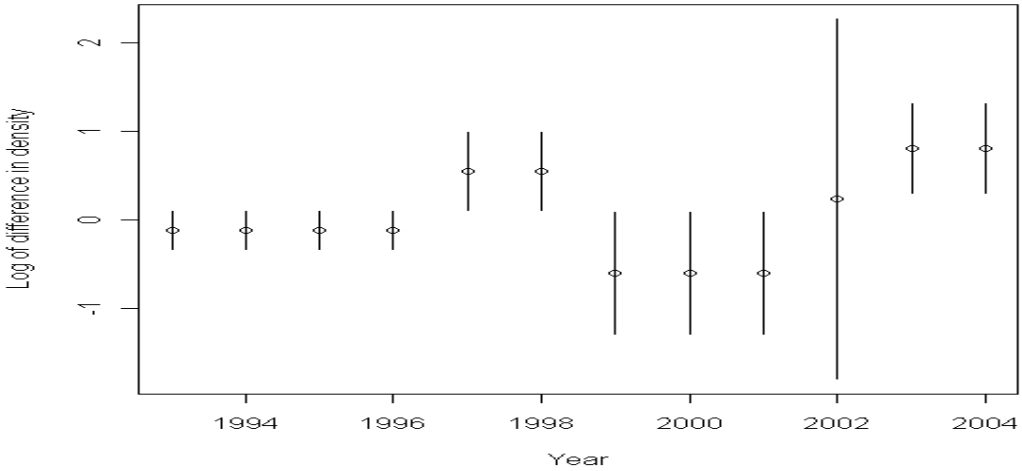


Figure 4.32: The difference in predicted log density between pairs of years, on the log scale, with 95% confidence intervals.

Each of the covariates was then included in a precision-weighted linear regression. The response variable was the difference in sperm whale log density between pairs of years, and

a stepwise selection procedure based on the AIC value selected a final model containing a covariate of the number of seismic surveys conducted (Table 4.18).

Table 4.18: The best three linear models of population change of sperm whales in FAO 21 + 31.

Model	Covariates retained	AIC
1	nSurvey	-32.5
2	nSurvey + NAO	-31.1
3	kmSurvey	-31.0

The multiple R-squared value of the final model was 0.54, indicating that 54% of the deviance in the weighted data was captured by the model. There was strong evidence (p-value = 0.00613) that the rate of change of sperm whale density in FAOs 21 and 31 is influenced by the number of seismic surveys conducted. The parameter estimate for nSurvey was positive, suggesting that the rate of population change of sperm whales in FAO 21 and 31 is positively influenced by the number of seismic surveys conducted (Figure 4.33).

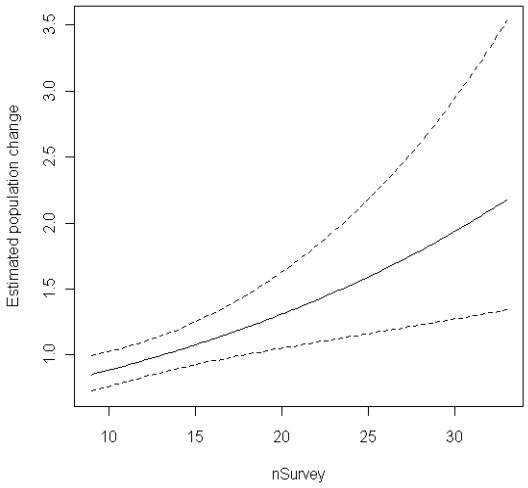


Figure 4.33: Estimated population change for sperm whales in FAO 21 + 31 over a range of nSurvey values.

Predictions of the difference in density between pairs of years (on the log scale) of the best linear model are shown in red alongside the density data (predicted by the best generalised model for years with data and linearly interpolated for years without) in black in Figure 4.34. The bars show 95% confidence intervals.

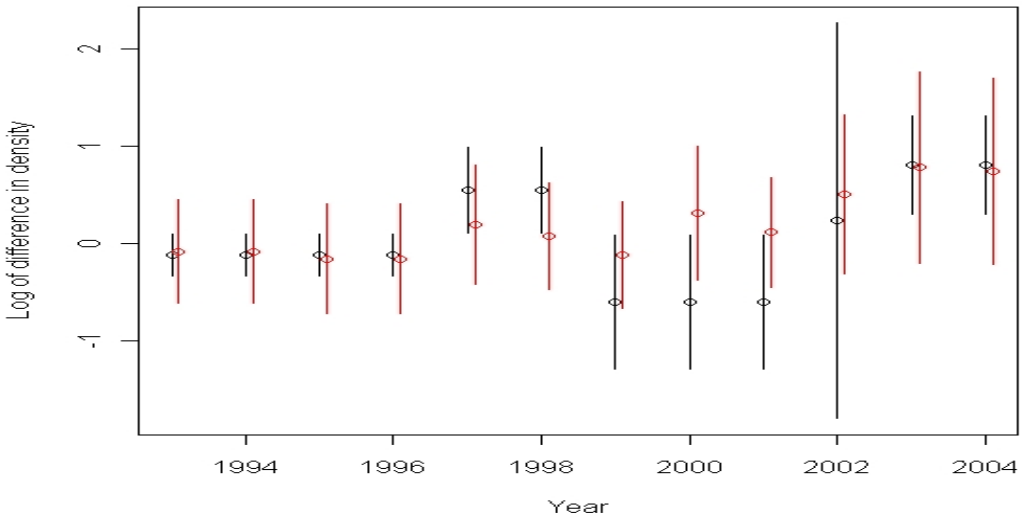


Figure 4.34: Predictions, from the best linear model, of population change (in red) alongside the log density data (predicted and interpolated) in black for sperm whales in FAO 21 + 31.

5. Overall Summary

When considering potential relationships between trends in cetacean populations and oil and gas exploration and production (E&P) sound it is necessary to consider the behavioural and physiological responses of cetaceans to E&P sound. As reviewed above, the range of high source level sounds produced during oil and gas E&P could potentially have a range of effects on cetaceans including, direct physical, chronic stress, perceptual, behavioural and indirect effects. These effects could then have the potential to lead to population effects through a number of mechanisms, including energetic deficiencies, reductions in viability, direct injury and mortality, but currently sufficient data does not exist to verify this. Direct assessment of population effects of E&P sound is generally prohibited by a lack of knowledge of cetacean population trends; studies of behavioural responses of cetaceans to seismic survey sound have more commonly be used to infer potential impacts of E&P sound on individuals.

As reviewed above, avoidance of seismic survey areas by fin and minke whales has been demonstrated, with individuals surfacing further from the vessel during seismic surveys and orientating themselves away from the sound source. Observers in Australia noticed that humpback whales demonstrated avoidance of seismic survey vessels whilst the seismic survey was being conducted, but concluded that gross disruption of the whale's migration route did not occur. Studies of the response of sperm whales to seismic survey sound have produced contradictory behavioural observations ranging from strong avoidance of areas where seismic exploration is occurring to no response to sound generated by seismic surveys. More consistent results have been reported when investigating responses of sperm whales to vessel traffic, with strong behavioural reactions recorded.

The IHS seismic survey database formed the basis of the review of global E&P sound. This database is not a complete record of all seismic surveys undertaken and closer inspection of the spatial spread of the records indicated that the database may not be truly representative of the global distribution of seismic survey effort.

Globally, the number of seismic surveys undertaken peaked in the 1980s but a different trend in seismic effort can be seen in the Northwest Atlantic where the number of seismic surveys started each year showed a general increase until the early 2000s.

Of the records contained within the database, the majority of seismic surveys had been conducted in the Northeast Atlantic, particularly in the North Sea and around the coasts of the United Kingdom, Ireland and Norway. Seismic exploration (measured by total number of shots fired) peaked in this area in the mid 1990s, which coincides with a peak in the number of 3D surveys started. Across all survey types, the number of seismic surveys started each year has been declining since the late 1980s, but the total number of shots fired each year started to increase again in 2003, a reflection of the increase in the number of 4D surveys being conducted each year. The Northeast Atlantic is also the region where most systematic cetacean surveys have been conducted (Jewell *et al*., 2008).

Seismic survey effort has also been intense on the west coast of Africa, around Australia and offshore of Brazil, which are three areas with a distinct lack of systematic cetacean surveys (Jewell *et al*., 2008).

The primary objective of this project was to determine the potential relationships between trends in cetacean populations and E&P sound data within JIP areas of interest but, as has been outlined previously, trends are often extremely difficult to detect due to low statistical power. Task 2 showed that combining published cetacean density estimates led to very poor statistical power for detecting population trends (Quick *et al*., 2008). During this task a combination of Generalised Additive Models (GAM) similar to those developed during Task 2 and a weighted stepwise linear regression analysis was applied to the four most data-rich species in the global cetacean database to try and determine relationships between cetacean trends and E&P sound data. Including the covariates from Task 2 in the generalised additive models attempted to take account of confounding variables in the density estimates (for example, differences in cetacean density arising from conducting surveys at different latitudes). However, the methodology used for this analysis has a number of limitations and should be considered exploratory at best. The data used for the generalised additive models were sparse – minke whales had density estimates from the most years, with 11 years of data, but there were data from only two consecutive years for humpback and sperm whales in FAO 21 + 31. Nevertheless, the models were used to generate predictions of cetacean density for the years that had density estimates and predicted density values were treated as known values, rather than estimated values. For years that did not have density values, it was assumed that the population had either increased or decreased exponentially and density values were interpolated; these values were also treated as known, rather than estimated. The linear interpolation created a continuous time series of data that allowed linear regression, using annual population change as the response variable, to be undertaken (otherwise there would not have been enough data) to investigate relationships between the rate of population change and the environmental, anthropogenic and seismic survey covariates. By weighting the rate of population change by the precision of the estimated population change we attempted to account for the variation

in the precision of the estimates. Modelling population change without any weighting reduced the amount of variation in the data explained by the models. However, the additional uncertainty associated with predicting missing years of population change according to the exponential population change model was not incorporated into the linear regression so remained unaccounted for.

During the linear regression, the distribution of the response variable (log of population change) was assumed to be normally distributed: in truth the underlying distribution of the response variable was not known but no compelling evidence of non-normality was found. It was not investigated whether the time-series nature of the data (i.e. the serial correlation between population change estimates) impacted heavily upon the results of the analysis and this is something that should be considered further. In addition to this, the use of linear models to investigate the relationships between the response variable and the explanatory covariates means that they were assumed to be linear. This assumption may be reasonable: for two of the species, generalised additive models were fitted in addition to the linear models and the results were very similar. This suggests that for those species the linear models adequately detected relationships in the data, or that the sample size was too small to support a non-linear model. While this analysis did produce some interesting results, the limitations of the methodology used mean the results should only be considered exploratory.

For three of the four species modelled the best generalised additive model of cetacean density contained a smooth of the minimum latitude of the survey areas in addition to the compulsory covariate year. For the fourth species (humpback whales in FAOs 21 and 31) density was best explained by the observation platform used during the survey, in addition to year.

Environmental covariates were the covariates most frequently retained in the linear regression models of population change; sea surface temperature was retained in the final models of three species. For fin and humpback whales in FAOs 21 and 31, sea surface temperature had a positive effect on population change, whereas for minke whales in FAO 27 SST had a negative effect on population change although the relationship was not significant at the 5% probability level. The North Atlantic Oscillation index was retained in the final models for fin and minke whales and had a positive influence on population change in both instances.

The only cause of anthropogenic mortality included in the models was the commercial catch of minke whales in FAO 27. However, the annual catch was small relative to population size (<1%). This covariate did not improve the fit of the model to the data and was not retained.

The number of seismic surveys undertaken was retained as an explanatory covariate for three species. For fin and humpback whales in FAOs 21 and 31 the number of seismic surveys conducted had a negative effect on population change, although the relationship was not significant at the 5% probability level for humpback whales. The negative relationship found between the rate of population change of fin whales and the number of seismic surveys conducted was statistically significant at the 5% probability level (p-value=0.0487). The number of seismic surveys conducted was also retained in the final linear model of sperm whale population change, although this time the relationship was a positive one. The relationship was significant at the 5% probability level.

Overall, this task has described and implemented a modelling approach suitable for investigating possible relationships between E&P sound and cetacean stock trends using published cetacean density estimates and a global seismic survey database. Limitations of the data, primarily a lack of cetacean density estimates from consecutive years and imprecise density estimates, mean the results should be considered exploratory. A lack of comprehensive seismic survey data was also a further limitation. Many records in the IHS database were incomplete with respect to area surveyed, requiring a measure of extrapolation.

This study has highlighted that if the resolution in data in terms of consecutive comparable cetacean density estimates and complete information on environmental and E&P sound covariates were available, the modelling framework described would be a suitable method for exploring the factors that may influence changes in cetacean stock trends. With this in mind it is suggested that future cetacean studies would benefit from conducting regular surveys and using comparable methodologies across species and areas.

5.1 Further work and considerations

The distribution of seismic survey effort could be explored and mapped further. The creation of density surface maps based on the number of shot points fired during seismic exploration for all JIP areas of interest would provide a clearer representation of oil and gas exploration and production to date. In particular, mapping the overlap of areas of high seismic survey activity and areas where cetacean abundance surveys have been carried out would be informative. The possibility of this work being conducted by SMRU Ltd and submitted for publication is being discussed.

Given more time, a more refined statistical analysis may also be possible, but given the small quantity of data available it may not yield particularly robust results. Ideally, further analysis would:

- be an integrated analysis that modelled factors affecting the density estimates at the same time as factors affecting annual population changes,
- deal with spatial variation in density in a more refined way,
- automatically weight the population changes appropriately, rather than using the rather arbitrary weighting currently used in the Generalised Additive models,
- deal explicitly with the time series nature of the data.

One possible approach is a Bayesian implementation that would contain a density function (denoting density at point X at time t). One would then make an explicit temporal model for how this density changes as a function of previous density and explanatory covariates, and then try to fit all 3 levels (the observations, the underlying smooth density surface, and the model for temporal evolution of density at once). The feasibility of this approach should be considered further.

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7. Appendix

7.1 Alternative sources of seismic survey data

Numerous sources of data on oil and gas exploration surveys were investigated including the UK DEAL database, Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS)/Joint Nature Conservation Committee (JNCC) reports, the Brown Book, the National Archive of Marine Seismic Surveys, Norwegian Petroleum Directorate, the Petroleum Affairs Division of Ireland and the Netherlands Oil and Gas Portal.

UK DEAL database – www.ukdeal.co.uk

This database contains details of 3989 2-dimensional (2D) and 586 3-dimensional (3D) seismic surveys conducted in United Kingdom Continental Shelf waters. Limited information on these seismic surveys is available on the website, but in order to download a spreadsheet of the survey details and the corresponding shape files, a subscription must be paid.

ASCOBANS/JNCC Report - http://www.service-board.de/ascobans_neu/files/ac13-36.pdf

ASCOBANS produces reports on Information on Seismic Survey Activities by the United Kingdom using data from the UK DEAL database. Two reports span the years of 1997 – 2005 and monthly information on shot point density per 1° by 1° block is given, as is the size of the air guns used when this is easily obtainable. These reports are available for download from the internet, but relate only to UK waters.

The Brown Book – www.databydesign.co.uk

The Brown Book publication has now been replaced by the UK Petroleum Data CD, which contains information relevant to the UK Petroleum industry. This information was previously retrieved from the Department of Trade and Industry's annual report on the Development of the Oil and Gas Resources of the United Kingdom, and is now supplemented with material from other public sources. The CD is available for purchase.

National Archive of Marine Seismic Surveys - http://walrus.wr.usgs.gov/NAMSS/index.html

The U.S. Geological Survey maintains a national archive of marine seismic surveys conducted in US EEZ water. The database is searchable and open access, containing information on the year of the survey, kilometres shot during the survey, and the bounding coordinates of the survey area, but unfortunately it is far from a complete.

Norwegian Petroleum Directorate - http://www.npd.no/English/Frontpage.htm

Maps showing the location of exploration wells in Norwegian waters are available for each year since 1999 (to 2006) from the Norwegian Petroleum Directorate. Data on drilling activity between 1999 and 2008 is also given in tabular form, and searchable maps showing oil and gas fields, production licenses and seismic areas are available.

Petroleum Affairs Division of Ireland - http://www.informatic.ie/paddi/paddi.asp

The Petroleum Affairs Division website has a searchable database of 392 seismic surveys undertaken in Irish waters since 1965. For each survey, the year, survey location and survey location map of the lines surveyed is provided, amongst other information. This information is freely available but unfortunately only relates to small geographical area.

NL Oil and Gas Portal - http://www.nlog.nl/en/home/NLOGPortal.html

The NL Oil and Gas Portal provides information about oil and gas exploration and production in the Netherlands and Dutch sector of the North Sea continental shelf. The identity, number of lines, kilometres surveyed and year are available for 913 2D and 235 3D seismic

surveys conducted between 1960 and 2008. However, the spatial information that accompanies the seismic database must be purchased.

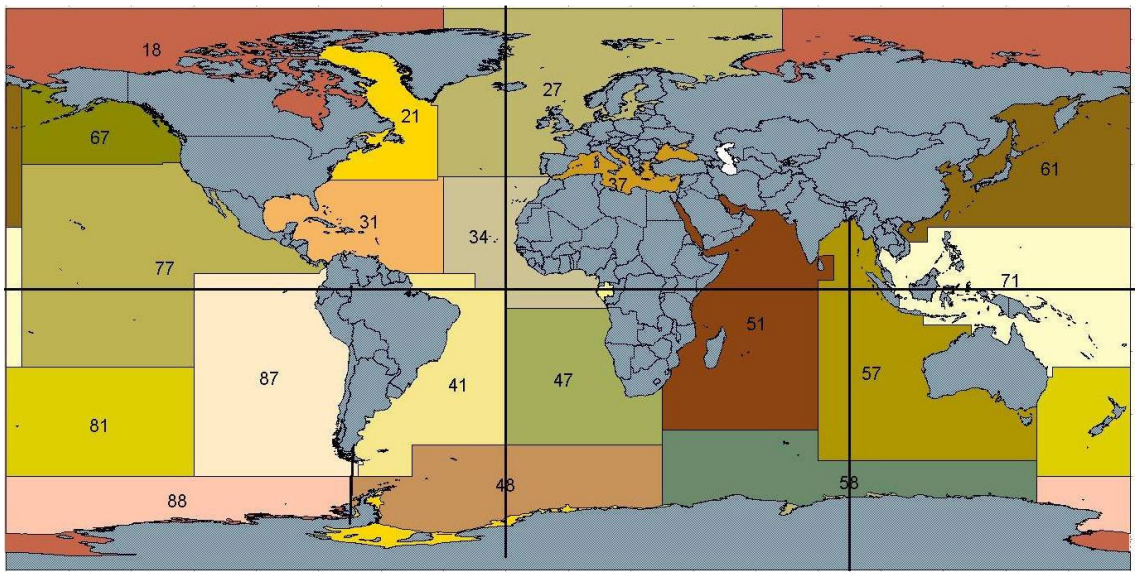

Figure 1.1: The FAO areas of the world.

Table 2.1: Potential broad-scale sensitivities of relevant cetacean species to oil and gas E&P sound (LOW, MED, and HIGH). The scale is based broadly on information on sound source characteristics, known (e.g. Nedwell *et al.*, 2004) or assumed (e.g. Houser *et al.*, 2001; Ridgeway and Carder, 2001) hearing sensitivity, and likely susceptibly to behavioural disturbance.

	Response	Sound source			
Species		**SEISMIC**	**VESSEL**	**DRILLING**	**ASSOCIATED**
Sperm whale					
	Physical	HIGH	MED	MED	MED
	Perceptual	HIGH	MED	MED	MED
	Behavioural	HIGH	HIGH	MED	HIGH
	Indirect	HIGH	LOW	MED	LOW
Fin whale					
	Physical	HIGH	LOW	LOW	MED
	Perceptual	HIGH	HIGH	MED	MED
	Behavioural	HIGH	HIGH	HIGH	HIGH
	Indirect	HIGH	MED	HIGH	LOW
Common minke whale					
	Physical	HIGH	LOW	MED	MED
	Perceptual	HIGH	HIGH	MED	MED
	Behavioural	HIGH	HIGH	HIGH	HIGH
	Indirect	HIGH	MED	HIGH	LOW
Harbour porpoise					
	Physical	HIGH	LOW	LOW	MED
	Perceptual	HIGH	LOW	LOW	MED
	Behavioural	HIGH	HIGH	MED	HIGH
	Indirect	HIGH	MED	HIGH	LOW
Humpback whale					
	Physical	HIGH	LOW	LOW	MED
	Perceptual	HIGH	HIGH	MED	MED
	Behavioural	HIGH	HIGH	HIGH	HIGH
	Indirect	HIGH	MED	HIGH	LOW
Striped dolphin					
	Physical	HIGH	LOW	LOW	MED
	Perceptual	HIGH	HIGH	LOW	MED
	Behavioural	HIGH	HIGH	HIGH	HIGH
	Indirect	HIGH	MED	HIGH	LOW
Long finned pilot whale					
	Physical	HIGH	LOW	LOW	MED
	Perceptual	HIGH	HIGH	MED	MED
	Behavioural	HIGH	HIGH	HIGH	HIGH
	Indirect	HIGH	LOW	HIGH	LOW

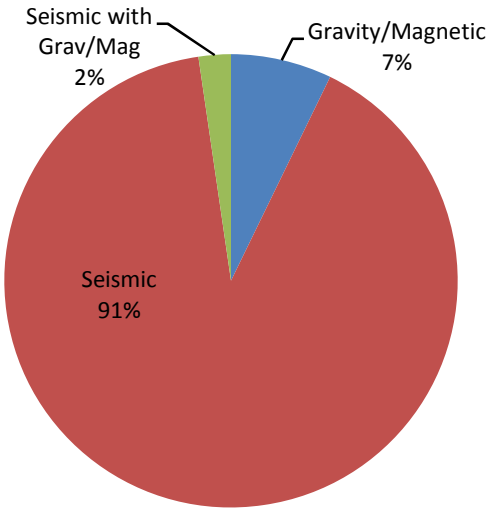

Figure 3.1: Proportion of seismic survey records utilising different methods of geophysical survey.

Table 4.1: Minke whale catch data from Norwegian and Icelandic waters made under objection to the zero catch limit set by the International Whaling Commission in 1985/6 and under Special Permit (http://www.iwcoffice.org/conservation/table_objection.htm and http://www.iwcoffice.org/conservation/table_permit.htm).

Year	No. minke whales caught in Norwegian and Icelandic waters
1986	379
1987	373
1988	29
1989	17
1990	5
1991	0
1992	95
1993	226
1994	280
1995	218
1996	388
1997	503
1998	625
1999	591
2000	487
2001	552
2002	634
2003	684
2004	569
2005	678
2006	606
2007	642

Table 4.4: Number of data points used to calculate the mean and total number of km surveyed each year using each seismic survey type for FAO 27. When none of the surveys of a particular type in a given year had the number of kilometres reported (see years with zeros highlighted in red) the mean number of kilometres surveyed (per survey) using that type of seismic survey across all years was used to calculate the number of kilometres surveyed that year.

Year	2D surveys		3D surveys		2+3D surveys		4D surveys	
	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with km info
1985	217	216	31	30	0	0	0	0
1986	174	172	19	18	0	0	0	0
1987	189	189	16	15	0	0	0	0
1988	215	215	21	15	1	0	0	0
1989	163	162	31	26	3	2	0	0
1990	183	181	48	31	1	0	0	0
1991	161	161	64	41	1	0	0	0
1992	149	149	76	49	2	1	0	0
1993	82	82	82	48	0	0	0	0
1994	60	59	79	21	5	0	0	0
1995	38	37	80	0	0	0	2	0
1996	64	64	87	0	1	0	0	0
1997	67	63	67	0	0	0	2	0
1998	34	32	56	1	2	0	1	0
1999	21	21	42	0	0	0	2	0
2000	24	22	26	0	0	0	8	0
2001	22	20	28	1	0	0	7	0
2002	9	9	23	0	0	0	10	0
2003	17	16	26	0	0	0	9	0
2004	14	12	19	0	2	0	12	0
2005	23	20	34	0	3	0	13	1
2006	37	26	31	0	1	1	14	1
2007	18	9	30	0	1	0	13	2

Table 4.5: Number of data points used to calculate the mean and total number of km surveyed in each year using each seismic survey type for FAO 21+31. When none of the surveys of a particular type in a given year had the number of kilometres reported (see years with zeros highlighted in red) the mean number of kilometres surveyed (per survey) using that type of seismic survey across all years was used to calculate the number of kilometres surveyed that year.

Year	2D surveys		3D surveys		2+3D surveys		4D surveys	
	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with km info
1985	11	9	4	4	0	0	0	0
1986	2	2	0	0	0	0	0	0
1987	3	3	0	0	0	0	0	0
1988	8	7	0	0	0	0	0	0
1989	8	8	2	2	0	0	0	0
1990	7	7	4	2	1	1	0	0
1991	9	9	7	1	1	1	0	0
1992	8	8	3	1	0	0	0	0
1993	6	6	3	0	2	0	0	0
1994	8	4	1	0	0	0	0	0
1995	4	4	5	1	0	0	0	0
1996	4	3	14	0	0	0	0	0
1997	4	4	9	0	2	0	0	0
1998	3	2	7	0	0	0	0	0
1999	13	13	8	0	0	0	0	0
2000	8	8	7	0	1	1	0	0
2001	10	10	15	1	1	0	0	0
2002	11	10	22	0	0	0	0	0
2003	13	13	16	0	2	0	1	0
2004	11	11	13	0	0	0	0	0
2005	9	9	9	0	1	0	0	0
2006	4	3	14	0	0	0	0	0
2007	11	9	8	0	0	0	0	0

Table 4.6: Number of data points used to calculate the total number of shot points fired in each year using each seismic survey type for FAO 27. When none of the surveys of a particular type in a given year had the number of kilometres reported (see years with zeros highlighted in red) the mean number of kilometres surveyed (per survey) using that type of seismic survey across all years was used to calculate the number of kilometres surveyed that year.

Year	2D surveys		3D surveys		2+3D surveys		4D surveys	
	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with sq km info	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with sq km info
1985	217	216	31	1	0	0	0	0
1986	174	172	19	1	0	0	0	0
1987	189	189	16	1	0	0	0	0
1988	215	215	21	0	1	0	0	0
1989	163	162	31	6	3	2	0	0
1990	183	181	48	6	1	0	0	0
1991	161	161	64	17	1	0	0	0
1992	149	149	76	22	2	1	0	0
1993	82	82	82	26	0	0	0	0
1994	60	59	79	48	5	0	0	0
1995	38	37	80	79	0	0	2	2
1996	64	64	87	87	1	0	0	0
1997	67	63	67	66	0	0	2	2
1998	34	32	56	53	2	0	1	1
1999	21	21	42	41	0	0	2	2
2000	24	22	26	24	0	0	8	7
2001	22	20	28	25	0	0	7	7
2002	9	9	23	23	0	0	10	9
2003	17	16	26	26	0	0	9	9
2004	14	12	19	19	2	0	12	12
2005	23	20	34	32	3	0	13	11
2006	37	26	31	27	1	1	14	10
2007	18	9	30	20	1	0	13	9

Table 4.7: Number of data points used to calculate the total number of shot points fired in each year using each seismic survey type for FAO 21+31. When none of the surveys of a particular type in a given year had the number of kilometres reported (see years with zeros highlighted in red) the mean number of kilometres surveyed (per survey) using that type of seismic survey across all years was used to calculate the number of kilometres surveyed that year.

Year	2D surveys		3D surveys		2+3D surveys		4D surveys	
	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with sq km info	No. Of surveys	Surveys with km info	No. Of surveys	Surveys with sq km info
1985	11	9	4	0	0	0	0	0
1986	2	2	0	0	0	0	0	0
1987	3	3	0	0	0	0	0	0
1988	8	7	0	0	0	0	0	0
1989	8	8	2	0	0	0	0	0
1990	7	7	4	2	1	1	0	0
1991	9	9	7	6	1	1	0	0
1992	8	8	3	2	0	0	0	0
1993	6	6	3	3	2	0	0	0
1994	8	4	1	0	0	0	0	0
1995	4	4	5	3	0	0	0	0
1996	4	3	14	14	0	0	0	0
1997	4	4	9	9	2	0	0	0
1998	3	2	7	6	0	0	0	0
1999	13	13	8	8	0	0	0	0
2000	8	8	7	5	1	1	0	0
2001	10	10	15	9	1	0	0	0
2002	11	10	22	22	0	0	0	0
2003	13	13	16	16	2	0	1	0
2004	11	11	13	12	0	0	0	0
2005	9	9	9	9	1	0	0	0
2006	4	3	14	8	0	0	0	0
2007	11	9	8	6	0	0	0	0

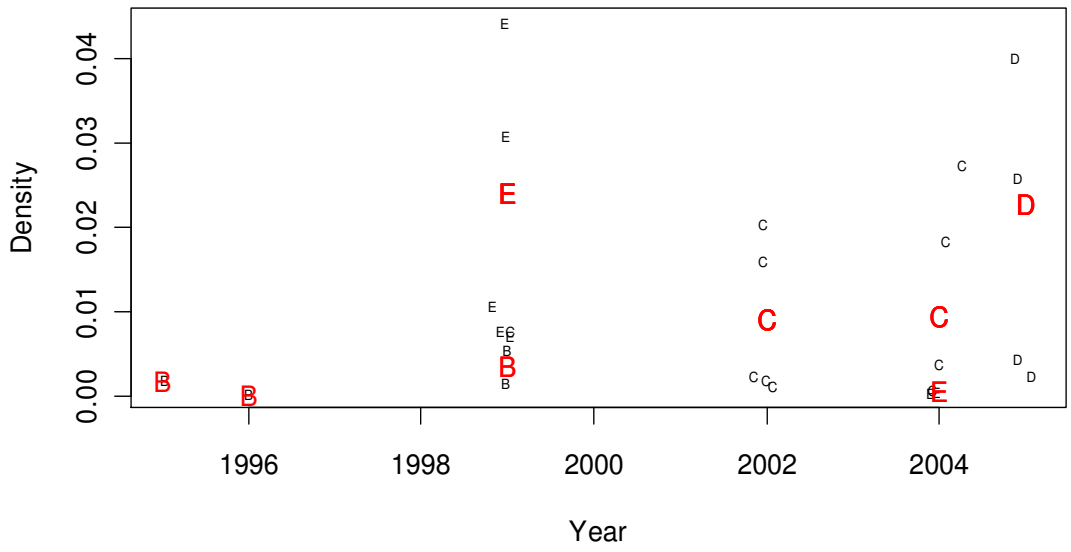

Figure 4.20: Density estimates predicted by model 3 (in red) shown alongside the observed density estimates (in black) of humpback whales in FAO 21 + 31. Confounding between the year 2005 and method D meant that predictions from this model could not be used as a reliable basis for further analysis.

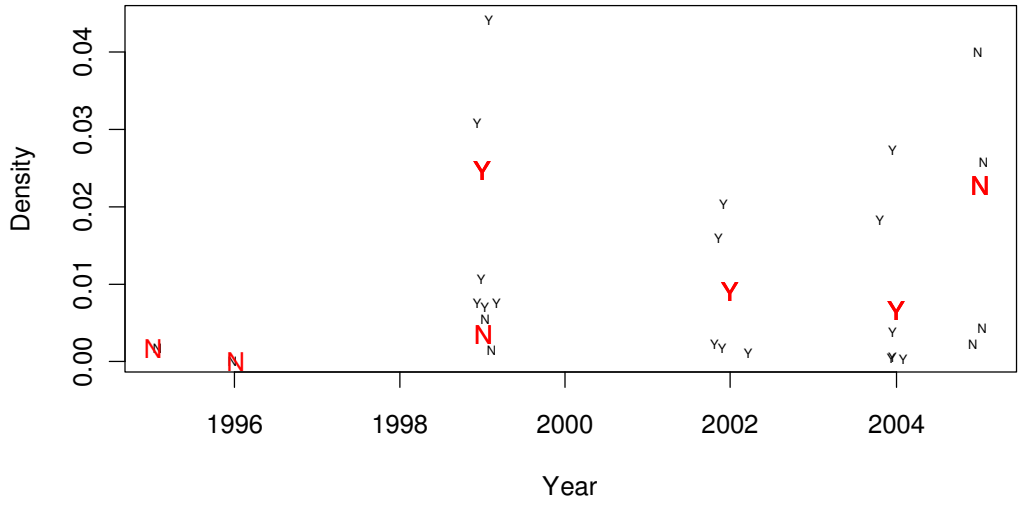

Figure 4.21: Density estimates predicted by model 5 (in red). Observed density estimates of humpback whales in FAO 21 + 31 are shown in black. Only one year, 1999, contains both density estimates that were corrected for g(0) and estimates that weren't corrected.

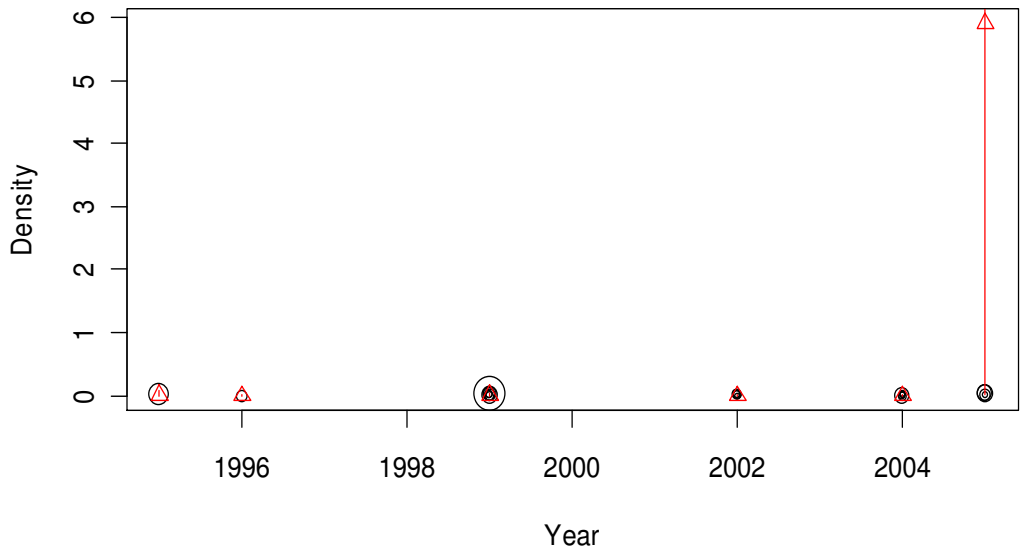

Figure 4.22: Density estimates predicted by model 6 (in red) for a mean value of maximum latitude, with confidence intervals for the predictions shown. Observed density estimates of humpback whales in FAO 21 + 31 are shown in black.

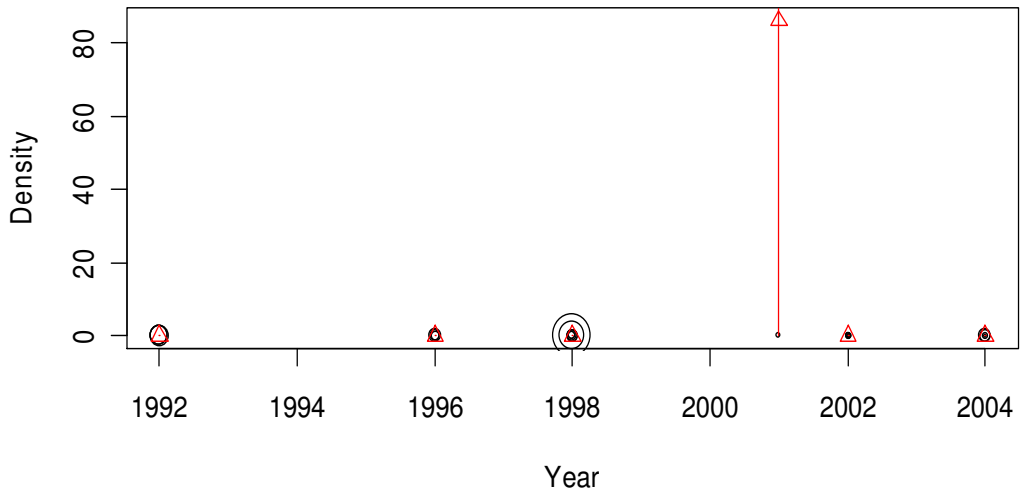

Figure 4.29: Predictions of sperm whale density (in red) generated by model 1 shown alongside the observed density estimates (in black). The prediction of sperm whale density in 2001 is unrealistically high and prompted the modification of model 1.

Report

Project Name:	JIP Cetacean Stock Assessment
Reference:	RA1007OGP
Project Manager:	Nicola Quick

Draft report drafted by:	Rebecca Jewell, Jonathan Gordon, Nicola Quick, Phillip Hammond	
Draft report checked by:	Nicola Quick, Phillip Hammond	
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Task 5 Deliverable:

Cetacean stock assessment in relation to Exploration and Production Industry sound

Species and areas recommended for further data collection and/or analysis

Summary

Task 5 summarises the work completed during the Cetacean Stock Assessment project and discusses both species and geographical areas suitable for further, or more detailed, data collection and analysis. To recommend one method for monitoring populations of all cetacean species would be misleading and impractical. Instead, different monitoring methods are briefly summarised and the main advantages and disadvantages of each are given to illustrate the factors that require consideration during the survey planning stage. The suitability of each method for different taxonomic groups is summarised. The importance of performing power analysis and cost-benefit analysis is highlighted and illustrated through two case studies. A framework for gathering relevant data other than on abundance is presented.

1. Introduction

The primary objective of this task is to summarise the outcomes of the previous tasks completed during the Cetacean Stock Assessment project with a view to recommending areas and species suitable for further data collection and analysis.

Knowledge of trends in cetacean populations is vital for informing management decisions, particularly regarding conservation and mitigation measures, and for assessing the effectiveness of any action taken. However, as demonstrated during Task 2, it is often extremely difficult to detect trends in cetacean populations due to low statistical power as a result of a lack of precise abundance estimates for most populations. Where cetacean surveys are conducted there are often estimates of abundance for the more frequently encountered species but, even for these species, time series of comparable abundance estimates are few.

Taylor *et al*. (2007) suggest that one approach to increase statistical power to detect trends is to design surveys with trend detection (rather than estimation of absolute abundance) in mind. These survey designs would seek to increase the precision of abundance estimates, which could be achieved by decreasing the size of the survey area and increasing effort within that area. This would result in increased precision of abundance estimates, which would fare better in detecting trends. However, surveying a smaller area makes this approach vulnerable to the effects of surveying a varying proportion of the population of interest (Taylor *et al*., 2007), which would generate variability and decrease precision.

2. Summary of the outcomes of previous tasks

2.1 Task One: Cetacean Populations in the Areas of Relevance

The aim of Task 1 was to review information on cetacean populations from the Areas of Relevance (AORs) described (Figure 1). The Environmental Risk Management Capability (ERMC) project database was used as a foundation for this task; it contains details of approximately 90% or more of the cetacean line transect surveys conducted globally. For each AOR a list of cetacean species likely to be present, information on the species for which quality data exist (with maps showing spatial representations of the surveys detailed) and knowledge of the stock structure of each population was provided (Jewell *et al*., 2008). The level of information on cetacean populations and stocks was highly variable between AORs; areas 4, 5A and 5B had the best resolution of information but for most species in other areas abundance and stock structure were unknown. Areas 5A, 5B and 6A contained the most systematic line transect surveys, which allow density information to be compared among years if multiple surveys have been conducted within the same areas. Following review of this report by the Joint Industry Program (JIP), it was agreed that areas 4 (Alaska), 5A (West coast of Canada, the USA and Mexico), 5B (East coast of Canada, the USA and the Gulf of Mexico) and 6A (NE Atlantic) would be taken forward for further analysis.

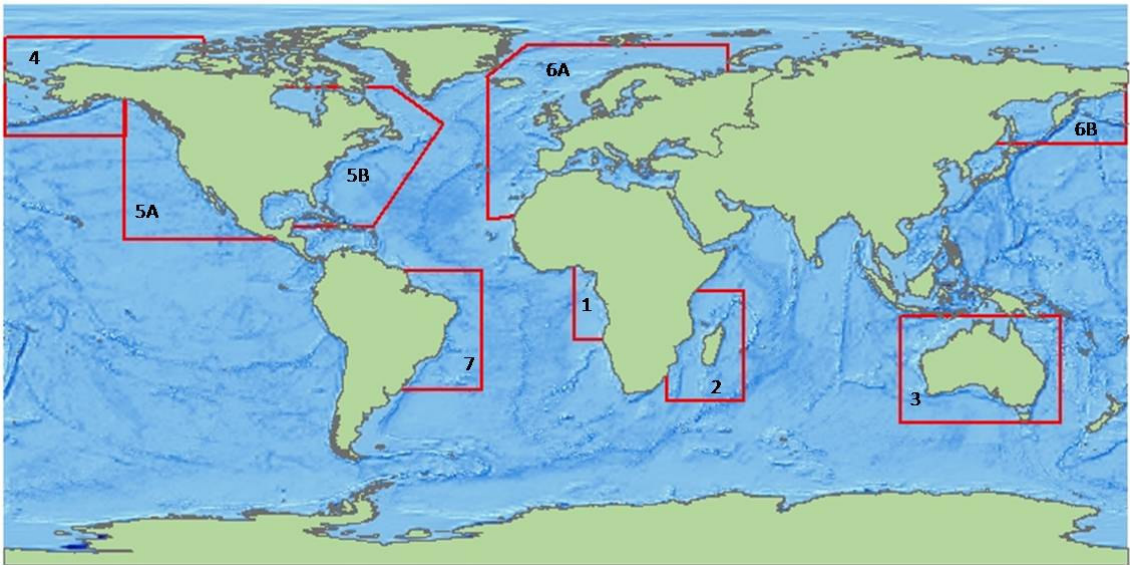

Figure 1.0: The nine Areas of Relevance identified for investigation of the availability of cetacean abundance data.

2.2 Task Two: Review of cetacean data

During Task 2 the cetacean survey data reviewed during Task 1 was used to explore the robustness of cetacean trend estimates. A global analysis using 1035 density estimates from 34 species of cetacean was conducted using Generalised Additive Models (GAMs) to determine which covariates most influenced the density estimates. Temporal (e.g. year, decade), taxonomic (species, family and family group), spatial (FAO area and ocean basin) and methodological (survey agency and method) explanatory variables were investigated. Taxonomic covariates were found to be the most important covariates, explaining more of the variability in the data than the other covariates. The next step was therefore to model,

again using GAMs, the most data-rich species individually. Seven species were selected (minke, fin, sperm, humpback long-finned pilot whale, striped dolphin and harbour porpoise). The explanatory variables explaining most of the variability in the data varied between species and area. Power analysis to reveal the level of population change detectable given the variability in the data showed that the power to detect population trends even for these most data-rich species was low. The mean coefficient of variation for the population change index was 0.85 and at this level of precision, only a change in population size of half an order of magnitude or greater would be detectable with a power of 0.8 (a common benchmark for acceptable level of power)(Quick *et al*., 2008).

2.3 Task Three: Environmental and anthropogenic factors linked to influencing or controlling cetacean population growth rates

Task 3 focused on collating data on the maximum population growth rates (R_{MAX}) for the most data-rich species as well as on environmental and anthropogenic factors influencing or controlling cetacean population growth rates. The main factors considered were anthropogenic mortality, food availability, disease, contaminants, and climate change. Task 3 also reviewed information on how stocks of cetacean species differ in rates of recovery, where data are available, and provided an assessment of the factors that may have limited the recovery of certain populations/stocks and the major gaps in the data.

Various factors were found to control cetacean population growth rates, and the extent of their control was linked to the status of the cetacean population and the severity of the effect. For populations of large baleen whales the most influential factor seemed to be large-scale anthropogenic removal of individuals by whaling, whereas for smaller cetaceans direct takes and incidental capture appeared to be the most influential factors (Murphy, 2008).

2.4 Task Four: Relating trends in cetacean populations to E&P sound data

The primary objectives of Task 4 were to map the distribution of seismic E&P sound, to review the behavioural and physiological responses of cetaceans to E&P sound and to determine the potential relationships between trends in cetacean populations and E&P sound data within JIP areas of interest.

Maps were created of the distribution of seismic surveys globally, and density surfaces of the number of shots fired during seismic surveys of different types were generated for two of the JIP areas of relevance (AOR_005B and AOR_006A). Seismic exploration for oil and gas, according to the IHS seismic database, has been most intense in the NE Atlantic both in terms of the number of surveys conducted and the intensity of those surveys (Jewell *et al*., 2009).

It is known that oil and gas E&P sound can have direct physical, chronic stress, perceptual, behavioural and indirect effects on cetaceans and that these effects have the potential to lead to population effects through a number of mechanisms, including energetic deficiencies, reductions in viability, direct injury and mortality. Avoidance of seismic survey areas by fin, humpback, minke and sperm whales has been demonstrated.

During Task 4 a combination of Generalised Additive Models (GAMs) similar to those developed during Task 2 and a weighted stepwise linear regression analysis was applied to the four most data-rich species in the global ERMC cetacean database (minke, fin, humpback and sperm whales) to investigate relationships between cetacean trends and E&P sound data. Covariates from Task 2 were included in the generalised additive models to take

account of confounding variables in the density estimates (for example, differences in cetacean density arising from conducting surveys at different latitudes and using different methods). Environmental (sea surface temperature and North Atlantic Oscillation index), anthropogenic (whaling catch data for minke whales) and seismic covariates (number of surveys conducted, number of kilometres surveyed and number of shot points fired) were included in the linear regression models of population change. Environmental covariates were the most frequently retained in the linear regression models of population change; sea surface temperature was retained in the final models of three species. The number of seismic surveys undertaken was also retained as an explanatory covariate for three species. However, the methodology used for this analysis has a number of limitations and should be considered exploratory at best.

3. Species and areas suitable for further data collection

The analysis conducted for Task 2 showed that very few of the species had data with sufficient power to detect trends. Only for three of the species, and in a limited number of AORs, could a 50% reduction of the population be detected with a power of 0.8 - these were the minke whale and long-finned pilot whale in AOR_006A and the striped dolphin in AOR_005A and AOR_005B. To detect trends in any population of cetaceans further data are required. The importance of the taxonomic covariates during the global analysis in Task 2 (Quick *et al*., 2008) indicates that when looking for trends, species should be considered singly, so sufficient data is needed for each species of interest if trend analysis is to be attempted.

There are a number of factors that might be considered when deciding which species should be the target of future data collection and analysis. One factor is which species are most endangered. Estimating abundance and trends of species with low density is extremely difficult so a large amount of resource will be required to obtain useful data if this is the prime consideration. A second consideration is which species are most vulnerable to the effects of oil and gas exploration and production sound. There is a large literature to inform this aspect. Another important factor is the amount and quality of available data on the species. It will be easier to investigate the effects of exploration and production sound for species for which good data already exist. A related factor is the ease with which future data can be collected. In this regard, Section 4 describes the importance of considering the appropriate method for data collection for each species. It is likely that in any area, it will be desirable to collect data on more than one species. Choosing target species thus requires consideration of the compatibility of the best methods for these species. For example, it may be possible and cost-effective to conduct visual and acoustic data collection simultaneously on survey ships as was done on the SCANS-II and CODA surveys (SCANS-II 2008; CODA 2009).

With regard to areas recommended for further data collection, again there are a number of considerations. In areas where oil and gas exploration is planned to commence and little is known of the cetaceans present in the area, surveys conducted prior to exploration would provide valuable baseline data. This is particularly true for offshore areas where developments in industry technology are enabling exploitation of deeper waters beyond the continental shelf. These areas have received little attention with regard to dedicated

cetacean surveys and without baseline data it will be much more difficult to investigate any effects of oil and gas exploration and production in those areas.

4. Methods for estimating cetacean abundance and monitoring temporal and spatial trends

A number of approaches can be used to estimate and/or monitor the abundance of a population or the use of a defined area by a cetacean species. These include fixed point sampling, line transect sampling, mark-recapture sampling, and modelling abundance as a function of environment variables. Platforms for data collection include fixed observation points on land or at sea, or ships and aircraft surveying at sea. Different data types include visual detections of animals at the surface, acoustic detections of vocalising animals, and photo-identification of individuals. Methods range from the very basic, yielding simple indices of abundance in limited areas, to the advanced, yielding accurate and precise estimates of absolute abundance across wide areas (e.g. Hammond *et al.,* 2002; SCANS-II 2008; CODA 2009).

Here we give a very brief description of these methods and tables that summarise the strengths and weaknesses of each, drawing particularly on Evans and Hammond (2004), the review of monitoring methods undertaken as part of the SCANS-II (2008) project, and Hammond (in press).

4.1 Visual survey data

4.1.1 Incidental sightings

In areas where little or no previous information is available, the collection of incidental sightings can give first indications of temporal and spatial distribution. Incidental sightings by non-specialists (e.g. bird watchers, ferry and other marine operators, coast guard, fishermen and recreational yachts) provide a low cost data source.

Strengths and weaknesses of using incidental sightings:

Strengths	Weaknesses
• Cheap way of collecting data on species presence in an area. • Provides qualitative information on temporal and spatial occurrence.	• Data will not allow estimation of abundance or trends.

4.1.2 Platforms of opportunity

Data for monitoring cetacean populations can be collected in conjunction with other research projects; so-called “platforms of opportunity” engaged in other activities, e.g. fish or bird surveys, ferries or cruise liners (e.g. Northridge *et al.,* 1995). The main advantage is the possibility of collecting a large amount of data for a fraction of the cost of a dedicated survey. The main disadvantage is that being unable to influence where and when the vessel travels typically results in a distribution of searching effort that precludes abundance

estimation. Data are usually visual sightings but acoustic data (see Section 4.2) can potentially also be collected.

Strengths and weaknesses of using platforms of opportunity:

Strengths	Weaknesses
• Cheap way of collecting data. • Can provide quantitative information on temporal and spatial distribution, depending on distribution of searching effort.	• Normally not possible to dictate time or area covered. • Estimation of abundance dependent on appropriate distribution of searching effort and usually not possible.

4.1.3 Fixed-point land-based sampling

Fixed-point land-based observations have been used to measure variation in density over time in inshore areas. Because the area observed is typically very restricted, these data cannot be used to infer changes in population abundance, only in the occurrence of animals in a limited area.

However, for coastally migrating cetacean species that use predictable near shore routes (bowhead whales, humpback whales and north-eastern Pacific gray whales) land-based fixed-point sampling is an appropriate method for estimating population abundance within a framework similar to line transect sampling (George *et al.,* 2004; Rugh *et al.,* 2005; Noad *et al.,* in press).

Strengths and weaknesses of fixed-point sampling from land:

Strengths	Weaknesses
• Inexpensive way of collecting data. • Provides information on temporal and spatial distribution and trends in the area covered. • Appropriate for estimating abundance for species with predictable and convenient migration routes.	• Limited area observed. • Population abundance monitoring not possible for most species.

4.1.4 Dedicated line transect sampling

Line transect sampling for cetaceans from ship or aerial surveys is well-developed and most robust information on cetacean abundance comes from using these methods. There is a large amount of published material on the theory and application of line transect sampling to cetacean populations (e.g. Hiby and Hammond, 1989; Garner *et al.,* 1999; Buckland *et al.,* 2001; Buckland *et al.,* 2004; IWC, 2005). Key aspects are the need for appropriate survey

design, and the need for accurate data on species identification, group size, angles and distances. Unless it is possible to take account of animals missed on the transect line and any movement in response to the survey platform, estimates will be of relative rather than absolute abundance.

Strengths and weaknesses of using line transect sampling:

Strengths	Weaknesses
• Well developed methodology allows robust estimation of absolute or relative abundance. • Can cover large areas, including potentially the entire range of a population.	• Data collection can be expensive. • Data collection sensitive to weather conditions.

4.2 Acoustic data

The collection of acoustic data on cetaceans has some significant advantages over visual methods. Acoustic data can be collected 24 hours per day, the methods are less dependent on observer skill and weather conditions, and data collection can be automated. However, only species that regularly make recognisable sounds can be targeted and it is not yet possible to use acoustic data to estimate absolute abundance, except for sperm whales, because the relationship between vocalisation and population density is typically unknown.

4.2.1 Towed hydrophone systems

The International Fund for Animal Welfare (IFAW) has developed a number of systems for automatic detection of odontocete vocalisations. In the last few years, these have been used to collect data with the eventual aim of estimating abundance for a number of species (Gillespie *et al.,* 2005; SCANS-II, 2008; Gillespie *et al.,* 2008; CODA, 2009). In the case of sperm whales, abundance has been estimated within a line transect framework (Lewis *et al.,* 2007; CODA, 2009). Collection of data on baleen whales from towed systems is challenging because the low-frequency vocalisations are masked by other sounds.

4.2.2 Static acoustic monitoring

Static acoustic data collection equipment has been used for some time to monitor the presence of cetaceans in particular areas. For example, a device called the T-POD (http://www.chelonia.demon.co.uk) has been used extensively to monitor harbour porpoises in coastal waters (e.g. Carstensen *et al.,* 2006). Static acoustic devices are also used to monitor baleen whales (e.g. Clark and Charif, 1998; McDonald and Fox, 1999; Marques *et al.,* in press). Such data can provide long time series of presence/absence data in a particular area and potentially estimates of density. Work on relating vocalisation rate to abundance on baleen and odontocete whales is being pursued by the Density Estimation for Cetaceans from passive Acoustic Fixed sensors (DECAF) project (http://www.creem.st-and.ac.uk/decaf/).

Strengths and weaknesses of using acoustic data:

Strengths	**Weaknesses**
• Data collection is independent of daylight and most weather conditions and can be automated. • Data can be used to monitor relative abundance if vocalisation rates are assumed to be constant over time. • Good method for estimation of sperm whale abundance.	• High frequency vocalisations have a limited detection range. • Performance of towed systems is dependent on the noise level of the vessel. • Species identification is currently difficult for many species. • Methods to estimate abundance are not well developed.

4.3 Mark-recapture using photo-identification

Mark-recapture methods are used to estimate the number of animals in the population from the capture histories of individual animals. For cetaceans, these methods typically rely on the photographic recognition of individuals from natural marks on their bodies (photo-identification) and sometimes on genetic identification of biopsied individuals. Mark-recapture analysis of photo-identification data is a widely used technique in cetacean research that can provide estimates of abundance and also population parameters e.g. survival and calving rate (Hammond, 1986; Hammond *et al.,* 1990; Wilson *et al.,* 1999).

Strengths and weaknesses of using mark-recapture:

Strengths	Weaknesses
• Typically cheap method of data collection. • Estimates of population size can be based on surveys made in discrete sampling areas within the population's range. • Methods can provide estimates of survival and calving rate as well as abundance.	• Relatively labour-intensive data collection, processing and management. • High probability of capture required for robust estimation of abundance. • Potential disturbance of animals by boats during data collection.

4.4 Spatial modelling of abundance

Spatial modelling, or density surface modelling, is based on fitting a model that describes density along the transect line as a function of habitat covariates and then using the model to predict density over the whole study area based on the value of the covariates in a spatial grid (Hedley *et al.,* 1999). Spatial modelling does not require an equal coverage probability survey design but it does require that the covariates used in the model are well sampled. Covariates commonly used include latitude, longitude, distance from land or ice, depth, bottom topography, sea surface temperature, chlorophyll concentration. Like all model-based estimates, inference from these spatial models is dependent on the models being

appropriate. The method is increasingly being used to estimate abundance from cetacean survey data (Cañadas and Hammond, 2006; 2008; SCANS-II, 2008; CODA, 2009).

Spatial modelling may also be used to study the influence of environmental and anthropogenic factors on distribution, abundance and habitat use (Cañadas and Hammond, 2008; Williams *et al*., 2006), and may thus be a useful tool for investigating the impact of oil and gas exploration and production sound.

Strengths and weaknesses of using spatial modelling in monitoring cetaceans:

Strengths	Weaknesses
• Transects do not need to be systematic or randomly selected. • Can produce abundance estimates for any defined area. • May produce estimates with higher precision than from design-based analysis. • Can be used to provide information on habitat use.	• Covariate data needed for entire survey area. • Methods are 'data-hungry'; caution is needed when applying models to small datasets.

4.5 Recommended methods

The above summary should make it clear that no single method is appropriate for estimating abundance or monitoring of all cetacean species. Determining which technique is best depends on a number of factors including: the objectives (e.g. area or population monitoring; absolute or relative abundance); characteristics of the species (e.g. whether or not it has natural markings suitable for photo-identification; its vocalisation characteristics); characteristics of the population (e.g. size; range; migration pattern); logistical issues (e.g. shipboard *vs* aerial; availability of platforms; multi-species and/or multi-purpose surveys; weather; equipment; accommodation); and available resources.

A useful checklist of steps to follow when considering which method is best is:

- Population monitoring or area use?
- Define population or area
- Define monitoring objectives
- Consider characteristics of the species/population
- Conduct statistical power analysis to find best method to meet objectives
- Consider logistics
- Conduct a cost-benefit analysis

We consider power analysis in Section 5 and cost-benefit analysis in Section 6.

The general suitability of a number of survey methods for different species groups is summarised in Table 4.1.

Table 4.1: General summary of potentially appropriate methods for monitoring the population abundance of different species groups (after SCANS-II, 2008). Whether or not they are actually appropriate will depend on the particular species and other considerations outlined in the text.

Species group	**Visual monitoring**			**Acoustic monitoring**		**Mark-recapture**
	Ship	*Aerial*	*Land*	*Towed*	*Static*	*Photo-ID*
Baleen whales	Yes	Yes	Yes	No	Yes	Yes
Toothed whales	Yes	Yes	No	Yes	Yes	Yes
Dolphins	Yes	Yes	No	Yes	Yes	Yes
Porpoises	Yes	Yes	No	Yes	Yes	No

5. Power to detect trends and cost-benefit analysis

Statistical power refers to the ability to correctly detect a trend in a population if that trend is present (Gerrodette, 1987). Power analysis conducted during the survey planning stage can increase the efficiency of research programmes by estimating the number and precision of samples needed to detect trends of different magnitudes with a given degree of confidence (Gerrodette, 1987; Steidl *et al*., 1997). Alternatively, power analysis used after the data have been collected can reveal the probability of detecting trends given the rate of population change and the number and variability of samples (Gerrodette, 1987).

5.1 Use of power analysis during survey design

Power analysis can reveal whether a particular survey design will generate data with sufficient power to answer the question posed (in this case whether or not population trends can be detected), preventing resources being spent on a monitoring scheme with very poor power (Gerrodette, 1987). Three factors must be considered: the number of samples (in this case abundance estimates), the precision of the estimates; and the probability of making a Type II error (i.e. concluding there is no trend when there actually is).

The higher the number and precision of abundance estimates the higher the likelihood of correctly identifying a trend will be. Ideally they should be spaced regularly in time and should have been obtained using the same method to ensure they have similar coefficients of variation (Gerrodette, 1987). Precision can be increased in some cases by improving methodology and by increasing the sample effort to increase the number of sightings and to decrease the variation arising from measurement errors. Details of the effort and coefficient of variation (CV) of estimates from previous surveys can be used to predict the CV of an estimate from planned survey providing similar methods are used (Holt *et al*., 1987). This involves calculating the proportionality constant linking the sample size to the CV of the abundance estimate of the previous survey; the effect of survey effort on the expected CV can then be investigated for future surveys (Wade and DeMaster, 1999). For this method to

yield accurate predictions variation in the distribution of group sizes encountered is assumed to remain unchanged from previous surveys (Wade and DeMaster, 1999).

The effect of altering survey frequency and interval can also affect power (Wade and DeMaster, 1999). For example, if the rate of change of a population is low, fewer surveys will be required to detect the change if they are conducted every second or third year rather than annually (Gerrodette, 1987).

The probability of making a Type II error can be set to be the same as the probability of making a Type I error (falsely rejecting the null hypothesis of no trend) as was done in a study by Thompson *et al.* (2000) to investigate how many years it would take to detect declines of 1% and 5% in a population of bottlenose dolphins in the Moray Firth, Scotland. However, requiring such a high power to detect a trend (0.95) is unrealistic in most cases, as discussed by Taylor *et al.* (2007). In Task 2 we followed common practice and chose a Type II error probability of 0.2, i.e. a power of 0.8.

5.2 Use of power analysis to inform data interpretation

Retrospective power analysis can assist with interpretation of the results, especially when the null hypothesis, that there is no trend in the population, is not rejected during the analysis (Gerrodette, 1987; Steidl *et al*., 1997). If the data had insufficient power to reject the null hypothesis (because there were too few abundance estimates or they were too variable) it would be a mistake to conclude from that test alone that there had been no significant change in population over the study period (Evans and Hammond, 2004). In this instance, the probability of successfully detecting trends of different magnitudes given the available data can be investigated (Gerrodette, 1987), as can the number of samples or the rate of population change that would have been necessary for the survey to have rejected the null hypothesis (Steidl *et al*., 1997). Retrospective power analysis conducted during the Task 2 analysis showed that only 4 of the 16 final models had sufficient power to detect a halving of the population with probability 0.8 (Quick *et al*., 2008).

5.3 Cost-benefit analysis

The cost of each survey type is an important consideration when designing cetacean surveys and it is important to undertake a cost-benefit analysis as part of the decision making process. The analysis consists of weighing the total cost of each survey method (costs of both data collection and analysis) against the total benefit of the method (the statistical power to detect trends). The survey method likely to provide the most precise abundance estimates for the lowest cost should be the one chosen (SCANS-II 2008). For cost-benefit analysis to be an informative exercise it is imperative that both the costs and the benefits of each survey method are estimated as accurately as possible. The daily costs must be known for hiring the survey platform, a suitable number of observers and other necessary staff and obtaining the equipment required for each survey type. The cost of the survey platform is likely to be the most influential factor in any cost-benefit analysis. It is also important to consider, based on average conditions at the appropriate time of year, how many hours of data are likely to be able to collected using each method, and how many kilometres might be surveyed, in order to work out costs per hour or km for each survey method. Power analysis should be conducted for each of the survey types to allow comparison of the cost of detecting a trend of given magnitude with a specified level of power.

As highlighted in Section 4, the recommended survey method depends on a number of factors, one of which is the species of interest. Cost-benefit analysis is therefore species specific and results from one analysis should not be applied to another non-target species.

5.3.1 Cost-benefit case study: Harbour porpoise in the North Sea

SCANS-II was a large, multi-platform survey conducted in 2005 to estimate the abundance of small cetaceans in European waters (http://biology.st-andrews.ac.uk/scans2/index.html). As part of the SCANS-II project, recommendations were developed for the best monitoring method to detect trends in harbour porpoise abundance and this was done using a cost-benefit approach (SCANS-II, 2008). The costs per hour and per km of searching effort were calculated, including the daily rental of the survey platforms and the hiring of observers. Difference in costs were driven primarily by the cost of hiring the survey platform but also by the data collection method (visual vs acoustic) because of differences in the number of observers required (fewer for acoustic) and the number of survey hours per day (more for acoustic). The cheapest method (per hour or per kilometre) was using a towed hydrophone from a vessel of opportunity. Of the dedicated survey platforms, using a small ship to tow a hydrophone was cheapest. Of the visual surveys, aerial survey was more expensive per hour than a small ship but cheaper than a large ship, but it was the cheapest method per kilometre because of the large number of kilometres that can be surveyed per hour.

To these costs were added the cost of equipment and data analysis, and the cost of each method to detect a 5% trend over 10 years from annual surveys (with a similar level of power) was then calculated and compared. The two cheapest methods to detect this trend utilised platforms of opportunity rather than dedicated survey platforms; however there are logistics issues with platforms of opportunity. Although they are very cheap, they typically cannot be directed to obtain representative survey coverage for cetaceans, so their value is limited for logistic reasons. Ship-based visual surveys were the most expensive options; aerial and ship-based acoustics were cheaper.

5.3.2 Cost-benefit case study: Sperm whales in the Gulf of Mexico

Sperm whale abundance has been estimated in the Gulf of Mexico using a number of different methods; population estimates have been made using visual ship-based line transect methods (Mullin and Fulling, 2004) and aerial surveys (Mullin *et al*., 2004). Dedicated acoustic line transect surveys have not been made in this area but towed hydrophone data have been collected from both large and small vessels during the Sperm Whale Seismic Study (SWSS, http://seawater.tamu.edu/SWSS/) in situations similar to those of a line transect survey to determine likely acoustic detection rates, and these data have been analysed. Photo-identification data, collected from both large and small research vessels have been used to estimate population size using mark recapture methods. Data from each of these sources have been used in a cost/benefit analysis for sperm whales in the Gulf of Mexico.

The CV associated with the population estimate obtained using mark recapture methods was used as a “target CV” and the amount and cost of survey effort required to obtain the same CV using the other methods was explored. For each of the survey types the length of transect that would need to be surveyed (in km) to obtain a target CV of 0.2 was calculated

using equation 1 (Buckland *et al*., 2001). L refers to the length of transect to be surveyed, L_0 is the length of transect surveyed during the pilot study, $CV(\hat{D})$ is the coefficient of variation of the density estimate resulting from the pilot study and $CV_t(\hat{D})$ is the target CV.

$$L - \frac{L_0(CV(\hat{D}))^2}{(CV_t(\hat{D}))^2} \quad (1)$$

The cost of each survey method was calculated based on the charter costs of each of the survey platforms and the research team costs (e.g. observers etc) (Table 5.1). The cost of chartering large vessels and aircraft were taken from SCANS-II, (2008) and converted to US dollars (at an exchange rate of €1=US$1.25). The cost of running a small vessel in the Gulf of Mexico was known from the SWSS project (Gordon pers comm.). The costs of equipment and data analysis, which can vary considerably among survey methods, were not considered.

As acoustic surveys to estimate abundance have not been conducted in the Gulf of Mexico, it was assumed that the CV associated with an abundance estimate derived from acoustic data would be similar to that derived from visual methods. Given that acoustic detection of sperm whales is less affected by factors such as weather and inter-observer variability, this is a conservative assumption. The mean acoustic detection rate of sperm whales from two large vessel surveys and two small boat surveys conducted for the SWSS project in the Gulf of Mexico was 0.0085 detections per km, which is 1.5 times higher than the mean visual detection rate (of 0.0055 groups per km) reported by Mullin and Fulling (2004). This information was used to estimate the number of kilometres that would need to be surveyed acoustically to achieve a CV of 0.2 when estimating abundance of sperm whales from acoustic data.

Different surveys methods are affected to differing extents by factors such as weather conditions and day length. We assumed that visual surveys would be conducted during daylight hours in wind speeds of 17 knots and less, while acoustic surveys would be conducted round the clock and in wind speeds of 28 knots or less. The expected average survey hours per day for a survey in summer (July) were calculated using data on day length at these latitudes and expected wind speed (Defence Mapping Agency, 1983). Applying an expected survey speed (18.5 km hour for boat-based surveys, 185 km hour for aerial surveys) allowed a cost per km to be calculated. Combining this with the number of kilometres required to attain the target CV resulted in a total survey cost to estimate relative abundance using each of the survey methods (Table 5.2).

Conducting a visual survey from a large vessel using a team of 6 observers should allow double-platform methods to be used and, therefore, absolute abundance to be estimated. Abundance estimates resulting from the other survey methods are likely to be systematically biased, by failing to account for animals missed on the track line during the survey and are therefore estimates of relative abundance.

The abundance estimate resulting from the photo-ID survey (Gordon et al., 2008)(Gordon *et al*., 2008b) was based on 834 good quality images of well-marked animals. During the SWSS survey, the large vessel averaged 8.6 good quality images of well-marked animals per day,

meaning 96 large vessel days would be required to obtain 834 of these images. The small survey vessel averaged 5.1 good quality images of well-marked animals per day, meaning 154 small vessel days would be required. These values were used to estimate the cost of estimating abundance with a CV of 0.2 using photo-identification methods from each of the vessel types.

Table 5.1: Comparison of the cost (in US$) of using different survey methods to measure relative abundance of sperm whales in the Gulf of Mexico in summer. This table is adapted from SCANS- II (2008).

Survey method	**Survey platform**	**Daily rental cost**	**No. Observers * daily cost**	**Hours on effort each day**	**US$/hr on effort**	**US$/km on effort**
Ship-based visual	Dedicated. Large ship	6250	6 * 200	12.88	578.33	31.3
	Dedicated. Small ship	1750	2 * 200	12.88	166.90	9.0
	Platform opportunity.	0	2 * 200	12.88	31.05	1.7
Acoustic survey	Dedicated. Large ship	6250	1 * 200	23.76	271.46	14.7
	Dedicated. Small ship	1750	1 * 200	23.76	82.07	4.4
	Platform of opportunity.	0	1 * 200	23.76	8.42	0.5
Aerial survey	Plane	4375	4 * 200	12.88	401.72	2.2
Photo-ID	Dedicated. Large ship	6250	6 * 200	N/A	N/A	N/A
	Dedicated. Small ship	1750	6 * 200	N/A	N/A	N/A

Table 5.2: Comparison of costs of conducting sperm whale surveys using different survey platforms to estimate relative abundance with a CV of 0.2. This table is based on the method used to estimate cost in SCANS-II (2008).

Survey method	**Survey platform**	**Survey effort required**	**Cost per km (US$)**	**Total cost for CV of 0.2 (US$)**
Ship-based visual	Dedicated. Large ship	16,084 km	31.3	502,800
	Dedicated. Small ship	16,084 km	9.0	145,103
	Platform opportunity.	16,084 km	1.7	26,996
Acoustic survey	Dedicated. Large ship	10,723 km	14.7	157,342
	Dedicated. Small ship	10,723 km	4.4	47,568
	Platform of opportunity.	10,723 km	0.5	4,879
Aerial survey	Plane	91,052 km	2.2	200,315
			Cost per day (US$)	
Photo-ID	Dedicated. Large ship	96 days	7450	715,200
	Dedicated. Small ship	154 days	2950	454,300

Tables 5.1 and 5.2 indicate some substantial differences in the costs of different types of survey to assess relative abundance of sperm whales in the Gulf of Mexico. Passive acoustic methods are notably less expensive than visual methods, primarily because survey effort can continue round the clock and in poorer weather conditions and the overall detection rate is slightly higher. Smaller vessels are less expensive to run than larger vessels and can provide very effective platforms for acoustic surveys, but they may not be adequate platforms for visual surveys. Note that although we have included costs for a small boat visual survey we are not aware of any successful visual surveys having been conducted from small vessels for sperm whales in offshore waters. Platform of opportunity surveys do not have associated vessel costs and so are extremely cost effective, provided of course they can provide good coverage of the survey area and can accommodate a research team and equipment. This is only likely to be the case in special cases, for example where a vessel is being used to conduct a fisheries survey in the area. Photo-identification surveys from small boats are more expensive than large boat acoustic surveys but less expensive than large boat visual surveys. As described in Section 6 however, an advantage of photo-identification studies is that they can provide information on many other population parameters that are relevant to understanding anthropogenic effects on marine mammals.

6. Population Consequences of Acoustic Disturbance framework

A focus on measuring changes in abundance may not be the only way to address the issue of monitoring, understanding and managing biologically significant effects of noise on marine mammals. This report has shown that densities and population abundance can be measured, but with poor precision. This means that very high levels of survey effort will be required to be able to show population trends within a reasonable amount of time. Even if these can be revealed, "problems" would not be evident until after they had occurred and consequently any management measures that might be necessary would be likely to be more draconian and onerous than those required, had the issue been revealed earlier. In addition, with so many factors affecting population trajectories, cause and effect relationships would be extremely difficult to establish. An alternative method of investigating the effects of noise on cetaceans exists in the form of a conceptual framework, known as the Population Consequences of Acoustic Disturbance (PCAD) model that was developed by the National Research Council (NRC) in 2005. This describes 5 stages that relate acoustic disturbance to population effects, linked by 4 transfer functions (NRC, 2005) (Figure 6.1), all of which vary in their measurability. For example, a complete lack of information about transfer function 3 – how changes to life functions (such as feeding and breeding) lead to changes in stage-specific vital rates – suggests that it will be difficult to determine how behavioural changes would translate to population effects without making measurements at some of the intermediary stages. However, the PCAD framework provides a good outline of the data that can be collected, how the information fits together and the processes that need to be studied to allow this method to elucidate the population effects of acoustic disturbance.

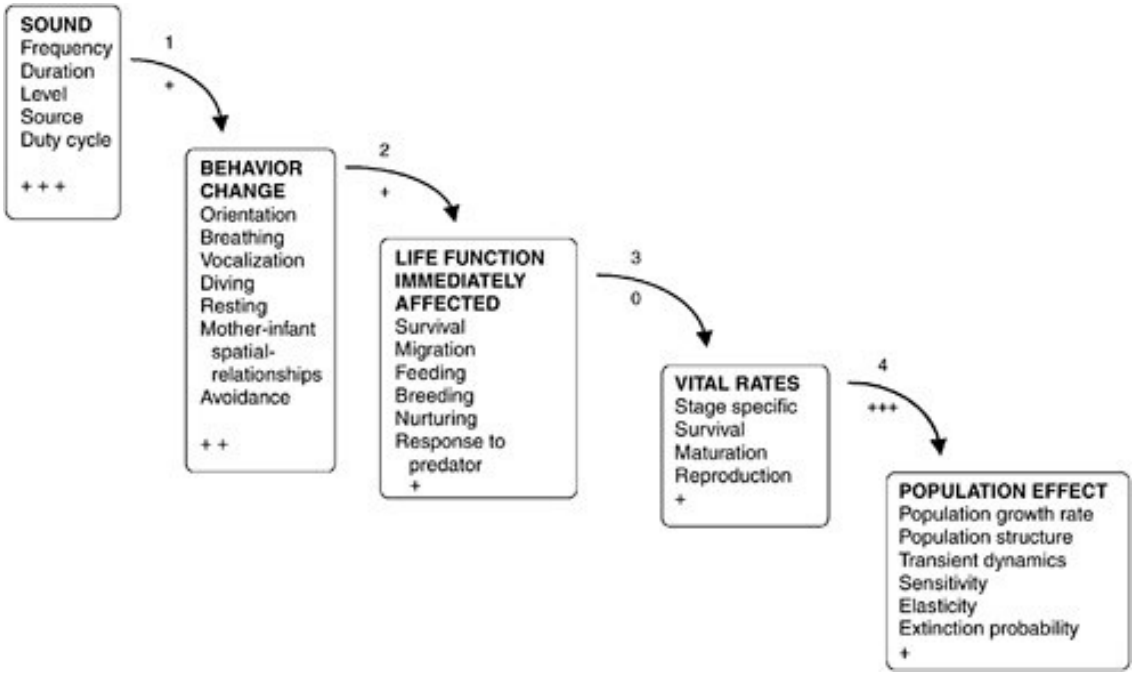

Figure 6.1: The stages and transfer functions of the Population Consequences of Acoustic Disturbance model (National Research Council, 2005).

For any population where there is likely to be concern about disturbance it will be sensible to establish a detailed population model (which will to some extent reflect the PCADs diagram) and to carry out a monitoring program that can provide data to address the key knowledge gaps in this model. A population modelling approach has a number of potential advantages. It provides a framework in which the biological significance of behavioural change and other impacts can be considered. It can be used in a sensitivity analysis to clarify the key unknown parameters and remaining uncertainties, a process that will be useful in prioritising research effort and spending. It provides a structure in which many different pieces of biological information can contribute to an overall understanding of the dynamics of a population leading to more precise measurements of population change, and allows predictions to be made and the effectiveness of different management scenarios to be explored. Approaches to modelling in this context are reviewed by Cabrelli *et al*. (2009) and there are many approaches that have been developed for other species and scenarios that are applicable here.

Cabrelli *et al*. (2009) revisit the PCADs model and suggest some simplifications and restructuring. Survival, which is already represented as a vital rate, is removed from the Life Function category and "Responses to Predator" is merged with the life function "Migration", leaving five life functions; feeding, mating, nurturing, physiology and migration. They argue that the links between these life functions and vital rates are in fact fairly clear. Reduced mating results in fewer offspring, disruption of nurturing will result in increased calf mortality risk and physiological disruption can, in the case of beaked whales at least, result in mortality (Figure 6.2).

Reduced feeding will result in both reduced energy intake and increased energy expenditure (as more effort is put into foraging) which will lead to reduced condition and growth, a lower level of nurturing and result in fewer offspring and greater adult mortality (Figure 6.3).

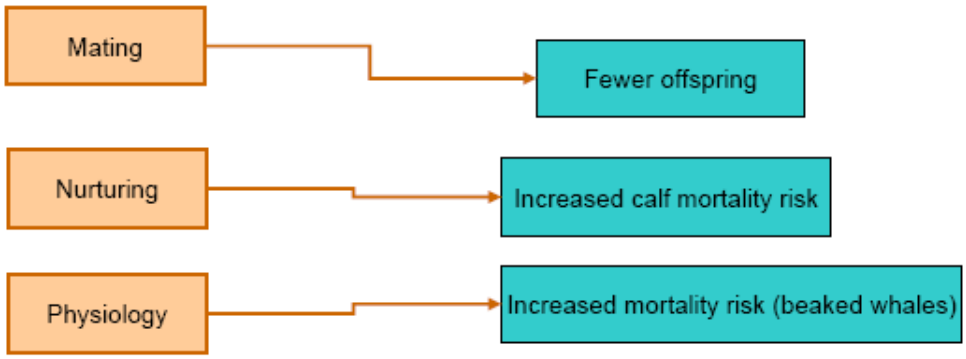

Figure 6.2: Hypothetical links between changes in three of the life functions in the PCAD model and vital rates. Figure taken from Cabrelli *et al*. (2009).

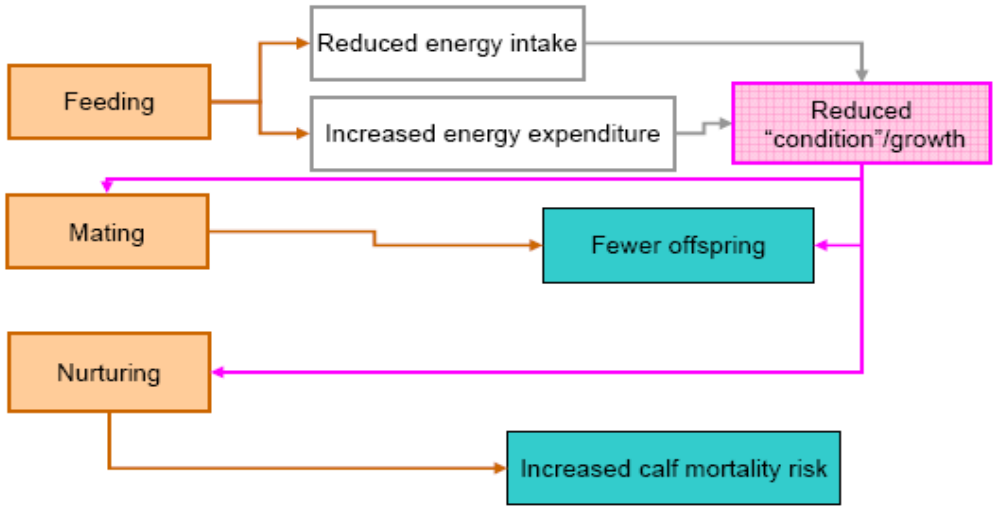

Figure 6.3: The consequences of changes in feeding behaviour on vital rates within the PCAD model. A. Direct effects of changes in feeding behaviour on offspring production. B. Indirect effects of changes in feeding behaviour on offspring production and calf mortality via mating and nurturing. Figure taken from Cabrelli *et al*. (2009).

In recent decades many advances have been made in studying marine mammals at sea making it feasible to measure much of the data required for the PCADs approach or to inform a detailed population model. As an example we outline how these data could be practically collected in the field from sperm whales in an area of interest to the E&P industry such as the Gulf of Mexico. The following sections give examples of the data required for four of the PCAD stages. Where possible the examples given relate directly to sperm whales in the Gulf of Mexico, but examples from sperm whales in other areas or other species are given where appropriate.

6.1 Collecting data for a PCAD based Population Model, case study: Sperm whales in the Gulf of Mexico

6.1.1 Individual Identification and Characterisation

6.1.1.1 Identification

Being able to identify animals individually in the field is important for mark-recapture methodologies which provide information on population size, vital rates and movements (e.g. see section 5.3), and for the application of many other research techniques, (see following sections). When linked with an assessment of an individual's characteristics (e.g. age, sex, body size) it is possible to use individual-based population models and thus avoid the "Ecological fallacy" of using models that assume that all member of a population respond equally to external factors (Cabrelli *et al.,* 2009).

Sperm whales can be readily identified from photographs of their tail flukes, an approach which has been used since the early 1980s (Whitehead and Gordon, 1984; Gordon, 1987b;

Review in Whitehead, 2003). The Gulf of Mexico is an ideal location for this approach, with around 80% of sperm whales being sufficiently well marked to allow long term re-identification (Gordon *et al*., 2008a).

Whales can also be identified individually using genetic fingerprinting methods which require a biopsy to be taken every time a whale is encountered. This technique raises ethical and permitting issues and requires more expensive analysis, but has been used effectively for mark recapture studies of humpback whales (Palsboll *et al*., 1997; Smith *et al.,* 1999), and to establish the gender of 58 of the 392 individuals in the Gulf of Mexico Sperm whale photo-id catalogue (Gordon unpublished data).

6.1.1.2 Characterising Individuals

Gender

Sperm whales are the most sexually dimorphic of cetaceans. Mature males are larger and more obvious in the field from their size and proportionally larger heads, but immature males and females are difficult to tell apart. Observations of suckling or the presence of a callus on the dorsal fin (which is more common in mature females (Kasuya and Ohsumi, 1966)) can also be used to determine gender.

Age and Body Size

In conjunction with information on gender, body size provides an indication of age. Body size can be measured photogrammetrically (Gordon, 1990; Jaquet, 2006) and from acoustic analysis (Gordon, 1991; Rhinelander and Dawson, 2004). An example from sperm whales in the Galapagos, illustrates how the former can be used to derive population and growth parameters (Waters and Whitehead 1990).

6.1.2 Behaviour change

Many researchers have made visual observations of surface behaviour and, in the case of sperm whales passive acoustics can be used to track groups over several days. Investigations of the effects of whale watching have assessed behavioural changes (Gordon *et al*., 1992; Richter *et al*., 2006), but observations made at the surface may shed little light on behaviours occurring underwater. Because sperm whales are a highly vocal species, passive acoustic monitoring can be used to reveal underwater vocal behaviour. Additionally, improved acoustic analysis tools, such as PAMGUARD make it easier to quantify vocal behaviour (Gillespie *et al*., 2008). Furthermore, more sophisticated techniques allied with increased processing power have made it easier to determine the underwater movements and dive behaviour of whales from hydrophone arrays of varying size. For example, Thode (2004, 2005), describes methods that utilise surface and bottom reflections to increase the effective dimensions of the hydrophone array, to track the underwater movements and dive behaviour of sperm whales. This method has now been incorporated into PAMGUARD making it more accessible to other researchers (http://www.pamguard.org/home.shtml).

Mother calf interactions and “baby sitting” by other group members, is a particularly important behaviour in sperm whales. Because the diving capabilities of sperm whale calves are limited, this behaviour often occurs close to the surface allowing easier observation and measurement. For example Gero (2007) provides detailed information on calf behaviour

based on direct surface and underwater observations of a well-studied group of sperm whales.

Telemetry offers another method for measuring underwater behaviour and significant advances in our understanding of sperm whale behaviour have been made in recent years through use of the DTag. This tag can be attached to sperm whales with suction cups, and records detailed information on depth, orientation and acoustics. The first applications of this device were in the Gulf of Mexico during a study of the effects of acoustic disturbance. Dtags have yielded much new information on underwater movements and vocal behaviour (Miller *et al*., 2004a; 2004b; Watwood *et al*., 2006) and have also been used to measure responses of sperm whales to airgun pulses (Tyack *et al*., 2008) during controlled exposure experiments.

A consequence of disturbance and behavioural change which may have biologically significant effects is stress (Cowan and Curry, 2008), and there is some potential for measuring biochemical markers for stress in biopsy samples (Forney *et al*., 2002).

6.1.3 Life Functions

6.1.3.1 Movements and Migrations

Once individuals can be identified in the field their movements can be measured over a range of time scales. Whitehead (2001) developed methods for analysing long term movements using mark recapture data from sperm whales and these methods have been made readily available in the SOCPROG program. In the Gulf of Mexico, Gordon *et al*. (2008a) analysed photo-id data to provide information on movements of sperm whales over time scales from a single dive cycle (~1 hour) to several years.
An alternative method for measuring movements is to use satellite telemetry, which has been successfully applied to sperm whales in the Gulf of Mexico (Mate *et al*., 2007). This revealed that most females and young remained within the Northern Gulf while some males ventured further into the North Atlantic. Photo-id and satellite telemetry have proven to be contrasting but complimentary methods. Photo-id has better spatial precision, is often cheaper, can provide data on a greater number of individuals and doesn't involve invasive tagging. However, it can only provide data from the study area and does not provide the same level of detail from individual animals as telemetry.

6.1.3.2 Feeding

Sperm whales feed at depths of hundreds or thousands of meters, and the process of feeding has never been directly observed. Several behaviours that might be useful proxies for feeding rates can be used, for example defecation rates can be measured reliably because sperm whales defecate at the surface. Defecation rates during tracking periods have been shown to correlate negatively with directivity of movement patterns over the same period. There are theoretical reasons for proposing that movements will be less direct when whales are feeding in patches of prey (Whitehead, 2003). Another potential proxy are vocalisations called "creaks" which sperm whales produce between bouts or regular clicking during their long feeding dives. These have similar characteristics to echolocation runs made by some other odontocetes. It has long been hypothesised that these are made during prey capture (Gordon, 1987a) and recent detailed observations using DTags strongly support this

(Miller *et al*., 2004a). Changes in creak rates have been observed in response to both whale watching activities (Gordon *et al*., 1992) and exposure to airgun noise (Tyack *et al.*, 2008).

If feeding conditions deteriorate then animals may need to increase their foraging effort. In sperm whales this might be manifested as a change in dive behaviour and/or a decrease in the proportion of time spent resting and socialising. Sperm whale groups cycle through two quite distinct behavioural phases. During the "foraging" phase groups split into small clusters of one or two individuals, spread out over a considerable area and make repeated long dives seeming to return to the surface only to recover oxygen supplies. During resting/socialising periods they come together into larger, tighter clusters that remain at or near the surface for periods of hours.

6.1.3.3 Body Condition

Changes in feeding rate may lead to changes in body condition, in particular the extent of fat reserves. The relative "fatness" of whales at different stages in a season was often clearly evident to whalers from visual observation and it's possible that some measure of this could be made using photography/photogrammetry, as has been done for gray whales (Perryman and Lynn, 2002) and right whales (Pettis *et al*., 2004). Because the acoustic fats in the sperm whale head cannot be metabolised the head becomes more prominent in starving sperm whales providing an additional cue for condition. Moore *et al*. (2002) used an ultrasound probe to measure blubber thickness on right whales, a species that is particularly easy to approach, but no attempts to adapt this technique to other species have been made. Because fat is lighter than other body tissues, changes in proportion of fat results in changes in buoyancy, which may be possible to measure with surface photogrammetry. However, it can also be revealed by detailed analysis of underwater swimming behaviour from telemetry. Biuw (2003) analysed a long term satellite telemetry dataset from southern elephant seals to determine changes in buoyancy and infer the feeding success of individual seal in different regions. Using DTag data Miller *et al*. (2004b) were able to measure the buoyancy of individual sperm whales. An extended series of such data from known individuals would allow changes in buoyancy (and fat reserves) to be monitored.

6.1.3.4 Growth

Repeated measurements of known individuals will allow growth rates to be measured and Pavan *et al*. (1998) claim to have measured growth between years in an individual male sperm whale from the Mediterranean. Furthermore, acoustic methods of measuring length potentially provide very high levels of precision for sperm whales (Gordon, 1991).

6.1.4 Vital Rates and Population Parameters

6.1.4.1 Stage Specific Survival

Survival rate can be estimated using mark recapture analysis of individual animals and this has been successfully done for a number of large whale species (bowhead - Zeh *et al*., 2002; humpback - Mizroch *et al*., 2004; gray - Bradford *et al*., 2006; blue - Ramp *et al*., 2006). Programs MARK and SOCPROG can be used for analysis.

6.1.4.2 Maturation, age at first reproduction

Long term photo-identification studies can provide information on maturation, age at first reproduction and birth intervals (e.g. Gabriele *et al*., 2007). These will be particularly valuable if linked with hormonal analysis of biopsy or faecal samples, though more development is required in this area (Smit 2003).

6.1.4.3 Reproduction

Calf production can be estimated by measuring the proportion of calves encountered at sea or by using photo-id data to construct the calving histories of individual females (e.g. Barlow and Clapham, 1997). However, methodologies will need to ensure that sampling is not biased with respect to calves, allows for the duration of the calving period and attempts to measure and account for calf mortality.

6.1.5 Population Effects

6.1.5.1 Population Size and Trends

Methods for measuring population size and trends for sperm whales are given in section 5.3.2. Mark-recapture analysis using photo-id data can be an effective method for estimating population size and trends and is compatible with many of the other data collection requirements outlined in this section.

6.1.5.2 Population Structure/ Social organisation

Sperm whales have one of the most complex social structures of any of the great whales and it will be important to incorporate information on their social structure into population models used for management. Sperm whale social organisation has been studied using photo-identification and the most detailed studies have been completed by Hal Whitehead and colleagues (Whitehead, 2003). This work has shown that females and immature males live in long term stable units of around 12 individuals which are often based on matrilines. Recently, a higher level of culturally mediated organisation has been discovered, based on "clans". Clans are made up of many social groups that exclusively associate with other members of the same clan (Rendell and Whitehead, 2003). Clan members have similar vocal repertoires and also share traits affecting their fitness (Marcoux *et al*., 2007; Whitehead and Rendell, 2004). Clan structure was first revealed through analysis of sperm whale codas (stereotyped vocalisations believed to be used for communication) and this is still the most efficient means of elucidating population structure at this level. Sperm whales produce coda vocalisations during periods of rest and socialising which occur approximately once per day, and these recordings can be readily collected in the course of fieldwork in which groups are tracked for periods of hours to days.

6.1.6 Conclusion

Much of the information required to parameterise a comprehensive population model based on the PCAD structure can be collected from live sperm whales using existing research techniques. Individually based research and photo-identification techniques are central to providing much of this data directly and many of the other data requirements could be addressed in the course of photo-identification studies with relatively little additional effort or expenditure.

7. Overall Conclusions

One of the most important findings of this project has been the huge variation in the number and size of systematic cetacean surveys conducted in different regions of the world. In several of the AORs, notably AOR_001 (West Africa), AOR_002 (East Africa) and AOR_003 (Australia), very few systematic cetacean surveys have been conducted (Jewell *et al*., 2008). Such an obvious absence of knowledge of cetacean abundance in these areas immediately precludes the possibility of detecting temporal trends in cetacean populations, and therefore the investigation of factors responsible for changes in population. Further research within these areas is of great importance if the conservation status of the species present in these areas is to be elucidated. In addition to this, it was found that even in the areas where systematic surveys had been conducted, there were very few species with sufficient numbers of density estimates to allow trends to be detected and the poor precision of those density estimates resulted in poor statistical power to detect trends over time (Quick *et al*., 2008). To maximise the chances of detecting trends, precise density estimates from comparable surveys in multiple years are required. Environmental and anthropogenic factors that could influence cetacean population growth rates were reviewed, and it was concluded that anthropogenic removal of individuals (for example, through whaling of large whale species and incidental capture of delphinids) was likely to have the most impact (Murphy, 2008). However, these influential factors are often difficult to quantify and, therefore, to incorporate into models of population change. Environmental conditions, and levels of seismic survey activity, were the covariates most frequently retained in the models of the rate of population change for the four most data rich species (Jewell *et al*., 2009). However, the poor precision of the cetacean density estimates on which the models of population change were based resulted in high levels of uncertainty being associated with the model output, and the results should only be considered exploratory. Further to this, it is not known whether the seismic data used were representative of the actual activity in the areas of relevance. To enhance the models further in this respect, better details of seismic survey activity are needed.

Further systematic cetacean surveys are required, particularly in areas where no or little baseline data exist, to allow identification of changes in the size of the population in the future. If the primary purpose of the surveys is to detect population trends this should be reflected in the survey design and the aim should be to estimate density with high precision. In areas where systematic surveys have been conducted previously, it is important that comparable surveys continue to be conducted to provide time series of data suitable for looking at changes in the population over time. It is also recommended that, where possible, additional data on behavioural responses and life functions (such as rates of breeding, feeding and survival) of cetaceans are gathered as they may shed light on how acoustic disturbance can result in population effects, as the PCAD model framework seeks to do.

7.1 Potential further work/publications by SMRU Ltd

There are a number of ways in which the work completed during this project can be extended and several publications are being discussed. The modelling of global cetacean density carried out during Task 2 is currently being refined and prepared for publication. This publication would cover the modelling approach used and highlight existing gaps in the

knowledge required for assessing trends in cetacean abundance. The role of power analysis during survey design is also being considered for further analysis and possible publication. The ability to have enough power to detect changes with a reasonable level of certainty is vital for future management decisions. This publication would address issues associated with using power analysis in survey designs for cetaceans. The review of anthropogenic and environmental factors that influence cetacean population growth rates may be revisited and submitted as a review publication. More extensive mapping of seismic surveys during oil and gas exploration and production, including generation of density surface maps of the number of shots fired during seismic surveys on a global scale, would be highly informative. Visual displays of the overlap between seismic surveys and systematic cetacean abundance surveys would be of particular interest to marine mammal scientists. This would also highlight the gaps in knowledge of cetacean abundance and allow more dedicated research effort in areas of poor survey coverage but high seismic activity Finally, the modelling process utilised during Task 4 to relate changes in cetacean density to environmental, seismic and anthropogenic mortality covariates may be developed further with a view to publishing the findings. This publication would further describe the modelling approach developed during the Task 4 analysis and outline the assumptions and difficulties with using this approach.

Overall, this project has provided an in depth look at the published data available for estimating abundance of cetaceans and at the possibility of developing a modelling approach to assess impacts of E&P noise on cetacean stock trends. It has drawn on a number of data sources and methodologies to explore analysis techniques and review the existing data for global cetacean abundance. Although a number of interesting results have been determined, it is clear that large gaps in data and knowledge exist and further data collection would provide a better understanding of what is a complex problem.

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