Scientific Advice on Matters Related to the Management of Seal Populations: 2010

Contents

Scientific advice

Background
Under the Conservation of Seals Act 1970 and the Marine (Scotland) Act 2010, the Natural Environment Research Council (NERC) has a duty to provide scientific advice to government on matters related to the management of seal populations. NERC has appointed a Special Committee on Seals (SCOS) to formulate this advice so that it may discharge this statutory duty. Terms of Reference for SCOS and its current membership are given in ANNEX I.

Formal advice is given annually based on the latest scientific information provided to SCOS by the Sea Mammal Research Unit (SMRU), a NERC Collaborative Centre at the University of St Andrews. SMRU also provides government with scientific reviews of applications for licences to shoot seals, information and advice in response to parliamentary questions and correspondence, and responds on behalf of NERC to questions raised by government departments about the management of marine mammals in general.

This report provides scientific advice on matters related to the management of seal populations for the year 2010. It begins with some general information on British seals, gives information on their current status, and addresses specific questions raised by the Scottish Government Marine Directorate (SGMD) and the Department of the Environment, Food and Rural Affairs (DEFRA). Appended to the main report are briefing papers, used by SCOS, which provide additional scientific background for the advice.

As with all publicly funded bodies, SMRU’s long term funding prospects are at present unclear and will depend to a large extent on NERC’s response to the Government’s comprehensive spending review. The long term funding picture and its implications for SMRU’s ability to carry out its monitoring functions should be clear by the time of SCOS 2011 (1 September).

General information on British seals
Two species of seal live and breed in UK waters: grey seals (*Halichoerus grypus*) and harbour (also called common) seals (*Phoca vitulina*). Grey seals only occur in the North Atlantic, Barents and Baltic Sea with their main concentrations on the east coast of Canada and United States of America and in north-west Europe. Harbour seals have a circumpolar distribution in the Northern Hemisphere and are divided into five sub-species. The population in European waters represents one subspecies (*Phoca vitulina vitulina*). Other species occasionally occur in UK coastal waters, including ringed seals (*Phoca hispida*), harp seals (*Phoca groenlandica*), bearded seals (*Erignathus barbatus*) and hooded seals (*Cystophora crystata*) all of which are Arctic species.

Grey seals
Grey seals (*Halichoerus grypus*)
Grey seals are the larger of the two resident UK seal species. Adult males can weigh over 300kg while the females weigh around 150-200kg. Grey seals are long-lived animals. Males may live for over 20 years and begin to breed from about age 10. Females often live for over 30 years and begin to breed at about age 5.

They are generalists, feeding mainly on the sea bed at depths up to 100m although they are probably capable of feeding at all the depths found across the UK continental shelf. Their diet varies both seasonally and geographically but comprises mainly small demersal fish species, i.e. fish that live on or close to the seabed. In the UK, their diet is composed primarily of sandeels, whitefish (cod, haddock, whiting, ling), and flatfish (plaice, sole, flounder, dab). Food requirements depend on the size of the seal and fat content (oiliness) of the prey, but an average consumption estimate is 7 kg of cod or 4 kg of sandeels per seal per day.

Grey seals forage in the open sea and return regularly to haul out on land where they rest, moult and breed. They may range widely to forage and frequently travel over 100km between haulout sites. Foraging trips can last anywhere between 1 and 30 days. Compared with other times of the year, grey seals in the UK spend longer hauled out during their annual moult (between December and April) and during their breeding season (between August and December). Tracking of individual seals has shown that they can feed up to several hundred kilometres offshore although most foraging probably occurs within 100km of a haulout site. Individual grey seals based at a specific haulout site often make repeated trips to the same region offshore, but will occasionally move to a new haulout site and begin foraging in a new region. Movements of grey seals between haulout sites in the North Sea and the Outer Hebrides have been recorded.

There are two centres of grey seal abundance in the North Atlantic; one in Canada and the north-east USA, centred on Nova Scotia and the Gulf of St Lawrence and the other around the coast of the UK especially in Scottish coastal waters. Populations in Canada, USA, UK and the Baltic are increasing, although numbers are still relatively low in the Baltic where the population was drastically reduced by human exploitation and reproductive failure probably due to pollution. There are clear indications of a slowing down in population growth in UK and Canadian populations in recent years.

Approximately 45% of the world's grey seals breed in the UK and 90% of these breed at colonies in Scotland with the main concentrations in the Outer Hebrides and in Orkney. There are also breeding colonies in Shetland, on the north and east coasts of mainland Britain and in SW England and Wales. Although the number of pups throughout Britain has grown steadily since the 1960s when records began, there is clear evidence that the growth is levelling off. The numbers born in the Hebrides have remained approximately constant since 1992 and growth has been levelling off in Orkney and possibly at some colonies in the northern North Sea

In the UK, grey seals typically breed on remote uninhabited islands or coasts and in small numbers in caves. Preferred breeding locations allow females with young pups to move inland away from busy beaches and storm surges. Seals breeding on exposed, cliff-backed beaches and in caves may have limited opportunity to avoid storm surges and may experience higher levels of pup mortality as a result. Breeding colonies vary considerably in size; at the smallest only a handful of pups are born, while at the biggest, over 5,000 pups are born annually. In general grey seals are highly sensitive to disturbance by humans hence their preference for remote breeding sites. However, at one UK mainland colony at Donna Nook in Lincolnshire, seals have become habituated to human disturbance and over 70,000 people visit this colony during the breeding season with no apparent impact on the breeding seals.

UK grey seals breed in the autumn, but there is a clockwise cline in the mean birth date around the UK. The majority of pups in SW Britain are born between August and September, in north and west Scotland pupping occurs mainly between September and late November and eastern England pupping occurs mainly between early November to mid December.

Female grey seals give birth to a single white coated pup which they suckle for 17 to 23 days. Pups moult their white natal coat (also called "lanugo") around the time of weaning and then remain on the breeding colony for up to two weeks before going to sea. Mating occurs at the end of lactation and then adult females depart to sea and provide no further parental care. In general, female grey seals return to the same colony to breed in successive years and often breed at the colony in which they were born. Grey seals have a polygynous breeding system, with dominant males monopolising access to females as they come into oestrus. The degree of polygyny varies regionally and in relation to the breeding habitat. Males breeding on dense, open colonies are able to restrict access to a larger number of females (especially where they congregate around pools) than males breeding in sparse colonies or those with restricted breeding space, such as in caves or on cliff-backed beaches.

Harbour seals

Harbour seals (*Phoca vitulina*) are found around the coasts of the North Atlantic and North Pacific from the subtropics to the Arctic. Five subspecies of harbour seal are recognized. The European subspecies, *Phoca vitulina vitulina*, ranges from northern France in the south, to Iceland in the west, to Svalbard in the north and to the Baltic Sea in the east. The largest population of harbour seals in Europe is in the Wadden Sea.

Harbour seals come ashore in sheltered waters, typically on sandbanks and in estuaries, but also in rocky areas. They give birth to their pups in June and July and moult in August. At these, as well as other times of the year, harbour seals haul out on land regularly in a pattern that is often related to the tidal cycle. Harbour seal pups are born having shed their white coat and can swim almost immediately.

Adult harbour seals typically weigh 80-100 kg. Males are slightly larger than females. Like grey seals, harbour seals are long-lived with individuals living up to 20-30 years. Harbour seals normally feed within 40-50 km around their haul out sites. They take a wide variety of prey including sandeels, gadoids, herring and sprat, flatfish, octopus and squid. Diet varies seasonally and from region to region. Because of their smaller size, harbour seals eat less food than grey seals; 3-5 kg per seal per day depending on the prey species.

Approximately 30% of European harbour seals are found in the UK; this proportion has declined from approximately 40% in 2002. Harbour seals are widespread around the west coast of Scotland and throughout the Hebrides and Northern Isles. On the east coast, their distribution is more restricted with concentrations in the major estuaries of the Thames, The Wash, Firth of Tay and the Moray Firth. Scotland holds approximately 85% of the UK harbour seal population, with 11% in England and 4% in Northern Ireland.

The population along the east coast of England (mainly in The Wash) was reduced by 52% following the 1988 phocine distemper virus (PDV) epidemic. A second epidemic in 2002 resulted in a decline of 22% in The Wash, but had limited impact elsewhere in Britain. Counts in the Wash and eastern England have failed to demonstrate any recovery since the epidemic, in contrast to the adjacent European colonies which have experienced rapid growth since 2002.

Major declines have now been documented in harbour seal populations around Scotland with declines since 2000 of 66% in Orkney, 50% in Shetland, 36% in the Outer Hebrides, 46% in the Moray Firth and 84% in the Firth of Tay. These declines are not thought to be linked to the 2002 PDV epidemic that seems to have had little effect in Scotland.

Historical status

We have little information on the historical status of seals in UK waters. Remains have been found in some of the earliest human settlements in Scotland and they were routinely harvested for meat, skins and oil until the early 1900s. There are no reliable records of historical population size. The Grey Seal (Protection) Act 1914, providing the first legal protection for any mammal in the UK because of a perception that there was a need to protect seals. Harbour seals were heavily exploited mainly for pup skins until the early 1970s in Shetland and The Wash. Grey seal pups were taken in Orkney until the early 1980s, partly for commercial exploitation and partly as a population control measure. Large scale culls of grey seals in the North Sea, Orkney and Hebrides were carried out in the 1960s and 1970s as population control measures.

Grey seal pup production monitoring started in the late 1950s and early 1960s and numbers have increased consistently since. In recent years, there has been a significant reduction in the rate of increase.

Boat surveys of harbour seals in Scotland in the 1970s showed numbers to be considerably lower than in recent aerial surveys, which started in the late 1980s, but it is not possible to distinguish the apparent change in numbers from the effects of more efficient counting methods. After harvesting ended in the early 1970s, regular surveys of English harbour seal populations indicated a gradual recovery, punctuated by two major reductions due to PDV epidemics in 1988 and 2002 respectively.

Legislation protecting seals

In the UK seals are protected under the Conservation of Seals Act 1970 (England, Scotland and Wales) and The Wildlife (Northern Ireland) Order 1985. In Scotland, the legislation has been superceded by the new Marine Bill (Scotland). The Wildlife (Northern Ireland) Order is also currently under review.

The Conservation of Seals Act prohibits taking seals during a close season (01/09 to 31/12 for grey seals and 01/06 to 31/08 for harbour seals) except under licence. The act allows for specific Conservation Orders to extend the close season to protect vulnerable populations. At present, after consultation with NERC, three such orders have been established providing year round protection to grey and harbour seals on the east coast of England and in the Moray Firth and to harbour seals in Shetland, Orkney and the east coast of Scotland between Stonehaven and Dunbar (effectively protecting all the main concentrations of harbour seals along the east coasts of Scotland and England).

The Marine (Scotland) Act 2010 (Section 6) prohibits the taking of seals except under licence. Licences can be granted for the protection of fisheries, for scientific and welfare reasons and for the protection of aquaculture activities. In addition, in Scotland it is now an offence to disturb seals at designated haulout sites. NERC provide advice on all licence applications and haulout designations.

Both grey and harbour seals are listed in Annex II of the EU Habitats Directive, requiring specific areas to be designated for their protection. To date, 16 Special Areas of Conservation (SACs) have been designated specifically for seals. Seals are features of qualifying interest in seven additional SACs.

1. What are the latest estimates of the number of seals in UK waters?

Current status of British grey seals

- UK grey seal pup production in 2009 was estimated to be 47,540
- Pup production remains stable in Orkney and the Hebrides
- Pup production continues to increase rapidly in the North Sea
- A new independent estimate of population size allows us to select between competing population estimation models
- Total grey seal population at the start of the 2009 breeding season is estimated to have been 106,200 (95% CI 82,000-138,700)

Variation in the number of pups born in a seal population can be used as an indicator of change in the size of the population and with sufficient understanding of population dynamics may allow estimation of total numbers of seals. Each year, SMRU conducts aerial surveys of the major grey seal breeding colonies in Britain to determine the number of pups born (pup production). The annually surveyed sites account for approximately 90% of all grey seal pups born throughout Britain. The remaining sites producing around 10% of the pups are surveyed less frequently. The total number of seals associated with the regularly surveyed sites is estimated by applying a population model to the estimates of pup production. Estimates of the total number of seals at other breeding colonies that are surveyed less frequently are then added in to give an estimate of the total British grey seal population. Further details are given in SCOS-BP 10/1 and SCOS-BP 10/2.

Pup production

The total number of pups born in 2009 at all annually surveyed colonies was estimated to be 42,296. Regional estimates were 3,396 in the Inner Hebrides, 12,113 in the Outer Hebrides, 19,150 in Orkney and 7,637 at North Sea colonies (including Isle of May, Fast Castle, Donna Nook and Farne Islands). A further 5,247 pups were estimated to have been born at other scattered colonies throughout Scotland, Northern Ireland, South-west England and Wales.

1.1 Trends in pup production

Overall, there has been a continual increase in pup production since regular surveys began in the 1960s. In both the Inner and Outer Hebrides, the rate of increase declined in the early 1990s and production has been relatively constant since the mid 1990s. The rate of increase in Orkney has declined since 2000 and pup production has been relatively constant since 2004. Overall pup production at colonies in the North Sea continues to increase exponentially, although it appears to have levelled off at the Isle of May and Farne Islands and the increase is due to expansion of newer colonies on the mainland coasts in Berwickshire and East Anglia. The differences in pup production between 2008 and 2009 are shown in Table 1. Total pup production at annually monitored colonies increased by 1.9% between 2008 and 2009, in contrast to the 6.9% increase between 2007 and 2008. Technical problems with the camera mount resulted in minimal surveys at the start of the survey season. Rather than extrapolating, the 2008 production estimates were used for all colonies in the Inner Hebrides and for seven out of 15 colonies in the Outer Hebrides.

The relatively large and widespread increase in 2008 was not evident in the Hebrides or Orkney in 2009. However, the North Sea colonies again increased by between 2% and 21%, with an overall increase of 15%.

On a longer timescale, during the most recent 5-year period (2004-2009) the total pup production for all annually monitored colonies in the Inner and Outer Hebrides and Orkney has remained almost constant. However, as previously reported, pup production at colonies in the North Sea continued to increase at around 8.7% p.a. over the same 5 year period.

Table 1: Grey seal pup production estimates for the main colonies surveyed in 2008

Location	2009 pup production	Change in pup production from 2008-2009	Average annual change in pup production from 2004-2009
Inner Hebrides	**3,400**	n.a.	+0.1%
Outer Hebrides	**12,110**	-4.7%	-0.3%
Orkney	**19,150**	+2.0%	0.0%
Isle of May + Fast Castle	**4,050**	+21.0%	+9.2%
All other colonies incl Shetland & mainland	3,250 **	-5.5%	
Total (Scotland)	41,950	+0.8%*	+0.4%*
Donna Nook +East Anglia	**2,240**	+14.9%	+15.8%
Farne Islands	**1,350**	+2.3%	+3.5%
SW England (last surveyed 1994)	250		
Wales ***	1,650		
Total (England & Wales)	5,490	+6.3%*	+8.8%*
Northern Ireland	100		
Total (UK)	47,540	+1.4%*	+1.0%*

*Average annual change in pup production calculated from annually monitored sites only

** estimate from several surveys in Shetland to provide most up-to-date estimate

*** estimate from indicator sites in 2004-05, multiplier derived from 1994 synoptic surveys

1.2 Population size

Because pup production is used to estimate the total size of the grey seal population, the estimate of total population alive at the start of the breeding season depends critically on the factors responsible for the recent deceleration in pup production.

Pup production can be used to estimate total population size with appropriate estimates of pup and non pup survival and age specific fecundity rates. Until the late 1990s the population grew exponentially, implying that the demographic parameters were on average constant over the period of data collection. Thus, single maximum likelihood estimates of the demographic parameters were available from a simple population model fitted to the entire pup production time series.

The recent levelling off in pup production must be a result of some combination of reductions in the reproductive or survival rates of pups, juveniles or adults (SCOS-BP 09/2 & 10/2). To date, the available data in the form of time series of pup production estimates have not contained sufficient information to allow us to quantify the relative contributions of these factors (SCOS-BP 06/7, 09/2). However, additional information is now available in the form of an independent estimate of population size based on counts of the numbers of grey seals hauled out during the summer and information on their haulout behaviour (SCOS-BP 10/4). Two Bayesian state-space models of grey seal population dynamics were fitted to the English and Scottish regional estimates of pup production from 1984 to 2009 (SCOS-BP 10/2), and to the independent estimate of total population size just before the 2008 breeding season. One model (EDDSNM) allowed for density dependence in pup survival, while the other (EDDFNM) allowed for density dependence in female fecundity. Both models had flexible forms of density dependence, but allowed no movement of recruiting females between regions. As in 2008 and 2009, the models directly estimated observation (i.e. counting) error which had previously been set to an arbitrarily high fixed value with a C.V. of 25%.

Until 2007 SCOS presented the lower (EDDSNM) estimate as the conservative estimate of the total grey seal population, but used the combined confidence limits of both models to reflect the degree of confidence in the population estimate. In 2008 and 2009 SCOS presented model weighted average estimates of the population to better represent the level of uncertainty in model selection. As the models had similar weights the resulting estimates were equal to the averages of the two model estimates.

The estimated population size associated with all annually monitored colonies in 2009 was 106,200 (95% CI 82,000-138,700) for the EDDSNM model and 206,700 (95% CI 181,400-243,000) for the EDDFNM model.

A comprehensive survey of data available from the less frequently monitored colonies is presented in SCOS-BP 10/1. Total pup production at these sites was estimated to be approximately 5,250 in 2009. Using the average ratio of pup production to population size for the annually monitored sites based on the EDDSNM model estimate, and assuming proportionally similar confidence intervals, produces a population estimate of 13,200 (approx C.I. 10,500 to 17,500) for these sites. Combining these with the annually monitored sites gives a 2009 estimated UK grey seal population of 119,400 (95% CI 92,500-156,200).

Incorporating the independent estimate of population size influenced estimates of population size for the entire time series in both models, and strongly facilitated model selection. The posterior model probabilities were 1.0 and 0.0 respectively; hence the model-averaged estimate of total population size was identical to that for the EDDSNM model. Therefore, the best estimate of the total population associated with all annually monitored colonies in 2009 was 106,200 (95% CI 82,000-138,700)

The trajectory of the EDDSNM model indicates that the grey seal population increased by around 0.4% between 2008 and 2009 and has been increasing at around 0.5% pa for the past five years. Almost all of the increase has occurred in the North Sea population. The population in the Northern and Western Isles has increased by less than 0.25% p.a. since 2000.

The population estimate for the annually monitored sites in 2008 published in the 2009 SCOS report was 183,000 based on a simple average of the two models. The apparently large decrease is not real, it is entirely a consequence of changes in the treatment of the model outputs in response to problems of model selection. The newly available independent population estimate increases our ability to discriminate between the models and means that the problem of model selection has been effectively overcome. This and a programme of continually updating the independent estimate means that such changes in treatment of model outputs are unlikely to be repeated. SCOS emphasizes the importance of this independent estimate for answering these crucial questions.

In addition to resolving the model selection problems, the independent estimate has also allowed us to dramatically reduce the magnitude of the confidence intervals from 47% to 196% of the mean estimate last year down to 78% to 129% of the mean estimate this year.

In 2008 and 2009 SCOS recommended that additional studies to obtain independent estimates of population size, fecundity and both pup and adult survival should be given high priority. SCOS discussed and approved a series of studies to provide additional insight into the dynamics of the grey seal population:

- A detailed analysis of the haulout behaviour of a large sample of grey seals determined by satellite telemetry was reviewed. Results indicate that approximately 35% of the grey seal population is hauled out at the time of the annual harbour seal surveys and that there are no significant regional, sex or age differences in haulout probability. These results were combined with regional haulout counts of grey seals obtained during the harbour seal moulting and breeding surveys throughout Scotland and on the east coast of England. The resulting independent population estimate was 74,223 (95% confidence interval: 54,300 – 118,300) (SCOS BP 10/4). This estimate was incorporated into the population modelling process allowing effective model selection and concomitant reductions in confidence intervals (SCOS BP 10/2).
- A preliminary version of a complementary modelling approach is presented in SCOS BP 10/5. A simple Bayesian method, using generalised additive models to smooth a series of pup production estimates followed by matrix models to scale their results up, was used to estimate the trajectories of four British grey seal populations. A uniform prior on the relative importance of density dependence in fecundity and first year survival is applied to produce an overall estimate and credibility (Bayesian confidence) interval for each population. This approach requires fewer assumptions than the current State Space Models while producing similar population estimates and credibility intervals. SCOS recommends that this and other modelling approaches should be investigated further.
- SMRU have continued the analysis of data from the long-term studies on the Isle of May and North Rona to extract information on fecundity, age at first reproduction and adult survival and the effects of co-variates on population parameters. Preliminary results were presented to SCOS 2010. SCOS recommends that the studies to improve priors on demographic parameters should be encouraged.
- An extensive program of methodology development and data extraction from pelage photographs of seals on the breeding beach has been established. SCOS acknowledged the exciting potential for synergy between the photo identification and long term demographic studies and requested additional presentations at SCOS 2011.

In light of the improvements in model fitting provided by the independent non breeding season estimate, the level of uncertainty in the population estimates associated with the relationship between numbers of pups and adults has been greatly reduced. However, there are also uncertainties associated with the estimates of pup production, which were believed to lie within a range of –10% to +13% of the values provided. Since 2006 the model used to generate total population estimates provides an independent estimate of the measurement errors in pup production estimates. The fitted estimate of the CV of the pup production estimates was 8.3% (95% credibility interval 6.8-10.1%). There are additional unknown uncertainties associated with the estimates of pup production at colonies that are not surveyed annually.

There are also uncertainties about the value used for adult male survival, about which little is known. This may now represent the main source of uncertainty in the grey seal population estimation process. The magnitude of this problem will be investigated and reported to SCOS 2011.

1.3 Population Trends

The long term average rates of change suggest that the growth of pup production in the Inner and Outer Hebrides has effectively stopped with little change in the Inner Hebrides and possibly a small decrease in the Outer Hebrides since the mid 1990s. Pup production in Orkney also appears to have levelled off since the end of the 1990(SCOS-BP 10/1 & 10/2; SCOS-BP 06/4). The independent population estimate suggests that density dependence is acting mainly on pup survival. This also implies that the overall population will closely track the pup production estimates. It is therefore likely that the total populations of grey seals in the Hebrides and Orkney will have followed similar trajectories to those shown by the time series of pup productions.

1.4 UK grey seal population in a world context

The UK grey seal population represents approximately 38% of the world population on the basis of pup production. The other major populations in the Baltic and the western Atlantic are also increasing, but at a faster rate than in the UK (Table 2). If the difference in growth rate is due to reduced pup survival in the UK population compared to the Baltic and the western Atlantic, the UK will hold less than 38% of the total all age population.

Table 2. Relative sizes of grey seal populations. Pup production estimates are used because of the uncertainty in overall population estimates

Region	Pup Production	Years when latest information was obtained	Possible population trend[2]
UK	47,500	2009	Increasing
Ireland	1,600	2005	Unknown[1]
Wadden Sea	400	2008	Increasing [2]
Norway	1,200	2003	Unknown[2]
Russia	800	1994	Unknown[2]
Iceland	1,200	2002	Declining[2]
Baltic	4,000	2003	Increasing[2,4]
Europe excluding UK	**9,200**		**Increasing**
Canada - Sable Island	62,000	2008	Increasing[3]
Canada - Gulf St Lawrence + Eastern Shore	14,400	2007	Declining[5]
USA	1,100	2002	Increasing
WORLD TOTAL	**134,200**		**Increasing**

[1] Ó Cadhla, O., Strong, D., O'Keeffe, C., Coleman, M., Cronin, M., Duck, C., Murray, T., Dower, P., Nairn, R., Murphy, P., Smiddy, P., Saich, C., Lyons, D. & Hiby, A.R. 2007. An assessment of the breeding population of grey seals in the Republic of Ireland, 2005. Irish Wildlife Manuals No. 34. National Parks & Wildlife Service, Department of the Environment, Heritage and Local Government, Dublin, Ireland.

[2] Data summarised in:- Grey Seals of the North Atlantic and the Baltic. 2007 Eds: T. Haug, M. Hammill & D. Olafsdottir. NAMMCO Scientific publications Vol. 6

[3] Bowen, W.D., McMillan,J.I. & Blanchard, W. 2007. Reduced Population Growth Of Gray Seals At Sable Island: Evidence From Pup Production And Age Of Primiparity. Marine Mammal Science, 23(1): 48–64

[4] Baltic pup production estimate based on mark recapture estimate of total population size and an assumed multiplier of 4.7

[5] Thomas,L.,Hammill,M.O. & Bowen,W.D. 2007 Estimated size of the Northwest Atlantic grey seal population 1977-2007 Canadian Science Advisory Secretariat: Research Document 2007/082 pp31.

Current status of British harbour seals

- approximately 25,650 harbour seals were counted in the U.K:
  - 79% in Scotland; 16% in England; 5% in Northern Ireland
- Compared with the mid 1990s, some populations have declined by:
  - 50% in Shetland; 67% in Orkney; 35% in the Outer Hebrides; 40% in the Moray Firth and 85% in the Firth of Tay.
- Other populations show do no show consistent declines:
  - Strathclyde is unclear having declined slightly after an apparent increase around 2000
  - The west coast of Highland region appears to be stable
  - The 2009 English East coast counts were 20% higher than in 2008 and only 7% below pre epidemic levels

Each year SMRU carries out surveys of harbour seals during the moult in August. Recent survey counts and overall estimates are summarised in SCOS-BP 10/3. It was considered to be impractical to survey the whole coastline every year and SMRU aimed to survey the whole coastline across 5 consecutive years. However, in response to the observed declines around the UK the survey effort has been increased and an attempt was made to survey the entire Scottish and the English east coast populations during 2007.

Seals spend the largest proportion of their time on land during the moult and they are therefore visible during this period to be counted in the surveys. Most regions are surveyed by a method using thermographic aerial photography to identify seals along the coastline. Conventional photography is used to survey populations in the estuaries of the English and Scottish east coasts.

The estimated number of seals in a population based on these methods contains considerable levels of uncertainty. A large contribution to uncertainty is the proportion of seals not counted during the survey because they are in the water. We cannot be certain what this proportion is, but it is known to vary in relation to factors such as time of year, state of the tide and weather. Efforts are made to reduce the effect of these factors by standardising the time of year and weather conditions and always conducting surveys within 2 hours of low tide.

Combining the most recent counts (2006-2009) at all sites, approximately 25,650 harbour seals were counted in the U.K: 79% in Scotland; 16% in England; 5% in Northern Ireland (Table 3). Including 2,900 seals counted in the Republic of Ireland produces a total of 28,550 harbour seals for the British Isles.

Not all individuals in the population are counted during surveys because at any one time a proportion will be at sea. The survey counts are normally presented as minimum estimates of population size. Telemetry-based, mark-recapture estimates suggest that approximately 60-70% of the population are counted during the moult surveys, leading to an estimate for the total British population of 40,000-46,000 animals. There is some debate about the validity of this multiplier and SMRU are currently undertaking a telemetry study of haulout behaviour to estimate the proportion of the population hauled out during the moult surveys. SCOS recognises the importance of this work in providing a robust multiplier to be applied to harbour seal population estimates in future.

Apart from the population in The Wash, harbour seal populations in the UK were relatively unaffected by PDV in 1988. The overall effect of the 2002 PDV epidemic on the UK population was even less pronounced. However, again The Wash was the most affected region and counts since 2002 did not indicate a recovery following the epidemic until 2009 when a large increase was observed. Counts by region for the 2009 season are given in Table 3. These are minimum estimates of the British harbour seal population. Results of surveys conducted in 2009 are described in more detail in SCOS-BP 10/3.

Table 3 Counts of harbour seals by region

Harbour seal Management Area	Current estimate (2007-2009)	Previous estimate (2000-2005)	Earlier estimate (1996-1997)
Shetland	**3,003** 2009	**4,883** 2001	**5,991** 1997
Orkney	**2,874** 2008, 2009	**7,752** 2001	**8,523** 1997
Highland North coast	**112** 2008	**174** 2005	**265** 1997
Outer Hebrides	**1,804** 2008	**2,067** 2003	**2,820** 1996
West Scotland, Highland (Cape Wrath to Ardnamurchan Point)	**4,696** 2007, 2008	**4,665** 2005	**3,160** 1996, 1997
West Scotland, Strathclyde (Ardnamurchan Point to Mull of Kintyre)	**5,834** 2007, 2009	**7,003** 2000, 2005	**5,651** 1996
South-west Scotland, Firth of Clyde (Mull of Kintyre to Loch Ryan)	**811** 2007	**581** 2005	**923** 1996
South-west Scotland, Dumfries & Galloway (Loch Ryan to English Border at Carlisle)	**23** 2007	**42** 2005	**6** 1996
East Scotland, Firth of Forth (Border to Fife Ness)	**148** 2007	**280** 2005	**116** 1997
East Scotland, east coast Fife Ness to Fraserburgh	**228** 2007	**406** 2005	**648** 1997
East Scotland, Moray Firth (wider) Fraserburgh to Duncansby Head	**871** 2007	**959** 2005	**1429** 1997
TOTAL SCOTLAND	**20,404**	**28,812**	**29,532**
	(2009)	(2005)	(1997)
Blakeney Point	**372**	**709**	**311**
The Wash	**2,829**	**1,946**	**2,461**
Donna Nook	**267**	**421**	**251**
Scroby Sands	**165**	**57** 2004	**65**
Other east coast sites	347	**153** 1994-2003	**137** 1994 –1997
South and west England (estimated)	**20**	**20**	**15**
TOTAL ENGLAND	**4,000**	**3,306**	**3,240**
TOTAL BRITAIN	**24,404**	**32,118**	**28,485**
TOTAL NORTHERN IRELAND	**1,248** 2002	**1,248** 2002	
TOTAL BRITAIN & N. IRELAND	**25,652**	**33,366**	**29,733**
TOTAL REPUBLIC OF IRELAND	**2,905** 2003	**2,905** 2003	
TOTAL GREAT BRITAIN & IRELAND	**28,557**	**36,271**	**32,638**

Population trends

A complete survey of Shetland in 2009 counted the same number of seals as in 2006, equivalent to 50% of the mid 1990s counts. Counts from a partial survey of Orkney were 2.2% higher than the same areas in 2008, but were 64% lower than the same areas in 2001. These latest results suggest that the Orkney harbour seal population declined by 67% since the late 1990s and has been falling at an average rate >13% p.a. since 2001.

Counts in the Outer Hebrides in 2008 were 35% lower than the peak count in 1996. Regular surveys over the intervening period suggest that there has been a sustained but gradual decline of around 3% pa since 1996.

Counts of parts of the Strathclyde region in 2009 were 15% higher than counts of the same areas in 2007. A count of the entire Strathclyde region in 2007 was 25% lower than in 2000 but similar to counts in the mid 1990s. If the subsample counted in 2009 was representative, the overall Strathclyde population will be intermediate between the 1990s and early 2000 counts.

Surveys in 2007 confirmed that the west coast of Highland Region has not show any decline and surveys in 2008 confirmed that the North coast of Highland Region has declined by 35% since the 2005 survey and is approximately 60% lower than in 1997.

Surveys of the east coast populations in 2009 showed a continuing rapid decline in the Firth of Tay population (SCOS-BP 10/3), a slight increase in the Moray Firth and a large increase in the English East coast populations. The Firth of Tay count continued the recent trend of rapid decline. This SAC population has declined at an average rate of 20% p.a. since 2002 with the 2010 count 84% lower than the peak count in 2000.

Overall, the combined count for the English East coast population (Donna Nook to Scroby Sands) in 2009 was 21% higher than the 2008 count and was <7% lower than the pre-epidemic count in 2002 (SCOS-BP 10/3, Figure 10, Table 4). The 2009 pup production estimate for the Wash was also 14% higher than the estimate for 2008. Despite these large increases, the English population has not kept pace with the rapid growth in the nearest European population in the Wadden Sea which increased by 12% between 2008 and 2009 and has grown by approximately 13% pa since the 2002 PDV epidemic.

Response to harbour seal declines

These widespread declines give clear cause for concern and have resulted in the implementation of area-specific Conservation Orders by the Scottish Government, providing harbour seals with year-round protection. A targeted research programme has been established including increased monitoring to confirm the magnitude and geographical extent of the declines and comparative studies of pup survival in areas of contrasting population dynamics.

In 2008 SCOS recommended that a survey of the harbour seal population of Shetland be given a high priority, that repeat surveys of Orkney and other regions would be desirable. Additional studies to obtain independent estimates of the proportions of the population ashore during surveys and any improvement in our knowledge of demographic parameters should be encouraged. In response, SMRU, with funding support from NERC, Scottish Government Marine Directorate, Scottish Natural Heritage and Natural England, has established a research programme which includes:

1. planned thermal image surveys of harbour seal moulting populations in Shetland and repeat surveys in Orkney,
2. continuation of the annual fixed wing survey of the English and Scottish east coast moulting populations,

3. continuation of the pup production surveys in the Moray Firth and East Anglian populations,
4. a satellite-telemetry based study of proportion of time seals spend hauled out during the moult in two populations with contrasting dynamics, i.e. Orkney and the west coast,
5. completion of analysis of pup survival rates in two populations with contrasting dynamics, i.e. Orkney and the west coast.
6. continued investigations into disease and environmental factors affecting survival in harbour seals

Results from 1 to 5 were presented to SCOS in 2010.

In 2009 a previously unidentified source of anthropogenic mortality was identified in harbour and grey seals in Scotland. In 2010, severely damaged seal carcasses have been found on beaches in eastern Scotland (St Andrews Bay, Tay and Eden Estuaries and Firth of Forth), along the North Norfolk coast in England (centred on the Blakeney Point nature reserve), and within and around Strangford Lough in Northern Ireland. All the seals had a characteristic wound consisting of a single smooth edged cut that starts at the head and spirals around the body. In most cases the resulting spiral strip of skin and blubber was detached from the underlying tissue. In each case examined so far the wound would have been fatal. The extremely neat edge to the wound strongly suggests the effects of a blade with a smooth edge applied with considerable force, while the spiral shape is consistent with rotation about the longitudinal axis of the animal.
The injuries are consistent with the seals being drawn through a ducted propeller such as a Kort nozzle or some types of Azimuth thrusters. Such systems are common to a wide range of ships including tugs, self propelled barges and rigs, various types of offshore support vessels and research boats. All the other explanations of the injuries that have been proposed, including suggested Greenland shark predation, are difficult to reconcile with the observations and, based on the evidence to date, seem very unlikely to have been the cause of these mortalities. A detailed description of the mortalities will be presented to SCOS 2011. (A preliminary analysis is available from the SMRU web site (http://www.smru.st-and.ac.uk/documents/366.pdf). The population consequences of these mortalities is unknown, but SMRU are investigating the events and will report results to SCOS 2011.

In 2008, SCOS recommended that a programme of research be developed to address specific hypotheses about the causes of the decline and that SMRU should seek additional funds to support such a research programme. A summary of the issues to be addressed was discussed by SCOS in 2009. Briefly, the following questions were identified as the priorities for research.

1. Is it likely that an artefact of the survey methodology or any of the following changes in the seals' behaviour could account for the observed changes in counts without a population change?
   - Changes in timing of peak counts during the moult,
   - Changes in patterns of haulout behaviour,
   - Movement, e.g. migration to neighbouring regions

2. Is reduced food availability causing any of the following effects? If so are they sufficient to account for the observed declines through:
   - Reduction in pup survival
   - Reduction in adult survival
   - Reduction in fecundity

3. Is the decline due to competition between harbour and grey seals?

- Do grey and harbour seals compete for food
- Do grey seals exclude harbour seals from certain habitats
- Do grey seals prey on young harbour seals

4. Are any of the following direct mortality effects having a significant impact on the harbour seal population?
   - Disease
   - Biotoxins
   - Pollution
   - predation
   - By catch
   - Deliberate killing

Table 4 Sizes and status of European populations of harbour seals.

Region	Number of seals counted[1]	Years when latest information was obtained	Possible population trend[2]
Outer Hebrides	1,800	2008	Declining
Scottish W coast	11,400	2007-2008	None detected
Scottish E & N coast	1,400	2008	Declining
Shetland	3,000	2009	Declining
Orkney	2,900	2008	Declining
Scotland	**20,400**		
England	**4,000**	2008	Recent decline[4]
Northern Ireland	**1,200**	2002	Decrease since '70s
UK	**25,600**		
Ireland	2,900	2003	Unknown
Wadden Sea-Germany	9,400	2008	Increasing after 2002 epidemic
Wadden Sea-NL	4,100	2008	Increasing after 2002 epidemic
Wadden Sea-Denmark	2,000	2008	Increasing after 2002 epidemic
Lijmfjorden-Denmark	1,400	2003	Recent decline [3]
Kattegat/Skagerrak	11,700	2003	Recent decline[3]
West Baltic	300	1998	Recent decline[3]
East Baltic	300	1998	Increasing
Norway	3,800	1996-98	Declining
Iceland	19,000	?	Unknown
Barents Sea	700	?	Unknown
Europe excluding UK	**55,600**		
Total	**81,200**		

1 –counts rounded to the nearest 100. They should be considered to be minimum estimates of total population size as they do not account for proportion at sea and in many cases are amalgamations of several surveys.
2 – There is a high level of uncertainty attached to estimates of trends in most cases.
3 – Declined as a result of the 2002 PDV epidemic, no recovery.
Data sources: www.smru.st-and.ac.uk; ICES Report of the Working Group on Marine Mammal Ecology 2004;, Harding *et al.* submitted to Ecology Letters

2. *What is known about the population structure, including survival and age structure, of grey and harbour seals in European, English and Scottish waters? Is there any evidence of populations or sub-populations specific to local areas?*

Grey seals

Within Europe there are two apparently reproductively isolated populations, one that breeds in the Baltic, usually pupping on sea ice in the spring, and one that breeds outside the Baltic, usually pupping on land in Autumn and early winter. These populations appear to have been reproductively isolated at least since the Last Glacial Maximum[1,2]. The vast majority (85%) of European grey seals breeding outside the Baltic breed around Britain. On the basis of genetic differences there appears to be a degree of reproductive isolation between grey seals that breed in the south-west (Devon, Cornwall and Wales) and those breeding around Scotland[3] and within Scotland, there are significant differences between grey seals breeding on the Isle of May and on North Rona[4]. Until 2002, SMRU treated this last group as a single population for the purpose of estimating total population size. Estimates of the numbers of seals associated with different regions were obtained by dividing up the total population in proportion to the number of pups born in each region.

Since 2003, a spatially-explicit model has been used to estimate the British grey seal population from geographically structured pup production estimates. A preliminary application of this model (SCOS-BP 03/4) indicated that there was little movement of breeding animals between Inner Hebrides, Outer Hebrides, Orkney and North Sea. This suggestion is further supported by recent results from grey seal population models that indicate an absence of large scale redistribution of breeding females between regions (SCOS-BP 09/02 & 10/2), again implying a high degree of philopatry. However, these results apply to large geographical regions, Outer Hebrides, Inner Hebrides, Orkney and North Sea. The lack of large scale redistribution is supported by the results of detailed studies at breeding colonies and re-sightings of photo-identified individuals that indicate breeding females tend to return to their natal breeding colony and remain faithful to that colony for most of their lives[5]. A NERC funded project to continue and extend the photo identification work began in 2009. A recognition system for pelage developed for identifying seals from head patterns has been modified to identify seals from pelage patterns on the flank, neck chest and abdomen. The catalogue now contains around 19000 distinct IDs. The current project is focussing on the breeding season photographs from North Rona. Initial results are encouraging and SCOS recommends that this work and further analysis of data from the long term demographic studies be given high priority.

At a finer scale, i.e. within these sub-populations, there may be substantial movement or recruitment of breeding females to colonies other than their natal sites. This is thought to be the explanation for the rapid initial growth of colonies in the North Sea and at specific

---

[1] Boskovic, Kovacs,K.M., Hammill,M.O. & White,B.N. (1996) Geographic distribution of mitochondrial DNA haplotypes in grey seals (*Halichoerus grypus*) Canadian Journal of Zoology 74 pp 1787-1796

[2] Graves, J.A., Helyar, A., Biuw, M., Jüssi, M., Jüssi, I. & Karlsson, O. (2008) Analysis of microsatellite and mitochondrial DNA in grey seals from 3 breeding areas in the Baltic Sea. *Conservation Genetics*

[3] Walton M. & Stanley, H.F. 1997. Population structure of some grey seal breeding colonies around the UK and Norway. European Research on Cetaceans. Proc 11th annual conference of European cetacean society. 293-296

[4] Allen, P. J., W. Amos, et al. (1995). Microsatellite variation in grey seals (Halichoerus grypus) shows evidence of genetic differentiation between two British breeding colonies." Molecular Ecology **4**(6): 653-662.

[5] Pomeroy, P.P., Twiss, S. & Redman,P. (2000). Philopatry, site fidelity and local kin associations within grey seal breeding colonies. Ethology 106 (10): 899-919

sites in the Hebrides and Orkney. In this respect, the grey seals at all of the English North Sea breeding sites are considered to have been relatively recently derived from other North Sea colonies and as such are unlikely to show any significant differentiation. This North Sea group is thought to show a degree of reproductive isolation from those breeding in Devon, Cornwall and the Scilly Isles.

Age structure

While the population was growing at a constant rate, i.e. a constant exponential change in pup production, the stable age structure for the female population could be calculated. However, since the mid 1990s this has not been possible since changes in pup production growth rates imply changes in age structure. In the absence of a population wide sample or a robust means of identifying age-specific changes in survival or fecundity, we are unable to accurately estimate the age structure of the female population. The independent population estimate (SCOS-BP 10/4) strongly suggests that the density dependent effect is operating through reduced pup survival (SCOS-BP 10/2). A consequence of a gradually increasing level of pup mortality would be a relative reduction in the size of young age classes. This density dependent effect has been apparent since the mid 1990s in the Hebridean populations, implying that at least the youngest 15 to 20 year classes will be reduced. The effect is more recent in Orkney so fewer year classes will be reduced. In the North Sea, the continued exponential growth implies that there will have been little or no perturbation of the stable age structure. Although there has never been any reliable information on age structure for the male component of the population the fact that the independent estimate is well below the mean predicted population size from the EDDSNM model may be an indication that male survival is low or has perhaps declined relative to female survival.

Survival rates

Survival rates and fecundity estimates for adult females breeding at North Rona and the Isle of May have been estimated from re-sightings of permanently marked animals and have previously been presented to SCOS. An analysis of these data has been submitted for publication. An integrated analysis of resightings, post-partum mass and reproductive success data was used to explore the relationship between mass and probability of breeding (individual fecundity). Results suggest important differences between the Isle of May and North Rona, with adult apparent survival rates generally higher and more consistent at IM 0.950 (0.933,0.965), and lower in some years on NR (0.75 – 0.99). There was no evidence of mass dependent survival, but there was annual variation in mass gain at IM. Overall fecundity estimates were different (0.63 NR, 0.76 IM) and fecundity declined rapidly with decreasing maternal mass at the end of a breeding episode. These estimates are lower than previous estimates for UK grey seals of 0.94 for the Farne Islands, and 0.83 for the Hebrides[6].
Both results are consistent with the differing dynamics at these two colonies and suggest that differences in vital rates among colonies may be widespread.

Harbour seals

Samples from seals in Northern Ireland, the west and east coasts of Scotland, the east coast of England, Dutch and German Wadden Sea, Kattegat/Skagerrak, Norway, Baltic Sea and Iceland have been subjected to genetic analysis. This analysis suggested that there may be significant genetic differentiation between harbour seal populations in European waters[7] [8]. The Irish-Scottish, the English east coast and the Wadden Sea

---

6 Boyd, I. L. (1985). "Pregnancy and ovulation rates in grey seals ({IHalichoerus grypus}) on the British coast." Journal of Zoology **205**(A): 265-272.

7 Goodman, S.J. (1998) Patterns of extensive genetic differentiation and variation among European harbour seals (Phoca vitulina vitulina) revealed using microsatellite DNA polymorphisms. Molecular Biology and Evolution, 15, 104-118.

harbour seals were identified as distinct population units. There is probably little movement of breeding animals between these populations although satellite telemetry reveals some interchange between the Wadden Sea and the English east coast populations outside the breeding season. Within the Ireland-Scotland population there is probably occasional movement of animals between regions, but there is no evidence from satellite telemetry of any long-range movements (for example, between the east and west coasts of Scotland) comparable to those observed in grey seals.

In 2010 Scottish Government provided additional funding for a study of the degree of genetic differentiation and spatial structure within the Scottish grey and harbour seal populations. This project is due to report in March 2011 and a briefing paper will be presented to SCOS 2011.

Satellite tracking of pups showed some dispersal from Orkney to the Outer Hebrides and down the east coast as far as the Firth of Tay. However pups in the sample that moved long distances did not survive.

In other European populations there is also little information on population scale movements. Studies of the movements of branded seals in the Kattegat/Skagerrak[9] indicate that there is only limited movement within the western Scandinavia population. However, in both 1988 and 2002 phocine distemper spread rapidly among European harbour seal populations, suggesting that substantial movement of individuals can occur, although the genetic studies suggest these movements do not result in large numbers of seals reproducing in locations they visit temporarily.

Age structure

The absence of any historical cull data or a detailed time series of pup production estimates means that there are no reliable data on age structure of the UK harbour seal populations. Some age structure data were available from seals found dead during the PDV epidemics in 1988 and 2002. However, these were clearly biased samples and could not be used to generate population age structures.

In the absence of consistent long time series of pup production or any systematic sampling of the population for age data, we are unable to define the age structure of the UK harbour seal population. With a sufficiently long time series of both pup production estimates and overall population indices (moult counts) the harbour seal population modelling approach under development at SMRU will be capable of generating age structures for the female component of the harbour seal population. Methods for estimating pup production from sparse survey data are being developed and a series of repeat surveys during the breeding seasons in the Wash and Moray firth have been carried out to enable SMRU to estimate pup production and assess the errors in the developing time series of pup production estimates.

SCOS recognise the importance of these and other studies based on haulout behaviour in assessing the status of harbour seal populations.

Survival rates

SMRU have recently conducted a comparative study of survival rates of harbour seal pups in the declining Orkney and apparently stable West Coast populations. Results suggest that both populations have similar but high mortality rates and that differential

[8] Stanley, H. F., S. Casey, et al. (1996). "Worldwide patterns mitrochondrial DNA differentiation in the harbour seal (iPhoca vitulina)." Molecular Biological Evolution **13**(2): 368-382.

[9] Härkönen, T. & Harding, K.C. (2001) Spatial structure of harbour seal populations and the implications thereof. Canadian Journal of Zoology, 79, 2115-2127.

pup mortality is unlikely to be responsible for the observed demographic patterns.

Current work

Work is currently underway to develop recommendations for spatial management units and to connect these to population structure. This is partly built from studies of movements and habitat use (SCOS-BP 05/3 and 05/5). Defining optimal management areas for UK seals requires an arrangement of relatively isolated groups of colonies. The motivation behind this requirement is that management actions taken in one unit should have minimal impact on the others. Clustering algorithms have been developed to subdivide grey seal breeding colonies into maximally isolated groups according to at-sea distance (SCOS-BP 06/5) and a method for optimal design of marine SACs based on at sea location data was presented in 2007 (SCOS-BP 07/8)

SCOS 2009 recommended additional effort to improve the estimates of harbour seal population size including improved estimates of the proportion hauled out during the moult, inclusion of high resolution digital imagery of all seals during thermal image surveys and the acquisition and use of new, reliable thermal imaging equipment. In addition, complementary modelling activities to support the collection of data should be given high priority. A telemetry study to address the question of haulout proportion started in summer 2009. The proportion of time spent hauled out did not differ between seals tagged in the stable west coast and declining Orkney populations and the overall proportion of time spent hauled out during the moult was similar to previous estimates. A full analysis of the results will be submitted for publication before SCOS 2011. Digital photography has been included throughout the harbour seal surveys to improve and confirm species identification. A harbour seal population model is under development and a full analysis will be presented to SCOS 2011.

Harbour Seal Population

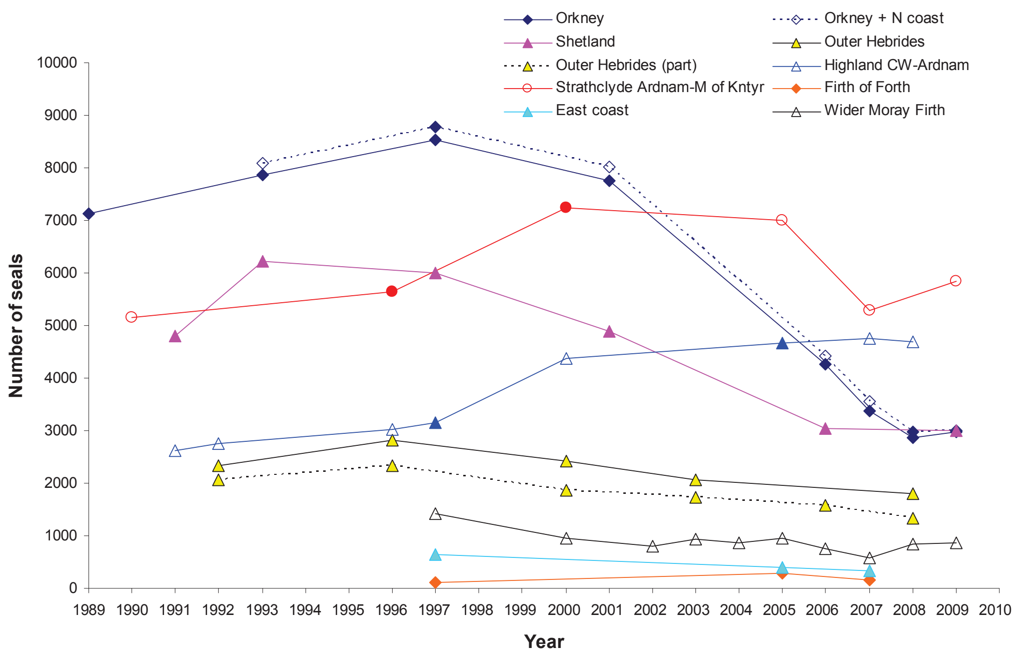
3. Is the existing harbour seal decline recorded in several local areas around Scotland continuing or not and what is the position in other areas?

Details of surveys carried out and the counts obtained are give above in answer to Question 1. Figure 1 below shows the population trends in the different survey regions around Scotland. The latest survey results confirm that:

- the Orkney harbour seal population declined by approximately 65% since the late 1990s and has been falling at an average rate of approximately13% p.a. since 2001.
- the Shetland harbour seal population declined by approximately 50% since the late 1990s However, the Shetland survey in 2009 produce an identical count to that in 2006. This may be an early indication that the rapid declines are slowing. Additional data will be required to test this.
- the Outer Hebrides harbour seal population declined by approximately 35% since the mid 1990s, indicating a sustained but gradual decline of around 3% pa since 1996.
- the Strathclyde harbour seal population has shown wide fluctuations but recent surveys indicate little overall change since the mid 1990s.
- the population in the Firth of Tay has declined dramatically, by approximately 85% in the last 10 years.

- The population in the Moray Firth has remained steady for the last 5 years, following a rapid decline oif approximately 50% in the previous 10 years.
- the harbour seal populations of the west and north coasts of Highland Region have not shown any significant decline since the late 1990s.
- the English East coast population declined after the 2002 PDV epidemic but the 2009 count suggests that it is only 7% below its pre-epidemic level.
- the nearest European population, in the Wadden Sea, has continue to grow at approximately 13% pa since the 2002 PDV epidemic.

Fig. 1. Trends in counts of harbour seals around Scotland.



4. *In light of the latest reports, should the Scottish Government consider additional conservation measures to protect vulnerable local harbour seal populations in any additional areas?*

Conservation orders are currently in place for the Northern Isles and down the east coast as far as the border. In 2009 SCOS recommended additional data collection and monitoring to further investigate the requirement for extending these orders.

The recent survey results for a sub-sample of the Strathclyde haulout sites showed a 15% increase over the 2007 counts of the same sites/areas. The overall 2007 count for Strathclyde was approximately 30% lower than the peak of 7,900 in 2000. If the sub-sample is representative of the whole area, then the 2009 estimate would be higher than counts in 1988, 1993 and 1996 suggesting that there has been little change over the longer term. As Strathclyde region now holds the largest component of the Scottish harbour seal population, SCOS firmly recommends that repeat counts of Strathclyde should be carried out as soon as practicable within the constraints imposed by the overall harbour seal survey requirements.

It is worth noting that although the Outer Hebridean population has not undergone the same rapid declines observed in the Northern Isles and East coast populations, the counts for the Outer Hebrides have shown a consistent gradual decline of approximately 3.5% p.a. that has been maintained since the mid 1990s. Following the same precautionary approach, SCOS recommends that consideration should also be given to extending conservation measures to the Outer Hebrides.

The continued, dramatic decline in the population of harbour seals in the Firth of Tay and Eden Estuary SAC is a clear cause for concern. The emergence of unusual anthropogenic mortalities, primarily of pregnant female harbour seals close to the SAC adds to the level of concern (see Q5 below). SCOS strongly recommends that this cause of mortality be investigated and if identified should be removed or effective mitigation measures be put in place as soon as possible.

5. What is the latest understanding of the causes of the recent decline in harbour seals?

In response to the reported declines, SMRU convened an internal workshop to identify the salient features of the declines and develop a research programme to address the most likely candidate causal factors. The report of the workshop was considered by the Scottish Seals Working Group and a proposed work package was developed. A list of questions to be addressed is presented above (Question 1, page 15).

A preliminary step in the process was to develop a modelling tool to gauge the relative importance of real or perceived trends in demographic rates. A preliminary demographic model for harbour seal population dynamics combined with a model for the aerial observation process has been implemented within a Bayesian estimation framework as a single state-space model. This approach has been further developed using a multi-year series of repeated counts within the breeding and moulting periods in the Moray Firth and modified to incorporate the effects of an extensive seal shooting program.

In addition, because of the urgency of the problem SMRU implemented six data collection projects and added another urgency project in 2010:

1. An extensive air survey programme, supported by intensive ground observation studies, was carried out in summer 2007 and continued in summer 2008 to identify the geographical extent and confirm the magnitude of the declines around the UK. Results were presented in SCOS-BP 08/3, 09/3 and 10/3 and are discussed above. These studies have determined the scale and geographical extent of the declines and have been the basis for establishing and maintaining conservation orders.
2. A comparative study of pup mortality patterns in a declining population (Orkney) and a stable population (Lismore) was carried out in 2007. Pre-weaning mortality was negligible in both regions. A model incorporating a normal time to tag failure and independent survival estimates in each region was fitted. Survival did not follow a simple exponential decay and was best fitted by a gamma distribution that allows for a gradually increasing probability of death, consistent with results in Danish seals that

show higher winter mortality. Results indicate that pup mortality was similar in the two samples and was therefore not identified as the main determinant of differences in observed population dynamics between Orkney and Lismore populations. However, this is based on a single year's pups and repetition of the pup survival study would be valuable in confirming this.

3. Archived blood samples from grey and harbour seals were screened to assess prevalence of anti-leptospira, toxoplasma and phocine distempter virus antibodies over the period 1991-2005. The results suggested it is unlikely that these infections played a major role in the decline of Scottish harbour seals (SCOS-BP 08/6). A follow-up comparative study of declining and stable populations was carried out between August and October, 2008. There was no evidence, in our sample of captured animals, of differences in levels of acute disease, no signs of infection, no abnormal parasite infestations, no evidence of a recurrence of PDV infections and no signs of nutritional stress. Thus ruling these out as possible causes for the decline. Detailed results were presented in SCOS-BP 09/6.
4. Samples of faeces, urine and blood serum from harbour seals in Orkney and on the east and west coasts of Scotland were screened for the biotoxin, domoic acid. Levels consistent with chronic exposure levels in other pinnipeds were detected in all areas, but were most prevalent in Orkney and the Firth of Tay. A study, funded by the Scottish Government, is now being carried out to investigate the spatial extent of this exposure (particularly in the spring and summer months) and to determine exposure to other biotoxins (particularly saxitoxin) which are also found in Scottish waters. Preliminary findings of this study and evidence that DA is found in intertidal waters are presented in SCOS-BP 10/6. Detectable levels of DA were found in water samples, in various fish species that are major prey species of both grey and harbour seals and in faecal samples collected from seal haulouts during the spring and early summer. Saxitoxin (paralytic shellfish poisoning toxin) was found at low levels in some of the seal fish prey items, but no positive faecal samples have so far been detected.
5. A satellite telemetry based study of proportion of time seals spend hauled out during the moult in two populations with contrasting dynamics, i.e. Orkney and the west coast was started in summer 2009.
6. An ongoing study of killer whale behaviour in Shetland has provided an opportunity to estimate predation rates. Results from 2008 & 2009 included direct observations of 4 successful kills and 2 recent kills. Extrapolating from these observations produced estimated takes similar to those based on assumptions about the degree of reliance on seals as prey and energetic requirements. Results suggest that killer whales may be a contributory factor in the declines (SCOS-BP 10/7).

Corkscrew injuries

An additional project was started in 2010 in response to a novel mortality event that has recently been identified in UK seal populations. A number of severely damaged seal carcasses have washed ashore in eastern Scotland and Eastern England. A total of 14 grey and harbour seals have been found in St Andrews Bay, Tay and Eden Estuaries and Firth of Forth between 2008 and September 2010 and a total of 38 grey and harbour seals have been found along the North Norfolk coast between December 2009 and September 2010. The seals have all apparently been killed by a characteristic wound consisting of a single smooth edged cut that starts at the head and spirals around the body. In most cases the resulting spiral strip of skin and blubber is detached from the underlying tissue. The wound is clearly the cause of death in each case examined so far. Similar injuries have been described on seals in Strangford Lough in Northern Ireland and at two locations on the Scottish west coast and a re-examination of pathology reports indicates that the mechanism is the same as that operating on the east coast.

The extremely neat edge to the spiral wound strongly indicates a cut made by a rotating blade within a channel or cowling of some sort or by the seal rotating past some form of static blade. The presence of additional facial wounds that match the shape of propeller rope cutter blades strongly suggests that the wounds were caused by some form of ducted propellers such as Kort drives or some types of azimuth thruster. SMRU are currently investigating the mechanism of injury to narrow down the range of potential vessels.

The relatively small numbers of seals found so far are unlikely to have a significant impact on large seal populations. However, in St. Andrews Bay and the Firth of Tay the harbour seal population has undergone a significant decline in the past decade and, if maintained, the current level of observed mortality due to this cause may prove unsustainable. We do not know if this mortality is a local inshore problem or a more widespread problem that has come to light because the recent mortalities have occurred close to shore.

In response, SMRU have begun to investigate potential causal mechanisms in collaboration with the RSPCA and Scottish Marine Mammal Stranding network, with support from Scottish Government, Scottish Natural Heritage and Natural England. Due to the seriousness of this development, results and progress will be reported to SCOS as and when they become available during 2010 and a briefing paper summarising the state of knowledge will be presented to SCOS 2011.

6. *What progress has been made in improving the ability to estimate with more certainty the size of current and future grey seal populations?*

Inclusion of an independent population estimate based on a complete census of the hauled out population in August and an extensive analysis of haulout records from telemetry studies has allowed the model selection process to un-equivocally select the EDDSNM model as the most appropriate. A description of this process and the concomitant improvement in confidence (c.i. reduced by >60%) is described in answer to Q1 above and in SCOS-BP 10/2 & 10/4.

7. *What are the current best estimates on seal consumption of salmon and sea trout?*

We do not have sufficient information to allow us to estimate predation on salmonids at a national scale. There are however three separate research projects that do or will provide information on local predation rates.

- Diet studies in the Firth of Tay and St Andrews Bay suggested that seals hauling out inside the estuary were preying heavily on salmon during spring and summer and on sea trout during the autumn. Harbour seal faecal samples from St Andrews Bay outside the Firth of Tay did not contain sea trout and had very few salmon otoliths. Salmonid otoliths appeared in only a few samples, producing very wide confidence intervals on the consumption estimates. The results are therefore of limited value for management purposes. The continued rapid decline of harbour seal populations will have reduced the estimated consumption, but the proportions of the local harbour seal population hauling out in the Firth of Tay and St Andrews Bay have also changed from approximately 30%:70% before 2003 to 55%:45% since 2003 [10].

[10] Sharples, R.J., Arrizabalaga, B. and Hammond, P.S. (2009) Seals, sandeels and salmon; diet of harbpur seals in St Andrews Bay and the Tay estuary, South East Scotland. M.E.P.S. 390:216-276

- Observations of seals consuming salmonids in rivers during regular surveys between 2005 and 2008 allow estimation of total consumption by month. Fish can not usually be identified to species level so figures are given for salmonids. Numbers fish consumed peaked in winter in all three rivers, thought to be the result of targeted predation on kelts[11].
- Additional attempts were made to directly monitor predation by harbour seals on sea trout using PIT tags and a purpose built seal-borne recorder and transmitter system. Unfortunately a series of technical problems meant that SMRU were unable to catch seals in the study area. Further attempts will be made in spring 2011.

Seals and Salmon Netting Stations

8. ***What is the current state of knowledge of interactions between seals and salmon netting stations and possible mitigation measures?***

A series of observations of seal activity and a photo i.d. project has been initiated at netting stations in both the Moray Firth and the Angus coast south of Montrose.
Trials of commercial ADDs at salmon bag nets are currently underway. During the 2009 salmon netting season, a seal scarer or Acoustic Deterrent Device (ADD) was operated at one netting station to evaluate its performance experimentally. Preliminary results suggest that the ADD significantly reduced the presence of seals seen around the netting station and to have reduced the incidence of damaged salmon in the net. Further trials are underway will be conducted this summer and final results will be available to SCOS in 2011.

Seals and Fish Farms

9. ***What is the current state of knowledge of interactions between seals and fin fish farms and possible mitigation measures?***

This has been recognised as a problem for some time in terms of the damage caused to cages and fish, but also in terms of secondary effects because of salmon escaping from cages and mixing with local wild populations. More recently, however, the potential effects of methods used to control seals around fin fish farms, involving acoustic deterrent devices (ADDs) and/or shooting seals in the vicinity of farm cages, have been increasingly viewed as a concern. This is partly because of potential effects of ADDs on other marine mammals and partly because the decline of common seals has focussed attention on ways in which it may be possible to reduce unnecessary killing of seals by man.

Telemetry studies of seals caught in rivers and seals tagged in the outer banks of the Firth of Tay are providing information on seal activity in salmon rivers and show that

[11]. Graham, I.M & Harris, R.N. (2010) Investigation of Interactions Between Seals and Salmon in Freshwater. Final Report to Scottish Government and SNH 102pp (Copies available from the Sea Mammal Research Unit e-mail dt2@st-and.ac.uk)

targeted telemetry studies can provide information at spatial scales relevant to the seal/fishery interaction.

SMRU have recently completed a study funded by the Scottish Aquaculture Research Forum (S.A.R.F)[12]. to investigate the management of interactions between seals and salmon farms and to specifically investigate the extent to which the Acoustic Deterrent Devices (ADDs) used in Scottish fish farms exclude or affect the distribution of cetaceans, how effective they are in preventing seals from damaging fish pens and damaging farmed fish or allowing fish to escape.

Results show that porpoises generally avoid sources of loud noise but at least some porpoises seem tolerant of the noise of ADDs and are able to forage quite close to such sound sources. This conclusion supports observations made by farm site managers over many years. Previous observations from Canada showing clear cut exclusion in response to ADDs measured shorter term exposures and were not made at fish farm sites so that any potential attractive effects of farms sites would have been missing. The extent to which this degree of exclusion may have significant effects on the foraging success or the conservation status of porpoises remains to be answered.

Long term seal survey data and fish farm distribution were compared to investigate the possibility that fish farms were implicated in the observed population declines. In all regions except Strathclyde the number of seals counted at haul out sites close to fish farm sites as a proportion of the total number counted in each region remained effectively constant suggesting that there have not been disproportionate declines at haul out sites closest to farm sites. The relative decline in seal numbers close to fish farm sites in Strathclyde requires further investigation.

A combined observation, video monitoring and photo i.d. study was carried out at several farms. Preliminary results indicate that photo-identification is possible at fish farm sites and can be used to explore the behaviour of individual animals.

Seal Haul Outs

10. ***Does the Committee consider that the process for selecting key seal haul-out sites for additional protection is appropriate?***

The Marine (Scotland) Act 2010 introduced an offence of harassment of seals at listed haul-out sites. Marine Scotland and SMRU are developing a flexible method for selecting key haul-out sites. The method defines haul-out sites taking into account local, small-scale changes in locations of hauled-out seals observed during the harbour seal monitoring surveys. A time-weighted average of the numbers of seals counted in predefined segments of the coast during the last 15 years of moult surveys can then be used to identify “key haul-out sites”. The selection process is carried out at regional level and allows Marine Scotland to assess the effect of changing the haulout size threshold or changing the regional breakdown on the number and location of key haul-out sites. The development is ongoing and the final version will be presented to SCOS in 2011.

……..………………………………………………………………………………………
……………

[12] Northridge, *et.al*. 2010. Assessment of the impacts and utility of acoustic deterrent devices. Final Report to the Scottish Aquaculture Research Forum, Project Code SARF044. 34pp. copies available at :
www.sarf.org.uk/Project Final Reports/SARF044 - Final Report.pdf

DEFRA QUESTIONS

1. *What are the latest estimates of the number of seals in English waters?*
   See answer to Scottish Government Q1 above.

2. *What is known about the population structure, including survival and age structure, of grey and common seals in European and English waters?*

   See answer to Scottish Government Q2 above.

3. *Is there any evidence of populations or sub-populations specific to local areas within English waters?*

   See answer to Scottish Government Q2 above.

4. *What is the latest estimate of consumption of fish by seals in English waters?*

   Answer deferred until after discussion at SCOS 2010 meeting.

5. *Have there been any recent developments, in relation to non-lethal methods of population control, which mean that they could now effectively be applied to English seal populations where appropriate?*

   Controlling seal populations could potentially be achieved by non-lethal reduction of the birth rate or by excluding seals from sensitive habitats and regions. These sorts of interventions have been attempted on a trial basis, on small scales in the past by the Department of Fisheries and Oceans, Canada. Neither SMRU nor the Department of Fisheries and Oceans, Canada, have carried out any recent research on this issue. Different forms of chemical sterilization are available and some are known to be effective in seals. In the past, the technology for delivering chemicals has been deficient and, while this remains the case, we are aware that progress is being made. Nevertheless, the main uncertainties surround the potential secondary effects of this type of intervention on colony structure, which could have the unintended consequences of stimulating population growth.

   Answers to Scottish Government Q8 & Q9 above provided information about current research, funded by Scottish Government, being undertaken to use acoustic deterrent devices (ADDs) to exclude seals from sensitive regions. During 2007 a programme of laboratory and field based tests of aversive sounds specifically designed to act as seal deterrents with minimal impacts on non target species have been conducted. Initial results are promising and may lead to more effective local control.

   Trials of the effectiveness of commercially available ADDs for deterring seals from specific areas and as barriers to upstream movement of seals were described in answer to Scottish Government Q8 above

6. *What are the latest results from satellite tagging in respect of usage of specific coastal and marine areas around England by grey and common seals and whether or not these suggest potential foraging sites?*

   Substantial data sets on movements and foraging behaviour have been collected from both grey and common seals over the past 10 years. When combined with aerial survey information on distribution of haulout sites and relative abundance of each species at these sites, the tracking data allows us to develop population scale habitat usage maps for the entire UK. A detailed description of habitat preference modelling based on grey seals in the North Sea has recently been published[13] .

   In the absence of direct measures of food ingestion we can not unequivocally identify foraging sites, but on the basis of dive and movement patterns we believe that foraging occurs throughout the movement range. Individuals of both species show behaviour indicating a mixture of periods of wide ranging foraging movements with little or no concentration on particular areas and regular repeated foraging in discrete patches. Overall, the intensity of habitat useage is assumed to indicate level of foraging activity and allows identification of foraging hotspots.

7. *Are there any disease outbreaks which are likely to have a significant impact on English seal populations within the next 12 months and, if so, what practical mitigation measures might be possible and appropriate?*

   No disease outbreaks likely to impact on English seal populations have been identified in 2009. The discovery of 9 dead adult common seals in St Andrews Bay in June/early July of 2008 was an unusual event, but the pathology was unclear and no further disease related mortality has been observed. The unidentified disease outbreak in Swedish and Danish waters in 2007 has apparently ended and did not extend to the North Sea populations. Preliminary results of blood tests from harbour and grey seals caught at the Farne Islands and in St Andrews Bay suggest that PDV is not currently circulating in the UK.

Seal populations

8. *What progress has been made in integrating grey seal population abundance models or selecting between these models using grey seal survey work undertaken in 2009?*

   See answer to Scottish Government Q1 above.

9. *What progress has been made in improving monitoring methods and abundance estimates of the common seal population?*

   See answer to Scottish Government Q1 above.

10. *Is the decline in common seal numbers in specific local areas continuing or not and what is the position in other areas?*

    See answer to Scottish Government Q3 above

[13] Aarts et al. (2008) Estimating space use and habitat preference from wildlife telemetry data. Ecography **31**:140-160

11. *What are the latest results from research investigating the causes of the recent decline in common seals and how has this improved understanding of potential causes?*

    See answer to Scottish Government Q3 above

12. *What are the key questions about seal populations that remain to be addressed to better inform practical seal management issues?*

    The most urgent issues are those surrounding the rapid, widespread decline of common seal populations around the UK. The pertinent questions and suggested work programs to address them are described above (See answer to Scottish Government Q3 above) and in SCOS-BP 08/5.

    Additional questions concerning the relationship between harbour seal populations in the southern North Sea and the apparent southward shift in foraging effort by grey seals in summer months are likely to become more important in future.

    The reduction in size of the confidence intervals around the grey seal population estimate and the identification of pup mortality as the likely mechanism of density dependence means that understanding the patterns and causes of pup mortality is the main requirement for understanding and predicting future trends in grey seal populations.

The transient links between seal populations

13. *Any evidence that seals move between protected sites and have any passages been identified*

    Extensive studies of movements by both grey and harbour seals have been conducted over the past 20 years. Results indicate that a large proportion of the grey seals made extensive movements between protected areas. For example it is not uncommon for grey seals tagged in the Firth of Tay to move to the northern Isles and/or the southern North Sea, a range that encompasses several protected areas. For harbour seals, both the frequency and extent of movements are more restricted. There are however records of movements of adult seals between Orkney and Shetland, Orkney and Moray Firth and between all the English east coast sites. Pup movements may be more extensive, within the small sample satellite tagged in Orkney, individuals moved to Shetland, the Outer Hebrides and the Moray and Tay Firths.

    The rest of the answer depends on the meaning of ‘passages’. If ‘passages’ is interpreted to mean movement from one site to another, then the answer is given above. If ‘passages’ is interpreted to mean corridors, the answer is more complicated. Grey seals’ movement patterns are highly variable and the routes between distant foraging and/or haulout sites are not clearly defined nor apparently are they tightly constrained. For harbour seals in England there are frequent recorded movements between the Wash and both the Thames and Donna Nook sites. In addition, there are recorded movements of pups between all English east coast sites and some records of movements between the Wadden Sea and the English east coast.

14. *Is there any evidence of any risks posed to seals between protected areas that they move between*

    There is little information on risks in general and no information on risks specific to movements between protected areas.

Seal diet

15. *What work might be done to follow up and maintain the detailed picture of grey seal diet obtained from the major survey in 2002, given the infrequent opportunities for such surveys, and how useful would this be in informing seal management?*

    A Scotland wide, seasonally structured study of harbour and grey seal diet is underway with funding from Scottish Government. Sample collection has begun and initial results will be reported to SCOS 2011. A small amount of funding from Natural England has allowed SMRU to expand this study to haulout sites on the east coast of England.
    Additional laboratory based feeding trial will be required to estimate biases due to otoliths digestion and differential otoliths recovery rates.
    After the results of this study have been analysed, consideration should be given to a structured smaller scale continuous monitoring programme.

16. *How is the research into quantifying the consumption of salmon and sea trout smolts and salmon kelts by seals progressing?*

    See answer to Scottish Government Q7 above.

Seal legislation

17. *Does the Committee consider that there is a significant scientific requirement to change the current close seasons for each native seal species?*

    This question is not relevant to Scotland where the new Marine Bill has superseded the Conservation of Seals Act and does not include a specifically defined close season. In England and Wales the current close season for grey seals is 1st September to 31st December and for common seals it is 1st June to 31st August. The close season was designed to cover the breeding season for each species. There have been changes in the timing of breeding in grey seals but they have not moved outside the close season (with the exception of some colonies in SW Britain that have an extended breeding season). SCOS does not see a need to change the definition of the close season for grey seals. However, in some locations common seal pups will be born before 1 June and females in the late stages of pregnancy could be more vulnerable. SCOS recommends that the close season for common seals should be extended from 1 May until 31 August each year.

The Wash

18. *What is the latest estimate of seal population numbers in the Wash?*

    Results of surveys conducted in the Wash in 2009 are reported in SCOS-BP 10/3 and described briefly in answer to Scottish Government Q1 & Q3. The mean moult count in 2009 was **2,829** and represented an increase of approximately

40% over the mean 2008 count. Estimated pup production also increased, by 14% over the 2008 estimate.

19. *What are the latest results from research investigating the causes of the failure in the common seal population to recover from pre 2002 PDV outbreak numbers and how has this improved understanding of potential causes?*

    There has been no specific research to identify reasons for the failure to recover from the 2002 epidemic. Results of annual air surveys during the harbour seal moult (August) show that since 2000 the number of grey seals counted at haulout sites has increased dramatically, by an average of >25%p.a. This exceeds the growth in population associated with the rapidly expanding grey seal breeding populations in the southern North Sea. This increase means that the total amount of seal foraging effort by both species in combination has increased rapidly in the south-western part of the North Sea. This increase is due mainly to grey seal redistribution and may be partly responsible for the lower growth rates of English harbour seal populations compared to neighbouring European populations in the Wadden Sea. Direct competition has not been documented, but SMRU are assessing diet of the two species for overlap. Simultaneous telemetry tracking data are available in some locations and SMRU are examining those for evidence of foraging site overlap. Results from both studies will be reported to SCOS 2011.

Seals and salmon netting stations

20. *What research is currently available on interactions between seals and salmon netting stations and what new research might usefully be done in this area?*

    See answer to Scottish Government Q8 above

Seals and fish farms

21. *What research is currently available on interactions between seals and fin fish farms and what new research might usefully be done in this area?*

    See answer to Scottish Government Q9 above

Occurrences of seals in fresh water in relation to seasonal salmon runs

22. *What is the regularity of such an occurrence?*

    SCOS is not aware of any information on the frequency or timing of such occurrences in English rivers. The results of a study of this issue in Scottish rivers have recently been reported to Scottish Government and are described briefly in answer to Scottish Government Q7 & Q8 above.

23. *Where are the common freshwater locations of such occurrences?*

    Seals are regularly seen in freshwater in several Scottish rivers and English east coast rivers such as the Tyne, Humber and Great Ouse.

24. *What are effective deterrents in such freshwater locations?*

    Trials of the use of ADDs to deter seals in fresh water are underway, funded by Scottish Government. These are described briefly in answer to Scottish Government Q8 & Q9 above.

25. *What damage to salmon stocks is there as a result of seals in fresh water*

    SCOS is not aware of any information on the scale of damage to salmon stocks in English rivers. The results of studies in Scottish east coast rivers are described briefly in answer to Scottish Government Q7 above.

26. *What information, if any, do you have on numbers of complaints of seal damage in England?*

    SCOS is not aware of any information on numbers of complaints of seal damage in England.

27. *What information, if any, do you have on seals being killed in England to prevent damage to fisheries during the 'open seasons'?*

    SCOS is not aware of any information on numbers of seals being killed in England to prevent damage to fisheries during the 'open seasons'. No licence is required to kill seals outside the close season or for protection of fishing operations. There are no reporting requirements in the Conservation of Seals Act except for seals killed under licence.

28. *What information, if any, do you have on seals being killed under the 'fisherman's defence' provided by s.9(1)(c) of the Act?*

    SCOS is not aware of any information on numbers of seals being killed in England under the 'fisherman's defence'. Again, as this does not require a licence under the Conservation of Seals Act there are no reporting requirements in England and therefore no reliable records.

    The same information for Scotland and Wales would also be of interest if not available for England or for comparison with figures from England.

    All seal killing in Scotland must now be carried out under licence under the new Marine Bill and all such events, for whatever purpose must be reported.

Shooting

29. *How effective are the current firearm and ammunition minima stipulated in the act in relation to the termination of a seal?*

    Answer deferred pending the reports of the recent discussions in Edinburgh between Scottish Government, BASC, SSPCA and SMRU.

30. *What is the likelihood of someone killing a seal with the first shot if they are not a trained marksmen? – taking into account distance of the shot, an appropriate point of impact and stability of firing position.*

    This is impossible to answer. The level of training required will depend on the shooter's innate abilities. Shooting from unstable platforms and long range will dramatically reduce the likelihood of hitting a vital target and will obviously reduce the likelihood of a clean kill. The Scottish code of practice sets a range of 150 metres as the maximum allowable range for shooting at seals. Shooting from unstable/ unsuitable platforms is illegal.

31. *Is there any evidence of the noise from such firearms effectively deterring seals from a net?*

    No. There is anecdotal evidence that individual seals will habituate to the sound of gun fire. Evidence from seal haulout sites in Air Force bombing and gunnery ranges suggests that they can habituate to extreme fire arms noise.

Marine renewables

32. *What research is currently underway in relation to possible impacts of marine renewable energy development (offshore wind, wave or tidal) on seals?*

    Large amounts of research are underway in the UK and throughout the world. Telemetry based studies of movements and behaviour in the areas of high tidal and wave energy have recently been funded by both Scottish and Welsh Assembly Governments. Similar detailed telemetry and population survey studies have been conducted with funding from both public and industry bodies in Scotland and Northern Ireland with the specific aim of investigating fine scale movements in relation to tidal energy devices to inform collision risk models. Research into collision risk models is being conducted by Scottish Association of Marine Science.
    Background/baseline information studies of movements and population status and distribution of both species have been carried out throughout the UK as part of the SEA process with funding from DECC (previously DTI) and Scottish Government and SNH.

33. *What value might there be in developing guidance on possible mitigation measures to avoid disturbance to seals (and other marine mammals) during marine renewable construction or installation along the lines of the JNCC "Guidelines for Minimising Acoustic Disturbance to Marine Mammals from Seismic Surveys"? (see link - http://www.jncc.gov.uk/pdf/Seismic_survey_guidelines_200404.pdf)*

    All marine renewable energy projects have to meet assessment requirements of the national/local permitting process. These usually require an extensive environmental impact assessment that should include risk assessment and proposed mitigation measures. Information on effectiveness of a range of such measures would be useful to both the industry and the regulators. Unlike the marine seismic industry, most tidal devices will have significant individual requirements due to local conditions and device characteristics. It will therefore be a more difficult task than that faced by the authors of the seismic survey guidelines.

Climate change

34. *Is there any evidence of significant impacts on seal populations from climate change and are there practical adaptation measures that might be considered to alleviate these?*

    At present there is no direct evidence of significant effects of climate change on seal populations. However, indirect effects including new biotoxins, disease agents and parasites and possible changes in prey availability, which are difficult to detect and document, are a potential factor in the recent declines in common seals in Shetland, Orkney and along the northern North Sea coasts.

    The precautionary position would be to assume that climate change is more likely to add stresses to populations than to be either neutral or beneficial. In these circumstances, practical measures to actively manage human factors that may either intentionally or inadvertently add additional stress to seal populations need to be encouraged.

    In practice, we need to maintain or improve our power to detect effects through maintenance and improvement of data collection and ensuring that, whenever practical, we have the capacity quickly to introduce new management approaches. Some of changes suggested to the Conservation of Seals Act will help to enhance data flow and the power to detect changes. Depending upon how they are implemented, they could also result in a more rapid response to evidence of effects.

    SCOS recommends that a study of the effects of environmental factors on aspects of the breeding biology and reproductive success of grey and common seals should be made a priority.

ANNEX I

NERC Special Committee on Seals

Terms of Reference

1. To undertake, on behalf of Council, the provision of scientific advice to the Scottish Government and the Home Office on questions relating to the status of grey and harbour seals in British waters and to their management, as required under the Conservation of Seals Act 1970.

2. To comment on SMRU's core strategic research programme and other commissioned research, and to provide a wider perspective on scientific issues of importance, with respect to the provision of advice under Term of Reference 1.

3. To report to Council through the NERC Chief Executive.

Current membership

Professor Marc Mangel (Chair),	University of California, Santa Cruz;
Dr J Armstrong,	Fisheries Research Services;
Professor IL Boyd,	University of St Andrews;
Dr S Wanless	N.E.R.C. C.E.H, Edinburgh;
Dr J. Greenwood,	CREEM, University of St Andrews;
Professor J. Pemberton,	University of Edinburgh;
Professor D. Bowen,	Bedford Institute of Oceanography, Canada;
Dr A. Bjørge,	Institute of Marine Research, Bergen, Norway;
Dr G. Englehardt,	CEFAS, Lowestoft;
Dr S. Reid (Secretary),	NERC, Swindon

ANNEX II

Briefing papers for SCOS

The following briefing papers are included to ensure that the science underpinning the SCOS Advice is available in sufficient detail. *Briefing papers* provide up-to-date information from the scientists involved in the research and are attributed to those scientists. *Briefing papers* do not replace fully published papers. Instead, they are an opportunity for SCOS to consider both completed work and work in progress. It is also intended that current *briefing papers* should represent a record of work that can be carried forward to future meetings of SCOS.

<u>List of briefing papers appended to the SCOS Advice, 2010</u>

10/01 Grey seal pup production in Britain in 2009
C.D. Duck

10/02 Estimating the size of the UK grey seal population between 1984 and 2009, and related research.
L. Thomas

10/03 The Status of British Common Seal Populations in 2008
C.D. Duck & D. Thompson

10/04 An estimate of the size of the UK grey seal population based on summer haulout counts and telemetry data.
M. Lonergan, B. McConnell, C.D. Duck & D.Thompson.

10/05 Scaling up from pup counts to population trajectories for British grey seals.
M. Lonergan, D.Thompson, L.Thomas & C.D. Duck

10/06 The trophic transfer of biotoxins to Scottish harbour and grey seals.
A.J. Hall

10/07 The impact of killer whale predation on harbour seals in nearshore Shetland waters: evidence for dietary specialisation and estimated predation rates
Volker B. Deecke, Andrew D. Foote, Sanna Kuningas

10/08 Potential Biological Removal as a method for setting the impact limits for UK marine mammal populations.
L. Boyd, D. Thompson & M. Lonergan.

C.D. Duck and C.D. Morris

Grey seal pup production in Britain in 2009

NERC Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, St Andrews KY16 8LB

Summary
Between September and December 2009, repeat aerial surveys of 59 grey seal breeding colonies in Scotland were attempted by SMRU. Staff from Scottish Natural Heritage (SNH), National Trust, Lincolnshire Wildlife Trust and Natural England ground counted pups born at colonies in Shetland, Orkney (South Ronaldsay) the Farne Islands, Donna Nook, Blakeney Point and Horsey (East Norfolk).

Severe turbulence, while photographing Berneray, the most southerly island in the Outer Hebrides on the first survey of the season, damaged the camera mounting and we were unable to collect any data between 20 September and 23 October, missing two or three surveys of the inner and Outer Hebrides, the first two of the early Orkney colonies and the first of the late Orkney, Firth of Forth and Helmsdale colonies. As a result, we had insufficient data to derive new pup production estimates for all colonies in the Inner Hebrides, for seven out of 15 colonies in the Outer Hebrides and for two colonies on the North mainland coast of Scotland. For these colonies the 2008 production figures have been used as a proxy.

The total number of pups born at annually monitored colonies was estimated to be **42,296,** 1.94% higher than the 2008 total of 41,490.

The annually monitored colonies account for approximately 90% of grey seal pups born in the UK. A number of colonies are monitored less frequently for a number of reasons including difficulty of access (Wales, SW England) and the relatively small numbers of pups born (Table 2).

1. Surveys conducted in 2009

The locations of the main grey seal breeding colonies in the UK are shown in Figure 1.

Each year SMRU conducts aerial surveys of the major grey seal breeding colonies in Scotland to determine the number of pups born. On the first survey on 20 September 2009, while photographing the colony on Berneray, at the southern tip of the Outer Hebrides, severe turbulence damaged the camera motion-compensating system. We did not discover the problem until completion of the second round, when the processing laboratory (Kenton Photographic Colour Lab) informed us that the films had not been exposed. Locating the problem and developing an alternative system took approximately three weeks and the next successful survey was on 23 October.

As a result, at least two rounds of colonies in the Inner and Outer Hebrides and the first round of colonies in Orkney and in the Firth of Forth were missed. Colonies in the Inner Hebrides and seven colonies in the Outer Hebrides were only surveyed twice and Loch Eriboll and Eilean nan Ron, Tongue, were not surveyed at all. All other colonies were surveyed a minimum of three and mostly four times, allowing production estimates to be calculated. For colonies with two or fewer surveys, the 2008 production estimates were used as a proxy. There were no surveys of any of the minor colonies that are less frequently surveyed.

A small number of colonies are monitored annually by different organisations: National Trust staff count pups born at the Farne Islands (Northumberland) and at Blakeney Point (Norfolk), staff from the Lincolnshire Wildlife Trust count pups born at Donna Nook and staff from English Nature count pups born at Horsey, on the east Norfolk coast. Scottish Natural Heritage (SNH) staff coordinated a fifth survey of grey seal pups born in Shetland and SNH Orkney staff ground counted pups born on South Ronaldsay.

2. Estimated pup production

Numbers of pups born (pup production) at the regularly surveyed colonies is estimated each year from counts derived from the aerial photographs using a model of the birth process and the development of pups. The method used to obtain pup production estimates in 2009 was similar to that used in previous years. A lognormal distribution was fitted to colonies surveyed four or more times and a normal distribution to colonies surveyed three times.

The 2009 total pup production estimate for the annually monitored colonies was **42,296**, an increase of **1.94%** from 2008 (**41,490**; Table 1). The trajectory of pup production with 95% confidence limits at all the major breeding colonies in England and Scotland (excluding Loch Eriboll, Helmsdale and Shetland) between 1984 and 2009 is shown in Figure 2a. Figure 2b shows the long-term pup production trajectories at the main island groups from 1960 to 2009. Pup production from the main island groups since 1987 is shown in more detail in Figures 3a (Inner and Outer Hebrides and Orkney) and 3b (North Sea colonies). The time series of production estimates for the four regional island groups is given in Table 3.

For colonies not surveyed by air, pups were counted directly from the ground. Ground counts are conducted annually at the Farne Islands, Donna Nook, Blakeney Point, Horsey and South Ronaldsay in Orkney but less frequently in SW England and Wales due to the inaccessibility of breeding colonies (Figure 3b). SNH staff count pups in Shetland in a manner compatible with counts from aerially surveyed colonies and, for colonies with sufficient counts, production was estimated using the same modelling procedure.

In 2009, as in 2008, aerial surveys were carried out from an altitude of 335m rather than the usual 365m (1,100 rather than 1,200 feet). The increased resolution of the images improved the quality of counts, although the area covered on each photograph was reduced. Because of the improved counts, the model was run using the standard fixed 50% misclassification parameter (allowing for the misclassification of moulted pups as whitecoats), and re-run using a fixed 90% misclassification proportion as there were insufficient counts in 2009 to allow the misclassification parameter to be estimated by the model. Due to the improvement in model fitting, productions derived using the 90% classification proportion were used.

3. Trends in pup production

The differences in pup production at the main island groups are shown in Table 1. Between 2008 and 2009, total pup production at annually monitored colonies was estimated to have increased by +1.94% overall with the change varying from –4.71% in the Outer Hebrides to +21.0% at the Isle of May, Fast Castle and Inchkeith in the Firth of Forth (Figure 3a).

Pup production estimates in 2009 were good for most colonies in Orkney, Helmsdale and in the firth of Forth (Isle of May, Fast Castle and Inchkeith).

Figure 2a and 2b and Table 1 show that pup production at the annually monitored colonies is stabilising. Over the past five years, the only colonies that showed any significant increase were at the southern end of the North Sea, at Donna Nook, Blakeney Point and at Horsey (Table1). Since 2001, the increase at the Isle of May and Fast Castle was entirely due to the Fast Castle contribution.

Between 1984 and 1996, pup production estimates from annually monitored colonies showed a fairly consistent annual increase, with the notable exception of 1988 (Figures 2 and 3). More recently, there were declines in pup production in 1997 (mainly due to a reduction in the number of pups born in the Outer Hebrides), in 1999 (in all island groups), in 2002 (mainly in the Outer Hebrides) and in 2005 (primarily in the Orkney colonies). In the years following each of these declines, there was a marked increase in production the following year (of 9.5%, 11.5%, 7.4% and 3.9% in 1998, 2000, 2003 and 2006 respectively). The recovery in 2006 was considerably smaller than on previous occasions.

The overall annual percentage change in pup production at each of the main island groups over the past five years (between 2004 and 2009) is shown in Table 1. The overall annual change, for all colonies combined, was +1.41%. Locally, the change varied from –0.54% in the Inner Hebrides to +14.18% at the relatively small colonies of Donna Nook, Blakeney Point and Horsey. Changes for the two preceding five-year intervals, 1994 to 1999 and 1999 to 2004, are also shown in Table 1. These changes in five-yearly intervals are probably the best indication of the current trends in grey seal pup production.

4. Pup production model assumptions

The model used to estimate pup production from aerial survey counts of whitecoated and moulted pups assumes that the parameters defining the distribution of birth dates are variable from colony to colony and from year to year, but that those defining the time to moult and the time to leave the colony remain constant. The pup production estimates are sensitive to the value used for the latter parameter and there is, therefore, an argument for allowing this parameter to vary between colonies.

Previously (in 2001), we considered the effect of allowing the time-to-leave parameter to vary. However, although the resulting pup production trajectory is slightly lower, the variations in production are consistent between the two methods. The results presented here are consistent with the Advice provided in previous years and incorporate a fixed mean time-to-leave (and a variable standard deviation) derived from studies on the Isle of May.

Similarly, the proportion of white pups misclassified as moulted (or vice versa) can vary. Variation may be counter dependent or may be simply a function of the quality of the aerial photograph, the prevailing light conditions under which the photograph was taken and the orientation in which any pup might be lying. In 2008, there were sufficient counts (minimum of five) to allow the estimation model to select the most appropriate misclassification proportion. In 2009, there were insufficient counts (maximum of four) so the misclassification proportion was fixed at 0.75 (correctly classified).

When counts of pups from the ground were used to populate the model, using a higher percentage of correctly classified pups (90%) produced a better fit with lower confidence intervals. This is because individual pups can be observed for longer and the classification is very likely to be more accurate.

5. Confidence limits

Ninety-five percent confidence limits on the pup production estimates were 2.2% for the Inner Hebrides (the same as 2008), 3.6% for the Outer Hebrides, 4.9% for Orkney and 5.4% for colonies in the Firth of Forth (Figures 3a and 3b).

6. Pup production at colonies less frequently surveyed

Approximately 10% of all pups are born colonies not surveyed annually (Tables 2 and 4). Confidence intervals cannot be calculated for most of the estimates provided because they represent single counts. Loch Eriboll, Eilean nan Ron (Tongue) and the coast between Duncansby Head and Helmsdale are exceptions. Loch Eriboll and Eilean nan Ron were not surveyed in 2009 while the Helmsdale colonies were surveyed four times (Table 2). The 95% confidence interval for the production estimate for the Helmsdale colonies was 10.3% of the point estimates. Table 2 includes the total count for the colonies listed individually in Table 4 (Other colonies). These and other potential breeding locations are surveyed when flying time, weather conditions and other circumstances permit. Table 2 indicates that at least **5,247** pups were born at colonies in the U.K. that are not surveyed annually.

Note that Oronsay Strand is now included with the Inner Hebrides total and Inchkeith is included with the Isle of May and Fast Castle total.

Note that the surveys described here do not account for seals breeding in caves. Small groups of grey seals breed in caves in the Outer Hebrides, along the Sutherland coast, in Orkney and in Shetland.

7. Pup production in Shetland

In Shetland, SNH staff coordinated a team of volunteers who carried out boat and ground counts of a number of breeding colonies.

In 2009, five colonies were counted four times: Uyea, Rona’s Voe, Whalsay Skerries, part of Dale of Walls and Mousa. Papa Stour was counted twice and North Fetland once. This was the first opportunity to obtain repeat counts at Uyea, with acceptable weather coinciding with low tides. The pup production estimate for Shetland (Table 1)

As with previous surveys, the model was run using both a 50% and a 90% moulter classification. The model produced better fits to the counts, with lower confidence intervals, using the 90% classification. These estimates are in Table 5. Moulted pups are more likely to be correctly classified during ground counts because the counters are relatively close to the pups and can assess more accurately whether a pup has fully moulted or not.

The minimum pup production for Shetland in 2009 was **831** pups (Table 1). This figure is a combination of estimates from 2009 (Mousa, Uyea, Whalsay Is., Rona's Voe and Dale of Walls), 2007 (Papa Stour, South Bressay and NE Unst) and from 2004 (S Havra, Fitfull Head and Muckle Roe) and is a combination of modelled estimates, of maximum counts and of the most recent counts from previous surveys. This is likely to be an underestimate of grey seal pup production in Shetland, since a number of colonies were either not surveyed, or were not surveyed in their entirety. The frequently severe weather conditions during the autumn months may limit any potential increase in grey seal pup numbers on the restricted and exposed breeding beaches and caves in Shetland.

8. Grey seal pup production in Ireland

In the 2005 season, there was a major effort to determine the number of grey seal pups born in the Irish Republic, coordinated by Oliver O'Cadhla from the Coastal Monitoring Research Centre in Cork. Pup production was estimated to be 1,574 (O'Cadhla et al., 2007). Including an estimate of 100 pups born in Northern Ireland, this gives a total of just under 1,700 pups born in Ireland.

To complete the production estimate for the whole of the island of Ireland, in 2005 SMRU surveyed the breeding colonies on the east and south coast of Northern Ireland, as an extension of the existing grey seal survey of Scotland. Four surveys were carried out; the first has to be abandoned due to poor visibility. SMRU previously surveyed breeding grey seals in Northern Ireland in 2002.

In addition, the National Trust and the Northern Ireland Environment Agency (formerly the Environment and Heritage Service, Northern Ireland) conduct monthly boat surveys of seals in Strangford Lough. Approximately 40 grey seal pups are born inside Strangford Lough and here, grey seals appear to breed some 3-4 weeks earlier than those breeding on the small islands to the east of the Ards Peninsula.

Outside Strangford Lough, the main breeding colonies were on the Copeland Islands at the mouth of Belfast Lough and on the North Rocks off the east coast of the southern end of the Ards Peninsula. In 2005, on the Copeland Islands, the maximum pup count was 16 and on North Rocks the maximum count was 9 pups. These numbers were considerably lower than counts made in 2002 (14 and 26 pups respectively). These surveys suggest that approximately 100 grey seal pups were born in Northern Ireland in 2005 and Table 2 shows this estimated number.

9. Proposed surveys for 2010

In the 2010 breeding season, we propose to continue the current survey protocol and obtain between three and five counts for each of the main grey seal colonies in Scotland.

10. Acknowledgements

We are grateful to all those who helped collect or provided the data presented in this report. These include: John Walton and colleagues (National Trust, Farne Islands), Rob Lidstone-Scott (Lincolnshire Wildlife Trust, Donna Nook), David Wood (National Trust, Blakeney Point), Ron Morris and Bill Bruce (Forth Seabird Group, Forth inner islands). SNH Orkney and Shetland and volunteers for their 2009 grey seal pup survey data. We are grateful to Bill Giles and Gordon Smith who, once again, enthusiastically and expertly piloted the grey seal survey aircraft.

References

Ó Cadhla, O., Strong, D., O'Keeffe, C., Coleman, M., Cronin, M., Duck, C., Murray, T., Dower, P., Nairn, R., Murphy, P., Smiddy, P., Saich, C., Lyons, D. & Hiby, A.R. (**2007**). An assessment of the breeding population of grey seals in the Republic of Ireland, 2005. Irish Wildlife Manuals No. 34. National Parks & Wildlife Service, Department of the Environment, Heritage and Local Government, Dublin, Ireland.

Table 1. Pup production estimates for colonies in the main island groups surveyed in 2009. The overall average annual changes, over successive 5-year intervals are also shown. These annual changes represent the exponential rate of change in pup production. The total for the North Sea represents the combined production estimates for the Isle of May, Fast Castle and Inchkeith in the Firth of Forth and for the Farne Islands, Donna Nook, Blakeney Point and Horsey in east England. There were insufficient surveys of all colonies in the Inner Hebrides and for seven colonies in the Outer Hebrides in 2009, production estimates from 2008 were used.

Location	2009 production	2008 production	Overall annual change in pup production: From previous year	Overall annual change in pup production: For previous 15 years, in 5 year intervals		
			2008-2009	1994-1999	1999-2004	2004-2009
Inner Hebrides	**3,396**	**3,396**	n/a	+0.5%	+4%	-0.1%
Outer Hebrides	**12,113**	**12,712**	-4.71%	-0.5%	+1%	-0.3%
Orkney	**19,150**	**18,765**	+2.05%	+6.65%	+4.29%	+0.49%
Isle of May, Fast Castle, Inchkeith	**4,047**	**3,346**	+21.0%	+11.84%	+4.79%	+8.01%
Farne Islands	**1,346**	**1,318**	+2.12%	-0.52%	+4.90%	+3.57%
Donna Nook + Blakeney Pt + Horsey	**2,244**	**1,953**	+14.9%	+11.28%	+17.86%	+14.18%
North Sea (i.e. previous 3 areas)	**7,637**	**6,617**	+15.41%	+5.0%	+7.8%	+9.2%
Total	**42,296**	**41,490**	**+1.94%**	**+2.8%**	+3. 5%	**+1.3%**

Table 2. Pup production estimates for breeding colonies surveyed less regularly.

Location	Location and year of most recent survey	Pup production
[1]Mainland Scotland	[1]Helmsdale (Duncansby Head to Helmsdale, 2008	1,098
	[1]Loch Eriboll, Eilean nan Ron (Tongue) 2008	557
Other colonies	Various, see Table 4	761
[2]Shetland	2009	831
South-west England	South-west England (incl Lundy),	250 (est.)
Wales	All Wales, 1994-2005	1,650 (est.)
Northern Ireland	2005	100 (approx.)
Total		**5,247**

[1]Loch Eriboll, Eilean nan Ron and Helmsdale are surveyed annually with production estimates derived using the same modelling process as for the main breeding colonies.

[2]See Table 5 for details of grey seal pup production in Shetland.

Table 3. Estimates of pup production for colonies in the Inner and Outer Hebrides, Orkney and the North Sea, 1960-2009.

YEAR	Inner Hebrides	Outer Hebrides	Orkney	North Sea	Total
1960			2048	1020	
1961		3142	1846	1141	
1962				1118	
1963				1259	
1964			2048	1439	
1965			2191	1404	
1966		3311	2287	1728	7326
1967		3265	2390	1779	7434
1968		3421	2570	1800	7791
1969			2316	1919	
1970		5070	2535	2002	9607
1971			2766	2042	
1972		4933		1617	
1973			2581	1678	
1974		6173	2700	1668	10541
1975		6946	2679	1617	11242
1976		7147	3247	1426	11820
1977			3364	1243	
1978		6243	3778	1162	11183
1979		6670	3971	1620	12261
1980		8026	4476	1617	14119
1981		8086	5064	1531	14681
1982		7763	5241	1637	
1983				1238	

Table 3 continued.

YEAR	Inner Hebrides	Outer Hebrides	Orkney	North Sea	Total
1984	1332	7594	4741	1325	14992
1985	1190	8165	5199	1711	16265
1986	1711	8455	5796	1834	17796
1987	2002	8777	6389	1867	19035
1988	1960	8689	5948	1474	18071
1989	1956	9275	6773	1922	19926
1990	2032	9801	6982	2278	21093
1991	2411	10617	8412	2375	23815
1992	2816	12215	9608	2437	27075
1993	2923	11915	10790	2710	28338
1994	2719	12054	11593	2652	29018
1995	3050	12713	12412	2757	30932
1996	3117	13176	14273[1]	2938	33504[1]
1997	3076	11946	14051	3698	32771
1998	3087	12434[2]	16367	3989	35877[2]
1999	2787	11759	15462	3380	33388
2000	3223	13396	16281	4303	37210
2001	3032[3]	12427	17938	4134	37531[3]
2002	3096	11248	17942[4]	4520[4]	36816[4]
2003	3386	12741[5]	18652[5]	4805[5]	39584[5]
2004	3385	12319	19123[3]	4921	39748
2005	3387	12297[6]	17644[6]	5132	38460[6]
2006	3461	11612	19332	5322	39727
2007	3071	11189	18952	5560	38772
2008	3396	12712	18765[7]	6617	41450
2009	3396[8]	12113[8]	19150	7637[8]	42296

[1]Calf of Flotta included with Orkney total (start in 1996).
[2]Berneray and Fiaray (off Barra) included in the Outer Hebrides total (start in 1998).
[3]Oronsay included with Inner Hebrides (start in 2001).
[4]South Ronaldsay included in the Orkney total; Blakeney Point and Horsey (both Norfolk) included with North Sea (start in 2002).
[5] North Flotta, South Westray, Sule Skerry included with Orkney; Mingulay included with Outer Hebrides (start in 2003)
[6] Pabbay included with Outer Hebrides; Rothiesholm (Stronsay) included with Orkney (start in 2005).
[7] New colony on Hoy included with Orkney
[8] 2008 production estimates were used as a proxy for all colonies in the Inner Hebrides and for 7 colonies in the Outer Hebrides for which new production estimates could not be derived. Oronsay Strand included with Inner Hebrides; Inchkeith included with North Sea.

Table 4. Scottish grey seal breeding sites that are not surveyed annually and/or have recently been included in the survey programme. Most recent data are in bold type. There were no 2009 updates to this Table.

	Location	Survey method	Last surveyed	Number of pups counted
Inner Hebrides	Loch Tarbert, Jura	SMRU visual	2003, 2007	10, **4**
	West coast Islay	SMRU visual	1998, every 3-4 years	None seen
	Oronsay Strand	SMRU photo	2005, 2006, 2007, 2008	40, 9, 47[1]
	Ross of Mull, south coast	SMRU visual	1998, infrequent	None seen
	Treshnish small islands, incl. Dutchman's Cap	SMRU photo & visual	annual	~20 in total
	Staffa	SMRU visual	1998, every other year	~5
	Little Colonsay, by Ulva	SMRU visual	1998, every 3-4 years	6
	Meisgeir, Mull	SMRU visual	1998, every 3-4 years	1
	Craig Inish, Tiree	SMRU photo	1998, every 2-3 years	2
	Cairns of Coll	SMRU photo	2003, 2007	22, **10**
	Muck	SMRU photo	1998, 2005	36, **18**
	Rum	SNH ground	2005, annual	10-15
	Canna	SMRU photo	2002, 2005	54, **25**
	Rona	SMRU visual	1989, infrequent	None seen
	Ascrib Islands, Skye	SMRU photo	2002, 2005, 2007, 2008	60, 64, 42, **64**
	Fladda Chuain, North Skye	SMRU photo	2005, 2007, 2008	73, 43, **129**
	Trodday, NE Skye	SMRU photo	2008 New	**55**
	Heisgeir, Dubh Artach, Skerryvore	SMRU visual	1995, 1989, infrequent	None None
Outer Hebrides	Sound of Harris islands	SMRU photo	2002, 2005, 2007, 2008	358, 396, (194)[2], **296**
	St Kilda	Warden's reports	Infrequent	Few pups are born
	Shiants	SMRU visual	1998, every other year	None
	Flannans	SMRU visual	1994, every 2-3 years	None
	Bernera, Lewis	SMRU visual	1991, infrequent	None seen
	Summer Isles	SMRU photo	2002, 2003, 2005, 2006, 2007, 2008	50, 58, 67, 69, 25, **73**
	Islands close to Handa	SMRU visual	2002	10
	Faraid Head	SMRU visual	1989, infrequent	None seen
	Eilean Hoan, Loch Eriboll	SMRU visual	1998, annual	None
	Rabbit Island, Tongue	SMRU visual	2002, every other year	None seen
Orkney	Sanday, Point of Spurness	SMRU photo	2002, 2004, 2005, 2006, 2007, 2008	10, 27, 34, 21, 8, **17**
	Sanday, east and north	SMRU visual	1994, every 2-3 years	None seen
	Papa Stronsay	SMRU visual	1993, every 3-4 years	None seen
	Holm of Papa, Westray	SMRU visual	1993, every 3-4 years	None seen
	North Ronaldsay	SMRU visual	1994, every 2-3 years	None seen
	Eday mainland	SMRU photo	2000, 2002	8, 2
Others	Firth of Forth islands esp. Inchkeith & Craigleith (by North Berwick)	SMRU photo, Forth Seabird Group	Infrequent, 1997 2003, 2004, 2005, 2006, 2007, 2008	<10, 4 86, 72, 110, 171, 206, 50, **34**
Total				**761**

[1]Pup production calculated from four counts

[2] 2005 count used in total as pups were missed in 2007

Table 5. Pup production estimates and maximum pup counts for grey seal colonies in Shetland from 2004 to 2008. Frequent severe gales in 2005 restricted the opportunity to count and probably removed significant numbers of pups from some of the breeding beaches. The estimated pup productions for Uyea in 2005 and 2006 are clearly underestimates as only those breeding on beaches that were visible from the mainland could be counted. These data were provided by SNH staff (assisted by SMRU in 2004) and by a team of hardy volunteers.

Shetland colony	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**
	Estimated production (90% moulter classification)	**Estimated production** (90% moulter classification)	**Estimated production** (90% moulter classification)	**Estimated production** (90% moulter classification)	**Estimated production** (90% moulter classification)	**Estimated production** (90% moulter classification)
Papa Stour	196	135	153	168	107 (max count)	88 (max count)
Dale of Walls	66	43	18 (max count)	36 (max count)	10 (max count)	33
Muckle Roe	23	no count	no count	no count	no count	no count
Rona's Voe	106	83	50	57	45 (max count)	82
Mousa	140	117	156	128	122 (max count)	178
Fetlar	50	32	21 (max count)	23 (max count)	no count	10 (max count)
Whalsey Islands	102 (max count)	72	77	103	119	95
South Havra	4 (max count)	no count	no count	no count	no count	no count
Fitful Head	18 (max count)	no count	no count	no count	no count	no count
Uyea (N. Mainland)	238 (max count)	122 (part only)	114 (part only)	101 (part only)	69 (max count, part only)	215 (all)
NE Unst				3 (max count)	no count	no count
Noss				2 (max count)	no count	no count
Total max counts	**362**	**260**	**299**	**324**	**324**	**37**
Modelled total	**581**	**505**	**459**	**479**	**495**	**794**
Estimated production (combination using most recent accurate estimates)	**943**	**765**	**758**	**803**	**819**	**831**


Grey seal breeding colonies
in Britain
Figure 1
Colonies asterisked are potential
Special Areas of Conservation
Major colonies encircled are surveyed annually
Shetland
*Faray & Holm of Faray
*North Rona
Orkney (27)
Outer
Hebrides
(15)
Loch Eriboll
Mainland (3)
Helmsdale & Berriedale
*Monach
Isles
Pabbay
Scotland
Major colony
Minor colony
*Treshnish Isles
Inner
Hebrides
(11)
*Isle of May
Fast Castle
*Farne Islands
North Sea
(7)
Ireland
Donna Nook
Blakeney
Point
*North Pembroke
Coast
England
Horsey
Wales
*Ramsey
*Skomer
Lundy
Isles of Scilly

Figure 2a. Total estimated grey seal pup production, with 95% confidence limits, at all the major, annually monitored colonies in Scotland and England from 1984 to 2009.

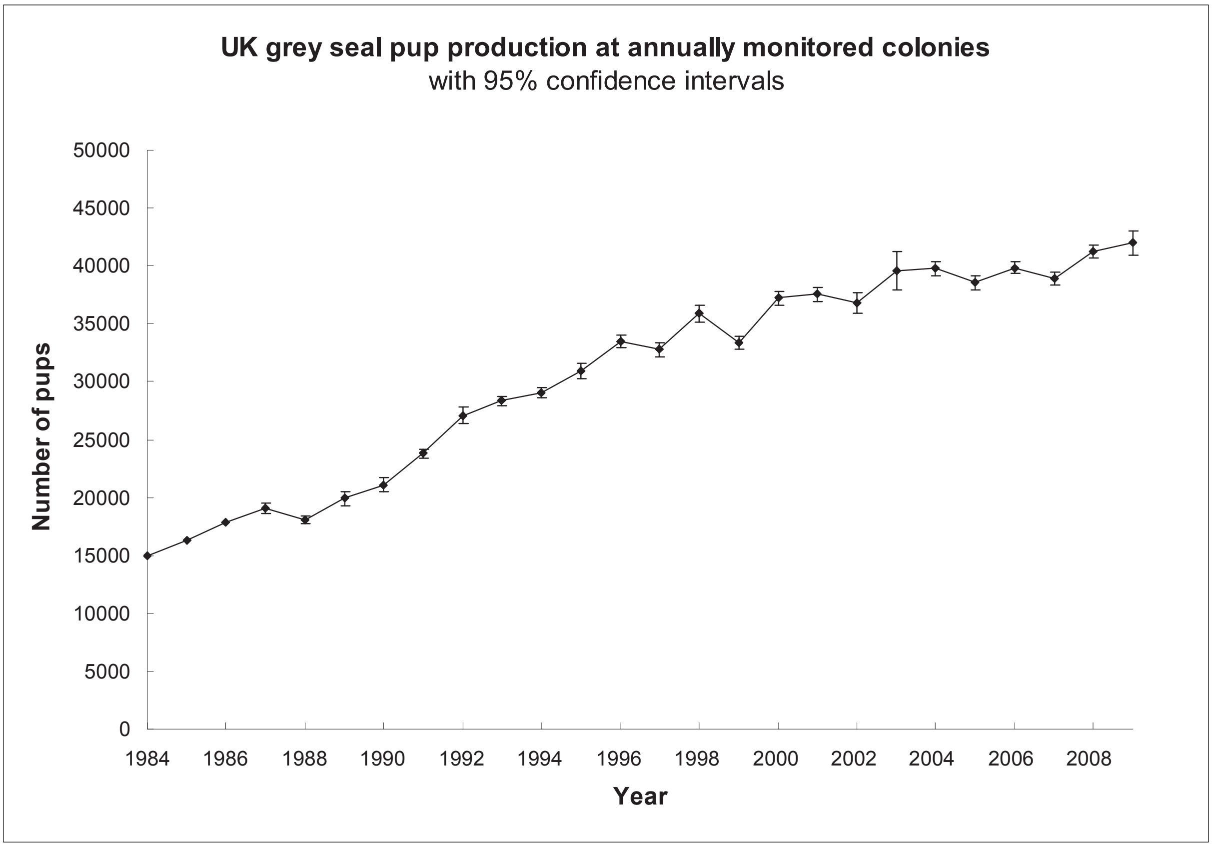


Figure 2b. Grey seal pup production trajectories from 1960 to 2009.

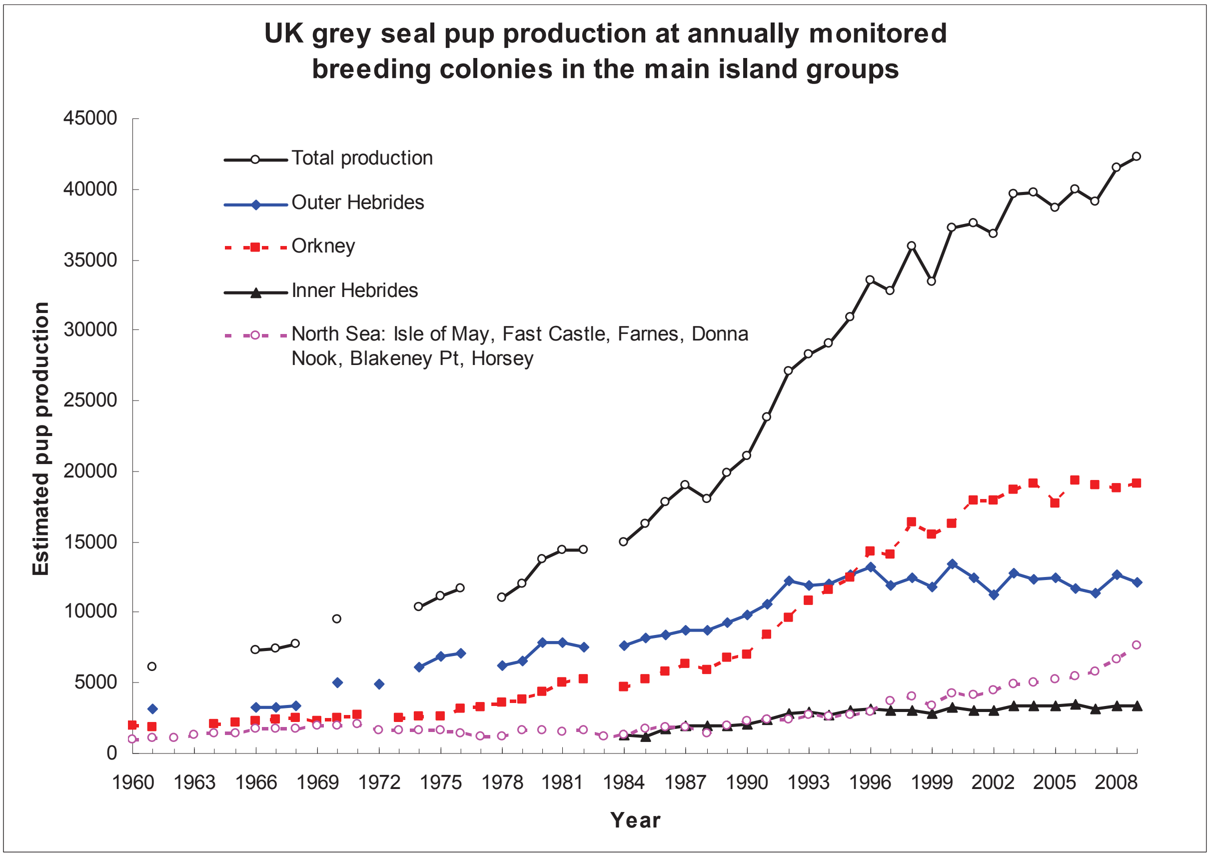
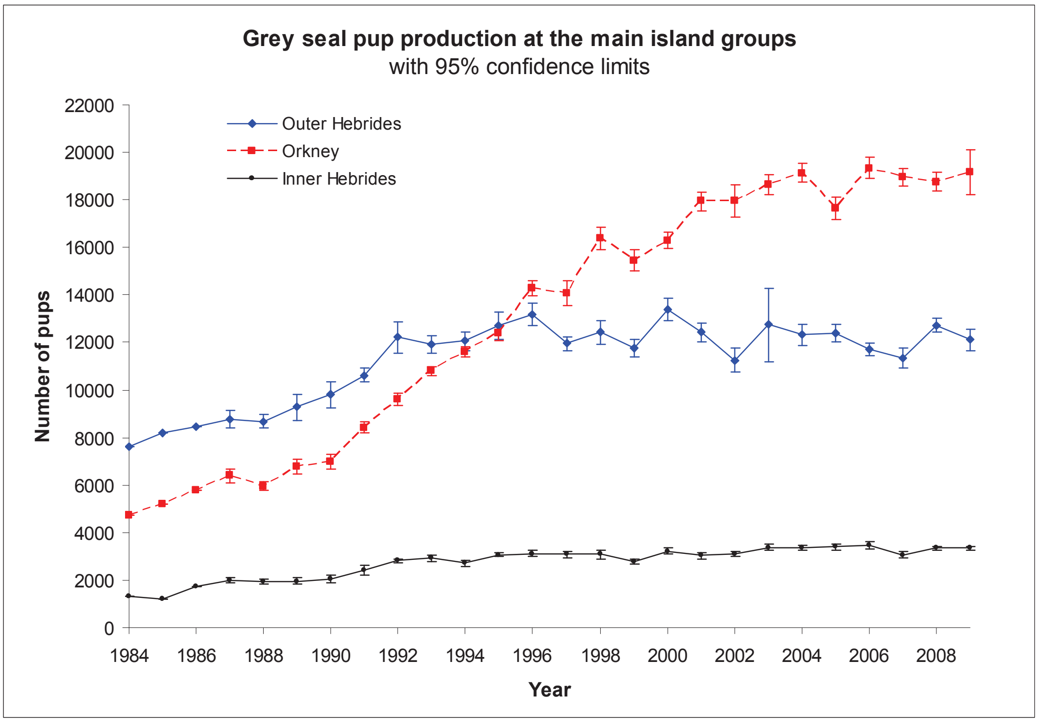
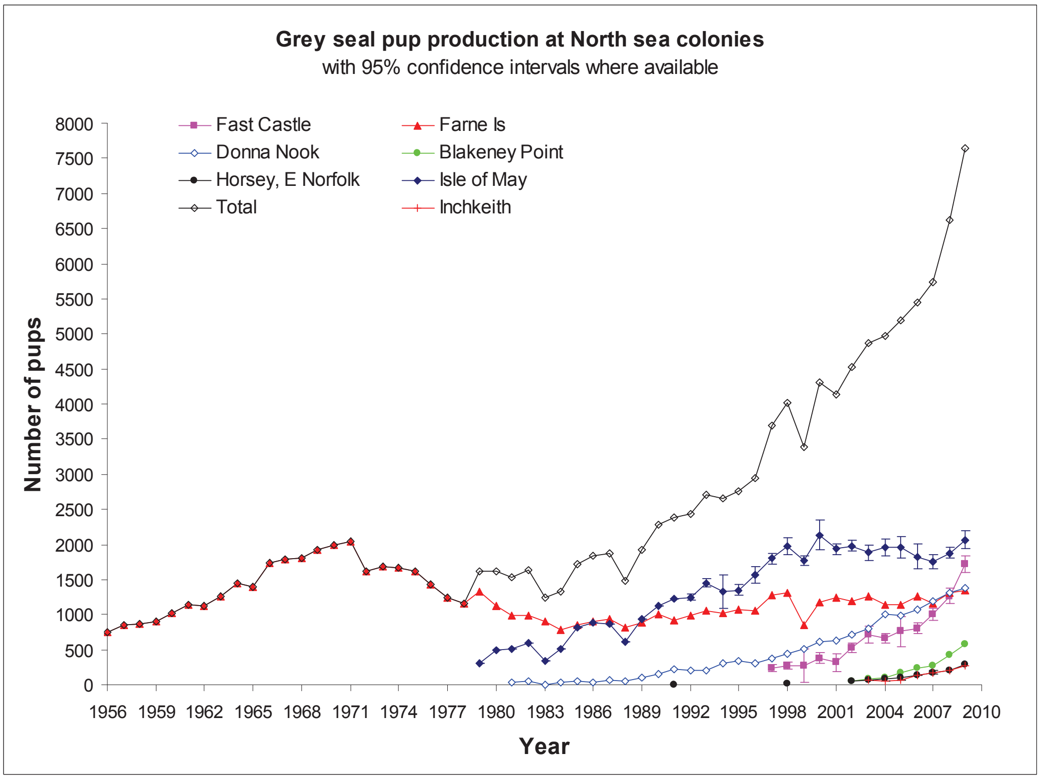


Figure 3. Trends in pup production at the major grey seal breeding colonies since 1984. Production values are shown with their 95% confidence limits where these are available. These limits assume that the various pup development parameters involved in the estimation procedure remain constant from year to year. Although they therefore underestimate total variability in the estimates, they are useful for comparing the precision of the estimates in different years. Note the difference in scale between Figures 3a and 3b.

3a) Outer Hebrides, Orkney and Inner Hebrides



3b) North Sea colonies



Len Thomas

Estimating the size of the UK grey seal population between 1984 and 2009.

NERC Sea Mammal Research Unit and Centre for Research into Ecological and Environmental Modelling, University of St Andrews, St Andrews KY16 8LB

Summary

We fitted two Bayesian state-space models of British grey seal population dynamics to two sources of data: (1) regional estimates of pup production from 1984 to 2009, and (2) an independent estimate assumed to be of total population size just before the 2008 breeding season. One model (EDDSNM) allowed for density dependence in pup survival, while the other (EDDFNM) allowed for density dependence in female fecundity. Both models had flexible forms of density dependence, but allowed no movement of recruiting females between regions. Including the independent estimate of population size influenced estimates of population size for the entire time series in both models, and strongly facilitated model selection. The estimated adult population size in 2009 was 106,200 (95% CI 82,000-138,700) for the EDDSNM model and 206,700 (95% CI 181,400-243,000) for the EDDFNM model. The posterior model probabilities were 1.0 and 0.0 respectively; hence the model-averaged estimate of total population size was identical to that for the EDDSNM model. These results assume an adult sex ratio of 57.8% females. If, instead, a uniform prior of between 50% and 100% females is used, then the estimated adult population size becomes closer to the independent estimate, particularly for the EDDSNM model, but the posterior estimated adult sex ratio becomes unfeasibly female-biased (84% (95% CI 58-99%) for EDDSNM and 93% (77-100%) for EDDFNM).

Introduction

This paper presents updated estimates of population size and related demographic parameters, based on the models and fitting methods of Thomas and Harwood (2009), but updated to include 2009 pup count estimates (Duck 2010), and additional data from an independent estimate of population size obtained from summer haulout counts and telemetry data (Lonergan *et al.* 2010). Models are specified using a Bayesian state space framework, and fitted using a Monte Carlo particle filter. Only the two best models from previous years' briefing papers are used: one assumes density dependent pup survival and the other density dependent fecundity. Both allow extended forms of Beverton-Holt-like density dependence and assume no movement of females between regions; hence they are abbreviated EDDSNM and EDDFNM respectively. Informative priors are used on many model parameters. We compare the fit of the two models by calculating posterior model probabilities, and make joint inference about population size from the two models combined. We also investigate the consequences of allowing for uncertainty in the adult sex ratio, a quantity that has been assumed to be fixed in previous analyses.

Materials and Methods

Process Models

The population dynamics models are described fully in Thomas and Harwood (2008) and papers cited therein. In summary, they track seal population numbers in 7 age groups (pups, age 1-5 females and age 6+ females) in each of four regions (North Sea, Inner Hebrides, Outer Hebrides and Orkney). There are three population sub-processes: (1) survival, (2) ageing and pup sexing and (3) breeding. (The models of Thomas and Harwood 2008 also included movement of recruiting females between regions, but we assume no movement in the current models.) The two models each have 8 parameters. They share 6: adult survival, ϕ_a, one carrying capacity parameter-related parameter for each region, β_1 - β_4, and a parameter, ρ, that dictates the shape of the density-dependent response. The model with density dependent survival (EDDSNM) has a parameter for maximum pup survival $\phi_{p\max}$ and another for constant fecundity α, while the model with density dependent fecundity (EDDFNM) has a parameter for maximum fecundity $\alpha_{\max}$ and constant pup survival ϕ_j.

Neither model describes the dynamics of adult male seals. To obtain an estimate of total adult population size, we follow previous briefing papers in multiplying the estimate of adult female population size by a fixed value of 1.73 (Hiby and Duck, unpublished) – i.e., assuming that females make up 57.8% of the adult population. For this briefing paper, we also have available an independent estimate of total population size, potentially allowing the sex ratio to be estimated. We make some initial steps in this direction, as detailed in a later section.

Data, Observation Models, and Priors
One source of input data was the pup production estimates for 1984-2009 from Duck (2010), aggregated into regions. This was assumed to be normally distributed with mean equal to the true pup production in each region and year, and constant coefficient of variation (CV). In previous briefing papers, the value for this CV was first estimated based on a run of a simple model (DDS), and then fixed at the estimated value to facilitate model comparison. For this paper, we used the estimated CV value of 10.64% from Thomas and Harwood (2009) to save time.

The second source of input data was the estimate of adult population size obtained by Lonergan *et al.* (2010) from summer haulout counts and telemetry data. The haulout data were collected between 2007 and 2009, with the majority from 2008. The telemetry data was collected between 1995-2008. Since it is not possible reliably to relate regional estimates of population size from the haulouts to regional estimates during breeding, only the total population estimate was used. This had to be attached to one year, and so we assumed it corresponded to the population size in 2008. For simplicity (and since mortality rates between summer and the start of the breeding season is likely very low) we assumed the estimate was of population size just before the start of the 2008 breeding season. Lonergan *et al.* (2010) gave an estimate of 77,427 with 95% CI 54,000-118,300. We approximated this by assuming the estimate comes from a right-shifted gamma distribution, with a shift of 44,482.15, a scale parameter of 8,506.34 and a shape parameter of 3.8889. This gave a distribution with the correct mean, and with lower and upper 2.5% lower and upper quantiles of 53,300 and 117,600 respectively. The above distribution implies a CV on the total population size estimate of 21.62%.

Prior distributions for the process model parameters were the same as those of Thomas and Harwood (2009) and are given in Table 1. We followed previous briefing papers in using a re-parameterization of the model to set priors on the numbers of pups at carrying capacity in each region, denoted χ_r for region *r*, rather than directly on the β s.

Prior distributions for the states were generated using the 1984 data, as described by Thomas and Harwood (2008).

Table 1. Prior parameter distributions

Param	Distribution	Mean	Stdev
ϕ_a	Be(22.05,1.15)	0.95	0.04
$\phi_{j\max}, \phi_j$	Be(14.53,6.23)	0.7	0.1
χ_1	Ga(4,2500)	10000	5000
χ_2	Ga(4,1250)	5000	2500
χ_3	Ga(4,3750)	15000	7500
χ_4	Ga(4,10000)	40000	20000
ρ	Ga(4,2.5)	10	5
$\alpha, \alpha_{\max}$	Be(22.05,1.15)	0.95	0.04

Fitting Method
We used the particle filtering algorithm of Thomas and Harwood (2008). This involves simulating samples from the prior distributions, projecting them forward in time according to the population model, and then resampling and/or reweighting them according to their likelihood given the data. An identical algorithm to that of Thomas and Harwood (2008) was used for the pup count data, and the additional adult data was included by reweighting the final output according to the likelihood of the estimated 2008 population size given the estimate of Lonergan *et al.* (2010).

The final output is a weighted sample from the posterior distribution. Many samples are required for accurate estimation of the posterior, and we generated 750 runs of 1,000,000 samples for each model (Table 2). Rejection control was used to reduce the number of samples from the posterior that were required to be stored, and the effective sample size of unique initial samples was calculated to assess the level of Monte Carlo error, as detailed in Thomas and Harwood

(2008). As an additional check, we divided the samples into 3 sets of 250 runs and examined the difference in results among these 3 sets.

Model comparison and model outputs

We calculated the posterior model probability for each model, assuming equal prior weights. (Since both models had the same number of parameters, the two prior weighting schemes used in Thomas and Harwood (2009) would both produce equal weights.)

We also present posterior estimates of the model parameters and estimated pup production from 1984-2009. Lastly, we present model averaged estimates of adult population size, combining the models according to their posterior model probabilities.

To evaluate the effect of the additional independent estimate of total population size, we calculated results both with and without this datum. These are based on the same set of particles, either reweighted to include the additional data or not.

Estimating the adult sex ratio

The population dynamics model fitted to pup production data allows estimation of the number of adult females, but this must be scaled by an assumed adult sex ratio to produce an estimate of total population size. In previous briefing papers, it was assumed the female proportion of the adult population was 0.58. The presence of an independent estimate of total population size potentially allows the sex ratio to be estimated. To illustrate this, we re-processed the simulation outputs for both models, allowing the female proportion to be a random variable with a uniform prior distribution with limits 0.5 (equal sex ratio) and 1.0 (all adults are females).

Results

Unique ancestral particle numbers

The number of particles retained and effective sample sizes (ESS) when only the pup count data was used (Table 2) were rather greater than those used in recent briefing papers, due to the larger number of simulations performed; however effective sample size was reduced by inclusion of the independent adult population size estimate. This is not surprising given that the estimate was some distance from that implied by the pup count data and priors alone, especially for the EDDFNM model (see later in *Results*).

Table 2. Number of particles simulated (K), number saved after final rejection control step (K), number of unique ancestral particles (U), effective sample size of unique particles from pup count data alone(ESS_{u1}), and with pup production data and the independent total population estimate (ESS_{u2}).*

Model	K ($\times10^7$)	K^* ($\times10^7$)	U ($\times10^4$)	ESS_{u1}	ESS_{u2}
EDDSNM	750	7.38	5.97	630.9	271.0
EDDFNM	750	3.53	2.22	372.5	108.1

Despite the small ESS, when the data was divided into three and key results examined, they were almost all found to be identical to 2-3 significant figures. For example, the estimates of total population size in thousands under the EDDFNM model, after the addition of the independent population estimate (i.e., the case with the lowest ESS) were 206.8, 206.2 and 203.7 respectively. An exception was the posterior model probabilities – these are reported on in a later section.

Comparison of models for density dependence with and without the total population estimate

Smoothed posterior means and 95% credible intervals for the two models are shown in Figure 1, both with and without the additional total population estimate. Both models showed similar fits to the pup production data, both between models and with or without the total population estimate. The models broadly provide a good fit to these data, but there are some clear deficiencies: neither adequately captures the rapid rise and sudden levelling off in pup production in the Hebrides during the early 1990s, nor the recent levelling off in Orkney; both over-fit pup production in the North Sea in the late 1990s and early 2000s. There is some evidence that the EDDSNM model tracks the observations slightly better than the EDDFNM, for example in the 2000s in the Inner Hebrides, but the differences between models are very slight.

Posterior parameter estimates are shown in Figure 2. Addition of the 2008 adult data changed the posterior estimates somewhat; there is also some evidence that not enough particles have been run in the non-smoothness of the posterior histograms (e.g., for ϕ_j).

Posterior model probabilities for the two models are shown in Table 3. Based on pup production data alone, there appears to be strong support for the EDDSNM model (difference in negative log integrated likelihood, -LnIL, of 5.3; posterior model probability 0.99), although there is some Monte Carlo error associated with this figure: dividing the simulations into three parts yielded changes in posterior model probability of up to 10 percentage points. When the total population estimate is included, support for the EDDSNM model becomes extremely strong, with the -LnIL values differing by 165 points and posterior model probability for the EDDSNM model of 1.0. With such a large difference in –LnIL, the posterior model probabilities are unlikely to be affected by Monte Carlo error, and indeed we obtained a weighting of 1.0 for EDDSNM in all three subsets of the simulations.

Table 3. Number of parameters, negative log integrated likelihood (-LnIL) and posterior model probabilities (p(M)) for fit to pup production data from 1984-2009 2009 and the additional total population estimate from 2008

Model	# params	-LnIL	p(M)
Pup production data alone			
EDDSNM	8	796.79	0.99
EDDFNM	8	802.09	0.01
Pup production and total population estimate			
EDDSNM	8	1080.20	1.00
EDDFNM	8	1245.53	0.00

Estimates of total population size

Estimates of total population size from the EDDFNM model were approximately twice those from the EDDSNM model, based on pup production data alone (Table 4 and Figure 3). Inclusion of the independent estimate of total population size from 2008 brought the estimates down by approximately 10% for the EDDSNM model and 20% for the EDDFNM model; the effect on the upper credibility interval of the EDDFNM model was particularly marked (Figure 3). Because the posterior model probability for the EDDSNM model was effectively 1.0, the model-averaged estimates of population size were identical to the EDDSNM estimates (given in the Appendix).

Estimating the adult sex ratio

For both population models, when adult sex ratio is not assumed fixed, the posterior sex ratio was considerably more female-based than the prior (Figure 4), and also than the previously-assumed value of 0.57: posterior mean (and 95%CI) was 0.84 (0.58-0.99) and 0.92 (0.77-1.00) for the EDDSNM and EDDFNM models respectively. Population size was thus made closer to the independent estimate: posterior mean (and 95% CI) total population size in 2009 was 81.5 (58.4-117.5) and 139.3 (118.0-172.2) thousand seals for the two models.

Table 4. Estimated size, in thousands, of the British grey seal population at the start of the 2009 breeding season, derived from models fit to pup production data from 1984-2009 and the additional total population estimate from 2008. Numbers are posterior means with 95% credibility intervals in brackets.

	EDDSNM	EDDFNM
Pup production data alone		
North Sea	22.4 (14.6 30.6)	30.7 (23.9 41.0)
Inner Hebrides	8.8 (6.9 10.7)	21.8 (17.7 29.7)
Outer Hebrides	32.3 (25.7 38.8)	87.0 (69.4 128.6)
Orkney	56.3 (40.7 83.4)	107.7 (86.6 145.2)
Total	119.8 (87.9 163.5)	247.1 (197.6 344.4)
Pup production and total population estimate		
North Sea	19.9 (13.3 28.2)	25.1 (21.2 30.4)
Inner Hebrides	8.1 (6.3 9.9)	18.4 (15.6 21.6)
Outer Hebrides	29.7 (23.9 36.2)	71.9 (63.5 84.3)
Orkney	48.7 (37.1 64.5)	90.0 (77.4 105.7)
Total	106.3 (80.6 138.8)	205.5 (177.6 242.0)

Discussion

Previous (unpublished) work has suggested that pup production data alone is not sufficient to distinguish between models of density dependent survival and fecundity. Yet, results given here show that given these models and priors, there is strong support for the EDDSNM model based on pup production data alone. Although Monte-Carlo error in model selection statistics may be a factor, this deserves further investigation.

Including the independent estimate of total population size caused the model selection to unambiguously favour the EDDSNM model. Population estimates were also reduced somewhat from those under that model based on pup production data alone, but the posterior mean estimate for 2008 from the model was still approximately 28,000 seals higher than the independent estimate. The extent to which the posterior estimates of total population size move towards the independent 2008 estimate depends upon the precision of this estimate relative to the precision of the estimate of total population size arising from the population dynamics model and pup production data. The latter is governed by the assumed precision of the pup production data and the amount of information about total population size contained in the priors on model parameters.

One obvious flaw in the initial models fitted here is that no uncertainty is attributed to the multiplier used to scale up adult female numbers to total population size. When the sex ratio parameter is estimated, rather than assumed fixed, the total population estimates for the two models more closely match the independent estimate – however, because the prior on sex ratio was wide (female proportion 0.5 to 1.0), quite unrealistic posterior estimates are required to produce move the population estimates even close to the independent estimate, particularly for the EDDFNM model. Careful thought will be required to specify appropriate priors on sex ratio. Until this is done, we should not investigate the effect of an uncertain sex ratio on model selection.

New estimates of other population parameters are becoming available – for example of fecundity at two intensively-monitored colonies (Smout et al. 2010). These could potentially be incorporated by revising the priors, or as observation data – the latter being more appropriate for parameters that vary through time such as through density dependence.

Other potential changes to the models have been discussed in previous briefing papers (e.g., allowing annual fluctuation in fecundity), but it is not clear what effect these would have on the model-based estimates of total population size.

Acknowledgement

Thanks to Mike Lonergan for suggesting a scaled gamma distribution to approximate the total population size estimate.

References

Duck, C.D. 2010. Grey seal pup production in Britain in 2009. *SCOS Briefing Paper 10/1.*

Hiby, L. and C.D. Duck. Unpublished. Estimates of the size of the British grey seal *Halichoerus grypus* population and levels of uncertainty.

Lonergan, M., B. McConnell, C. Duck and D. Thompson. An estimate of the size of the UK grey seal population based on summer haulout counts and telemetry data. *SCOS Briefing Paper 10/4*.

Smout, S., R King and P. Pomeroy. 2010. Colony specific implications of individual mass changes for survival and fecundity in female grey seals. *SCOS Briefing Paper 10/.*

Thomas, L. and J. Harwood. 2008. Estimating the size of the UK grey seal population between 1984 and 2007. *SCOS Briefing Paper 08/3*

Thomas, L. and J. Harwood. 2009. Estimating the size of the UK grey seal population between 1984 and 2008. *SCOS Briefing Paper 09/2*

Figure 1. Posterior mean estimates of true pup production from two models of grey seal population dynamics, fit to pup production estimates from 1984-2009 (circles) and a total population estimate from 2008. Lines show the posterior mean bracketed by the 95% credibility intervals for the EDDSNM (blue) and EDDFNM models (red).

(a) Pup production data only *(b) Pup production data and 2008 total population estimate*

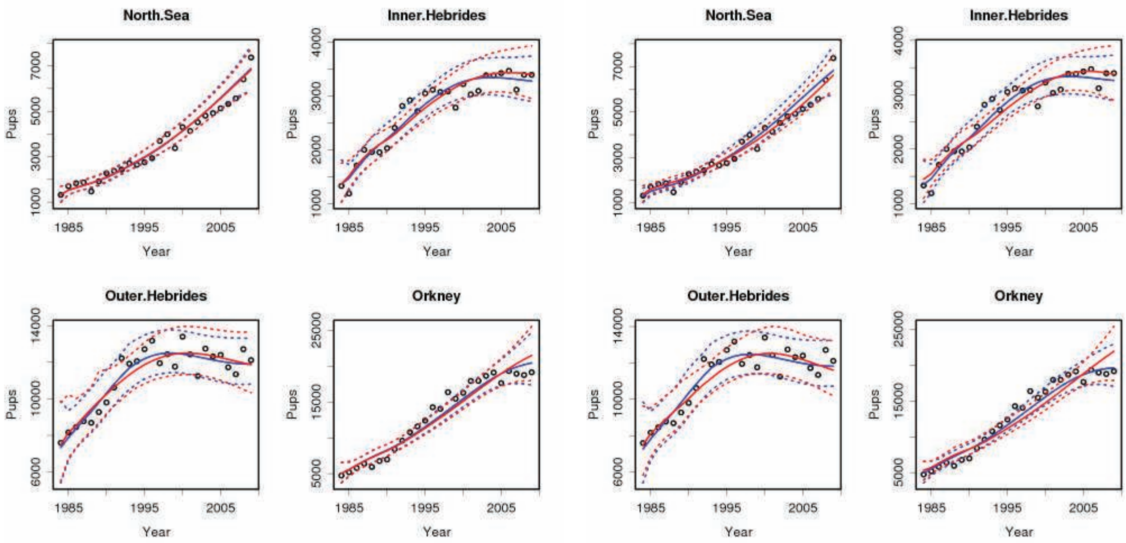
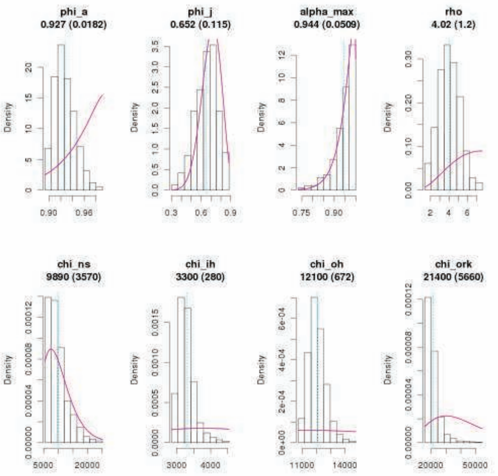


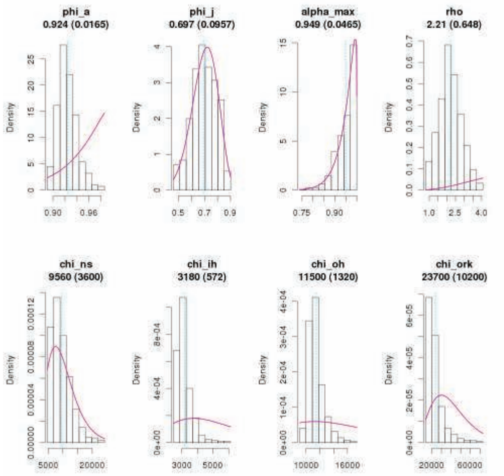
Figure 2. Posterior parameter estimates (histograms) and priors (solid lines) from two models of grey seal population dynamics, fit to pup production estimates from 1984-2009 (circles) and a total population estimate from 2008. The vertical line shows the posterior mean, its value is given in the title of each plot after the parameter name, with the associated standard error in parentheses.

Pup production data only

(a) Extended density dependent survival no movement (EDDSNM)

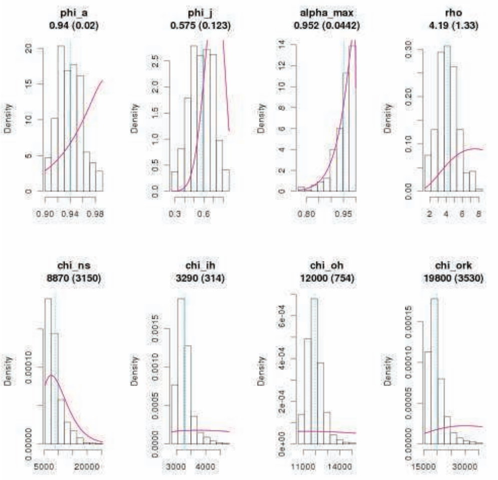


(b) Extended density dependent fecundity no movement (EDDFNM)



Pup production data and 2008 population estimate

(c) Extended density dependent survival no movement (EDDSNM)



(d) Extended density dependent fecundity no movement (EDDFNM)

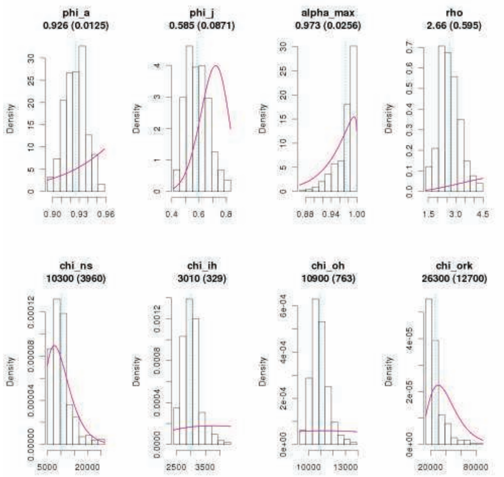


Figure 3. Posterior mean estimates of total population size from two models of grey seal population dynamics, fit to pup production estimates from 1984-2009 and a total population estimate from 2008 (circle, with horizontal lines indicating 95% confidence interval on the estimate). Lines show the posterior mean bracketed by the 95% credibility intervals for the EDDSNM (blue) and EDDFNM models (red).

(a) Pup production data only

(b) Pup production data and 2008 total population estimate

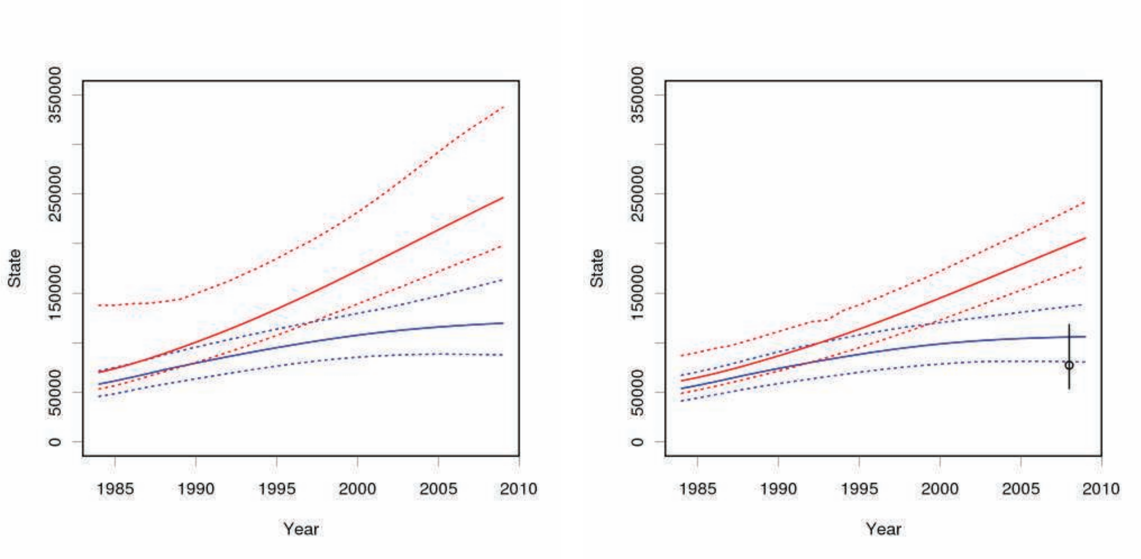
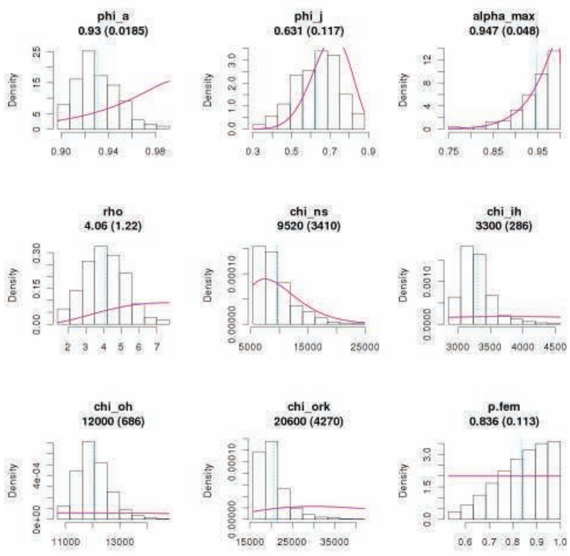
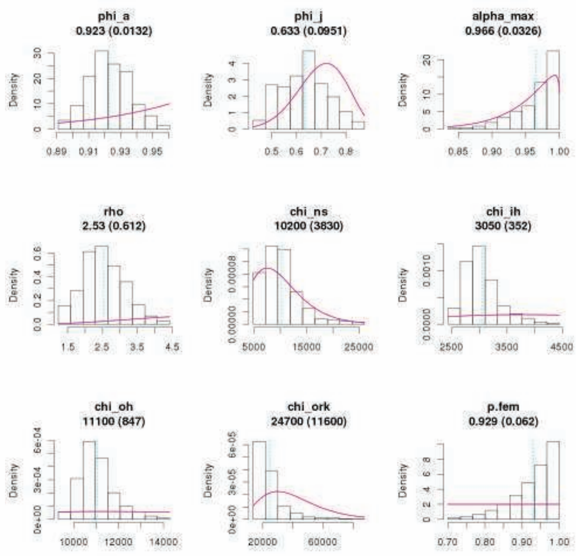


Figure 4. Posterior parameter estimates (histograms) and priors (solid lines) from two models of grey seal population dynamics, fit to pup production estimates from 1984-2009 (circles) and a total population estimate from 2008. The vertical line shows the posterior mean, its value is given in the title of each plot after the parameter name, with the associated standard error in parentheses. The models here differ from those in Figure 2 in that here the proportion of adult females in the population (p.fem) is an estimated parameter, while in the runs reported in Figure 2 it was assumed fixed.

(a) Extended density dependent survival no movement (EDDSNM)



(b) Extended density dependent fecundity no movement (EDDFNM)



Appendix

Estimates of total population size, in thousands, at the beginning of each breeding season from 1984-2009, made using the EDDSNM (extended density dependent survival with no movement) model of British grey seal population dynamics fit to pup production estimates and a total population estimate from 2008. Numbers are posterior means followed by 95% credibility intervals in brackets.

Year	North Sea	Inner Hebrides	Outer Hebrides	Orkney	Total
1984	4.9 (3.7 6)	5.2 (4.1 6.6)	24.5 (18.9 30.2)	19.4 (14.7 24.4)	54 (41.5 67.3)
1985	5.2 (4.1 6.4)	5.5 (4.3 6.9)	25.8 (20.2 31.8)	20.6 (15.8 25.7)	57.1 (44.4 70.7)
1986	5.6 (4.5 6.8)	5.8 (4.6 7.2)	27.1 (21.3 33.2)	22 (17.2 27.1)	60.5 (47.5 74.3)
1987	6 (4.9 7.3)	6.2 (4.9 7.6)	28.2 (22.2 34.6)	23.6 (18.5 28.8)	64 (50.4 78.2)
1988	6.5 (5.3 7.9)	6.5 (5.1 8)	29.3 (23 36.1)	25.3 (20 30.8)	67.7 (53.5 82.7)
1989	7 (5.7 8.4)	6.9 (5.4 8.4)	30 (23.7 36.9)	27.1 (21.6 33.1)	70.9 (56.3 86.8)
1990	7.5 (6.1 9)	7.2 (5.6 8.8)	30.5 (23.9 37.8)	29 (23.1 35.4)	74.1 (58.8 90.9)
1991	8 (6.6 9.6)	7.4 (5.9 9.1)	30.9 (24.2 38.2)	30.9 (24.7 37.6)	77.2 (61.3 94.5)
1992	8.6 (7 10.2)	7.7 (6 9.5)	31.1 (24.4 38.7)	32.9 (26.2 39.8)	80.2 (63.7 98.1)
1993	9.2 (7.5 11)	7.9 (6.1 9.8)	31.2 (24.5 38.8)	34.8 (27.8 42.1)	83.1 (65.9 101.6)
1994	9.8 (8.1 11.7)	8.1 (6.2 10)	31.1 (24.5 38.7)	36.8 (29.3 44.4)	85.8 (68.1 104.8)
1995	10.5 (8.6 12.5)	8.2 (6.3 10.2)	31 (24.6 38.5)	38.8 (30.8 46.8)	88.4 (70.3 107.9)
1996	11.1 (9.2 13.3)	8.3 (6.4 10.3)	30.8 (24.6 38.1)	40.7 (32.2 49.1)	90.9 (72.4 110.8)
1997	11.9 (9.7 14.1)	8.3 (6.4 10.4)	30.5 (24.6 37.6)	42.5 (33.5 51.4)	93.2 (74.2 113.5)
1998	12.6 (10.3 15)	8.3 (6.4 10.4)	30.3 (24.5 37.2)	44.1 (34.7 53.4)	95.3 (75.9 116)
1999	13.3 (10.8 15.9)	8.3 (6.4 10.3)	30.1 (24.5 36.8)	45.5 (35.7 55.2)	97.2 (77.3 118.3)
2000	14.1 (11.2 16.9)	8.3 (6.4 10.3)	29.9 (24.3 36.4)	46.6 (36.6 56.7)	98.9 (78.6 120.3)
2001	14.8 (11.8 17.9)	8.3 (6.4 10.2)	29.7 (24.3 36.1)	47.6 (37.3 58.2)	100.4 (79.8 122.4)
2002	15.6 (12.2 19)	8.2 (6.4 10.1)	29.6 (24.2 35.9)	48.3 (37.9 59.6)	101.6 (80.7 124.6)
2003	16.3 (12.5 20.1)	8.2 (6.4 10.1)	29.5 (24.1 35.8)	48.7 (38.3 60.7)	102.7 (81.3 126.6)
2004	17 (12.7 21.3)	8.1 (6.4 10)	29.5 (24 35.7)	49 (38.4 61.7)	103.6 (81.5 128.7)
2005	17.7 (12.9 22.5)	8.1 (6.3 10)	29.4 (24 35.7)	49.1 (38.2 62.5)	104.3 (81.4 130.7)
2006	18.3 (13.1 23.9)	8.1 (6.3 9.9)	29.5 (23.9 35.8)	49.1 (38.1 63.2)	104.9 (81.5 132.8)
2007	18.9 (13.2 25.2)	8.1 (6.3 9.9)	29.5 (23.9 35.9)	49 (37.9 63.8)	105.4 (81.3 134.8)
2008	19.4 (13.3 26.7)	8 (6.3 9.9)	29.6 (23.9 36)	48.9 (37.5 64.1)	105.9 (81 136.8)
2009	19.9 (13.3 28.2)	8.1 (6.3 9.9)	29.7 (23.9 36.2)	48.7 (37.1 64.5)	106.3 (80.6 138.8)

C.D. Duck, C.D. Morris & D. Thompson

The status of British harbour seal populations in 2009

NERC Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, East Sands, St Andrews, Fife, KY16 8LB

Summary

In August 2009, the Sea Mammal Research Unit (SMRU) completed the survey of harbour seals around the whole of the coast of Scotland which started in 2007 and to survey harbour seals between the Humber Estuary and east Norfolk in England.

In Scotland, helicopter thermal image surveys were restricted to part of Strathclyde including Mull, Lismore, Jura, Islay, Colonsay, Oronsay and west Kintyre to provide up-to-date information on seal numbers for potential renewable energy production in the Sound of Islay. The whole of Shetland was surveyed but only part of Orkney was completed due to unfavourable weather conditions. Out of 15 potential survey days, seven were lost to poor weather.

In England, harbour seals were surveyed from fixed-wing aircraft in Lincolnshire, Norfolk, Suffolk, Essex and Kent. The Tees Seal Research Programme kindly provided information on seals in the Tees Estuary (Woods, 2009).

Since 2007, most groups of harbour and grey seals were photographed using a hand-held digital camera to confirm numbers and species identity. The numbers used in this Briefing Paper are all from recounts, with the assumption that these are the more accurate.

From surveys carried out between 2007 and 2009, the minimum number of harbour seals counted in Scotland was **20,404** and in England **4000** making a total for Great Britain of **24,404** (Table 1). In 2002, **1,248** harbour seals were counted in Northern Ireland, making a UK total of **25,652**.

The number of harbour seals counted in Shetland was identical to the number counted in 2009 (3,003) excluding Foula. In Strathclyde, the 2009 count for Mull, Lismore, Islay, Jura, Colonsay, Oronsay and West Kintyre (4,090) was slightly higher (15.1%) than the 2007 count for the same area (3,552). The Orkney survey was only partially completed due to poor weather. The 2009 count for completed areas (1,384) was very similar (2.2% higher) to the equivalent count in 2008 (1,354) but 63.8% lower than the equivalent count in 2001 (3,824), prior to the decline. In the Moray Firth, both breeding season and moult counts were slightly higher than the 2008 counts. In the Firth of Tay, the decline appeared to continue with the 2009 count (111) 50% lower than the previous lowest ever count in 2008 (222).

During the 2009 breeding season, SMRU conducted repeat air surveys of harbour seals breeding in the Moray Firth, continuing the time series started by the University of Aberdeen. Breeding season surveys were also carried out in England, between the Humber Estuary and Scroby Sands.

Introduction

Most surveys of harbour seals are carried out during their annual moult, in August. At this time during their annual cycle, harbour seals tend to spend longer at haulout sites and the greatest and most consistent numbers of seals are found ashore. However, during a survey, there will be a number of seals at sea and not counted. Thus the numbers presented here represent the minimum number of harbour seals in each area and should be considered as an index of population size. Although harbour seals can occur all around the UK coast, they are not evenly distributed. Their main concentrations are in Shetland, Orkney, the Outer Hebrides, the west coast of Scotland and in east and south-east England, mainly around Lincolnshire and Norfolk (Figure 1)

Surveys of harbour seals around the Scottish coast are carried out on an approximately five-yearly cycle, with the exception of the Moray Firth and Firth of Tay which are surveyed annually. In 2006, significant declines in harbour seal numbers were found in Shetland and in Orkney and elsewhere on the North Sea coast on the UK (Lonergan *et al.* 2007). Between 2007 and 2009, we surveyed the entire Scottish coast and repeated some parts of Strathclyde and Orkney. Additional funding from Scottish Natural Heritage (SNH) allowed us to complete a third consecutive survey of Orkney. In 2009, the entire Scottish coast was completed with additional surveys in part of Strathclyde and part of Orkney.

In August 2010, with additional funding from SNH, surveys will be limited to Orkney.

In 2009, as in 2007 and 2008, most groups of seals were photographed with a high-resolution digital camera to confirm species identity and numbers in groups. These images were used to determine the classification of seals within haulout groups and will be used to determine the age and sex structure of grey seals. The grey seal data has been used to inform the models used to estimate the total grey seal population size (SCOS BP 10/4).

In England, the Lincolnshire and Norfolk coast, which holds over 95% of the English harbour seal population, is usually surveyed twice annually during the August moult and, since 2004, Natural England have funded breeding season surveys (in early July) of harbour seals in Lincolnshire and Norfolk, including The Wash.

Funding from Scottish Natural Heritage

Scottish Natural Heritage (SNH) has provided funding for harbour seals surveys in every survey year since 1996. Without this additional funding, we would not have known about the serious decline in numbers in Shetland and Orkney, as we would not have been able to carry out surveys of these island groups in either 2001 or 2006 and would not have detected the recent declines. SNH have also funded the annual surveys of Orkney since 2007.

Methods

Seals hauling out on rocky or seaweed covered shores are well camouflaged and difficult to detect. Surveys of these coastlines are by helicopter using a thermal-imaging camera. The thermal imager can detect groups of seals at distances of over 3km. This technique enables rapid, thorough and synoptic surveying of complex coastlines. In addition, digital images were obtained using a digital camera equipped with an image-stabilised zoom lens. Both harbour and grey seals were digitally photographed and the images used to classify group composition.

Surveys of the estuarine haulout sites on the east coast of Britain were made using large format vertical aerial photography or hand-held oblique photography from fixed-wing aircraft. On sandbanks, where seals are relatively easily located, this survey method is highly cost-effective.

To minimise the effects of environmental variables and to maximise the counts of seals on shore, surveys are restricted to within two hours before and after the time of local low tides (derived from POLTIPS, National Oceanographic Centre, NERC) occurring between approximately 12:00hrs and 18:00hrs. Surveys are not carried out in persistent or moderate to heavy rain as the thermal imager cannot ‘see’ through rain and because seals will increasingly abandon their haul-out sites and return into the water.

Results

1. Minimum estimate of the size of the British harbour seal population

The overall distribution of harbour seals around the British Isles from August surveys carried out between 2007 and 2009 is shown in Figure 1. For ease of viewing at this scale, counts have been aggregated into 10km squares.

Minimum population estimates, based on August surveys carried out between 2007 and 2009, 2000 and 2005 and in 1996 and 1997, are shown in Table 1. The Table includes numbers from both Northern Ireland and the Republic of Ireland from surveys in 2002 and 2003 respectively. For eastern England, where repeat counts were obtained (for The Wash, Donna Nook, Blakeney Point and Scroby Sands) the mean value has been used.

The most recent minimum estimate of the number of harbour seals in Scotland is **20,404** from surveys carried out between 2007 and 2009 (Table 1). This is 29.1% lower than the previous total for Scotland (28,812) from surveys carried out between 2000 and 2005 (Table 1). The most recent minimum estimate for England is **4,000**, which is 24% higher than the 2008 count of **3,230**. The 2009 count comprises 3,633 seals in Lincolnshire and Norfolk plus 347 seals in Northumberland, Cleveland, Essex and Kent between 2007 and 2008 and an estimated 20 seals from the south and west coasts. Including the **1,248** harbour seals counted in Northern Ireland in 2002, gives a UK total of **25,652.**

2. Harbour seals in Scotland: moult

In August 2009, the area surveyed for harbour seals using a thermal imager included: Mull. Lismore, Jura, Islay, Colonsay, Oronsay and west Kintyre in Strathclyde, Shetland (excluding Foula) and part of the northern isles in Orkney including Rousay and surrounding islands, Westray, Eday, Sanday, North Ronaldsay, Eday, the Green Holms and Faray, Holm of Faray and Rusk Holm. Eday, Stronsay, Copinsay and part of the north and east coast of Mainland was also surveyed but following prolonged heavy rain and strong winds. Data for these areas have been excluded in the totals provided for 2009.

The trends in counts of harbour seals in different areas (based on potential Management Units) of Scotland, from surveys carried out between 1988 and 2009 are shown in Figure 4 and Table 1. In 2009, the number of harbour seals counted in Shetland (3,003) was identical to the 2006 count, excluding Foula. Numbers in those parts of Strathclyde that were surveyed (4,090) were 13.2% higher than counts of the same area in 2007 (3,552). The harbour seal count for the area in Orkney surveyed in 2009 under appropriate conditions (1,384) was very slightly greater (2.2%) than the 2008 count for the same areas (1,354).

Moray Firth

Aberdeen University's Lighthouse Field Station, in Cromarty, obtained detailed annual breeding and moult counts of harbour seals in the Inner Moray Firth from June, July and August between 1988 and 2005. These counts of the inner Moray Firth are shown in Figure 5. SMRU's counts of a slightly larger area, including Loch Fleet and Findhorn, are also shown (SMRU Find-LF moult) along with counts of the outer Moray Firth, including the Brora coast up to Helmsdale (SMRU moult, outer MF).

SMRU's aerial surveys of the Moray Firth began in August 1992. The August counts are shown in Table 2 with the trends in different parts of the Moray Firth in Figure 6. This figure represents a combination of both thermal imaging and fixed wing surveys of the area. The 2009 count were lower than counts from both 2006 and 2007 (Table 2). It is not very clear whether harbour seal numbers in this area have stabilised following a period of decline between 1997 and 2002 or whether the decline is continuing at a reduced rate. These declines may, at least in part, have been due to a bounty system for seals which previously operated in the area (Thompson *et al*., 2007).

Firth of Tay

The 2009 count for the Firth of Tay was 111, exactly half the previous lowest count from 2008. Numbers in this Special Area of Conservation (SAC) are now 17.3% of the mean of counts between 1990 and 2002 (641). In 2007, 147 harbour seals were counted in the Firth of Forth. Previously we suggested that these seals were from the same population. Even if this is the case, numbers appear to continue to decline.

3. Harbour seals in Scotland: breeding season

Moray Firth

During the 2009 breeding season, SMRU conducted five air surveys harbour seals in the Moray Firth between mid June and mid July. The mean number of adults counted during these surveys, with the standard error, is shown in Figure 5. The mean count of harbour seals between Ardersier and Loch Fleet in 2009 was 671, 27.1% greater than the 2009 mean count of 528. The 2009 mean count between Findhorn and Helmsdale was 742, 21.8% greater than the 2008 mean count of 609.

4. Harbour seal surveys in England: moult

In 1988, the numbers of harbour seals in The Wash declined by approximately 50% as a result of the phocine distemper virus (PDV) epidemic. Prior to this, numbers had been increasing. Following the epidemic, from 1989, the area has been surveyed once or twice annually in the first half of August each year (Table 4, Figure 6).

Two aerial surveys of harbour seals were carried out in Lincolnshire and Norfolk during August 2009 (Tables 1 and 4). The mean count for The Wash (2,829) was a 40% increase over the mean of the 2008 counts which were similar to the counts over the previous 4 years. The 2009 counts were almost back to the the mean pre-epidemic 2002 count (2,976).

Overall, the combined count for the English East coast population (Donna Nook to Scroby Sands) in 2009 was 28% higher than the 2008 count and significantly higher than all years since 2002 epidemic (Figure 6, Table 4). This apparent sudden change from a continual decline to a rapid recovery is as yet unexplained. The English population is still lagging behind the rapid recovery of the Wadden Sea population that has been increasing consistently since 2002 and increased by 12% between 2008 and 2009. .

Harbour seals in the Tees Estuary are monitored by the Industry Nature Conservation Association (INCA). There appears to be a very slow recovery with numbers in August between 40 and 50 (Woods 2008; Woods 2009). Low but increasing numbers of pups are born (up to 12 in 2008 and 2009).

5. Harbour seals in England: breeding season

A total of 1130 pups and 2523 older seals (1+ age classes) were counted in The Wash during the 2009breeding season survey compared with 994 pups and 2,132 older seals in July 2008. Pups were widely distributed, being present at all occupied sites in 2009. The 2009 adult and pup counts were 14% and 18% higher than in the 2008, and similarly higher than in 2006 & 2007. The similarity of pup counts in 2006-2008 suggested that, like the moult counts, the production was not increasing rapidly as seen in the Wadden Sea. The 14% increase in pup count in 2009 is consistent with the large increase in the moult count.

6. Proposed harbour seal surveys 2010

Breeding season: Moray Firth

Five breeding season fixed-wing surveys were carried out in the Moray Firth between 18 June and 17 July 2010.

The Wash, Donna Nook and Blakeney Point

A series of five fixed wing surveys was carried out between 14th June and 13 July 2010 to provide data to estimate pup production in the Wash and adjacent sites. These data will be combined with a time series of pup counts to estimate pup production and develop a cost effective pup production monitoring strategy. This analysis will be presented to SCOS 2011.

Moult - Planned surveys

In Scotland, a full survey of Orkney is planned for August 2010, to obtain information on the stats of harbour seals in the islands, weather and equipment permitting. The same methods will be used as in previous years, incorporating digital still images.

In England, two fixed-wing surveys of the Lincolnshire and Norfolk coast will be carried out.

Acknowledgements

We are extremely grateful to all the Countryside Agencies for providing funding for carrying out surveys in their areas. SNH has provided very significant funding for Scottish surveys since 1996; Natural England funded recent surveys of The Wash and surrounding coasts. The Irish surveys were funded by the Northern Ireland Environment Agency (previously the Environment and Heritage Service) and the National Parks and Wildlife Service for northern and southern Ireland respectively.

We are very grateful for the technical expertise enthusiastically provided by the companies supplying the survey aircraft and pilots: PDG Helicopters and Giles Aviation.

References

Lonergan, M., C.D.Duck, D. Thomspon, B. L. Mackey, L. Cunningham and I.L. Boyd (2007). Using sparse survey data to investigate the declining abundance of British harbour seals. *J. Zoology*, **271**: 261-269.

Thompson P.M., Mackey B., Barton, T.R., Duck, C. &. Butler, J.R.A. (2007). Assessing the potential impact of salmon fisheries management on the conservation status of harbour seals (*Phoca vitulina*) in north-east Scotland. Animal Conservation **10**:48-56.

Woods, R. (2008) Tees Seals Research Programme, Monitoring Report No. 20. (1989–2008). Unpublished report to the Industry Nature Conservation Association, available at: http://www.adscreative.com/testbed/inca/downloads/Seals_Report_2008.pdf.

Woods, R. (2009) Tees Seals Research Programme, Monitoring Report No. 21. (1989–2009). Unpublished report to the Industry Nature Conservation Association.

Figure 1. **The August distribution of harbour seals in Great Britain and Ireland, by 10km squares. These data are from surveys carried out between 2007 and 2009.**

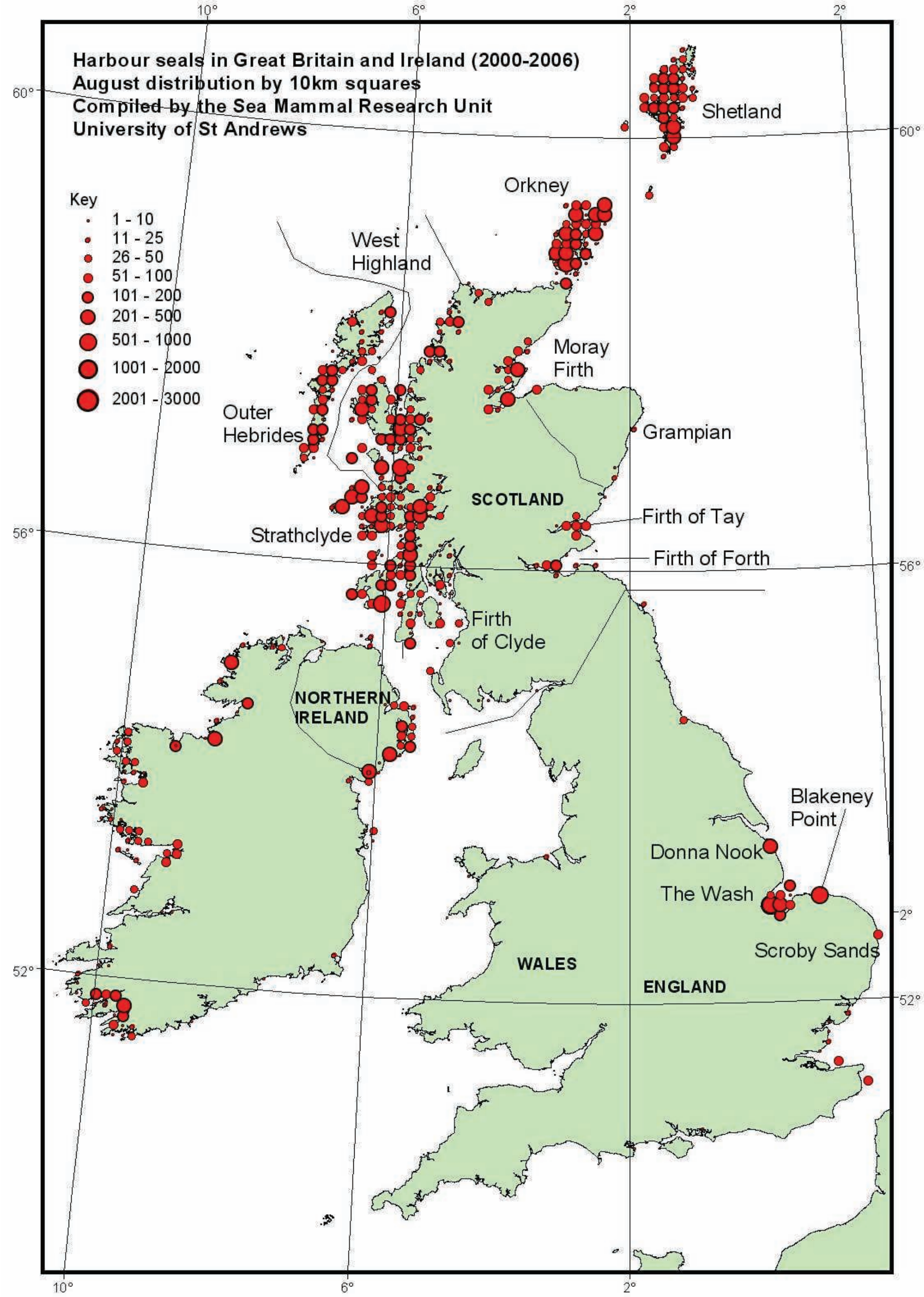


Figure 2. The number and distribution of harbour seals in Management Areas around the coast of Scotland, from surveys carried out between August 2007 and 2009. All areas were surveyed by helicopter using a thermal imaging camera.

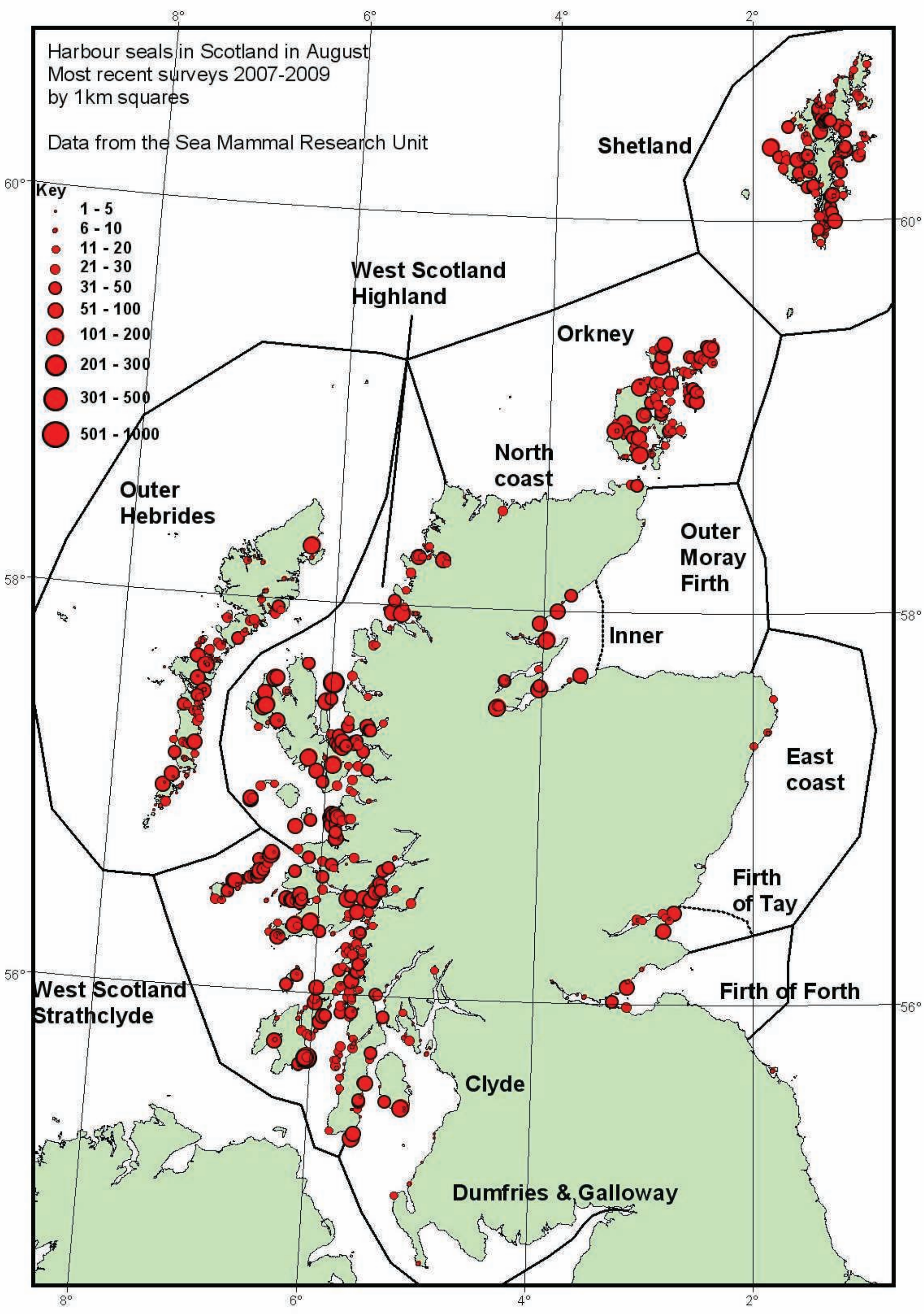


Figure 3. The number and distribution of grey seals in Management Areas around the coast of Scotland, from surveys carried out between August 2007 and 2009. All areas were surveyed by helicopter using a thermal imaging camera.

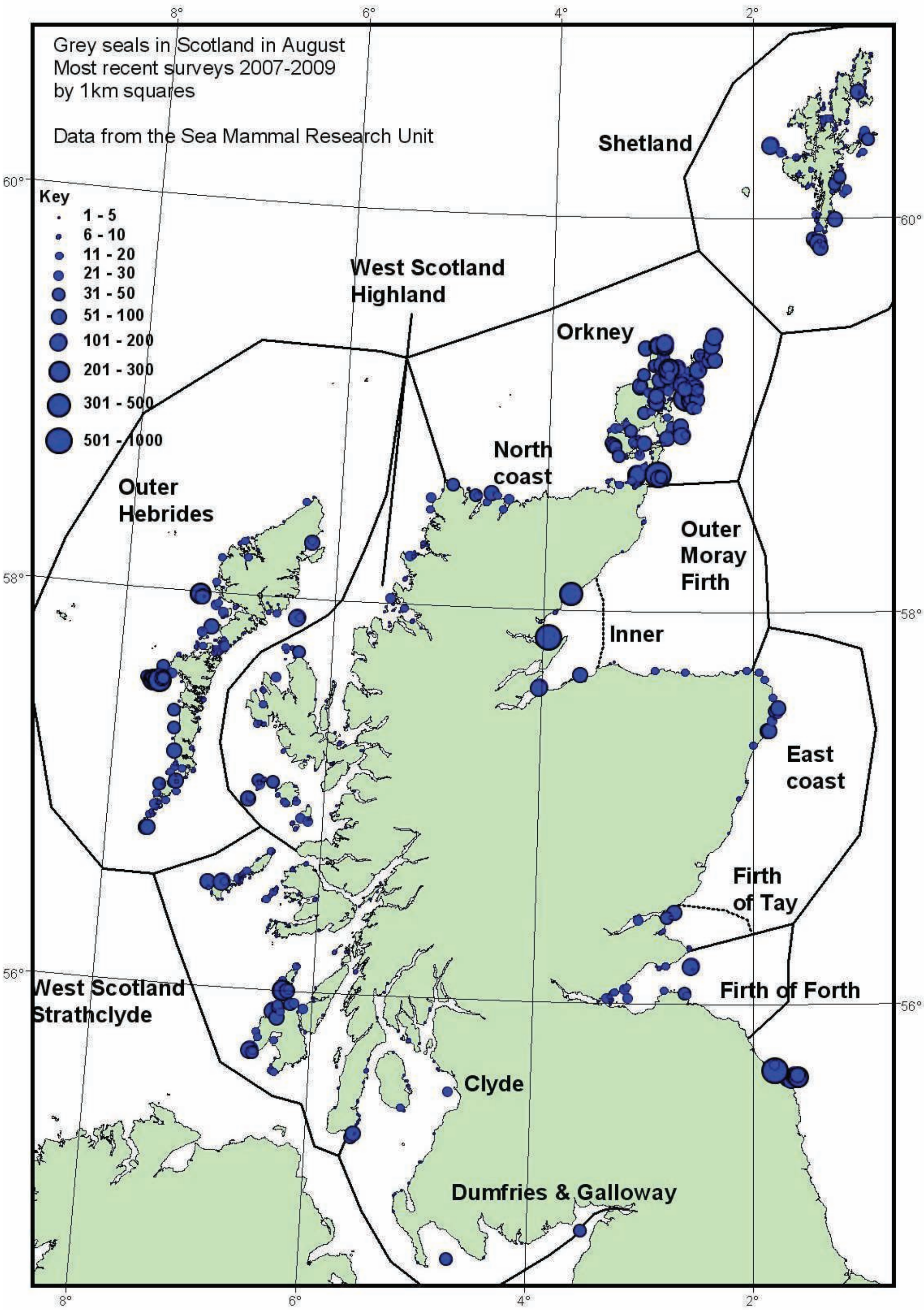


Figure 4. Trends in counts of harbour seals in Management Areas around Scotland. Data from the Sea Mammal Research Unit. Solid symbols show where data were from one or two years; open symbols show where data were collected over more than two years.

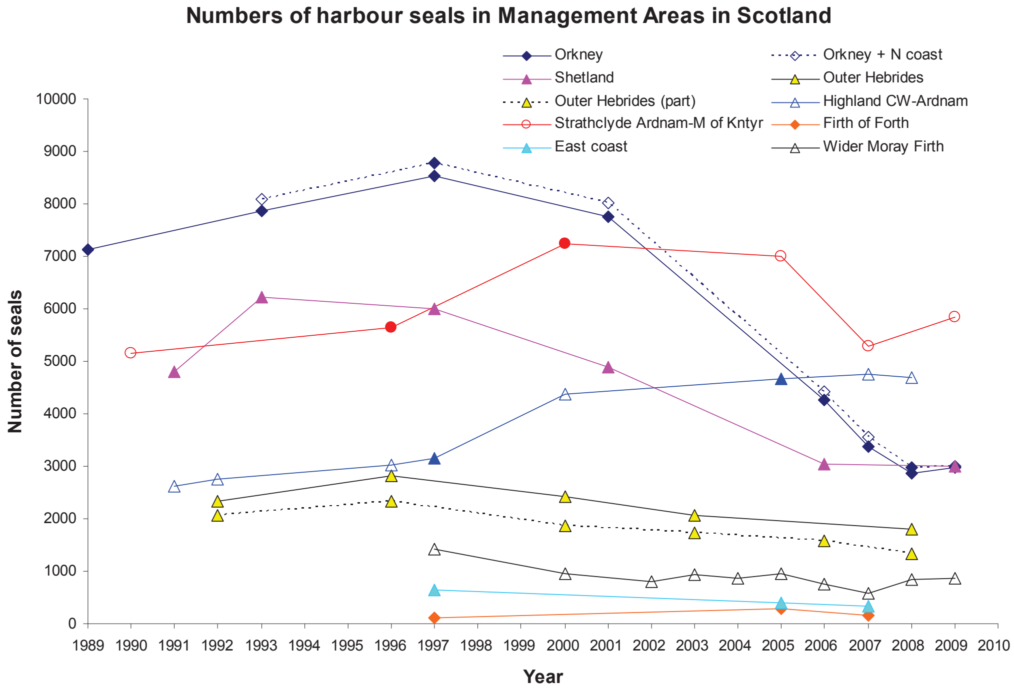


Figure 5. Trends in harbour seal numbers in the Moray since 1988. Seals were counted during their breeding season and during their moult by the University of Aberdeen's Lighthouse Field Station (LFS, Inner Firth) and more recently by SMRU (breeding season counts are for the Inner Firths plus Loch Fleet; the Outer Moray Firth includes Findhorn and the coast to Helmsdale).

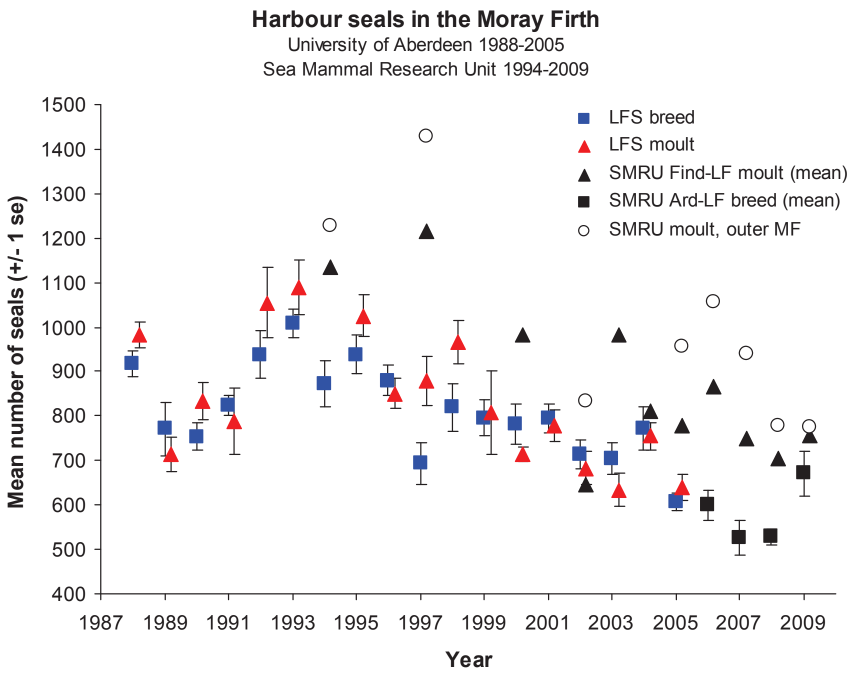


Figure 6. The number of harbour seals counted in areas within the Moray Firth between 1992 and 2009, by the Sea Mammal Research Unit.

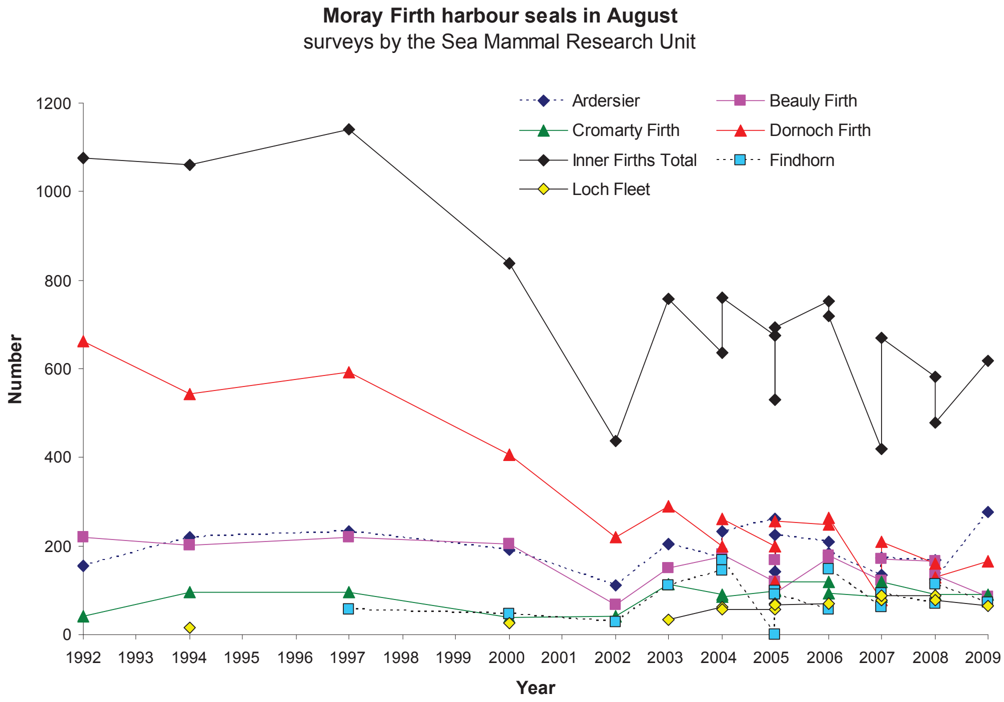


Figure 7. The number of harbour seals counted in the Firth of Tay between 1990 and 2009, by the Sea Mammal Research Unit.

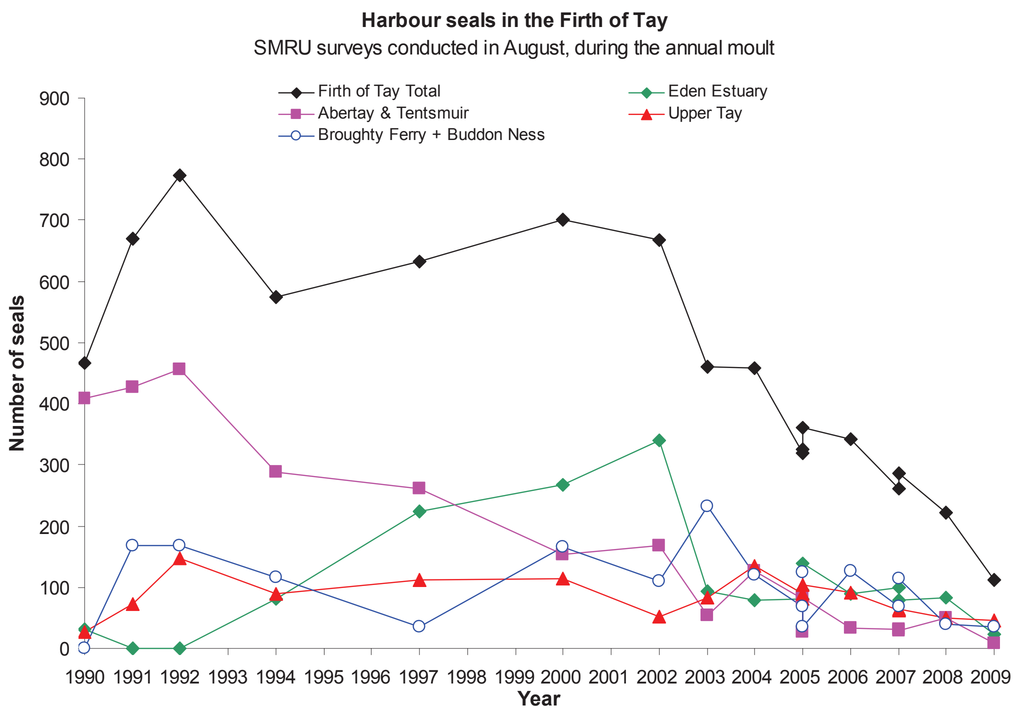


Figure 8. Counts of harbour seals in The Wash in August, 1967 - 2009. These data are an index of the population size through time. Fitted lines are exponential growth curves (growth rates given in text) and a 2nd order polynomial for post 2002 counts for illustration.

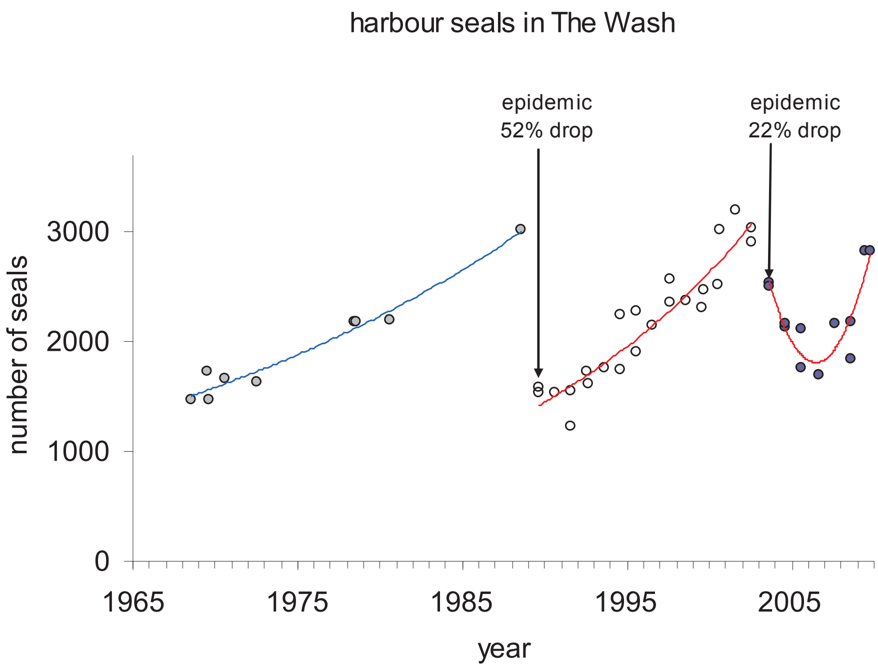


Table 1. Minimum estimates of the UK harbour seal population In Management Areas from the most recent surveys and from two previous surveys. The year of survey is underneath the number of seals counted.

Harbour seal Management Area	Current estimate (2007-2009)	Previous estimate (2000-2005)	Earlier estimate (1996-1997)
Shetland	**3,003** 2009	**4,883** 2001	**5,991** 1997
Orkney	**2,874** 2008, 2009	**7,752** 2001	**8,523** 1997
Highland North coast	**112** 2008	**174** 2005	**265** 1997
Outer Hebrides	**1,804** 2008	**2,067** 2003	**2,820** 1996
West Scotland, Highland (Cape Wrath to Ardnamurchan Point)	**4,696** 2007, 2008	**4,665** 2005	**3,160** 1996, 1997
West Scotland, Strathclyde (Ardnamurchan Point to Mull of Kintyre)	**5,834** 2007, 2009	**7,003** 2000, 2005	**5,651** 1996
South-west Scotland, Firth of Clyde (Mull of Kintyre to Loch Ryan)	**811** 2007	**581** 2005	**923** 1996
South-west Scotland, Dumfries & Galloway (Loch Ryan to English Border at Carlisle)	**23** 2007	**42** 2005	**6** 1996
East Scotland, Firth of Forth (Border to Fife Ness)	**148** 2007	**280** 2005	**116** 1997
East Scotland, east cosat Fife Ness to Fraserburgh	**228** 2007	**406** 2005	**648** 1997
East Scotland, Moray Firth (wider) Fraserburgh to Duncansby Head	**871** 2007	**959** 2005	**1429** 1997
TOTAL SCOTLAND	**20,404**	**28,812**	**29,532**
	(2009)	(2005)	(1997)
Blakeney Point	**372**	**709**	**311**
The Wash	**2,829**	**1,946**	**2,461**
Donna Nook	**267**	**421**	**251**
Scroby Sands	**165**	**57** 2004	**65**
Other east coast sites	**347**	**153** 1994-2003	**137** 1994 –1997
South and west England (estimated)	**20**	**20**	**15**
TOTAL ENGLAND	**4,000**	**3,306**	**3,240**
TOTAL BRITAIN	**24,404**	**32,118**	**28,485**
TOTAL NORTHERN IRELAND	**1,248** 2002	**1,248** 2002	
TOTAL BRITAIN & N. IRELAND	**25,652**	**33,366**	**29,733**
TOTAL REPUBLIC OF IRELAND	**2,905** 2003	**2,905** 2003	
TOTAL GREAT BRITAIN & IRELAND	**28,557**	**36,271**	**32,638**

Table 2. Numbers of harbour seals in the Moray Firth during August (SMRU surveys). See Figure 6.

Location	07 Aug 1992	30 July 1993[1]	13 Aug 1994	15 Aug 1997[1]	11 Aug 2000	11 Aug 2002[1]	7 Aug 2003	10 Aug 2004	13 Aug 2004	8 Aug 2005	9 Aug 2005	16 Aug 2005[1]	18 Aug 2005[1]	4 Aug 2006[1]	20 Aug 2006	15 Aug 2007[1]	24 Aug 2007	13 Aug 2008[1]	20 Aug 2008	6 Aug 2009
Ardersier	154	-	221	234	191	110	205	172	232	260	143	195	224	210	184	150	173	167	123	277
Beauly Firth	220	-	203	219	204	66	151	175	180	119	169	-	94	174	178	115	170	165	135	85
Cromarty Firth	41	-	95	95	38	42	113	90	86	98	101	-	118	119	93	67	118	90	90	90
Dornoch Firth (SAC)	662	-	542	593	405	220	290	199	262	199	118	-	256	249	264	153	209	160	130	166
Inner Moray Firth Total	**1077**	**-**	**1061**	**1141**	**838**	**438**	**759**	**636**	**760**	**676**	**531**	**-**	**692**	**752**	**719**	**485**	**670**	**582**	**478**	**618**
Findhorn	-	-	*58*	*46*	*111*	*144*	*167*	*0*	*98*	*90*	*58*	*148*	*74*	*63*	*68*	*82*	*94*	*69*	*115*	*73*
Loch Fleet	-	*16*		*27*	*33*	*62*	*56*	*58*	*70*	*68*	*70*	-	*76*	*79*	*53*	*85*	*87*	*87*	*77*	*65*
Loch Fleet to Dunbeath	**-**	*92*		*214*		*188*	-	-	-	-	-	-	*113*	*163*	*137*		*90*	*102*	*43*	*19*
Outer Moray Firth Total				**1428**		**832**							**955**	**1057**	**989**		**941**	**840**	**713**	**775**

[1]Thermal imaging survey

Table 3. Numbers of harbour seals in the Firth of Tay during August. See Figure 7.

Location	13 Aug 1990	11 Aug 1991	07 Aug 1992	13 Aug 1994	13 Aug 1997[1]	12 Aug 2000	11 Aug 2002	7 Aug 2003[2]	10 Aug 2004	8 Aug 2005	9 Aug 2005	14 Aug 2005[1]	14 Aug 2006	4 Aug 2007	7 Aug 2007[1]	29 Aug 2008	7 Aug 2009
Eden Estuary	31	0	0	80	223	267	341	93	78	81	95	139	90	99	79	83	22
Abertay & Tentsmuir	409	428	456	289	262	153	167	53	126	80	26	82	34	32	30	50	8
Upper Tay	27	73	148	89	113	115	51	83	134	90	80	104	91	62	64	49	45
Broughty Ferry & Buddon Ness	0	169	169	117	35	165	(109)	232	121	68	125	36.	127	68	114	40	36
Firth of Tay Total (SAC)	**-**	**670**	**773**	**575**	**633**	**700**	**(668)**	**461***	**459**	**319**	**326**	**361**	**342**	**261**	**287**	**222**	**111**

[1]Thermal imaging survey
[2]In August 2003 low cloud prevented the use of vertical photography; counts were from photographs taken obliquely and from direct counts of small groups of seals.

Table 4. Number of harbour seals counted on the east coast of England since 1988. **See Figure 8**. Data are from fixed-wing aerial surveys carried out during the August moult.

Date of survey	13/8	8/8 12/8	11/8	2/8 11/8	1/8 16/8	8/8	6/8 12/8	5/8 15/8	2/8	2/8 8/8	7/8 14/8	3/8 13/8	4/8 12/8	4/8	11/8 12/8	9/8 10/8	6/8 14/8	09/8	15/8	3/8	8/8 16/8	
Year	**1988**	**1989**	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**
Blakeney Point	701	- 307	73	- -	- 217	267	- 196	438 392	372	250 371	535 738	715 602	895 disturb	772	346 631	399	577 715	741 677	719	550	620 541	372
The Wash (SAC)	3087	1531 1580	1532	1226 1551	1724 1618	1759	2277 1745	2266 1902	2151	2561 2360	2367[1] 2381	2320 2474	2528 3029	3194	3037 2916	2529 2497	2126 2167	1768 2124	1695	2162	1846 2174	2835 2823
Donna Nook	173	- 126	57	- -	18 -	88	60 146	115 36	162	240 262	294 201	321 286	435 345	233	341 -	231	242 346	372 470	299	214	132 250	170 363
Scroby Sands	-	- -	-	- -	- -	-	61 -	- 49	51	58 72	52 -	69 74	84 9	75			49 64		71		60 101	100 230
The Tees	-	- -	-	- -	- -	-	- 35	- -	-	- -	- -	- -	- -	-	-		- -	-	-		41[3]	49[4]
Holy Island, Northumber-land	-	- -	-	- -	- -	-	13	- -	-	12[2]	- -	- -	10	-	-		- -	17[2]	-	7		
Essex, Suffolk & Kent	-	- -	-	- -	- -	-	- -	90 -	-	- -	- -	- -	- -	-	- 72	190	- -	101	-		299	

[1] One area used by harbour seals was missed on this flight (100 – 150 seals); this data point has been excluded from analyses. Totals are of means when more than one survey of any area in any year.
[2] Holy Island surveyed by helicopter using a thermal imaging camera.
[3] Tees data kindly provided by Robert Woods, INCA (Woods, 2008).
[4] Tees data kindly provided by Robert Woods, INCA (Woods, 2009).

Mike Lonergan, Bernie McConnell, Callan Duck & Dave Thompson

An estimate of the size of the UK grey seal population based on summer haulout counts and telemetry data.

NERC Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, St Andrews KY16 8LB

1. Summary

Scaling up from a count of hauled out seals to a population estimate requires allowance to be made for the proportion of animals at sea during the survey and therefore unobserved. We used historical telemetry data to assess the effects of age and sex on the proportion of British grey seals hauled out around August daytime low waters, and combined this with counts of grey seals made during the 2007-9 harbour seal moult surveys to estimate the total UK population of grey seals at that time as 74,223 (95% confidence interval: 54,300 – 118,300). These results show that the population can be estimated from this data without a need to classify images of individual animals by age and sex.

2. Introduction

British grey seal numbers are estimated by using Bayesian State Space models to scale up from pup production to population estimates. Unfortunately these data provide limited information to select between different models, which can produce very different population estimates depending on where in the lifecycle density dependence is assumed to operate (SCOS-BP 10/3 & 10/5). Counts of hauled out grey seals were therefore carried out in the summer, alongside harbour seal moult surveys. This briefing paper investigates the proportion of grey seals hauled out, and therefore available to be observed during the surveys, in order to scale the counts up to provide population estimates with appropriate confidence intervals. It also considers the variability of the haulout behaviour and the effect this could have on these results.

Specifically, we examine environmental and regional differences in the proportion of time around low tides that grey seals haul out for. We also look at how sex and length, which correlates with age (at least among younger animals) affect the probability of their being hauled out during the surveys.

Last year, we reported that overall, the animals hauled out for approximately 0.35 (95% CI: 0.32-0.38) of the time considered, and, while there was wide individual variation, neither sex nor length, a proxy for age, had a significant effect. It therefore seemed that unclassified census counts are sufficient for population estimation. We also found that neither region nor the exact timing of the survey within the summer had a substantial impact on the probability of being hauled out.

While the previous results were broadly correct, those confidence intervals were for the proportion of time animals hauled out. We have improved our summarisation of the telemetry data and also corrected a small bias that resulted from the way we had treated missing data. This paper gives revised estimates of, and confidence intervals around, the proportion of animals that would be observed during a survey and applies them to give an explicit population estimate.

3. Methods

<u>Aerial Survey data</u>

The August 2007, 2008 and 2009 harbour seal moult surveys were extended so that, between them, they covered almost all potential grey seal haulout sites. These aerial survey flights were carried out between 08:00h and 18:00h and within two hours of a local low water that fell within the same window. On rocky shores in Scotland the surveys were further restricted to afternoon tides. The data were aggregated into five regions (North Sea, Orkney, Inner Hebrides, Outer Hebrides, and Shetland – the first four of these correspond to regions used for population estimation).

Telemetry Data

All the telemetry data that has been collected from British grey seals over the past 20 years were examined. Only information collected within the August aerial survey windows was used for this analysis. This came from 107 animals, covered the years from 1995 to 2008 (table 1), and described a total of 6,500 hours of relevant animal behaviour, approximately 65% of the total that complete knowledge would have provided. The remainder of the information had been corrupted or lost during transmission.

Individual data

Each individual animal was handled to attach its tag. At this time its sex, mass and length were recorded and an individual identifier chosen. A small proportion of this information has subsequently been lost. Animals caught before they had left their natal beach, and some small individuals were recorded as juveniles, but accurate aging of seals beyond the first few months requires examining growth layers within an extracted tooth. Teeth were not generally extracted, and animals' weights can vary seasonally, so length was used as a proxy for animals' ages.

Environmental data

The location information provided by the tags was used to associate seals with haulout sites, named areas approximately 10km across. Each animal was linked with a series of sites, with time at sea being associated with the haulout site at which the animal had last been recorded Each haulout site was also associated with the nearest Secondary Tidal Prediction Port. From this we calculated the timing and height above datum of the low waters the animals experienced. Dates of full moons were used to represent the Neap-Spring tidal cycle. Daily rainfall, wind speed, and mean temperatures were obtained from nearby meteorological stations and associated with the individual animals. The haulout sites were grouped into the regions, though there was no data associated with haulouts in Shetland and additional data was available from the Irish Sea, an area that was not covered by these aerial surveys.

Haulout information

The tags have a conductivity sensor that indicates whether they are wet or dry. They record haulouts, defined to start when the tag is continuously dry for 10 minutes and end when it is continuously wet for 40 seconds. Haulouts are numbered consecutively enabling animals to be classified as hauled out, not hauled out, or of unknown status at particular times. Unfortunately transmitting the data as haulout records biases the recovered information on the proportion of time animals haul out, since the two at-sea periods around any missing haulout record will be classified as of unknown status. To remove this bias we treated the status of the first haulout period after each block of missing data as also being unknown. The pseudo-random and delayed transmission of the data mean that the probability of data being missing is unlikely to be related to seal behaviour (Fedak et al., 2002), so these periods are ignored in this analysis. The large total seal population meant that we could neglect stochastic effects, so we calculated the proportion of each survey window that each animal was hauled out.

This process resulted in a set of 1837 datapoints, each one of which contained an estimate of the proportion of one potential tidal survey window that one animal hauled out, the date and time of that low tide, its height, the region and local haulout site at which it occurred, and the animal's identifier. Most datapoints also contained the animal's length and sex as well as that day's local temperature, rainfall and wind speed.

Analysis

The analysis occurred in four phases: first the effects of the various potential explanatory factors were examined; then the independence of the behaviour of animals was assessed to examine the effects of unmeasured covariates and effective sample sizes; next the distribution of appropriate bootstrap resamples of the haulout data was generated; and finally the aerial survey data were multiplied by this to give an overall population estimate.

Mean proportions of animals hauled out, and standard errors around these estimates, were calculated and plotted by day of year, year, region, the height of low tide above datum, days since full moon (as proxies for the spring-neap cycle), mean daily temperature, daily rainfall. The proportion of time individual animals spent hauled out was also plotted out against their lengths and by sex. A Mann-Whitney U-test was used to look at sex differences, and the results compared to an empirical null distribution, generated by repeatedly permuting the sex of the animals, to account for the non-independence of the datapoints. Spearman's rank correlation coefficient was used, with similar empirical null distributions, to look at the effects of tidal height, temperature, wind speed, rainfall and animal length. The non-linear effects of time of low tide, day in year and days since full moon were investigated by using the Wald-Wolfowitz non-parametric runs test of autocorrelation, with an empirical null distribution, on daily mean data values arranged in order of the explanatory covariate. Two separate null distributions were created for the day of year tests. To consider temporal autocorrelation in behaviour, the ordering of the days was permuted, while cross correlation between animals was investigated by creating permuted datasets with each individual animal's data separately time-shifted.

Differences between regions were investigated by comparing the maximum differences between regional means to a distribution of equivalent values generated by repeatedly permuting the animals between regions. Similar permutations were carried out to compare years, but these had to discard years from which there was insufficient data.

A simple overall estimate of the expected proportion of the population hauled out during the survey window was calculated as the mean of the proportion of the time the individual animals were hauled out. A bootstrap estimate of the precision of this estimate was made from 10000 replicates with the animals as the unit of resampling. The validity of the resulting confidence interval depends on the independence of the data from the individual animals.

If all the datapoints were independent, and drawn from the same distribution, then a simple estimate of the expected variability in the proportion that would be seen during repeated surveys could come from repeatedly drawing sets of 107 datapoints from the dataset. There are three potential problems with such an estimate that result from: differences between the behaviour of individual animals, autocorrelation within each individual's behaviour, and correlation between the behaviour of nearby animals.

If individuals vary in the proportion of the survey windows that they haulout for, then a simple bootstrap resampling of the data is likely to underestimate the variability of real surveys. If this variability is correlated with the amount of data received from each animal, then the mean of the individual animals' mean haulout proportions will differ from a direct, unweighted, mean over the dataset. The two values were calculated over both the original dataset and bootstrap resamples generated with animals as the unit of resampling.

To investigate autocorrelation in individual behaviour, which would be expected to occur given that the species typically hauls out for periods between foraging trips lasting several days, Wald Wolfowitz runs tests were performed on the data from the individual animals, with permuted versions providing null distribution for significance testing, to examine this.

A simple way to deal with both variability between individuals and autocorrelation in individual behaviour is to calculate confidence intervals based on bootstrap resamples that take one datapoint from each individual. However that ignores correlation between the animals' behaviour. Substantial (positive) correlation between individuals' behaviour would make the width of this confidence interval an underestimate of the true value.

We tested for correlation between individuals using a weighted mean of the standard errors of groups of animals. We did this at the regional and local, haulout, level. In each case we divided the animals into groups, each of which contained data from one day and area. Lone animals were discarded, and the standard error (standard deviation divided by the number of animals) was calculated for the remaining groups. Since larger groups contain more data, and should therefore produce better estimates of the standard errors, we then took a weighted mean, with each datapoint weighted by the number of individuals in each group, as an overall estimate of "average standard error". We then created replicate datasets by permuting the datapoints between the groups and recalculating standard errors. The extent of the correlation was estimated by comparing the original values to the distributions from the permuted datasets.

A simple bootstrap will underestimate the variability of future samples taken from a structured dataset. We therefore generated an estimate of the variability of surveys by drawing replicate sets of 107 datapoints from each significantly different subset of the original data, and summed these into an overall distribution. To allow for the differing amounts of data available from the tags, the probability of each datapoint's inclusion in each replicate was weighted to ensure each every tag was approximately equally represented in the final, summed, distribution. These weightings were proportional to the number of datapoints in the category divided by the total number of datapoints contributed by that individual. The survey results were then divided by these figures to produce a distribution of population sizes consistent with these data.

4. Results

The summarised data is displayed in figures 1 & 2. The error bars shown on the plots are approximate 95% confidence intervals based on the standard errors of the estimates. Any lack of independence in the data, which comes from a limited number of individuals who are exposed to broadly similar environmental conditions, will lead these to understate the variability in the estimates.

The mean proportion of the survey windows individuals hauled out for was very variable (Figure 1), though the greatest variability was in the animals with most limited data. Taking the mean of these estimates produces a population estimate of 0.34 of the animals being hauled out during the surveys. The equivalent simple mean of the data was 0.31 (95% CI: 0.28-0.34). Simple bootstrap resamples of animals' overall haulout proportions gave a 95% confidence interval around the overall mean of 0.30-0.37. Dividing the survey results by these numbers gave an overall population estimate and confidence interval of 77,801 (95% confidence interval: 70,000 – 87,200) (Table 2).

There was no significant difference in the proportion of the survey windows males and females hauled out for (Mann-Whitney U-test; $p>0.5$; empirical null distribution) or animal length (Spearman's rank correlation coefficient; $p>0.5$; empirical null distribution). No effect of height above datum of the low tide, or daily temperature, rainfall, wind speed (Spearman's rank correlation coefficient; $p>0.2$, 0.8, 0.09, 0.07; empirical null distributions) was detected. Similarly, the timing of low tide and the number of days since full moon had no significant effect (Wald-Wolfowitz runs test; $p>0.1$, $p>0.8$; empirical null distributions).

The permutation test showed that the differences between the regional data are statistically non-significant ($p>0.1$). While the 45% of time the individuals in the Inner Hebrides hauled out was above the 95% confidence interval for permuted populations (0.25-0.43), the 16% difference between it and the animals in the Irish Sea lay well inside the appropriate confidence interval (0.04-0.21). Seven out of the 107 tagged individuals moved between regions (Table 1). Some of these also returned to their original regions within the period, further cautioning against considering them separately.

The apparent differences between the years seem to be due to the small sample sizes. Essentially any answer could appear for 1999 and 2001, but, even excluding them, the greatest difference between years (0.10, between the mean values for 2002 and 2003) was non significant ($p>0.9$; permutation test), and is actually less than would be expected from the permutation test (95% confidence interval on maximum difference: 0.12-0.43). Restricting the comparison to the four years containing more than 10 tagged animals, which all have mean haulout proportions in the range 0.32-0.34 produced very similar results ($p>0.9$; 95% confidence interval on maximum interannual difference 0.03-0.21; permutation test).

The only significant pattern detected was temporal. There is a visually striking pattern in the data when mean values are plotted by day (figure 1), which is statically significant against both the shifted and permuted versions (Wald-Wolfowitz runs test; $p>0.01$, $p<01$; empirical null distributions). This is particularly surprising given that the data comes from multiple years and regions, making it hard to see what common pattern could be driving it.

There is some evidence of negative autocorrelation in the individual data. There was insufficient data or variability in the data from 8 tags, and 7 of the remaining 99 tags generated empirical p-values below 0.05, though 4 of these are from tags that produced 5 or fewer datapoints. More interestingly, 23 tags produced test statistics greater than 95% of the relevant null distributions, indicating that these animals switched between hauling out for high and low proportions of the survey window more often than would be expected by chance.

Estimates of the proportion of animals hauled out were, unsurprisingly, less variable when they were based on more animals (Figure 3). However the average standard errors showed no indication of correlation between the behaviour of individuals (Figure 3). The difference between the two sets of average standard errors are due to the aggregation at regional rather than haulout level producing groups containing more individuals.

The large and statistically significant variability in the results between days, made it necessary to stratify the bootstrap resampling. Therefore 1000 replicate sets of 107 haulout proportions were drawn, with replacement from the data from each day, with appropriate weightings. These distributions were summed to provide an overall distribution. It was effectively a normal distribution with a mean of 0.3505 and a standard deviation of 0.0680, giving a 95% confidence interval of 0.22-0.48. Dividing the survey counts by these numbers produced a total population estimate of 74,427 individuals (95% confidence interval: 54,000 – 118,300).

5. Discussion

This analysis has shown that the probability of being hauled out in August, and thus being counted in an annual aerial census, is not substantially affected by seal age or sex. This suggests that efforts to sex and age animals observed during aerial surveys will have little effect on the resulting population estimates. Indeed, this nicely avoids the question of how accurately can sex and a proxy of age be determined from aerial photographs.

It also shows little evidence for geographical and environmental effects on the probability of animals hauling out. It appears that, while individuals differ, there are no obvious consistent patterns associated with any of the obvious potentially explanatory variables we considered. This, and the fact the data came from animals in many areas and many years, makes the substantial differences between different days particularly hard to explain. There seems to be little direct correlation between these animals' behaviour, but on some days of the month the animals' seem to haul out twice as much as on others. Only part of this seems explicable by the autocorrelation in individuals' behaviour.

Non-parametric bootstrapping and permutation tests make few assumptions about the patterns underlying data. They are generally less powerful than equivalent parametric methods, but, in cases like this, where the patterns of interdependence between datapoints are complex, they provide a robust methodology. They do, however, require the identification of appropriate units for data manipulation and analysis. The quadrupling in the dispersion, that results from respecting the variability between different days, provides a clear demonstration of the importance of checking the appropriateness of any data aggregation.

This analysis shows the power available from combining the data from large numbers of relatively small telemetry deployments. The biological implication of the remarkable consistency of these results and the daily fluctuations, and their extension to the rest of the year are beyond the scope of this investigation. Hopefully they will be further examined in due course.

6. References

Fedak, M., Lovell, P., McConnell, B. & Hunter, C. (2002) Overcoming the constraints of long range radio telemetry from animals: Getting more useful data from smaller packages. *Integrative and Comparative Biology,* **42,** 3-10.

McConnell, B. J., Fedak, M. A., Lovell, P. & Hammond, P. S. (1999) Movements and foraging areas of grey seals in the North

Sea. *Journal of Applied Ecology,* **36,** 573-590.

Tables

<table>
<tr><td rowspan="3">year</td><td colspan="10">Region</td><td rowspan="3">total</td></tr>
<tr><td colspan="2">North Sea</td><td colspan="2">Inner Hebrides</td><td colspan="2">Outer Hebrides</td><td colspan="2">Orkney</td><td colspan="2">Irish Sea</td></tr>
<tr><td>m</td><td>f</td><td>m</td><td>f</td><td>m</td><td>f</td><td>m</td><td>f</td><td>m</td><td>f</td></tr>
<tr><td>1995</td><td>-</td><td>-</td><td>1*</td><td>-</td><td>8*</td><td>3</td><td>2</td><td>1</td><td>-</td><td>-</td><td>14</td></tr>
<tr><td>1996</td><td>-</td><td>-</td><td>-</td><td>-</td><td>2</td><td>-</td><td>1</td><td>-</td><td>-</td><td>-</td><td>3</td></tr>
<tr><td>1997</td><td>1</td><td>1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>1</td><td>1</td><td>-</td><td>-</td><td>4</td></tr>
<tr><td>1998</td><td>3*</td><td>2</td><td>-</td><td>-</td><td>2</td><td>-</td><td>9*</td><td>4</td><td>-</td><td>-</td><td>19</td></tr>
<tr><td>1999</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td colspan="2">1</td><td>1</td></tr>
<tr><td>2000</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>2001</td><td>-</td><td>1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>1</td></tr>
<tr><td>2002</td><td>-</td><td>1*</td><td>-</td><td>-</td><td>-</td><td>-</td><td>1</td><td>1*</td><td colspan="2">2</td><td>4</td></tr>
<tr><td>2003</td><td>-</td><td>-</td><td>-</td><td>7</td><td>1*</td><td>-</td><td>1*</td><td>-</td><td>-</td><td>-</td><td>8</td></tr>
<tr><td>2004</td><td>-</td><td>-</td><td>5</td><td>3</td><td>1</td><td>-</td><td>-</td><td>-</td><td>8</td><td>7</td><td>24</td></tr>
<tr><td>2005</td><td colspan="2">4 | 2* | 5</td><td>-</td><td>-</td><td>-</td><td>-</td><td colspan="2">1*</td><td>-</td><td>-</td><td>11</td></tr>
<tr><td>2006</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>2007</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td></td><td>-</td></tr>
<tr><td>2008</td><td>9*</td><td>8*</td><td>-</td><td>-</td><td>-</td><td>1</td><td>1*</td><td>2*</td><td>-</td><td>-</td><td>18</td></tr>
<tr><td rowspan="2">totals</td><td colspan="2">17 | 2 | 18</td><td>6</td><td>10</td><td>14</td><td>4</td><td colspan="2">16 | 1 | 9</td><td colspan="2">8 | 3 | 7</td><td>56 | 5 | 46</td></tr>
<tr><td colspan="2">37</td><td colspan="2">16</td><td colspan="2">18</td><td colspan="2">26</td><td colspan="2">18</td><td>107</td></tr>
</table>

*a pair of asterisks on a row indicates that a single tagged animal was recorded in two regions during august.

Table 1. The number of tagged seals contributing data to this analysis, grouped by year region and sex. Seven of these grey seals (indicated by asterisks) moved between regions during the study periods. Data on the sex of five animals has been mislaid.

region	Survey		Population estimate (& 95% CI)	
	year	count	Simple bootstrap	Stratified bootstrap
North Sea	2008	9407	28,058 (25,200 – 31,400)	26,842 (19,600 – 42,600)
Inner Hebrides	2007, 2009	2852+350 =3202	7,979 (7,200 – 8,900)	7,633 (5,600 – 12,100)
Outer Hebrides	2008	3396+301 =3697	10,129 (9,100 - 11,400)	9,690 (7,100 – 15,400)
Orkney	2008	9251+137 =9388	27,593 (24,800 – 30,900)	26,396 (19,300 – 41,900)
Shetland	2009	1355	4,042 (3,600 – 4,500)	3866 (2,800 – 6,100)
Irish Sea	-	-	-	-
total	2007-2009	26261(+350+301+137) =27049	77,801 (70,000 – 87,200)	74,223 (54,300 – 118,300)

Table 2. The number numbers of grey seals counted during the summer surveys and population estimates resulting from scaling these up using the proportion of tagged animals hauled out. The first column comes from a simple bootstrap of the proportion of time each animal hauled out for, using the mean over the datapoints increases all the values by approximately 10%. The righthand column bootstraps the data from each day separately, with the datapoints weighted to balance the contribution of each animal, and sums the resultant distributions. The difference in the precisions comes from the simple bootstrap sampling hiding the large differences in the proportion of animals hauling out each day.

Figures

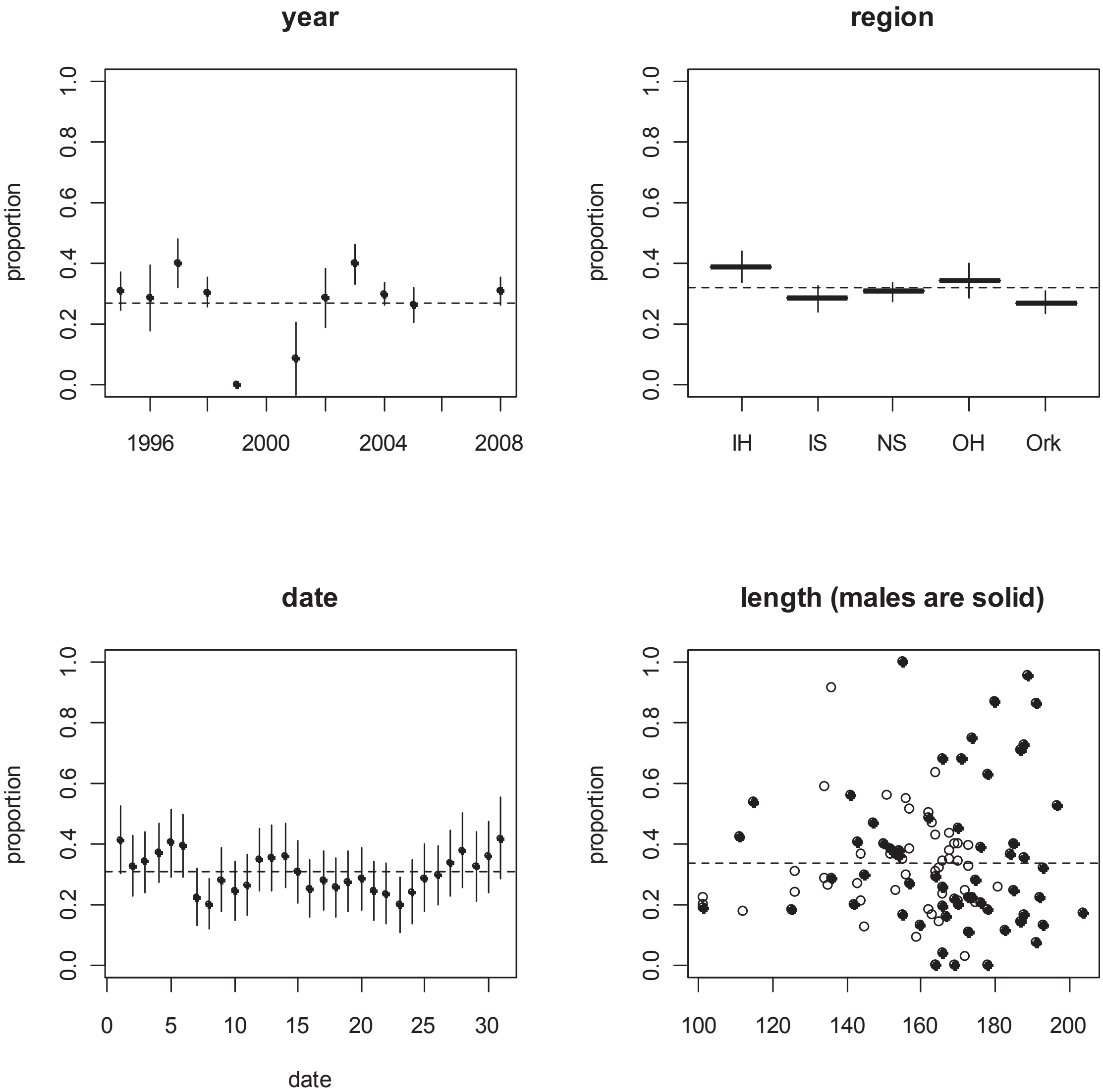


Figure 1. Proportion of the survey window (2 hrs each side of daytime low tide in August) that animals were hauled out. Means and standard errors (assuming normality and independence, so probably underestimated) for data grouped by day in august, year and region. The lower right plot shows individuals for whom lengths were recorded. Solid circles are males, hollow one females. The broken line in each pane is the mean of that set of points.

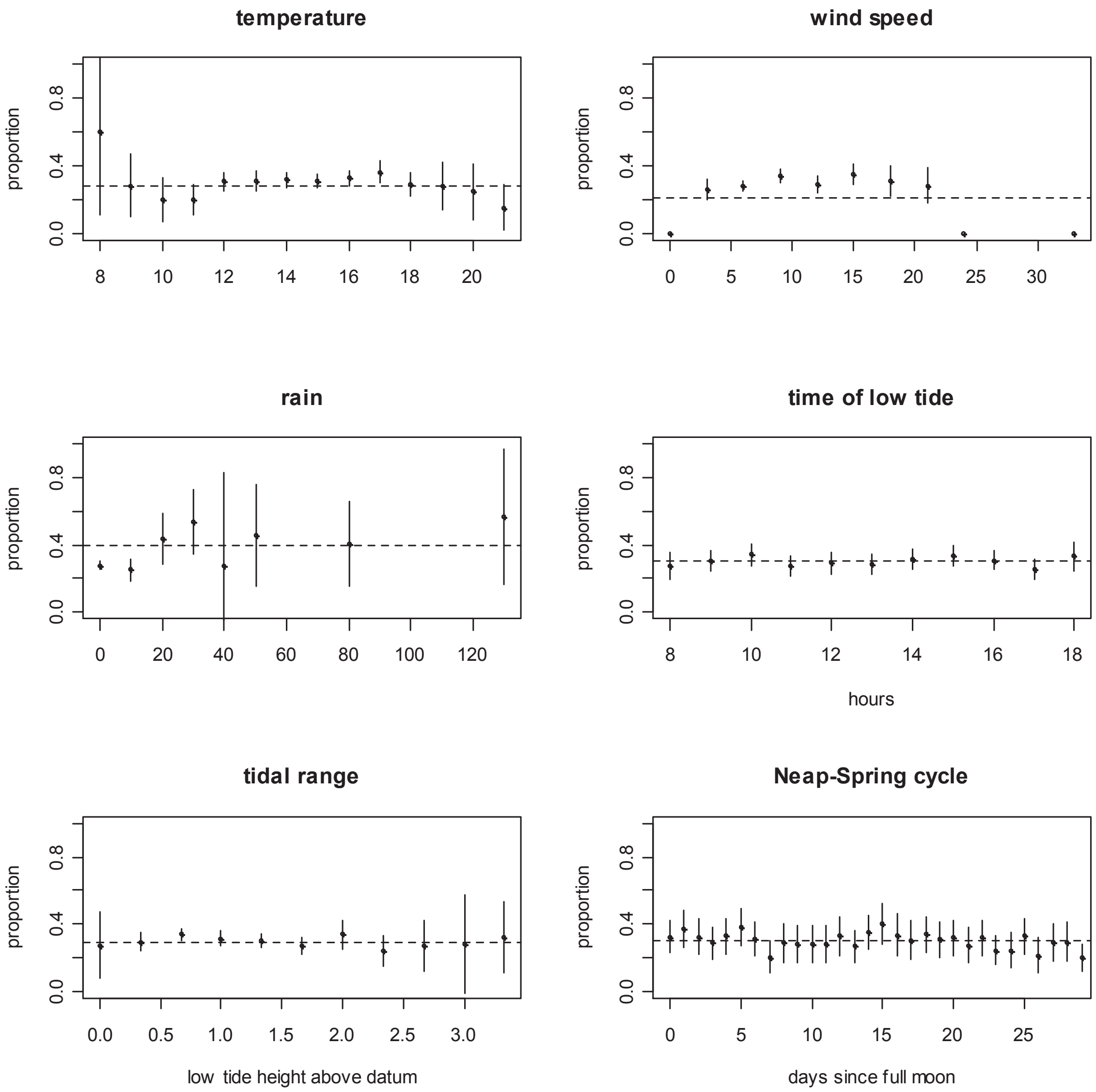


Figure 2. Proportion of the survey window (2 hrs each side of daytime low tide in August) that animals were hauled out. Means and standard errors (assuming normality and independence, so probably underestimated) for data grouped by temperature, wind speed and rain, time of day of low tide; the size of the tide; and the number of days since the previous full moon. The broken line in each pane is the mean of that set of points.

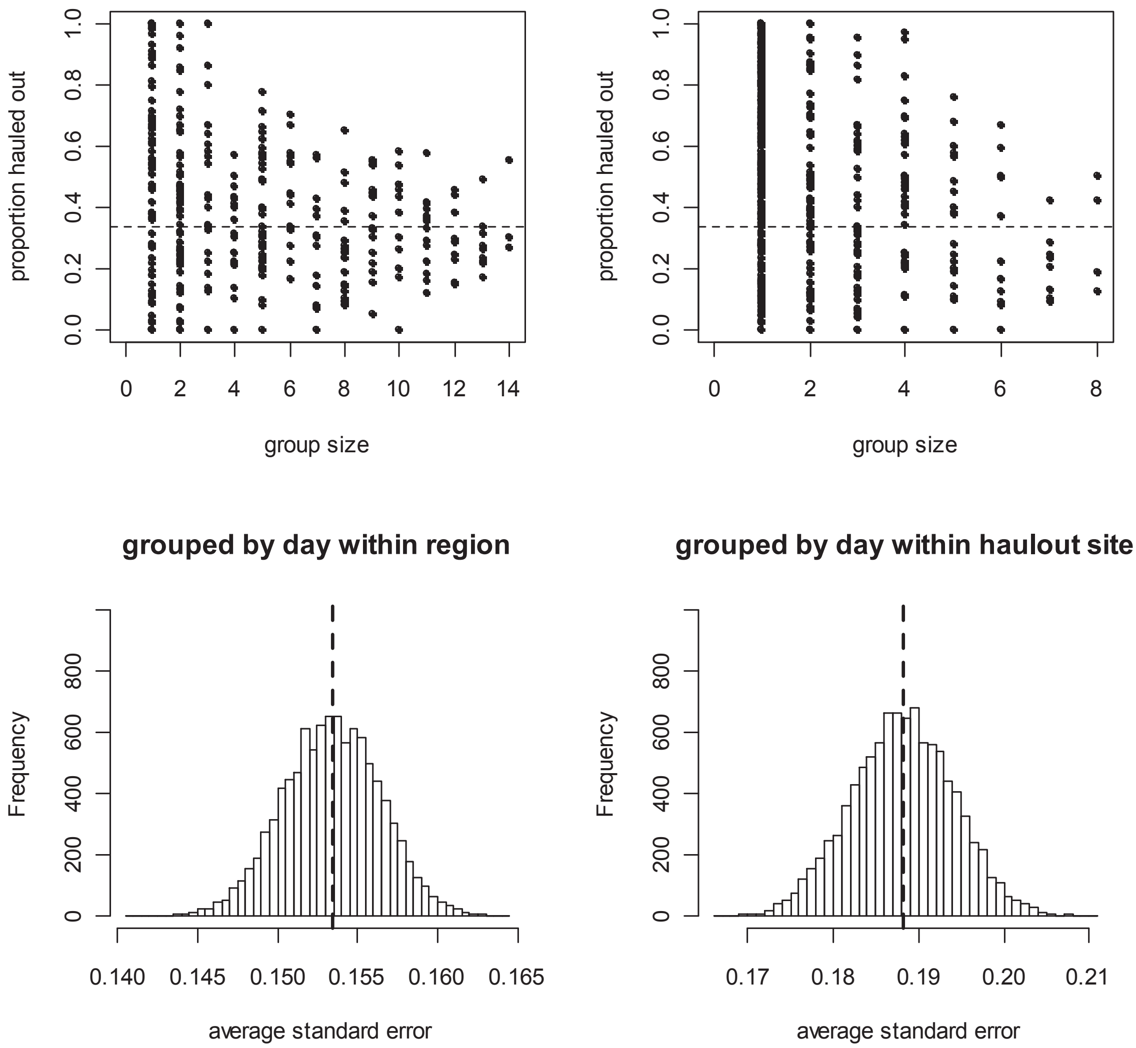


Figure 3. The variability in the proportions of animals hauled out. Each datapoint in the upper left pane is the mean haulout proportion, over the survey window, for animals in one region on one day. The upper right is the equivalent but grouped by local haulout rather than region. The lower plots compare the average standard error of the groups (broken lines) to the distribution of values resulting from permuting the values. Correlation in the animals' behaviour could be expected to produce a line lying to the left of distribution.

Mike Lonergan, Dave Thompson, Len Thomas & Callan Duck

Scaling up from pup counts to population trajectories for British grey seals.

NERC Sea Mammal Research Unit University of St Andrews, St Andrews KY16 8LB

Abstract

For British grey seals, as with many pinniped species, population monitoring is implemented by aerial surveys of pups at breeding colonies. Scaling pup counts up to population estimates requires assumptions about population structure; this is straightforward when populations are growing exponentially, but not when growth slows, since it is unclear whether density dependence affects pup survival or fecundity. We present a simple approximate Bayesian method for fitting pup trajectories, estimating adult population size and investigating alternative biological models. The method is equivalent to fitting a density dependent Leslie matrix model, within a Bayesian framework, but with the forms of the density dependent effects as outputs rather than assumptions. It requires fewer assumptions than the state space models currently used, and produces similar estimates. It is approximate because we do not explicitly define prior probability distributions for the population numbers or the density dependent effects; rather we use the pup production data in an ad-hoc, but intuitive, way to refine prior estimates of the other parameters and generate the population trajectories. We discuss the potential and limitations of the method and suggest that this approach provides a useful tool for at least the preliminary analysis of similar datasets.

Introduction

Grey seals are colonial breeders. Females mature at around six years of age and give birth to a single pup in the autumn. The pups are born on land and remain ashore for several weeks, initially with their mothers then alone. This behaviour, along with their neonatal white coats, makes the pups relatively easy to observe. Counting the other components of these populations is much less straightforward, since, while they do haul out on land, the animals spend most of their time at sea and submerged. Grey seal population estimation therefore effectively comes down to scaling up from numbers of pups.

The species is abundant around Britain and on the eastern seaboard of North America. There are also smaller numbers of animals in the Baltic Sea and around the northern European coastline. In 1914 a, pessimistic, estimate that the British population of grey seals was down to 500 individuals lead to the Grey Seal (Protection) Act. This gave some legal protection to the species (Lambert 2002). From there exponential growth at around 6-7% brought the population to around 70,000 in the 1970s.(Summers 1978). Similar exponential growth has been recorded in the grey seal population breeding on Sable Island off Nova Scotia (Bowen, McMillan et al. 2003; Bowen, McMillan et al. 2007)

Since 1984, pup production at the main Scottish grey seal colonies has been monitored by series of aerial surveys carried out throughout the breeding season. Each year between 3 and 6 flights are made over each colony, using a fixed-wing aircraft with a vertically fitted large format camera (Hiby *et al.* 1987). The numbers of animals in each photograph are counted and used to estimate the total numbers of pups that were born at each colony. Equivalent counts are made directly by observers on the ground at the colonies in England. A consistent methodology has been used to estimate total numbers throughout this study and, where sufficient surveys have been completed, calculate the estimates' precision (Thomas *et al*. 2005; SCOS 2007). Previous analyses have summed the data within each of four regions: the North Sea (effectively defined as the eastern coastline of the UK from the Firth of Forth to the Thames), Orkney, the Inner Hebrides and the Outer Hebrides. We follow this and use the total pup production estimates from each area (figure 1) as inputs to our models.

While the population was growing exponentially, scaling up from pup production estimates to total population size was relatively straightforward, requiring only estimates of the proportion of females breeding and the sex ratio. However, around 1995, the previously steady growth started to slow in some regions (figure 1). This has been interpreted as indicating the onset of density dependent population regulation. More recent estimates of population size depend critically on the assumptions made about where in the species' lifecycle density dependent effects occur. A set of Bayesian state space models has been used to model the population and advise government agencies involved in its management (SCOS 2007). State space modelling involves specifying two linked sub-models: the first, the "process model", represents the evolution of the true, but unknown, state of the system (i.e., the number of seals of different ages in each region in each year); while the second, the "observation model", represents the relationship between observations (pup production estimates) and states (true numbers of pups). The process sub-model can be viewed as a type of stochastic, age structured matrix model of the species population dynamics. For grey seals, various sub-models have been investigated, containing different assumptions about the component of the population subject to density dependent regulation, and about movement of animals between regions. Each model was fitted to the pup production data, with prior distributions specified for population numbers in the initial year, and for all model parameters (fecundity, the survival of pups and adults, carrying capacity, animal movement, and observation error). More details of the approach are given in Newman et al. (2006, 2009), Buckland et al. (2009) and Thomas et al. (2007). Their analysis of this dataset assumed that environmental carrying capacity was the only parameter to vary between regions. Fitting the models is computationally intensive, requiring statistical expertise, customised software and many hours on a fast PC to fit each model to 23 years of pup production data (see comparison of algorithms in Newman et al. 2009). This has led us to attempt a simplification that may be accessible to a wider range of biologists and conservation managers.

Material and Methods

We fitted generalised additive models, with log link functions and gamma (a parameter that reduces the tendency of these model to overfit data) set to 1.4 (Wood, 2006), separately to each of the four regional pup production time series. These were simple empirical models that used a cubic spline to smooth the observed data. The mgcv library within the R statistical environment (R Development Core Team, 2006) was used for this. A quasi-Poisson error structure was adopted for all the models as the data is based on counts..

Pup production in each region showed a period of exponential growth, though with different annual growth rates (figures 1 and 2). Separate deterministic age structured matrix models were therefore fitted for the female population in each region. 10,000 replicate pup production trajectories were calculated for each one, using the Bayesian covariance matrices for the gam models to allow for the dependencies between the parameters of their smooth terms (Wood 2006). Each trajectory's maximum annual growth rate was then calculated.

Two sets of incomplete age-structured matrix models of the females within each population were then constructed, with one assuming that all the density dependence was in fecundity and the other putting it all into pup survival. These were incomplete, and differed from those in Newman *et al*. (2009), by not characterising the step within which density dependence occurred. Five annual age classes were used, along with a sixth that contained all the older animals. Only animals within the oldest category were considered to breed.. For the variable fecundity model, each replicate's one-year old females, $f_{1,t}$, were calculated by dividing half the previous year's pups, p_{t-1}, (so assuming an equal sex ratio at birth) by the pup survival parameter, s_p :

$$f_{1,t} = p_{t-1} /(2\ s_p)$$

A similar process was used to fill in the subsequent 2, 3, 4 and 5-year-old classes, but using an "adult" survival parameter, s_a:

$$f_{i,t} = f_{i-1,t-1} /s_a \qquad i=2,3,4,5$$

. The numbers of individuals in the older age groups during the early years of the study were estimated from the stable age structure for an exponentially growing population. The numbers of adult, six-plus, females were then projected forwards throughout the dataset:

$$f_{6,t} = (f_{5,t-1} + f_{6,t-1})/s_a$$

Each year's effective fecundity was then calculated.

Equivalent calculations were made for the model with density dependent pup survival, though these used the fecundities to calculate numbers of adults, then worked back down in age. Within this model, the recent younger age classes were filled in using pup survival estimates generated from the data from the years with most similar estimated adult numbers. Further details and code for these calculations are contained in the appendix.

Scaling the replicate pup production trajectories up into population trajectories requires them to be combined with suitable sets of demographic parameter values. Newman *et al*. (2009) provide separate prior distributions for maximum (low population density) fecundity and adult and pup survival (table 1). Each set of demographic parameter values would result in a particular exponential growth rate for a population at low density. The maximum growth rate within each pup production trajectory (figure 2) was taken as an estimate of the low density growth rate for that replicate population. The demographic parameter values then need to be drawn from their joint conditional probability distribution given the appropriate exponential growth rates. Explicitly calculating these distributions is not straightforward, but they can be approximated numerically by drawing from an unconditional joint probability distribution for the demographic parameters and discarding those results whose maximum growth rate falls outside a small neighbourhood of the required value. 10,000 sets of parameter values were drawn from Newman's priors and the rates of stable exponential growth that each would produce calculated. Each of the replicate pup trajectories was then associated with the set of demographic parameter values that produced the most similar exponential growth rate, and the sets of deterministic matrix models populated using these values.

Two different methods were used to combine the results of the two models of each region. In one the two posterior distributions of population estimates were simply summed, while the other defined an informal uniform prior on where the result lay between the two extremes and repeatedly drew two uniform random variables, using one to identify a replicate and the other to determine the weighting of the results of the two population models for that replicate, creating a distribution that effectively smeared across the two directly modelled extremes. Total, rather than female only, population estimate distributions were then calculated by multiplying each replicate by a draw from a normal distribution with mean 1.73, the value used in previous analyses of this data (SCOS 2007) and standard deviation of 0.1, to allow for the uncertainty in the sex-ratio within these populations.

Results

Figure 1 shows the smoothed pup production trajectories for each of the regions. It can be seen that the growth rates have at least slowed substantially everywhere except in the North Sea. In the Outer Hebrides the highest pup production estimate occurred before 2007 in all but 64 out of the 10,000 replicate trajectories, implying that a significant decline has occurred in that region. For 95% of these replicates, the highest values occurred within the period 1995-2002. Everywhere except the North Sea, the density dependent effects cause the pairs of matrix models to diverge (figure 3). Estimates of the 2007 population are given for each region in table 2, and have a slightly higher precision than those produced by the state space models (SCOS 2008). Adult survival is the only demographic parameter substantially altered by the model fitting (table 1).

The two sets of models show different patterns of density dependence. When all the density dependence is assumed to occur in fecundity, there is a clear overall decline in this parameter with population size (figure 4), while the models with only density dependence in pup survival show highly variable survival at intermediate populations (figure 5). This imprecision may suggest that these models are struggling to fit the data because they are misspecified. From this it might be tentatively hypothosised that the models with density dependent pup survival may be less realistic than the ones incorporating density dependent fecundity. However, testing this would be difficult without explicit assumptions about the functional form of the density dependence.

Discussion

The two models agree that there are probably slightly more than 20,000 grey seals that breed on the eastern coasts of England and Scotland (our North Sea region) and that this population is continuing to grow in a near exponential fashion. In all the other areas the predictions diverge rapidly with the models containing density dependent fecundity producing estimates for 2007 that are 2-3 times as large as the equivalent figures for density dependent survival. The confidence intervals of these pairs of models do not overlap. Outside the North Sea, the precision of the population estimates would be greatly improved if it were possible to distinguish where in the grey seal lifecycle density dependence impacts most strongly. Because they do not specify the functional form of the density dependence, the models presented here can give little information on this. It is also difficult to extract this information from the more complex state space models of this system, even though these do explicitly assume the form of the density dependence (Thomas et al, 2005; Newman 2009), probably because the connection between the data (pup counts) and the required information (location of density dependence in the lifecycle) is through the, initially unknown, population size. Additional information, independent of that used here, is therefore required.

The approach presented here is Bayesian in the sense that it uses prior distributions on the density-independent demographic parameters (adult survival, maximum pup survival, maximum female fecundity). However, it is approximate because there are no formal priors on the density dependent components of the model or the population sizes, and therefore no complete likelihoods for the results. Rather than a fully parametric model, we have opted for a simpler, semi-parametric approach with an informal approach to model fitting. The pup production data are used to derive a maximum growth rate for the early part of the time series, and this is used, with an assumption of the location of density dependence in the species' lifecycle, to derive the adult population sizes together with approximate graphical representations of the density dependent effects. This strategy has some similarities to that adopted in Approximate Bayesian Computation (Toni & Stumpf 2009), where explicit calculation of the likelihood is avoided by the use of easier-to-compute summary statistics. The process of sampling parameter values from prior distributions is stochastic and Monte Carlo error can therefore influence the outcome.

Repeating the model fitting produced population estimates and confidence intervals within 1% of the values reported here, suggesting that this effect is small. It could be further reduced by increasing the number of replicates used. The calculations reported in this paper took around 10 minutes to run, on a laptop with a 2.33 Ghz Intel Core 2 Duo processor and 2Gb of RAM.

This methodology effectively pushes all the uncertainty in the system into the error terms of the gams. These models therefore have lower precision than the colony based pup production estimates and estimate each year's expected, rather than actual, pup production. The uncertainty then passes through into the population estimates and could be expected to inflate their credibility intervals. The similarity of the credibility interval widths to those from the more detailed state space models suggests that the additional effects, such as demographic stochasticity and movement between areas, which are explicitly represented in those models, may have limited impact on the precision of their results in this case. This is to be expected, since demographic stochasticity should be small for such large populations, and the estimated amount of movement between colonies is also small (inspection of posterior movement parameter estimates reported in SCOS 2008). Alternatively, the extra parameters and assumptions about the functional forms for density dependence and movement within the state space models, may absorb a sufficiently large proportion of this small dataset to negate the benefits of their more accurate representation of the system. Another possibility is that the use of different demographic parameter values in each region, made possible by the other simplifications in model structure, is the key to the performance of these scaled gam models. Additionally, the process of matching each replicate's maximum growth rate to that of a set of demographic parameter values, rather than simply drawing directly from the priors, may actually extract most of the information available to the more complete Bayesian analysis.

The uniform prior on the relative impact of density dependence on fecundity and pup survival is clear and unambiguous. It is much easier to calculate than a set of intermediate models, and reflects the current state of ignorance as to the true balance between these factors. While it is straightforward to apply here, it might be harder to justify its combination with formal likelihood based model selection techniques, such as Akaike's Information Criterion (Burnham and Anderson 2001), which penalise models for including additional parameters. Such formal model selection techniques are inapplicable to the approach described here because it does not estimate an explicit likelihood.

Our approach could be seen as a retrograde step, since it does not attempt as complete a description of the system or utilisation of the data, as the state space models. It could also be criticised for its limited predictive and explanatory power. However, any projection of models requires extrapolation, and needs to be done cautiously. For these populations, the most obvious danger would be in the projection of density dependent effects beyond the range of existing data, which requires a belief that their functional forms have been adequately described. It is also possible that, if the state space models were modified in the light of these results, for example by modifying them to allow adult survival to vary between areas, the precision of their estimates would improve. However, as the most appropriate analysis of datasets will always depend on their size and the availability of resources, this sort of less demanding methodology may also be appropriate for other small datasets.

Supplementary material

The following supplementary material is available at SMRU: additional figures; technical details of the model, its fitting and diagnostics; and the code used to fit the models.

Acknowledgements
Collection of aerial survey data is funded by NERC and involves many people at the Sea Mammal Research Unit and elsewhere. Additional data were collected by National trust, Lincolnshire Wildlife Trust and SNH, The analysis and its interpretation have been discussed with many colleagues at St Andrews and the members of the NERC Special Committee on Seals. We are grateful to them all.

References

Bowen, W.D., McMillan, J. and Mohn, R. (2003) Sustained exponential population growth of grey seals at Sable Island, Nova Scotia. ICES Journal of Marine Science: 60(6):1265-1274
Bowen, WD; McMillan, JI; Blanchard, W (2007) Reduced population growth of gray seals at Sable Island: Evidence from pup production and age of primiparity Marine Mammal Science 23(1):48-64
Buckland, S. T., K. B. Newman, C. Fern´andez, L. Thomas and J. Harwood.(2007). Embedding population dynamics models in inference. Statistical Science 22, 44–58.
Burnham, K.P. and D.R. Anderson. (2001).Model Selection and Inference. 2nd Edition. Springer-Verlag, New York.
Hiby, A.R., Thompson D. and Ward, A.J.. (1987) Improved census by aerial photography - an inexpensive system based on non-specialist equipment. *Wildl. Soc. Bull.* **15**: 438-443.
Lambert, R. (2002) The Grey Seal in Britain: a twentieth century history of a nature conservation success. Environment and History 8(4):449-474.
Newman, K.B., S.T. Buckland, S.T. Lindley, L. Thomas & C. Fernández. (2006). Hidden process models for animal population dynamics. Ecological Applications 16: 74-86.
Newman, K.B., C. Fernández, S.T. Buckland and L. Thomas. (2009) Monte Carlo inference for state-space models of wild animal populations. Biometrics. doi:10.1111/j.1541-0420.2008.01073.x
R Development Core Team (2006). R: A language and environment for statistical computing. Vienna: R Foundation for Statistical Computing, URL http://www.R-project.org
SCOS 2007. Scientific advice on matters related to the management of seal populations: 2007. Reports of the UK Special Committee on Seals, available at http://smub.st-andrews.ac.uk
SCOS 2008. Scientific advice on matters related to the management of seal populations: 2008. Reports of the UK Special Committee on Seals, available at http://smub.st-andrews.ac.uk
Summers, C.F. (1978) Trends in the Size of British Grey Seal Populations. Journal of Applied Ecology 15:2:395-400.
Thomas, L., S. T. Buckland, K. B. Newman and J. Harwood. (2005). A unified framework for modelling wildlife population dynamics. Australian and New Zealand Journal of Statistics 47, 19–34.
Toni, T., and M.P.H. Stumpf (2009) Simulation-based model selection for dynamical systems in systems and population biology. Bioinformatics, Advance Access published on October 29, 2009; doi:10.1093/bioinformatics/btp619
Wood, S.N. (2006) Generalized Additive Models: An Introduction With R, Chapman and Hall/CRC

Figure captions

Figure 1: Grey seal pup production estimates (points) and smoothed estimates (with 95% credibility intervals) for each of the four regions.

Figure 2: Annual growth rates for smoothed pup productions. Each point (black = North Sea; red = Orkney; green= Inner Hebrides; blue = Outer Hebrides) is the mean of the annual change in the 10,000 replicate draws from the gam model matrix, the vertical bars contain 95% of the runs). The increase in the growth rate in Orkney may be due to the ending of culling there in the early 1980s.

Figure 3: Population trajectories (mean values and 95% credibility intervals) for each region. In each case the lowest (dotted) set of lines are the smoothed pup production estimates; the middle (solid) sets are the total population estimates from the density

dependent pup survival models and the upper (dashed) sets of lines those from the models with density dependent fecundity.

Figure 4: Patterns of fecundity estimated within the model with density dependent fecundity. Each dot in the black clouds represents the estimated fecundity calculated for one year of one replicate in a matrix model. The red lines are the median and 2.5th and 97.5th percentiles of these results.

Figure 5: Patterns of pup survival estimated within the model with density dependent pup survival. Each dot in the black clouds represents the estimated fecundity calculated for one year of one replicate in a matrix model. The red lines are the median and 2.5th and 97.5th percentiles of these results.

Table captions

Table 1: Distributions of parameter values. The priors are taken from Newman et al. (2009), the posterior values are those from the replicate model runs in each region.

Table 2: Estimates (mean and 95% credibility intervals) of the total size of the grey seal populations in each region before breeding in 2007. The results for the two models are given along with those from simple (equally weighted) model averaging and applying the uniform prior across the two models. The numbers in italics are the equivalent estimates calculated from the best fitting state space models contained in the 2008 report of the UK Standing committee on Seals (SCOS 2008).

[1] all the CIs include uncertainty in the population sex-ratio.

[2] the CIs are estimated conservatively by summing those of the individual models.

Short title: British grey seal population trajectories.

Tables

Table 1:

	symbol	prior			posterior							
					North Sea		Orkney		Inner Hebrides		Outer Hebrides	
		distribution	mean	sd	mean	sd	mean	sd	mean	sd	mean	sd
max pup survival	s_p	Beta(14.53,6.23)	0.7	0.1	0.7	0.1	0.7	0.1	0.7	0.1	0.7	0.1
adult survival	s_a	Beta(22.05,1.15)	0.95	0.04	0.92	0.01	0.96	0.01	0.95	0.02	0.91	0.01
max fecundity	b	Beta(22.05,1.15)	0.95	0.04	0.95	0.05	0.95	0.05	0.95	0.05	0.95	0.05

Table 2:

Model	2007 Regional Population (in thousands, mean & 95%CIs[1])				
	North Sea	Orkney	Inner Hebrides	Outer Hebrides	Total[2]
Density dependent pup survival	20.9 (16.4-25.7) *17.1 (10.6-25.9)*	46.1 (35.6-58.0) *60.9 (40.9-93.5)*	8.0 (5.9-10.7) *8.3 (6.5-10.5)*	34.3 (27.0-42.0) *31.3 (24.0-39.1)*	109.4 (84.8-136.4) *117.6 (89.1-168.9)*
Density dependent fecundity	24.1 (19.7-29.0) *27.2 (20.7-38.2)*	124.6 (102.2-151.1) *103.0 (79.5-142.9)*	24.7 (18.3-34.2) *21.4 (16.5-32.1)*	69.6 (57.0-86.0) *88.1 (67.0-143.0)*	243.3 (190.8-277.9) *239.7 (188.8-356.2)*
model averaged	22.5 (17.1-28.3) *20.5 (11.1-33.6)*	85.4 (37.2-146.2) *75.4(40.4-130.0)*	16.4 (6.2-32.3) *12.9 (6.5-27.2)*	52.1 (28.1-82.5) *51.2 (23.8-111.5)*	177.3 (88.5-289.2) *160.1 (84.5-304.5)*
uniform prior	22.5 (18.0-27.3)	85.78 (45.8-131.3)	16.3 (8.0-27.7)	52.1 (33.1-74.3)	176.8 (104.6-260.8)

Figure 1:

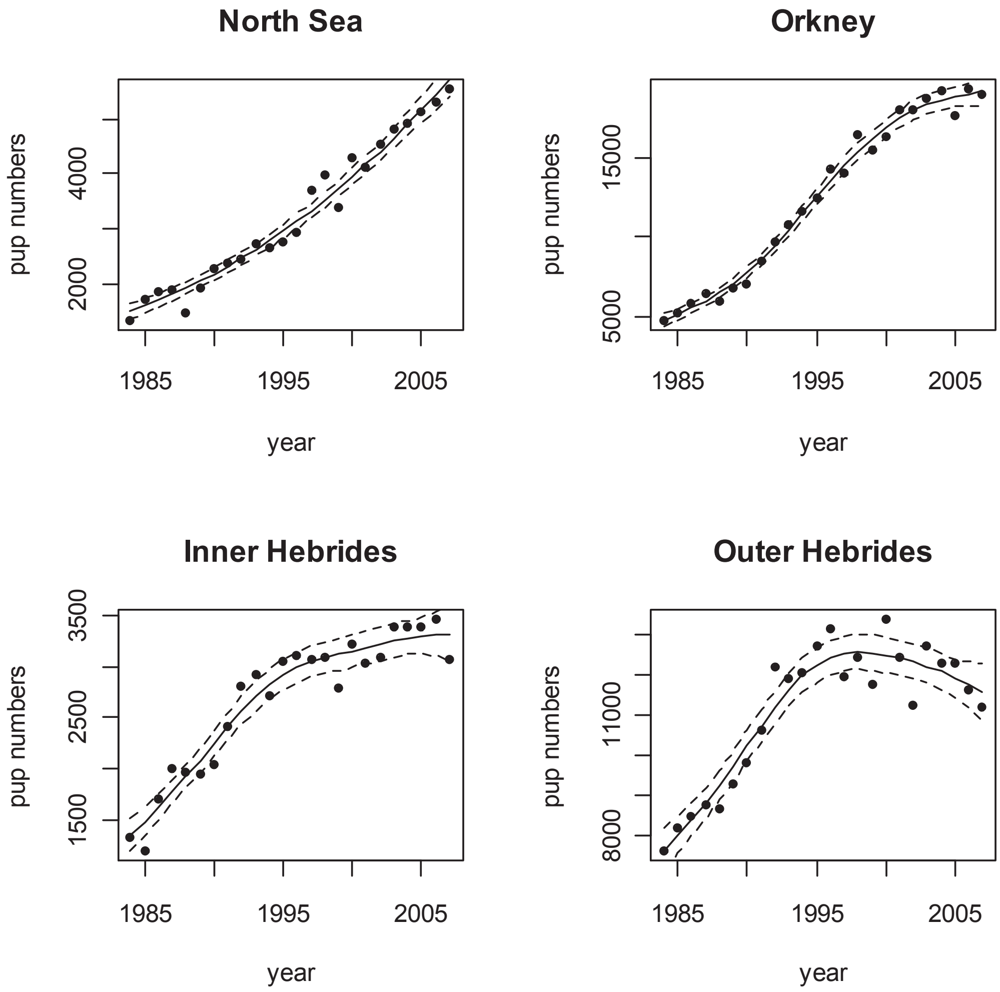

North Sea
pup numbers
2000
4000
1985
1995
2005
year
Orkney
pup numbers
5000
15000
1985
1995
2005
year
Inner Hebrides
pup numbers
1500
2500
3500
1985
1995
2005
year
Outer Hebrides
pup numbers
8000
11000
1985
1995
2005
year

Figure 2:

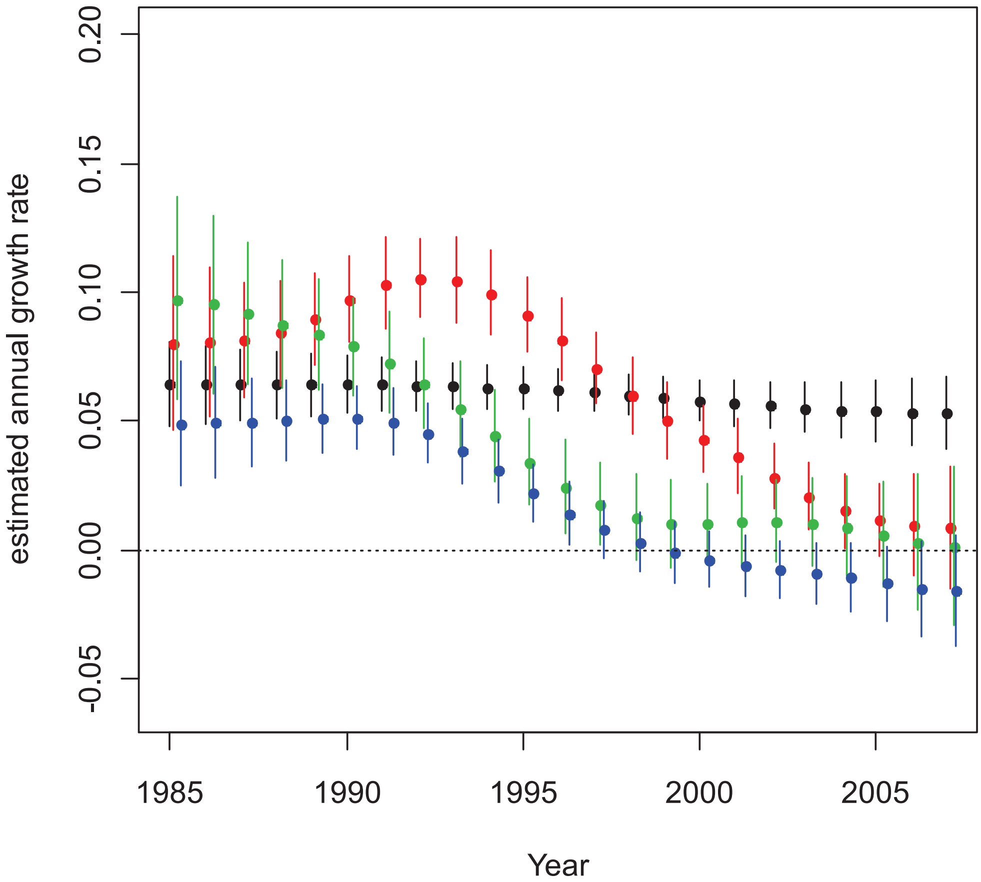

estimated annual growth rate
-0.05
0.00
0.05
0.10
0.15
0.20
1985
1990
1995
2000
2005
Year

Figure 3:

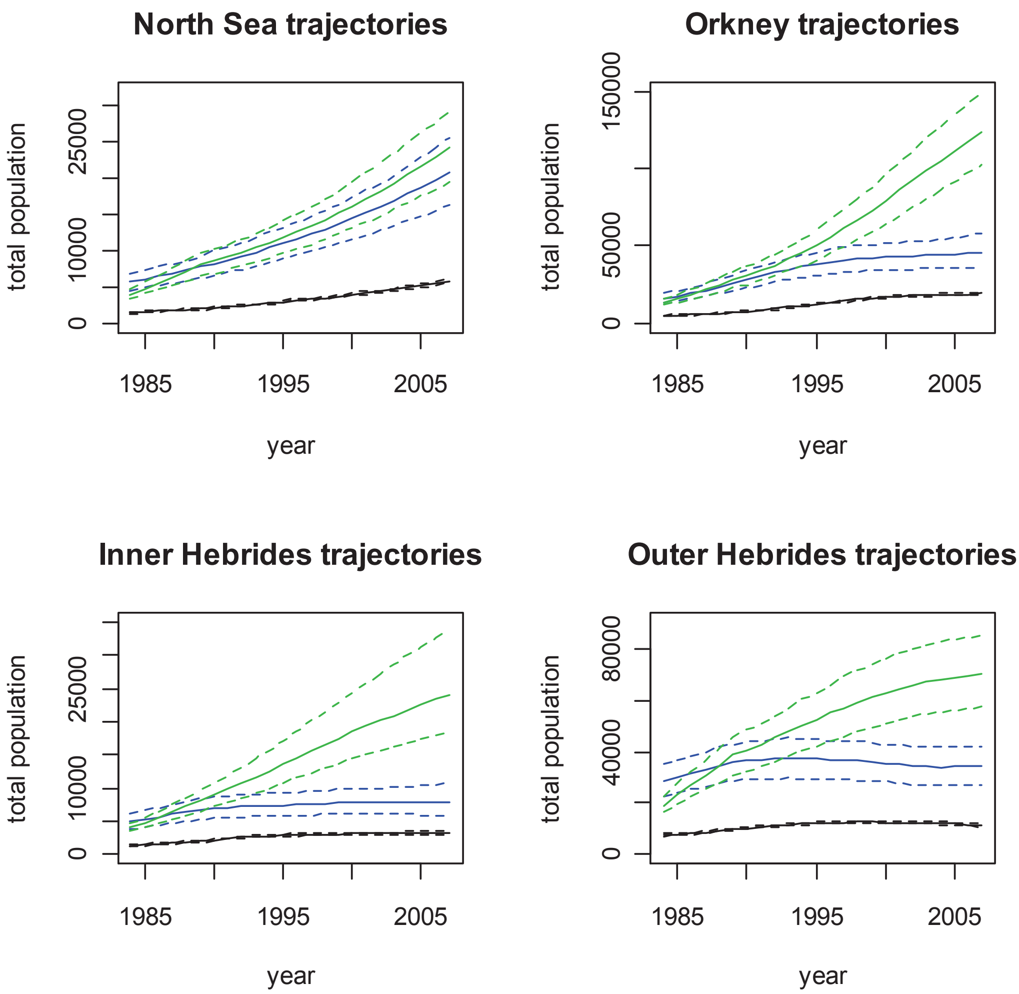

North Sea trajectories
total population
0
10000
25000
1985
1995
2005
year
Orkney trajectories
total population
0
50000
150000
1985
1995
2005
year
Inner Hebrides trajectories
total population
0
10000
25000
1985
1995
2005
year
Outer Hebrides trajectories
total population
0
40000
80000
1985
1995
2005
year

Figure 4:

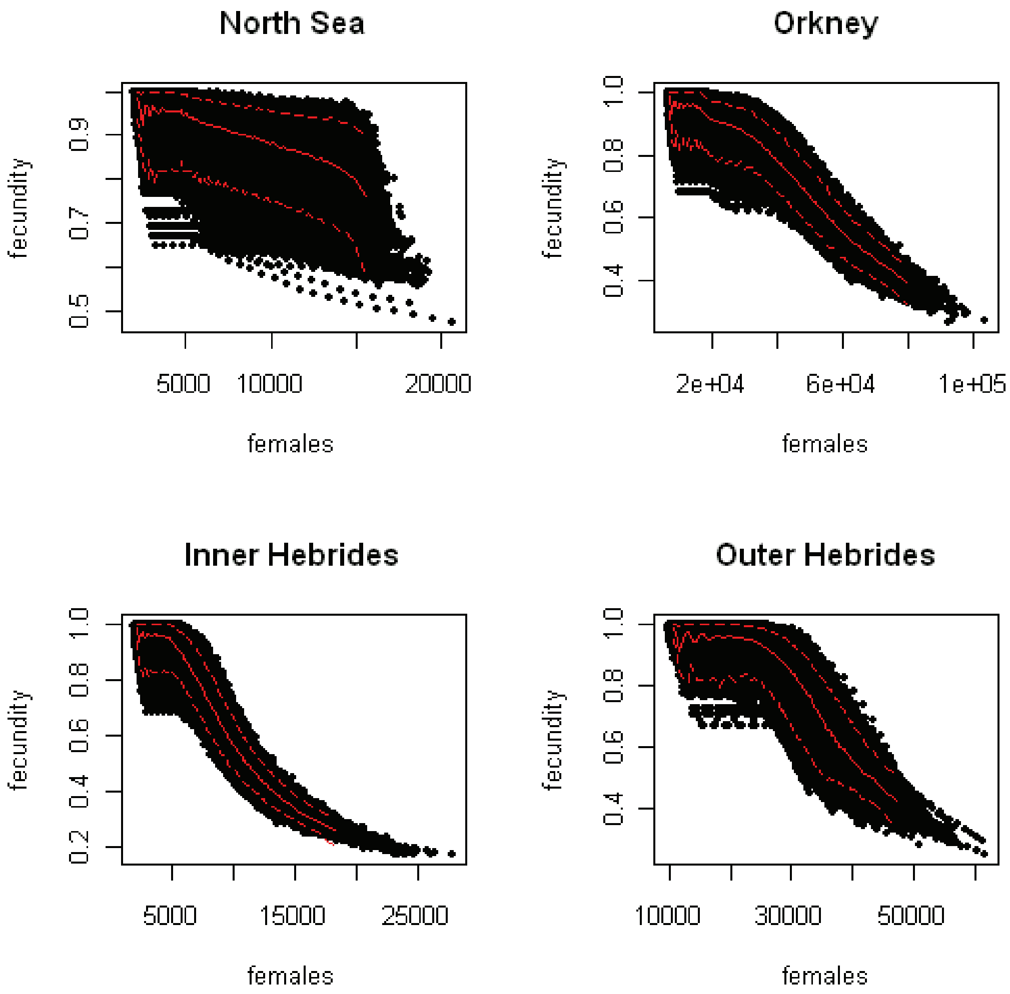
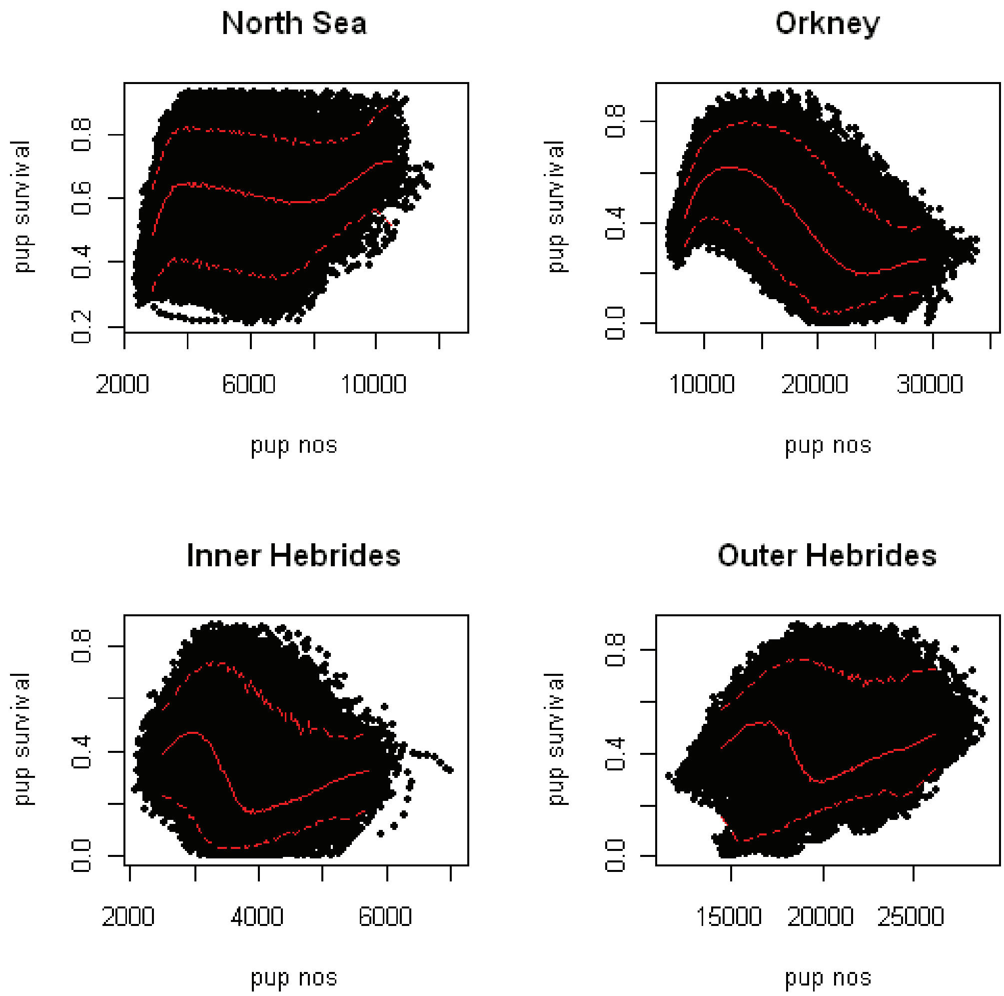

North Sea
fecundity
0.5
0.7
0.9
5000
10000
20000
females
Orkney
fecundity
0.4
0.6
0.8
1.0
2e+04
6e+04
1e+05
females
Inner Hebrides
fecundity
0.2
0.4
0.6
0.8
1.0
5000
15000
25000
females
Outer Hebrides
fecundity
0.4
0.6
0.8
1.0
10000
30000
50000
females

Figure 5:


North Sea
pup survival
0.2 0.4 0.6 0.8
2000 6000 10000
pup nos
Orkney
pup survival
0.0 0.4 0.8
10000 20000 30000
pup nos
Inner Hebrides
pup survival
0.0 0.4 0.8
2000 4000 6000
pup nos
Outer Hebrides
pup survival
0.0 0.4 0.8
15000 20000 25000
pup nos

Ailsa J. Hall

The trophic transfer of biotoxins to Scottish harbour and grey seals.

Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, St Andrews KY16 8LB

Summary

Following our recent finding that domoic acid (DA) exposure is occurring in Scottish harbour seals (Hall and Frame, 2010), a study, funded by the Scottish Government, is now being carried out to investigate the spatial extent of this exposure (particularly in the spring and summer months) and to determine exposure to other biotoxins (particularly saxitoxin) which are also found in Scottish waters. Here we present some preliminary findings of this study and evidence that DA is found in intertidal waters. Detectable levels were measured just above the seabed. In addition DA is taken up by various fish species, particularly benthic flatfish and squid. These fish and cephalopods are major prey species of both grey and harbour seals (particularly plaice and dab in which the highest levels were found). In addition we are continuing to find DA in faecal samples collected from seal haulouts during the spring and early summer often (although not exclusively) following major blooms of *Pseudo-nitzschia*. We also found that a proportion of faecal samples from grey seals on the east coast of Scotland (Tay estuary) were also positive. Although the foraging areas for grey seals are more offshore than those of harbour seals, studies at sites where both species occur have found overlap in the diet. Thus both grey and harbour seals are consuming contaminated prey but harbour seals may be more at risk if the finding that DA is more common in coastal and intertidal waters is borne out. Saxitoxin (paralytic shellfish poisoning toxin) was found at low levels in some of the seal fish prey items, but no positive faecal samples have so far been detected.

Introduction

We recently found that Scottish harbour seals (*Phoca vitulina*) are exposed to domoic acid (DA) (Hall and Frame, 2010), a biotoxin produced by diatoms of the *Pseudo-nitzschia* group of phytoplankton. DA, when consumed in sufficient quantities, is highly neurotoxic and can be lethal to humans and wildlife (Landsberg, 2002). Following this finding the Scottish Government are currently funding a study to:

(a) determine the extent of the exposure to DA, using subsamples of the faecal samples being collected to determine the diet of Scottish harbour (and grey) seals

(b) investigate exposure to other biotoxins also found in Scottish waters.

Although these toxins have a high turnover rate and short half life in body fluids, they may persist for several days in faecal material, making it a useful matrix for exposure estimation.

Exposure to DA in harbour seals was the focus of the initial study (Hall and Frame, 2010) as it was hypothesised that biotoxins may be a potential factor in the decline in abundance but samples from Scottish grey seals (*Halichoerus grypus*), collected during the spring and summer months following the spring algal blooms, have not yet been screened.

In addition, it is important to establish the trophic link between the biotoxins released into the seawater following a harmful algal bloom event, the fish prey and the biotoxins in the seals. Although there is a very effective monitoring scheme for determining biotoxins in shellfish to protect human health and the aquaculture industry in the UK, levels in fish are not generally analysed. The risk for human health through fish consumption is low as the biotoxins are concentrated in the viscera and are found at very low levels in the flesh (Lefebvre et al., 2002a). Studies in the US have found that DA is rapidly transported down through the water column, adsorbed onto particles. Detection of DA in bottom sediments also indicates that the toxin may persist long after the *Pseudo-nitzschia* blooms. Their results indicate that vertical fluxes of DA could explain the high levels of DA previously observed in benthic organisms (Sekula-Wood et al., 2009; Lefebvre et al., 2002b). A study of DA levels in benthic and benthopelagic fish species in Monterey Bay, California found concentrations were indeed significantly higher in the benthic species (Vigilant and Silver, 2007). DA was also detected in benthic flatfish when there were no or few toxic cells in the surface water.

Here we report the initial findings of DA in the water column in the mouth of the Eden Estuary, St Andrews Bay, in the spring and early summer, 2010. The presence of DA was determined using Solid Phase Adsorption Toxin Tracking (SPATT) resin bags (Mackenzie et al. 2004) developed for monitoring DA at shellfish farm sites (Fig 1.) and kindly donated by Marine Scotland, Aberdeen. These bags are being attached to buoys moored in the mouth of the Eden, placed at differing depths between May to August with the assistance of the Dept. Geography and Geosciences, University of St Andrews.

A box of discarded fish from a prawn fishing vessel working out of Anstruther harbour, Fife, was collected and the levels of DA and saxitoxin (STX) measured in the fish and cephalopod viscera. STX is a biotoxin often found in Scottish waters that causes paralytic shellfish poisoning (PSP) in humans. It is produced by dinoflagellates of the genus *Alexandrium*.

Finally, DA levels in faecal samples collected between March and July from four sites around Scotland; Orkney, the Eden / Tay Estuary; SE Islay and Shetland, have been determined. These sites included both grey and harbour seal and mixed species haulouts.

Methods

Domoic Acid ELISA

DA concentrations were determined using a direct competitive ELISA (ASP assay kits, Biosense, Norway).This method has been widely used for detecting DA in a variety of matrices including shellfish flesh, and blood, urine and faeces from marine mammals (Litaker et al., 2008) and is the Association of Analytical Communities (AOAC) official method for the measurement of DA in environmental samples. A monoclonal antibody labelled with horseradish peroxidase competes for binding to DA in the wells of a microtitre plate. A known amount of DA is bound to each well and in the presence of DA in the sample or standards competes for antibody binding sites. Following incubation and washing steps the bound proportion is detected following the addition of an enzyme that forms a coloured product whose intensity is measured using a microplate reader at a wavelength of 405nm.

SPATT bags

DA was extracted from the SPATT bags (Fig. 1) using the method supplied by Marine Scotland. Briefly, the SP-700 resin beads in the bags, onto which any toxins will have adsorbed, were rinsed with deionised water, shaken for 1 min and 4.7g transferred to a 25ml reservoir containing at 20µm frit. The beads were again rinsed with distilled water then extracted under vacuum. The toxins were then eluted with aqueous methanol (50:50) and the eluate diluted as recommend to 1:25. To date (end July 2010) three bags have been deployed and retrieved. They were suspended at depths of 1, 3 and 6m above the sea bed in the mouth of the Eden estuary.

Fish guts

The levels of DA in fish viscera were determined using the procedure for the extraction and quantification of DA in shellfish flesh. Fish were individually weighed and gutted and pools of up to 6 fish (depending on how many of each was available) were assembled. The pooled guts were then ground and 4g subsample was homogenized with a 1:4 dilution in 50% methanol for at least 1 min. Samples were centrifuged at 3000x*g* for 10 min and the supernatant retained. Samples were diluted to 1:2000 in assay buffer. Concentrations of DA and STX were measured in 19 and 15 different species respectively. All the fish were caught off the coast of Kirkcaldy, Fife, during a single fishing trip on 30th May, 2010.

Seal Faeces

Previous studies have found that tryptophan and an unidentified substance in some seal faecal samples can interfere with the antigen-antibody binding in the DA ELISA assay. Therefore the standard procedure for this matrix involves extracting the DA from the faeces (Lefebvre et al., 1999). Thus faecal DA was extracted using solid phase extraction (SPE), strong anion exchange (SAX) columns (Supelco, UK) and the method of Lefebvre et al 1999. Samples (between 2-4 g depending on how much was available) were ground then homogenized at high speed in 1:4 v/w dilution of 50% methanol and then centrifuged for 30min at 4000 x g. Supernatants were decanted and passed through a 0.22µm filter before SPE. SPE columns were first conditioned with 6ml 100% methanol followed by 3ml water and 3ml 50% methanol before addition of the sample (2ml) at a rate of one drop per second. Columns were washed with 5ml 10% acetonitrile and extracts were eluted with 0.5 M NaCl in 10% acetonitrile. The collected eulate was vortexed and further diluted 1:100 before analysis by the ELISA method (Garthwaite et al., 1998). All samples were analysed in duplicate and results calculated from a standard curve using the Excel spreadsheet macro supplied by Biosense. The limit of detection (LOD) was estimated at 2.5 mg/kg for fish, 5 ng/g for seal faeces and 1 ng/ml for the SPATT extracts.

Saxitoxin ELISA

Extracts from 15 species of fish viscera and 15 extracted faecal subsamples collected at the Eden estuary mixed haulout site (June, 2010) were also analysed for the presence of STX using a saxitoxin (PSP) ELISA kit (Abraxis LLC, Philadelphia, USA). The method is similar to the DA ELISA in which enzyme conjugate solution is incubated with anti-STX antibody in the presence of the samples or standards. After incubation the contents of the plate are discarded and an enzyme substrate solution is added. Following a further incubation the reaction is stopped and the absorbance read at 450nm. Evaluation and estimation of concentration from a standard curve was performed using a 4-parameter logistic fit using the program R (R Development Core Team, 2010)

Results

1. Domoic Acid

(a) SPATT bags

The concentration of DA in the extracts from three SPATT bags ranged from <LOD to 4.3 ng/ml. The one positive sample (SA3) was deployed at the outer Eden estuary buoy from 17th May to 28th June at a level of 1m above the seabed which is 9m deep at low water spring.

(b) Fish guts

DA was detected at levels above the maximum allowable concentration for human shellfish consumption (i.e. the concentration at which adverse health effects occur and at which the shellfish production areas are closed, namely 20mg/kg) in 9/19 species (47%) analysed. The highest levels were found in two species of flatfish (dab and plaice, 167 and 523 mg/kg respectively) and in squid (121 mg/kg Fig. 2). These species are among the main prey of both grey and harbour seals around Scotland.

(c) Seal Faeces

The number of faecal samples analysed, by region and month are given in Table 1. Also shown is whether the haulout was predominantly grey or harbour seals or if the samples were from a mixed species haulout. The samples listed as mixed have also been subsampled for the future determination of species using genetic markers. The frequency distribution of the concentration of DA in the samples by region is shown in Fig. 3. Concentrations ranged from <LOD to 34.8 ng/g. Positive samples were found in all regions although only one low positive was found in Orkney (Fig 4.), which was sampled in early spring. Interestingly 3/32 (12.5%) of the samples from an exclusively grey seal haulout site in the Tay estuary were also positive. The difference in the proportion of positive samples among locations was a significant with the Eden Estuary and Shetland having a higher proportion ($p<0.01$) than the other sites (Table 1, Fig. 4).

2. Saxitoxin concentrations in fish guts and seal faeces

Saxitoxin levels in 15 of the fish and cephalopod species were determined by ELISA. Levels were detectable but were low, generally <5 ug/100g viscera (Fig. 3). The action level for this biotoxin to protect human health is 80 ug/100g. However, it should be noted that this is a limited, random sample of mixed fish species and that further monitoring, particularly during a bloom event, may yield different results.

Fifteen faecal samples collected from the Eden Estuary on 7th June, 2010 were also screened for the presence of saxitoxin. All the samples were below the LOD. On the day the samples were collected the haulout site contained mixed species, both grey and harbour seals.

Discussion

This preliminary study has shown that DA can be detected in the intertidal water column at the mouth of the Eden estuary and in selected seal prey species collected at offshore sites, close to seal haulout sites. These results might therefore be indicative of levels in fish elsewhere in this region. The water sampling in the Eden is continuing and alhough the sample size reported here is very small, it is interesting to note that the single positive sample was found close to the seabed.

In addition the highest levels of DA were found in plaice and dab, both benthic feeding flatfish whose prey includes molluscs and invertebrates. The levels measured in the plaice viscera were very high, but comparable to the highest levels reported previously in offshore shellfish around Scotland (Stobo et al., 2008). Although this is a rather limited study, it may suggest that animals feeding on benthic prey could experience higher DA exposure. Indeed a number of other studies have also found higher DA levels in benthic than benthopelagic species (Vigilant and Silver, 2007; Costa et al., 2005). However intertidal and pelagic food webs still have the highest exposure, particularly during and just following large toxin-producing bloom events (Turner and Tester, 1997). Of some note is that a large bloom of *Pseudo-nitzchia* of 240,000 cells/L (warning level >50,000 cells/L) was reported at the shellfish monitoring site off the coast of Elie, Fife in late April, 2010. The extent of, or whether this bloom was producing DA, is not known.

DA is continuing to be found in faecal samples from harbour seals collected at various haulout sites around Scotland during the spring and early summer. In addition we have now found measurable levels in grey seal faecal samples. This is not unsurprising given the overlap in the diet of the two species in regions where they co-occur (Thompson et al., 1996). The proportion of positive samples was low (Table 1, 4/32), but comparable to the positive proportion on the West Coast at SE Islay (Table 1, 2/17) in April. In general harbour seal foraging areas tend to be closer inshore than grey seal foraging sites (Thompson et al., 1996; Cunningham et al., 2009) which may put them at greater risk if the finding that DA is higher in coastal than more offshore waters is borne out in future studies. The proportion of positive samples was significantly higher in Shetland and the Eden estuary (although the latter was a mixed haulout site). In addition to the bloom off the Fife coast in April there have been (as in previous years) numerous large blooms of *Pseudo-nitzschia* around Shetland since May 2010. The samples from Orkney were obtained very early in the season, and no major blooms have been reported from the West coast outside Loch Fyne and Loch Creran so far this year.

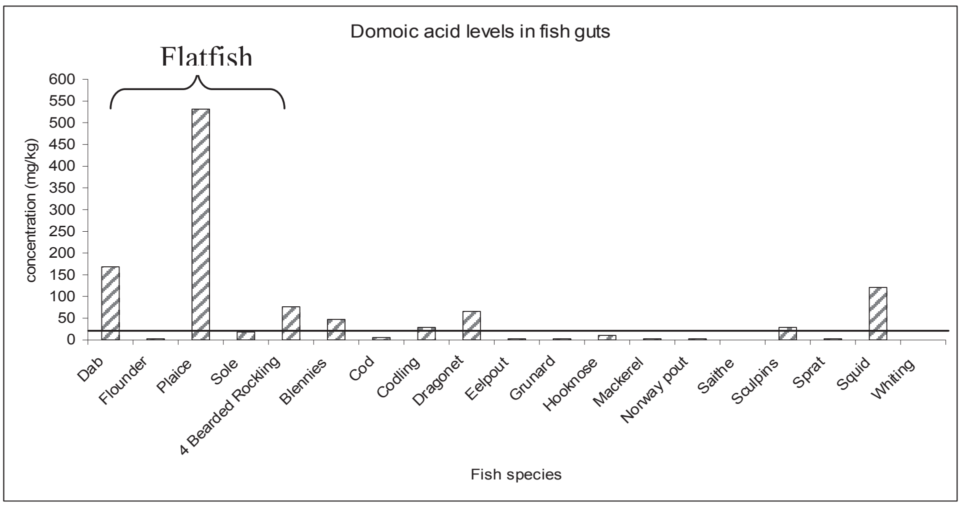
Very low levels of saxitoxin were found in various fish species and all the faecal samples collected from the Eden that were screened (n=15) were negative. Although a number of *Alexandrium* blooms have been reported this year at the Elie monitoring station, numbers of cells/L were low (around 20 cells /L) and again it is not known if any were toxin-producing blooms. The monitoring of saxitoxin in seal faecal samples will continue, particularly focussing on samples from sites where blooms are much larger such as Shetland and the Outer Isles.

Table 1. No. faecal subsamples analysed and proportion positive for DA (>5 ng/ml) by region, month and species on haulout.

Region	Month	Species	No. analysed	Proportion positive
SE Scotland, Tay Estuary	May	Grey	32	0.125
SE Scotland, Eden Estuary	June	Mixed	15	0.733
Orkney	March	Mixed	17	0.059
Shetland	June/July	Harbour	12	0.500
West Coast, SE Islay	April	Harbour	17	0.118
Total			93	0.258

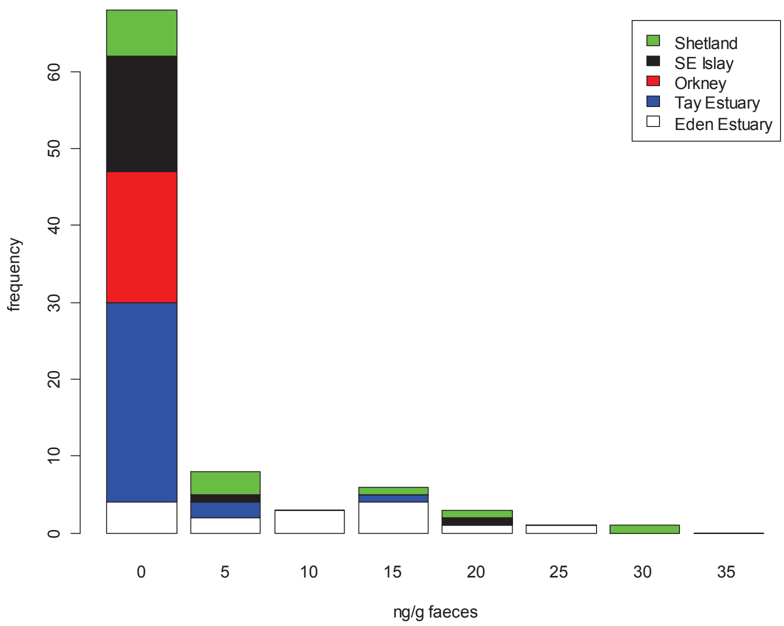
Fig. 1. Solid Phase Adsoprtion Toxin Tracking bags used to determine the presence of DA in seawater.

Fig. 2. Domoic acid levels in the guts of 19 species of fish and cephalopods, June 2010. Line shows the maximum permissible concentration in shellfish (20mg/kg) for



human consumption.

Fig. 3. Frequency distribution of DA in faecal samples from seal haulout sites around



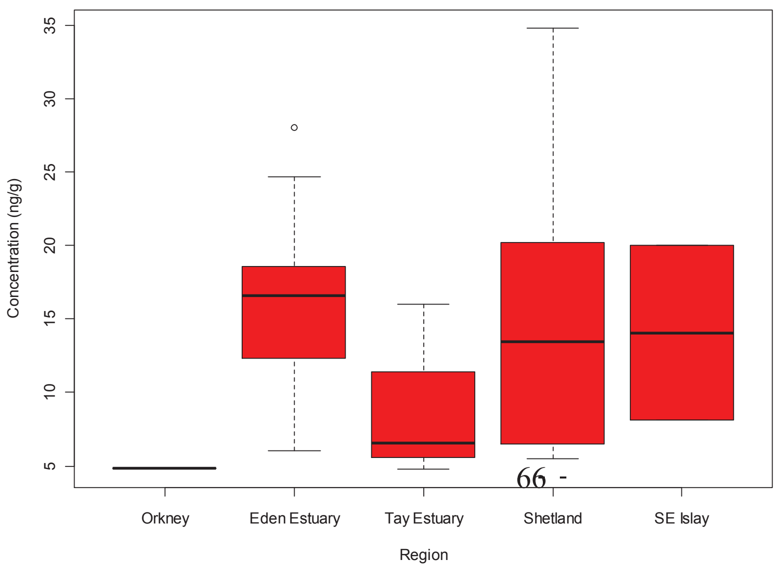
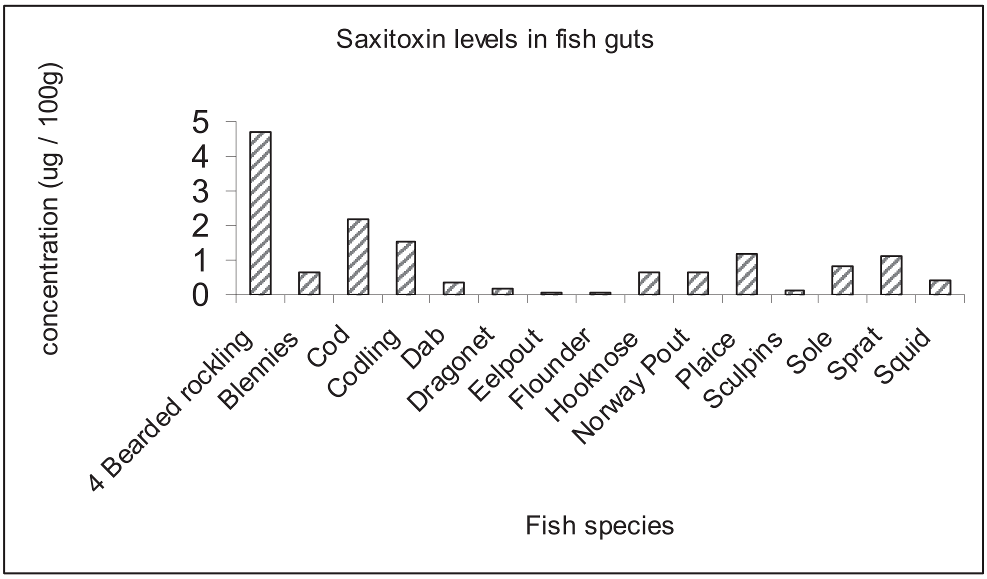


Fig. 4. Boxplot showing 25th and 75th percentiles, median and range of concentrations of DA in positive (>5ng/g) faecal samples by region (see Table 2 for

season and species

Fig. 5. Saxitoxin levels in the guts of 15 species of fish and cephalopods. June 2010. The recommended action level is 80 µg STX eq/100 g tissue.
Scotland



References

Costa P., Rosa R., Pereira J. Ad Sampayo M (2005) Detection of domoic acid, the amnesic shellfish toxin, in the digestive gland of Eledone cirrhosa and E. moshata (Cephalopoda, Octopoda) from the Portuguese coast. Aquatic Living Resources 18: 395-400

Cunningham, L, Baxter JM, Boyd IL, Duck CD, Lonergan M, Moss SE, McConnell B. 2009. Harbour seal movements and haul-out patterns: implications for monitoring and management. Aquat Conserv 19:398-407.

Garthwaite, I, Ross KM, Miles CO, Hansen RP, Foster D, Wilkins AL, Towers NR. 1998. Polyclonal antibodies to domoic acid, and their use in immunoassays for domoic acid in sea water and shellfish. Nat Toxins 6:93-104.

Hall, AJ, Frame E. 2010. Evidence of domoic acid exposure in harbour seals from Scotland: A potential factor in the decline in abundance? Harmful Algae 9:489-493.

Landsberg, J. H. (2002) The effects of harmful algal blooms on aquatic organisms. *Reviews in Fisheries Science,* 10, 113-390.

Lefebvre, K. A., Powell, C. L., Busman, M., Doucette, C. J., Moeller, P. D. R., Sliver, J. B., Miller, P. E., Hughes, M. P., Singaram, S., Silver, M. W. et al.(1999) Detection of domoic acid in northern anchovies and California sea lions associated with an unusual mortality event. *Natural Toxins,* **7,** 85-92.

Lefebvre, K.A., Bargu,. S., Kieckhefer, T. and Silver, M.W. (2002b). From sanddabs to blue whales: the pervasiveness of domoic acid Toxicon, 40(7), 971-977.

Lefebvre, KA, Silver MW, Coale SL, Tjeerdema RS. 2002a. Domoic acid in planktivorous fish in relation to toxic Pseudo-nitzschia cell densities. Marine Biology 140:625-631.

Litaker, R. W., Stewart, T. N., Eberhart, B. T. L., Wekell, J. C., Trainer, V. L., Kudela, R. M., Miller, P. E., Roberts, A., Hertz, C., Johnson, T. A., et al. (2008) Rapid enzyme-linked immunosorbent assay for detection of the algal toxin domoic acid. *Journal of Shellfish Research,* **27,** 1301-1310.

MacKenzie, L, Beuzenberg V, Holland P, McNabb P, Selwood A. 2004. Solid phase adsorption toxin tracking (SPATT): a new monitoring tool that simulates the biotoxin contamination of filter feeding bivalves. Toxicon 44:901-918.

R Development Core Team, 2010. A Language and Environment for Statistical Computing, Vienna, Austria

Sekula-Wood, E, Schnetzer A, Benitez-Nelson CR, et al. 2009. Rapid downward transport of the neurotoxin domoic acid in coastal waters. Nat Geosci 2:272-275.

Stobo, L. A., Lacaze, J. P. C. L., Scott, A. C., Petrie, J. & Turrell, E. A. (2008) Surveillance of algal toxins in shellfish from Scottish waters. *Toxicon,* **51,** 635-648.

Stobo, L. A., Lacaze, J. P. C. L., Scott, A. C., Petrie, J. & Turrell, E. A. (2008) Surveillance of algal toxins in shellfish from Scottish waters. *Toxicon,* 51, 635-648.

Thompson, PM, McConnell B, Tollit DJ, Mackay A, Hunter C, Racey PA. 1996. Comparative distribution, movements and diet of harbour and grey seals from the Moray Firth, N.E. Scotland. J Applied Ecology 33:1572-1584.

Turner, J. and Tester P. (1997) Toxic marine phytoplankton, zooplankton grazers and pelagic food webs. Limnol. Oceanogr. 42: 1203-1214

Vigilant, V. and Silver M (2007). Domoic acid in benthic flatfish on the continental shelf of Monterey Bay, California, USA. Marine Biology 151: 2053-2062

Acknowledgements

The author would like to acknowledge the considerable help of Lindsay Wilson and Donald Malone for their dedicated faecal collection efforts. Elizabeth Turrell at Marine Scotland kindly donated the SPATT bags and Melanie Chocholek is doing a great job deploying them for us in the Eden Estuary. The fish discards were also very kindly donated by the skipper of the Providence.

Volker B. Deecke[1], Andrew D. Foote[2], Sanna Kuningas[1]

The impact of killer whale predation on harbour seals in near shore Shetland waters: evidence for dietary specialisation and estimated predation rates

[1]Sea Mammal Research Unit, Scottish Oceans Institute. University of St Andrews, Fife, KY16 8LB
[2]University of Aberdeen, School of Biological Sciences, Lighthouse Field Station, Cromarty, IV11 8YJ

Summary

Predation by killer whales has been suggested as a potential cause of recent harbour seal declines in Shetland. Here we report the findings of two field seasons conducting group follows on killer whales in nearshore Shetland waters to investigate feeding strategies, and to quantify predation rate and composition of the diet.

Introduction

Recent studies indicate that there have been significant declines of harbour seals *Phoca vitulina* around much of the UK over the last decade (Lonergan et al. 2007). Work is now underway to investigate possible causes for these changes, and to identify appropriate conservation action. It is not currently known whether declines are the result of a single common factor, or multiple drivers. However, declines in the Northern Isles have been the most severe with an approximate decrease of 40% in Orkney and Shetland (Lonergan et al. 2007), suggesting that these exaggerated declines around the Northern Isles may result from an additional, localized cause. Bolt et al. (2009) showed both a spatial correlation around the UK with killer whale occurrence and harbour seal declines, and a temporal correlation with the timing of killer whale occurrence around Shetland and the harbour seal pupping season. Bolt et al. (2009) then used a simple bioenergetics model to estimate the potential numbers of harbour seals predated by killer whales sighted annually in nearshore waters around Shetland. The estimates suggested up to 800 seals could be consumed in one year if the whales were exclusively feeding on harbour seal pups. However a key piece of missing information that was the source for the most uncertainty in our values was the composition of the diet of the killer whales sighted in nearshore waters around Shetland (Bolt et al. 2009). Here we use data collected from group follows of killer whales in nearshore Shetland waters to quantify the predation rate and composition of the diet and to investigate dietary specialisation among different groups.

Methods

Fieldwork was conducted in Shetland between 22 May-17 August 2008 and between 29 April-11 July 2009. We opportunistically scanned from elevated points on land to detect groups of killer whales and set up an extensive sightings network to obtain information on killer whale movements from ferry crews, fishermen, and members of the general public. When groups were located we launched a 6m rigid-hulled inflatable powered by two 40hp outboard engines. Once sufficient photographs of all group members were obtained to allow identification of the group, we followed the animals at a distance of 100-500m to record underwater vocalisations, and to document behaviour and any feeding activity for a maximum of 4 hours or as long as conditions permitted.

Acoustic behaviour was recorded during focal follows using a hydrophone array consisting of 2 Benthos AQ-4 transducers with Magrec HP-02 pre-amplifiers towed 60m behind the boat at a depth of approximately 5-20m. Sound was recorded using a Marantz PMD671 solid state recorder at a sampling rate of 96kHz. Voice notes noting the distance and behaviour state of the group on each surfacing were recorded onto a separate channel. Behaviour state was classified as 'Milling', 'Slow Travel', 'Travel' and 'Milling after Kill' using the criteria of Deecke et al. (2005). Whenever the animals' surface behaviour or characteristic prey-handling sounds on the hydrophone array indicated a predation event, we approached the group to recover prey remains for molecular identification of prey species. In addition we also attempted to take photographs of any prey animals for species identification. Only data recorded within 500m of at least one animal were included in the analysis of vocal behaviour to ensure that most if not all of the vocalisations produced by the group could be detected.

Tissue samples were obtained from prey scraps following two predation events to verify the prey species. Samples were wrapped in aluminium foil and frozen at -20°C without preservative for storage.
The mitochondrial DNA (mtDNA) control region was amplified with two sets of primers, first using universal primers MTCRf and MTCRr (Hoelzel et al. 1998). Secondly using primers specifically designed for harbour seals L16371 and HI6571 (Lamont et al. 1996). Sequences were compared with sequences in the database GenBank using the BLAST algorithm.

Photo-identification data collected during dedicated surveys and opportunistically from photographs taken by the public in 2008 and 2009 were analysed using photo-identification techniques to determine population and social structure and using mark-recapture methods to estimate the number of individuals in near shore waters during these years.

Results

In almost all encounters in near shore waters, the killer whales exhibited behaviour consistent with hunting for seals e.g. hugging the coastline tightly, particularly around seal haul-outs. Evidence for feeding behaviour, including lunges towards seals, both grey and harbour, could be obtained in 9 encounters. Group size ranged from 1 to 6 for groups seen to attack sea mammals and from 25-50 estimated for groups documented to feed on fish. So far, none of the individuals involved in marine mammal predation have been observed feeding on fish, which may suggest some degree of dietary specialisation consistent with our characterisation of type 1 killer whales based on stable isotope values (Foote et al. 2009).

Further evidence of seals being primarily targeted as prey by killer whales in nearshore waters around Shetland came from analysing their acoustic behaviour. Conditions allowed us to conduct focal follows to document acoustic behaviour during 10 encounters (3 in 2008 and 7 in 2009). Focal follows lasted between 1:08 hrs and 3:50 hrs with an average of 2:39 hours and we spent between 0:20 hrs and 1:31hrs recording within range (500m of the animals). Group sizes during focal follows ranged between 5 and an estimated 25-50 animals.

Rates of vocal behaviour are given in Fig. 1a. Median rate of vocal behaviour for seal-hunting groups was 0.00 calls per animal per minute (interquartile range: 0.00-0.14) across all behaviour categories. Vocal rates for the two encounters with fish-feeding groups were 0.00 and 0.97 calls per animal per minute. Analysis of the vocal behaviour of seal hunting groups by behaviour state shows that these animals restrict vocal communication to a few narrowly-defined contexts (Fig. 1b). Significant numbers of pulsed calls were only recorded when animals where showing surface-active behaviour (tail slaps, pectoral slaps, breaches, etc.) behaviours often associated with social interactions between animals, or when the animals were milling after a confirmed marine mammal kill. During behaviour states typically associated with active search for prey (slow travel and travel) the animals were usually silent. The vocal behaviour of seal-hunting killer whales shows striking parallels to that of mammal-eating killer whales in the Northeast Pacific (Barrett-Lennard et al. 1996; Deecke et al. 2005) and the reduction of vocal communication may be part of a specialised hunting strategy to avoid detection by acoustically sensitive marine mammal prey.

Information on documented nearshore feeding events is given in Table 1. We did not observe any clear preference in species or age class of seal based on predation attempts. However, the small number of confirmed kills we documented were mainly harbour seals. We were only able to confirm age class in one case. We successfully sequenced 376 bp of the mtDNA control region of the two mammalian prey samples.

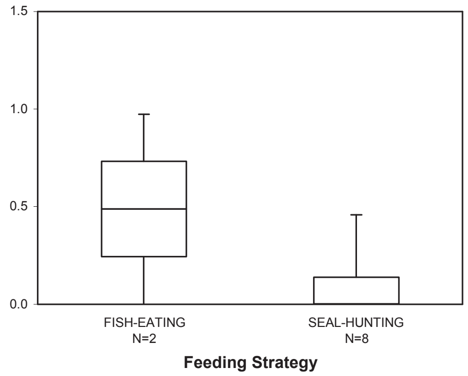


Figure 1a. Rates of vocal behaviour (pulsed calls emitted per individual per minute) for killer whale groups feeding on fish (left bar) and hunting seals (right bar) around Shetland.

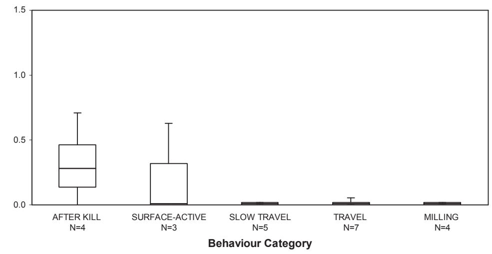


Figure 1b. Rates of vocal behaviour during different behaviour states for seal-hunting killer whale groups feeding around Shetland.

Both samples were identified as harbour seal by the BLAST algorithm. The prey sample collected on the 4th July 2008 scored 678 and was a 99% match with a 0.0 E value to sequence accession number U36354.1, which corresponds to haplotype G2, a haplotype with Europe-wide distribution and as far west as Iceland (Stanley et al. 1996). The prey sample collected on 25th May 2009 scored 689 and was a 99% match with a 0.0 E value to sequence accession number U36365.1, which corresponds to haplotype G3, previously found in Scotland and Northern Ireland (Stanley et al. 1996).

Mark-recapture estimates for pair of years was calculated using– simple two-sample Chapman estimator, suggesting approximately 30 individuals were in Shetland nearshore waters 2008-2009 (Fig.

2).

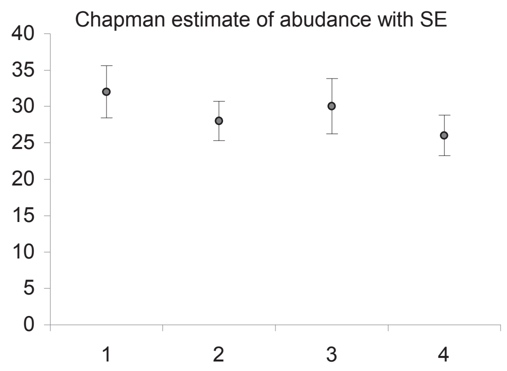


Figure 2. Estimates of the number of individual killer whales in nearshore waters 2008-2009. Estimates 1. & 2. used good and average quality photographs, estimate 1. was using both sides, estimate 2 with just left-hand sides. Estimates 3. & 4. used only good quality images, estimate 3. was using both sides, estimate 4 with just left-hand sides.

However, the total number of individuals within this community, which are also seen in waters around Caithness and Orkney, may be slightly larger based on discovery curves using a wider dataset. Identified individuals were annually site faithful to this area between 2005-2008 and linked by association (Foote et al. in press).

Discussion

Bioenergetic modelling suggests that each adult female/sub-adult male will require approximately one adult harbour seal a day, adult males will require twice this and juveniles approximately half this (Bolt et al. 2009). These data are consistent with observed predation rates from Glacier Bay, Alaska where killer whales feed predominantly on harbour seals (V.B. Deecke unpublished data). The group composition and the number of seals consumed during our follows (Table 1) averages out at 0.6 seals per day per adult female or sub-adult male. Although this is lower than our model estimate, if the two occasions where a predation event had taken place just prior to the start of our follows were harbour seals and are included then it would bring the predation rate up to 0.85 seals per F/SAM per day. Therefore the bioenergetics models used by Bolt et al. (2009) are reasonably consistent with our observed predation rate by nearshore killer whales around Shetland. However, we suspect that the sightings data used in that study significantly under-estimates the number of killer whales present in nearshore waters.

Our estimates suggest approximately 30 whales in Shetland waters during 2008-2009 and at present we have identified 36 individuals within this nearshore seal-eating community. They are primarily observed around Shetland, Orkney and Caithness from May-Aug (Bolt et al. 2009), e.g. 120 days, but identified individuals have been seen as early as March around Shetland. If these individuals take harbour seals at the observed predation rate throughout this time period then the number of harbour seals taken annually will be in the upper range of, or larger than, the estimates in Bolt et al. (2009). Based on our observations and the models in Bolt et al. (2009) killer whale predation should be factored in to any recovery plan for Scottish harbour seals.

Acknowledgements

This project was funded by grants from Marine Scotland, Scottish Natural Heritage, Carnegie Trust for the Universities of Scotland. AF was supported by a 6thC scholarship from the University of Aberdeen and a Marie Curie Actions fellowship and VBD was supported by a Marie-Curie Intra-European Fellowship..

Date	Length of follow (hrs)	Confirmed Kill (V = visual, M = Molecular)	Group Size	Group composition
31-May-08	1	*	6	1M, 4F/SAM, 1J
12-Jun-08	1.5		5	1M, 3F/SAM, 1J
12-Jun-08	0.25		1	1M
22-Jun-08	1		5	1M, 3F/SAM, 1J
30-Jun-08	3		5	1M, 3F/SAM, 1J
2-Jul-08	4	Harbour seal pup (V)	1	1M
4-Jul-08	2	Harbour seal (V, M)	5	1M, 3F/SAM, 1J
14-Jul-08	0.25		5	1M, 3F/SAM, 1J
14-Jul-08	0.25		1	1M
18-Jul-08	3		5	1M, 3F/SAM, 1J
15-Aug-08	0.25		4	3F/SAM, 1J
19-May-09	0.25		5	1M, 3F/SAM, 1J
20-May-09	2.75	*	5	1M, 3F/SAM, 1J
24-May-09	1	Harbour seal (M)	5	1M, 3F/SAM, 1J
31-May-09	4		5	1M, 3F/SAM, 1J
7-Jun-09	0.25		5	1M, 3F/SAM, 1J
20-Jun-09	2.75		4	3F/SAM, 1J
28-Jun-09	1.25		13	3M, 7F/SAM, 3J
28-Jun-09	2.5	Mammal (V)	6	2M, 3F/SAM, 1J
1-Jul-09	0.25		6	3M, 2F, 1J

*Table 1. Confirmed kills made during group follows of killer whales in nearshore waters around Shetland during summer 2008 and 2009. Details of follows of two groups feeding on fish further offshore (approx 5 miles) are not included. M =adult male, F/SAM = adult female or sub-adult male, J = juvenile. * indicates that a mammal kill appeared to have been made just prior to the start of our follow.*

References

Barrett-Lennard, L. G., Ford, J. K. B. & Heise, K. A. 1996. The mixed blessing of echolocation: Differences in sonar use by fish-eating and mammal-eating killer whales. *Animal Behaviour*, 51: 553-565.

Bolt, H. E., Harvey, P. V., Mandelberg, L and Foote, A. D. 2009. Occurrence of killer whales in Scottish inshore waters: temporal and spatial patterns relative to the distribution of declining harbour seal populations. *Aquatic Conservation: Marine Freshwater Ecosystem*, 19: 671-675.

Deecke, V. B., Ford, J. K. B. and Slater, P. J. B. 2005. The vocal behaviour of mammal-eating killer whales (*Orcinus orca*): Communicating with costly calls. *Animal Behaviour*, 69: 395-405.

Foote, A. D., Newton, J., Piertney, S. B., Willerslev, E. and Gilbert, M. T. P. 2009. Ecological, morphological and genetic divergence of sympatric North Atlantic killer whale populations. *Molecular*

Ecology, 18: 5207-5217.

Foote, A. D., Similä, T., Víkingsson, G. A. and Stevick, P. T. (in press) Foraging specialization influences movement and site fidelity in a top marine predator, the killer whale. *Evolutionary Ecology*, available online early.

Hoelzel, A. R., Dahlheim, M. and Stern, S. J. 1998. Low genetic variation among killer whales (*Orcinus orca*) in the Eastern North Pacific and genetic differentiation between foraging specialists. *Journal of Heredity*, 89: 121-128.

Lamont, M. M., Vida, J. T., Harvey, J. T., Jeffries, S., Brown, R., Huber, H. H., Delong, R. and Thomas, W. K. 1996. Genetic substructure of the Pacific harbor seal (*Phoca vitulina Richardsi*) off Washington, Oregon, and California. *Marine Mammal Science*, 12: 402-413.

Lonergan, M., Duck, C. D., Thompson, D., Mackey, B. L., Cunningham, L. and Boyd, I. L. 2007. Using sparse survey data to investigate the declining abundance of British harbour seals. *Journal of Zoology* 271: 261-269.

Stanley, H. F., Casey, S., Carnahan, J. M., Goodman, S., Harwood, J., Wayne, R. K. 1996. Worldwide patterns of mitochondrial DNA differentiation in the harbor seal (*Phoca vitulina*). *Molecular Biology & Evolution*, 13: 368-382.

I.L. Boyd, D. Thompson & M. Lonergan

Potential Biological Removal as a method for setting the impact limits for UK marine mammal populations

NERC Sea Mammal Research Unit, Gatty Marine Laboratory, University of St Andrews, St Andrews KY16 8LB

Summary

1. Introduction

The Potential Biological Removal (PBR) has been developed as a tool for guiding management decisions concerning marine mammal populations (Wade 1998). Although it was developed principally to satisfy the requirements of the US Marine Mammal Protection Act (MMPA), it has properties that lend itself to more general application in wildlife management and it has been used widely (including in the UK) for estimating limits to the number of marine mammals that can be taken from a population as a direct result of human activity (Johnson et al. 2000; Marsh et al. 2004; Thompson et al. 2007; Butler et al 2008; Okamura et al. 2008; Underwood et al. 2008; Williams et al. 2009). In addition, the method is beginning to be adopted for birds (Dillingham et al. 2008; Barbraud et al. 2009; Runge et al. 2009; Zydelis et al. 2009).

Wade (1998) undertook an assessment of the properties of the PBR using simulation. This suggested that the PBR could be used to estimate mortality limits and it concluded that, as a general rule, any marine mammal population with an estimate of human-caused mortality that is greater than the calculated PBR has a level of mortality that could lead to depletion. In this case, a depleted population is one that has fallen below it maximum mean net productivity level, which may be somewhere between 50% and 85% of the maximum sustained population size, or carrying capacity.

However, in testing this method the simulations (Wade 1998) were applied to populations with a direct density dependent control on population growth. Some marine mammal populations are probably below their historical carrying capacities and are declining, stationary or growing at less than their estimated maximum rates of increase. In many management scenarios this assumption of density dependent growth represents a weakness in the PBR method.

A prime consideration in choosing a management tool is its usefulness in dealing with real situations. Here we propose a simple method for determining the appropriate level of permissible biological removals from populations with a wide range of current/recent dynamics. However, we do this only as an interim solution to allow progress while a more robust structure for management scenario evaluation is developed.

2. Definition

A PBR is defined as the minimum population estimate of the stock (N_{MIN}), where N_{MIN} is the lower 20th percentile of the likely population size, multiplied by 50% of the maximum potential rate of increase of the population (R_{MAX}), and this is then multiplied by a recovery factor (F_R):

$$PBR = N_{MIN} \frac{1}{2} R_{MAX} F_R \qquad 1.$$

The rationale for this approach is that, in the absence of human impacts, N_{MIN} multiplied by R_{MAX} is the maximum net rate of change in the population in any time period (usually one year in the case of marine mammals). Although there are examples of pinniped populations increasing at close to the theoretical R_{MAX}, experience has shown that populations can normally achieve only half of the potential maximum rate (~6-7% per annum for most marine mammals) even under close-to-ideal conditions. In addition, even in situations when there is high certainty in the estimates of N_{MIN} and R_{MAX}, at least 50% of the production of the population would need to remain in order to ensure that the population was sustained above its maximum net productivity level.

The first two terms, which we call the *biological parameters*, in the BPR describe the underlying dynamics of a population in very simple terms. Although the use of the $\frac{1}{2} R_{MAX}$ is not precautionary, precaution is included through the use of the minimum population estimate and in the third term F_R, which is generally set at some level between 0.1 and 1.0. F_R allows management to further increase the level of precaution where populations are particularly sensitive or important, or the other parameter estimates are imprecise. We call F_R the *Management parameter*. Until now the value of F_R has been fixed in a fairly arbitrary way based on perceptions of levels of uncertainty in the parameters. The objective of this paper is to provide a rationale for deciding the level of F_R.

3. Selecting the parameters

The population management goal must be defined in terms of a target population size or an intended direction of change in the population.

All of the input parameters in the PBR are subject to different levels of certainty. In the case of the biological parameters this is usually measurement uncertainty whereas in the case of the management parameter this is normally uncertainty based upon the social, economic and political consequences of the issue being managed. However, if we denote the uncertainty around the parameter as n_{min}, r_{max} and f_r, in general we will find that

$$f_r \propto n_{\min}, r_{\max} \qquad 2.$$

and

$$F_R \propto \frac{1}{n_{\min}}, \frac{1}{r_{\min}} \qquad 3.$$

In other words, the level we use to set the management parameter must be designed to achieve the management objectives but it must also reflect the level of knowledge we have about the biological parameters.

Wade (1998) carried out simulations to show the probability of a particular management objective being met. Here, we investigate the decision process for F_R and simulate its effect using the same model as Wade.

Table 1: Classification of different sources of certainty associated with the biological parameters used within the PBR calculation.

N_{MIN}	Level of certainty
Population is completely open or no information about population boundaries	LOW
Estimates of population size have no CV; less than 2 counts available	LOW
Population is partially closed, likely to have <10% immigration/emigration per year	INTERMEDIATE
The CV is a post-hoc estimate from multiple serial counts	INTERMEDIATE
Population completely closed	HIGH
Estimates of population are designed to provide a CV within the sampling method	HIGH
Estimates of population are direct counts of minimum numbers (and are used as the estimate of N_{MIN})	HIGH
R_{MAX}	
Based upon general appraisal of life histories of taxonomic group	LOW
In open, or partially open, populations presence of potential adjacent sink populations	LOW
Presence of undefined, non-specific causes of mortality or low reduced reproductive capacity (indicated by, for example, a rapid long-term decline) that compromise population resilience	LOW
Same as above, but with supporting evidence from the population, or a closely linked population, that general life history traits apply in this case	INTERMEDIATE
Direct measurement of demographics allows estimation of R_{MAX}	HIGH
In open, or partially open, populations presence of potential adjacent source populations	HIGH

We are seeking a process that will define F_R using rules about the precision of N_{min} and R_{max}. These rules must be relatively easy to implement and robust in terms of the precaution they confer on the process. Since it is unusual for us to be able to support management of marine mammal populations with precise information, even about the uncertainties involved in estimates, we suggest the follow set of rules:

Each of the biological parameters are classified as of *high*, *intermediate* or *low* quality. A guide to classification of each parameter is provided in Table 1.

For each of the biological parameters, the corresponding classification with the highest level of precaution is applied. This means that, for example, even if a population is completely closed (meriting a classification of HIGH certainty for the parameter N_{MIN}) and if the population estimate has no CV associated with it (meriting a LOW classification) then the most precautionary estimate will be chosen, which would be LOW in this case.

Using this scheme, there are 9 possible uncertainty classifications for each PBR calculation. A critical decision for this approach is the mapping of this classification onto the range of F_R defined by Wade (1998). One possible mapping is shown in Table 2.

Table 2: Suggested relative levels of confidence in the biological parameters used to calculate the PBR to be used in helping to guide setting the management parameter, F_R

F_R		N_{MIN}		
		HIGH	INTER	LOW
R_{MAX}	HIGH	1.0	0.7	0.3
	INTER	0.7	0.5	0.2
	LOW	0.3	0.2	0.1

As a worked example, we provide an estimate of a PBR for harbour seals within The Wash population in one case using the uncertainty criteria and then taking account of recent population dynamics information.

The Wash harbour seal population was growing at a rate of approximately half the estimated R_{max} for harbour seals before being reduced by 25-30% in 2002. Since then it has not recovered at all despite the absence of any apparent anthropogenic mortality.

The latest count for this population is 2,010, which is a minimum count for the population and this would attract a classification of HIGH for the N_{MIN} parameter. However, it is a population which is likely to be only partially closed which would attract a classification of INTERMEDIATE for the same parameter. Thus, the classification for N_{MIN} would be INTERMEDIATE.

For the parameter R_{MAX} we have evidence from time-series of counts from this population and one nearby (in Sweden) that the general life history patterns for this species in terms of reproductive rates and survival rates fit our expectations for pinnipeds. However, we do not have specific data for the population in question. Thus, the classification for R_{MAX} would be INTERMEDIATE using the rules in Table 1. Nevertheless, there is evidence of decline in this population involving non-specific causes of mortality or reduced reproductive capacity, which would attract a classification of LOW. In combination, and based on Table 2, this would define F_R for the harbour seal population in The Wash as 0.2, giving a PBR of 24.

Given that this population has shown a lack of recovery to historical levels, some might argue that this is not sufficiently conservative and that a PBR of 0 would be more appropriate. However, it remains possible that the population is close to its carrying capacity (because this may have declined). Nevertheless, if the population began to decline further, this would result in reclassification of the quality of the N_{MIN} parameter as LOW because of the risk of extinction and this would produce a PBR of 12 animals, the lowest than can be set.

An important subsequent question is whether this classification is actually correct. The results of a simulation, using the same methods as used by Wade (1998), were used to examine the efficacy of the PBR based on the classification system described in the present study and also based upon the quantitative expression of this classification in Table 3. As can be seen from the result in Table 4, in all scenarios there was a very low probability of the population failing to satisfy the management objectives, as defined in this case.

Table 3: Allocation of qualitative estimates of standardized errors under each of the classification schemes where z is a standard normal variate applied to define N_{MIN}, CV is the coefficient of variation around a survey-based estimate of abundance, and σ is the standard error of the around R_{MAX}. Value for σ for cetaceans should be σ x 0.3 those shown here to account for the difference in R_{MAX} (~1.12 for pinnipeds and ~0.04 for cetaceans).

z, CV, σ		N_{MIN}		
		HIGH	INTER	LOW
R_{MAX}	HIGH	0.842, 0.2, 0.01	1.282, 0.5, 0.01	1.96, 0.8, 0.01
	INTER	0.842, 0.2, 0.03	1.282, 0.5, 0.03	1.96, 0.8, 0.03
	LOW	0.842, 0.2, 0.05	1.282, 0.5, 0.05	1.96, 0.8, 0.05

Table 4: Results of simulations showing the probability of populations failing to meet the management objectives for the population ($p_{default}$) and remaining above the mean net productivity levels (assumed here to be 50% of K) based upon the classification scheme proposed here and using the values in Table 3. Each result was based upon 10,000 iterations of the same simulation used by Wade (1989). Each run of the simulation lasted 300 years; the first 200 years were used to condition the time series and judgements about whether management objectives were met were based on the final 100 years.

$p_{default}$		N_{MIN}		
		HIGH	INTER	LOW
R_{MAX}	HIGH	0.0015	<0.0001	<0.0001
	INTER	<0.0001	<0.0001	<0.0001
	LOW	<0.0001	<0.0001	<0.0001

4. Discussion

The PBR may not always be the ideal technical solution for setting management limits on the number of marine mammals from a population, but with appropriate selection of a management factor, taking into account all the available population information it may provide a simple practical solution.

Any solution to this problem needs to capture the following characteristics:

- Biological plausibility
- Capacity for refinement through focused gathering of additional data
- Precaution without over-precaution
- Simplicity to encourage buy-in from those who will use the management tool and those who will be affected by decisions based upon its outcomes
- Robustness to cumulative impacts of multiple low-level effects
- Consistent and repeatable through time and across management regimes

The PBR has most of these characteristics. However, we advocate that, while the PBR could be given to non-experts to implement, there would always be a strong preference for expert debate about the exact level of the management parameter The present study has begun to build the framework within which that debate can take place.

The study has shown that there are a set of simple rules that can be applied in order to establish the appropriate recovery factor. In fact, of the characteristics listed above, to date, the PBR has suffered from potential lack of repeatability depending upon who was implementing the algorithm and how authors weighted the importance of difference factors affecting the uncertainties around the biological parameters in the PBR. The present study has added to the current knowledge of the properties of the PBR by providing the potential for judgements to be both consistent and robust.

Although this study has carried out basic simulations to examine the extent to which the values chosen are likely to be correct, it would be useful to extend this simulation exercise to investigate the parameter sensitivities in each case and to examine a range of different management objectives.

There is a danger that the present study will be used by managers as the defined methods for setting acceptable biological limits for human-induced mortality of marine mammals. We wish to emphasise that, while this approach may be the most practical solution available at the time of writing, there is a need to transition to approaches that are considerably more robust.

References

Barbraud, C; Delord, K; Marteau, C, et al. (2009) Estimates of population size of white-chinned petrels and grey petrels at Kerguelen Islands and sensitivity to fisheries. *Animal Conservation* 12: 258-265.

Butler, JRA; Middlemas, SJ; McKelvey, SA, et al. (2008) The Moray Firth Seal Management Plan: an adaptive framework for balancing the conservation of seals, salmon, fisheries and wildlife tourism in the UK. *Aquatic Conservation – Marine and Freshwater Ecosystems,* 18: 1025-1038.

Dillingham, P.W. & Fletcher, D. (2008) Estimating the ability of birds to sustain additional human-caused mortalities using a simple decision rule and allometric relationships. *Biological Conservation*. 141: 1783-1792

Johnston, DW; Meisenheimer, P; Lavigne, DM (2000). An evaluation of management objectives for Canada's commercial harp seal hunt, 1996-1998. *Conservation Biology* 14: 729-737.

Marsh, H., Lawler, I.R., Kwan, D., Delean, S., Pollock, K. & Alldredge, M. (2004) Aerial surveys and the potential biological removal technique indicate that the Torres Strait dugong fishery is unsustainable. *Animal Conservation* **7**, 435–443

Okamura, H; Iwasaki, T; Miyashita, T (2008) Toward sustainable management of small cetacean fisheries around Japan. *Fisheries Science* 74: 718-729.

Runge, MC; Sauer, JR; Avery, ML, et al. (2009) Assessing Allowable Take of Migratory Birds *Journal of Wildlife Management* 73: 556-565.

Thompson, PM; Mackey, B; Barton, TR, et al. (2007) Assessing the potential impact of salmon fisheries management on the conservation status of harbour seals (*Phoca vitulina*) in north-east Scotland Source. *Animal Conservation* 10: 48-56.

Underwood, JG; Camacho, CH; Aurioles-Gamboa, D, et al. (2008) Estimating sustainable bycatch rates for California sea lion populations in the Gulf of California. *Conservation Biology* 22: 701-710.

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

Williams, R; Lusseau, D; Hammond, PS (2009) The role of social aggregations and protected areas in killer whale conservation: The mixed blessing of critical habitat. *Biological Conservation* 142:: 709-719.

Zydelis, R; Bellebaum, J; Osterblom, H, et al. (2009) Bycatch in gillnet fisheries - An overlooked threat to waterbird populations. *Biological Conservation* 142: 1269-1281.