Scientific Advice on Matters Related to the Management of Seal Populations: 2015

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Executive Summary

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. Questions on a wide range of management and conservation issues are received from the UK government and devolved administrations. In 2015, 18 questions were addressed by SCOS.

Current status of British grey seals

Grey seal population trends are assessed from the counts of pups born during the autumn breeding season, when females congregate on land to give birth. The most recent surveys of the Scottish grey seal breeding sites were carried out in 2014. Results and pup production estimates for all colonies will be presented to SCOS in 2016. Thus the most recent available pup production estimates for 2015 are from surveys carried out in 2012. These resulted in an estimate of 56,988 (95% CI 56,317, 57,683). This is then converted to an estimate of total population size (1+ aged population) using a mathematical model. The population dynamics model trajectories were projected forward and added to the estimated numbers of animals associated with the less frequently surveyed breeding colonies to give an estimate of 111,600 (95% CI 91,400-139,200) UK grey seals in 2014.

Current status of British harbour seals

Harbour seals are counted while they are on land during their August moult, giving a minimum estimate of population size. Not all areas are counted every year but the aim is to cover the UK coast every 5 years.

Combining the most recent counts (2007-2014) gives a total of 29,109 animals in the UK. Scaling this by the estimated proportion of the population hauled out, produced an estimated total population for the UK in 2014 of 40,414 (approximate 95% CI 33,106, 55029).

Harbour seal counts were stable or increasing until around 2000 when declines were seen in Shetland (which declined by 30% between 2000-2009), Orkney (down 78% between 2000-2013) and the Firth of Tay (down 96% between 2000-2014). However, other regions have been largely continually stable (west coast of Highland region and the Outer Hebrides). Counts along the English east coast were very similar to those reported for 2013.

SCOS recommended that the measures to protect vulnerable harbour seal populations should remain in place.

Grey and harbour seal population structure

Information on vital rates, such as a time series of fecundity and survival rates would improve our ability to provide advice on population status . There is information from two breeding colonies with contrasting population trajectories, but these may not be regionally representative.

Knowledge of UK harbour seal demographic parameters is limited and therefore inferences about the population dynamics rely largely on count data from moulting surveys. Information on vital rates would again improve our ability to provide advice on population status.

Population trend detection

The ability to detect a significant trend in population abundance depends to some extent on the uncertainty associated with the count data, the environmental conditions and the available budget. Thus monitoring effectiveness (ability to detect trends) and efficiency (ability to do this at low cost) are key considerations in determining the temporal and spatial scale required, as is the management objective being considered.

Causes of the recent decline in common/harbour seals

Potential causes of the decline in Scottish harbour seals in some regions include interactions with grey seals (both indirect such as competition for resources and habitat and direct such as predation) and exposure to toxins from harmful algae. Funding granted to the Sea Mammal Research Unit (SMRU) from Scottish Government has enabled the integrated research project suggested by the Harbour Seal Decline workshop (see SCOS 2014) to be undertaken. Study site identification and initial data collection has now started. Reports on the findings from this study will be presented to SCOS annually.

Causes of the recent unusual seal mortalities

The latest understanding of the cause of the recent unusual spiral seal mortalities is that this is likely due to predation by male grey seals rather than ducted propellers. A study funded by Scottish Government is being carried out by SMRU to determine whether collisions with vessels remain a plausible explanation.

Potential biological removal

Options for changes in the Permitted/Potential Biological Removals (PBRs) approach for use in relation to the seal licence system were discussed extensively by SCOS. It was agreed that a workshop be held to bring together experts on seal population dynamics, population modelling and population management to provide a recommended approach or approaches to managing anthropogenic impacts on UK seals. This should be held as soon as possible in 2016.

PBR Estimates for 2016

The provisional regional Potential Biological Removal values for Scottish seals for 2016 for use in issuing seal licences were endorsed by SCOS.

Seals and Marine Renewables

Since reporting in 2014 there are a number updates on the interactions between seals and marine renewable devices (wind, wave, and tide).

Results of a behavioural study, funded by DECC and carried out by SMRU, during the construction of a wind farm using data from GPS/GSM tags on 24 harbour seals in the Wash suggests that seals were not excluded from the vicinity of the windfarm during the construction phase. Comparison of predicted received noise levels to exposure criteria suggests that half of the seals were exposed to noise levels that exceeded published auditory damage thresholds. However, the prediction of auditory damage in marine mammals is a rapidly evolving field and has a number of key uncertainties associated with it. The biological implications of this are currently unclear and the study provides no information to assess the possible effects on individual survival or fecundity. However, current evidence suggests there is no large scale displacement of animals from operational wind farms.

Seals and salmon netting stations

SMRU testing of acoustic deterrent devices (ADDs) at salmon bagnet stations with evidence concluded that they are effective under some conditions. Additional work also showed that net modifications are potentially an effective means of limiting seal predation, primarily by preventing whole fish being removed from the fish court, and further modifications seem promising to reduce predation in the outer parts of the net still further.

Seals and rivers

ADDs have been successfully trialled to limit the passage of seals up salmon rivers but there are concerns related to how they are deployed and maintained.

Seal bycatch

Estimates of seal bycatch are reported annually to the European Commission in the UK “Report on the implementation of Council Regulation (EC) No 812/2004”, which is produced annually by SMRU under contract to Defra (is it Defra or DEFRA) and the Scottish Government. Seal bycatch estimates for static net fisheries are included in the Annex to that report by ICES subdivision, but should be treated with caution as several caveats apply, and only point estimates are given.

Marine strategy framework directive (MSFD)

The latest available data from the UK were used to perform a preliminary assessment of MSFD indicators. M-3 and M-5 describe changes in grey seal and harbour seal population abundance and distribution. It was necessary to arbitrarily subdivide UK Assessment Units into smaller subareas to calculate distribution metrics for harbour seals. The distribution metrics showed no catastrophic contraction or shift in distribution has occurred for either grey or harbour seals in any Assessment Unit.

Effects of disturbance on seals

A series of controlled disturbance trials carried out by SMRU at harbour seal haulout sites in the Sound of Islay found that repeated disturbance by boats did not cause seals to abandon sites. Seals either hauled out again or went on typical foraging trips after disturbance events. Similar studies in Danish waters produced broadly similar results. Grey seals at several haulout and breeding sites have habituated to close approaches by pedestrians, low flying fighter jets and loud noises from nearby bombing ranges.

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). SMRU is an interdisciplinary research group at the University of St Andrews which receives National Capability funding from NERC to fulfil its statutory requirements and is a delivery partner of the National Oceanography Centre. SMRU also provides government with scientific reviews of licence applications 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 2013. It begins with some general information on British seals, gives information on their current status, and addresses specific questions raised by the Marine Scotland (MS) and the Department of the Environment, Food and Rural Affairs (Defra) and Natural Resources Wales (NRW).

Appended to the main report are briefing papers which provide the scientific background for the advice.

As with most publicly funded bodies in the UK, SMRU's long-term funding prospects involve a reduction in spending in cash terms that represents a substantial reduction in real terms into the foreseeable future. This reduction continues to have a negative impact on the underpinning scientific information on which this advice is based.

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 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 generalist feeders, foraging mainly on the sea bed at depths of up to 100m although they are probably capable of feeding at all the depths found across the UK continental shelf. They take a wide variety of prey including sandeels, gadoids (cod, whiting, haddock, ling), and flatfish (plaice, sole, flounder, dab). Amongst these, sandeels are typically the predominant prey species. Diet varies seasonally and from region to region. Food requirements depend on the size of the seal and fat content (oiliness) of the prey, but an average consumption estimate is 4 to 7 kg per seal per day depending on the prey species.

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 most foraging probably occurs within 100km of a haulout site although they can feed up to several hundred kilometres offshore. 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, the USA, the 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. However, there are clear indications of a slowing down in population growth in the UK and Canadian populations in recent years.

Approximately 38% of the world’s grey seals breed in the UK and 88% 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 population growth is levelling off in all areas except the central and southern North Sea where growth rates remain high. The numbers born in the Hebrides have remained approximately constant since 1992 and growth has been levelling off in Orkney since the late 1990s.

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 or three 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 more 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

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.

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 *in utero* and can swim almost immediately.

Harbour seals 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.

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 79% of the UK harbour seal population, with 16% in England and 5% 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 did not demonstrate any recovery from the 2002 epidemic until 2009 but have increased dramatically in the past four years. In contrast, the adjacent European colonies in the Wadden Sea have experienced continuous rapid growth since 2002 but that increase may be slowing.

Major declines have now been documented in several harbour seal populations around Scotland, with declines since 2001 of 76% in Orkney, 30% in Shetland between 2000 and 2009, and 92% between 2002 and 2013 in the Firth of Tay. However the pattern of declines is not universal. The Moray Firth count declined by 50% before 2005 remained reasonably stable for 4 years then increased by 40% in 2010 and has fluctuated since. The Outer Hebrides apparently declined by 35% between 1996 and 2008 but the 2011 count was >50% higher than the 2008 count. The recorded declines are not thought to have been linked to the 2002 PDV epidemic that seems to have had little effect on harbour seals 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. 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. However, 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 the 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

The Grey Seal (Protection) Act, 1914, provided the first legal protection for any mammal in the UK because of a perception that seal populations were very low and there was a need to protect them. In the UK seals are protected under the Conservation of Seals Act 1970 (England, and Wales), the Marine (Scotland) Act 2010 and The Wildlife (Northern Ireland) Order 1985.

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 issued by the Marine Management Organisation (MMO). The Act also allows for specific Conservation Orders to extend the close season to protect vulnerable populations. After consultation with NERC, three such orders were 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 the Outer Hebrides, 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 conservation orders in Scotland have been maintained under the Marine (Scotland) Act 2010.

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 (through SMRU) provides advice on all licence applications and haulout designations.

The Wildlife (Northern Ireland) Order 1985 provides complete protection for both grey and harbour seals and prohibits the killing of seals except under licence. In Northern Ireland it is an offence to intentionally or recklessly disturb seals at any haulout site.

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. The SAC reporting cycle required formal status assessments for these sites and these were completed in 2013.

Questions from Marine Scotland, Department for Environment, Food and Rural Affairs and Natural Resources Wales.

Questions for SCOS 2015 were received from all three administrations (Marine Scotland, MS; Department for Environment, Food and Rural Affairs, Defra; Natural Resources Wales, NRW) and are listed in Annex II. Some of these questions were essentially the same, requiring regionally specific responses in addition to a UK wide perspective. These very similar questions were therefore amalgamated, with the relevant regional differences in response being given in the tables and text. The question numbers by administration are shown in the boxes for cross reference. The remaining questions were therefore regionally unique, requiring responses that focussed on the issue for a given area. The questions are grouped under topic headings, in the order and as they were given from the administrations.

1. What are the latest estimates of the number of seals in UK waters?	MS Q1; Defra Q1; NRW Q1

Current status of British grey seals

Grey seal population trends are assessed from the counts of pups born during the autumn breeding season, when females congregate on land to give birth. Regional differences in population estimates do not necessarily reflect the abundance of animals in each region at other times of the year.

The most recent surveys of the Scottish grey seal breeding sites were carried out in 2014. Results and pup production estimates for all colonies will be presented to SCOS 2016. The most recent available pup production estimates are from surveys carried out in 2012. These resulted in an estimate of 56,988 (95% CI 56,317, 57,683). Pup production estimates by location are given in Table 1. These are then converted to estimates of total population size (1+ aged population) using a mathematical model.

To estimate the total population size in 2014, the population dynamics model trajectories were projected forward and added to the estimated numbers of animals associated with the less frequently surveyed breeding colonies to give an estimate of 111,600 (95% CI 91,400-139,200) UK grey seals (1+ aged population).

Table 1. ***Grey seal pup production estimates in 2012***.

Location	Pup production in 2012
England	**5,213**
Wales	**1,650***
Scotland	**50,025**
Northern Ireland	**100***
Total UK	**56,988**

*Estimated production for less frequently monitored colonies, see Table 2 for details.

Aerial surveys to estimate grey seal pup production were carried out in Scotland in 2012, using a new digital camera system. Details of the methods are given in SCOS-BP 14/01. Major colonies in Scotland are now surveyed biennially by air (see SCOS-BP14/01). Pup production is then converted to total population size (1+ aged population) using a mathematical model. The stages in the process (pup production → mathematical model → total population size) and the trends observed at each stage are given below.

Pup Production

Information on pup production at all major Scottish colonies was presented in SCOS–BP 14/01. The total number of pups born in 2012 at all UK colonies was estimated to be 57,000 (95% CI 53,900 - 60,100).

Regional estimates at annually surveyed colonies were 4,100 (95% CI 3,900, 4,300) in the Inner Hebrides, 14,100 (95% CI 13,300, 14,900) in the Outer Hebrides, 22,900 (95% CI 21,600, 24,100) in Orkney and 10,200 (95% CI 9,500, 10,883) at the North Sea colonies (including Isle of May, Fast Castle, Farne Islands, Donna Nook, Blakeney Point and Horsey/Winterton). A further 5,700 pups were estimated to have been born at less frequently surveyed colonies in Shetland and Wales as well as other scattered locations throughout Scotland, Northern Ireland and South-west England, producing a total UK pup production of 57,000.

An update on progress to estimate pup production for 2014 is given in SCOS-BP 15/01. In addition, an update on developments in the pup production model is given in SCOS-BP 15/03 which will be incorporated next year in conjunction with the new pup count data.

Trends in pup production

Details of the *trends* in pup production up to 2012 were presented in SCOS-BP 14/01. Briefly, this showed that there has been a continual increase in pup production since regular surveys began in the 1960s (Figure 1). In both the Inner and Outer Hebrides, the rate of increase declined in the mid 1990s. Production was relatively constant since the mid-1990s but between 2010 and 2012 showed an annual increase of ~10 and ~5% respectively, the first substantial increase since the 1990s. And although the rate of increase in Orkney has declined since 2000, pup production also increased at an annual rate of ~6% between 2010 and 2012.

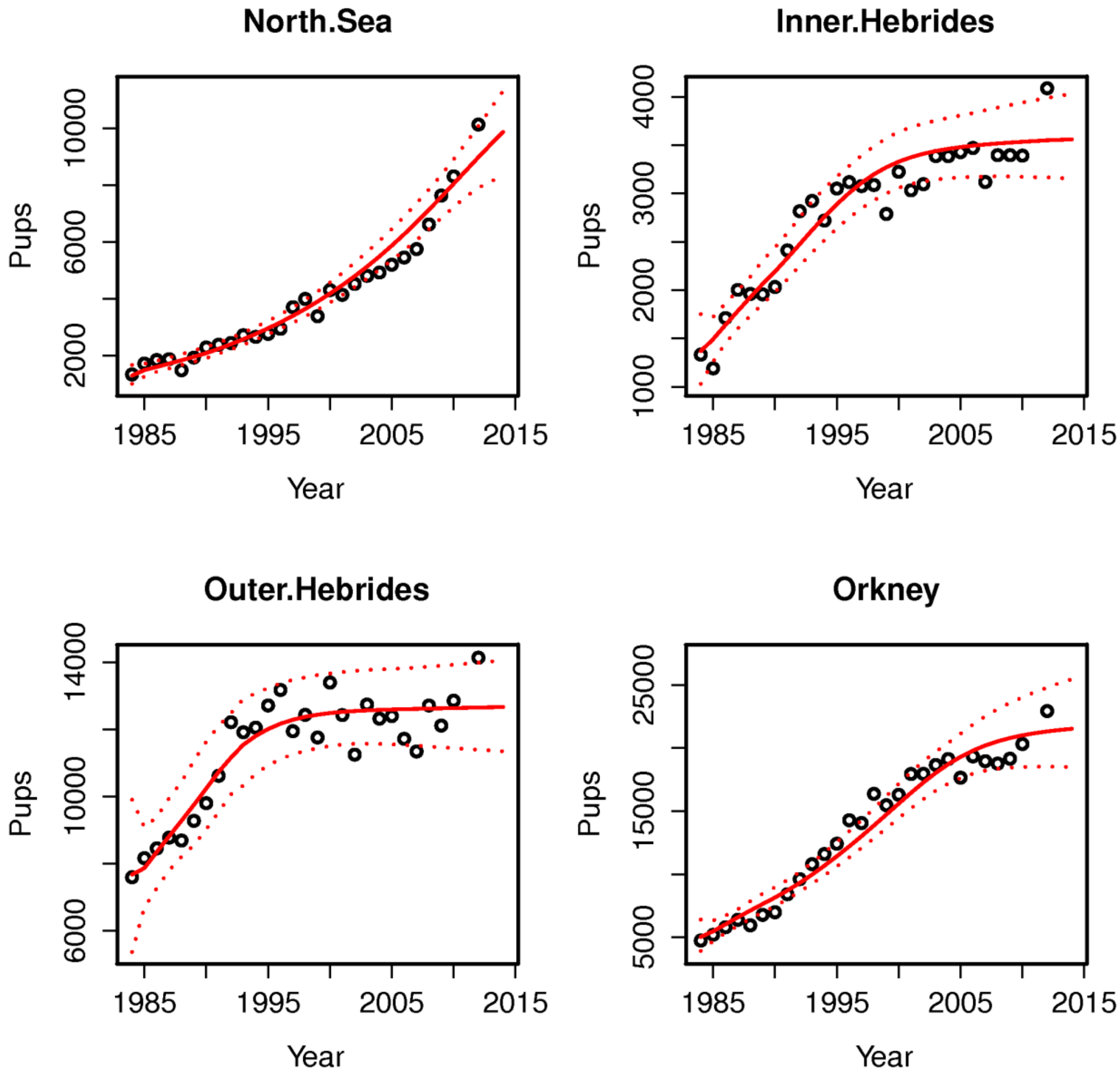

Figure 1. ***Mean estimates of pup production (solid lines) and 95% Confidence Intervals (dashed lines) from the model of grey seal population dynamics, fit to pup production estimates from 1984-2012 (circles) and a total population estimate from 2008.*** *The model projects forward to 2014 using the fitted demographic parameter estimates from 2012.*

Pup production at colonies in the North Sea continued to increase exponentially up to 2012 but majority of the increase was due to continued rapid expansion of newer colonies on the mainland coasts in Berwickshire, Lincolnshire, Norfolk and Suffolk. Interestingly, these colonies are all at easily accessible sites on the mainland where grey seals have probably never previously bred in significant numbers. Pup production in 2012 for the entire North Sea region is shown in Table 2. These show an annual increase of 10.4% p.a. between 2010 and 2012. Estimates for the ground counted colonies on the English east coast (Farne Islands, Donna Nook, Blakeney and Horsey) in 2013 and 2014 show that the rapid increase has continued. Although there was little change at the Farne Islands, the more southerly mainland colonies have increased by >22% p.a. between 2012 and 2014.

The most recent data for pup production from the major breeding sites in Wales are an estimate of 96 pups in North Wales[1], for Pembrokeshire 465 pups in 2005[2] and 379 pups born on Skomer and adjacent mainland sites in 2014.[3] The relative size of pup production at the different breeding colonies by region is shown in Figure 2.

[1]Stringell, T., Millar, C., Sanderson, W., Westcott, S. & McMath, A. (2014). When aerial surveys won't do: grey seal pup production in cryptic habitats of Wales. Journal of the Marine Biological Association of the United Kingdom, 94, 1155-1159.

[2]Strong, P.G., Lerwill, J., Morris, S.R., & Stringell, T.B. (2006). Pembrokeshire marine SAC grey seal monitoring 2005. CCW Marine Monitoring Report No: 26; unabridged version (restricted under licence), 54pp.

[3] http://wtswwcdn.8a1bc20d.cdn.memsites.com/wp-content/uploads/2014/07/Seal-Report-2014-final-.pdf

Table 2. ***Latest available grey seal pup production estimates for the UK (main colonies were surveyed in 2012)***.

Location	Pup production in 2010	Pup production in **2012**	Average annual change 2010 to **2012**	Average annual change 2001 and 2006	Average annual change 2006 to **2012**
Inner Hebrides	3,391	4,088	+9.8%	+2.8%	+3.1%
Outer Hebrides	12,857	14,136	+4.9%	+0.1%	+3.3%
Orkney	20,312	22,926	+6.2%	+0.1%	+3.0%
Firth of Forth	4,279	5,210	+10.3%	+3.9%	+11.6%
Regularly monitored colonies in Scotland	40,839	46,360	+6.5%	+1.0%	+3.9%
Other Scottish colonies [1] (incl. Shetland & mainland)	3,299 [1]	3,665 [1]	+5.4%		
Total Scotland	**44,138**	**50,025**	+6.5%		
Donna Nook +East Anglia	2,566	3,360	+14.4%	+15.6%	+15.1%
Farne Islands	1,499	1,603	+3.4%	+0.7%	+5.1%
Annually monitored colonies in England	4,065	4,963	+10.5%	+7.0%	+11.2%
SW England (last surveyed 1994)[3]	250 [3]	250 [3]			
Wales [2,3]	1,650 [3]	1,650 [3]			
Total England & Wales	**5,965**	**6,863**	**+7.3%**		
Northern Ireland[3]	100 [3]	100 [3]			
Total UK	**50,203**	**56,988**	**+6.5%**		

1 Estimates derived from data collected in different years

2 Multiplier derived from indicator colonies surveyed in 2004 and 2005 and applied to other colonies last monitored in 1994

3 Esimated production for colonies that are rarely monitored

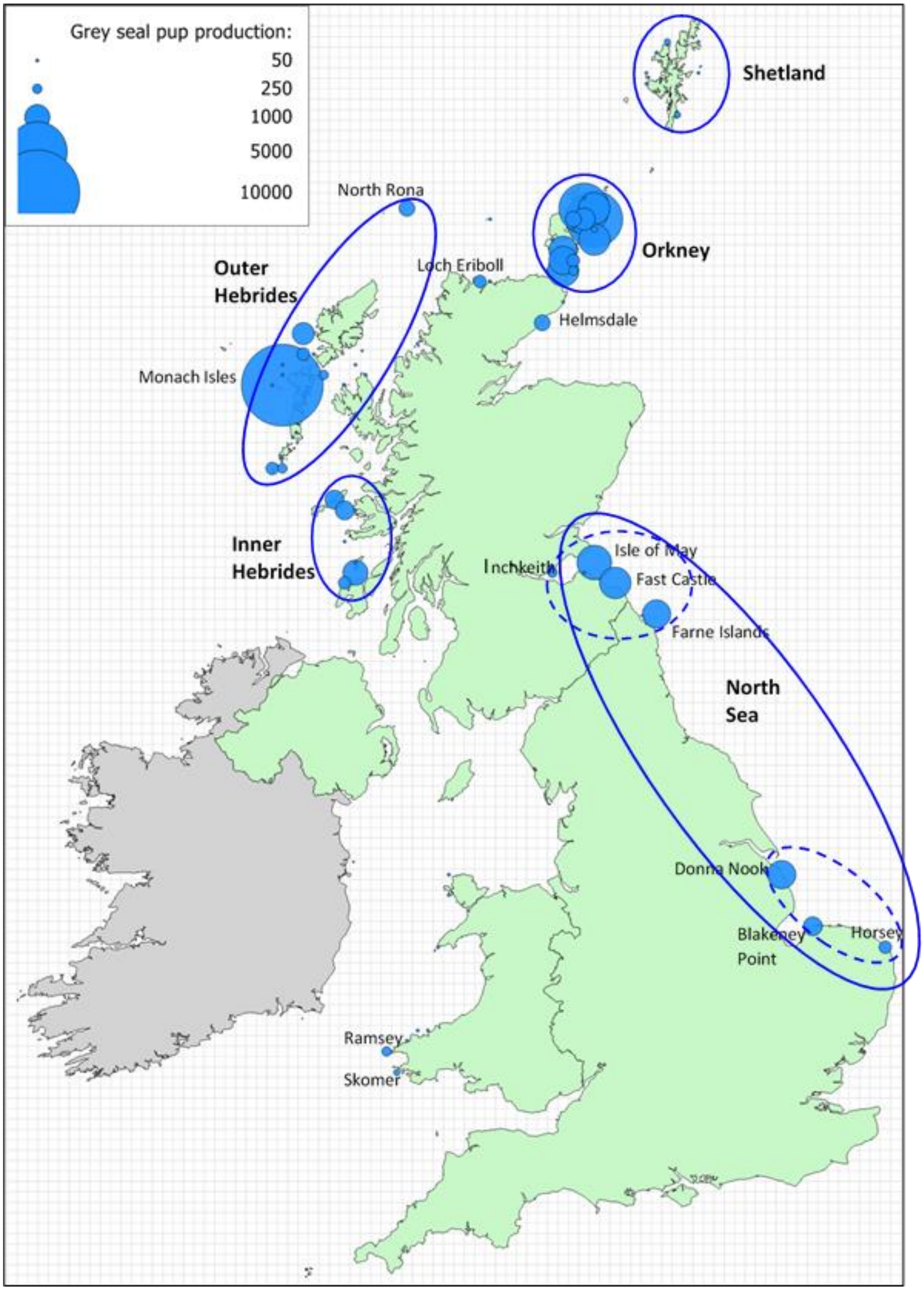

Figure 2. ***Distribution and size of grey seal breeding colonies.*** *Blue ovals indicate groups of colonies within each region.*

Population size

Converting pup counts from air surveys (i.e. biennially surveyed colonies) into a total population size requires a number of steps as shown in Figure 3.

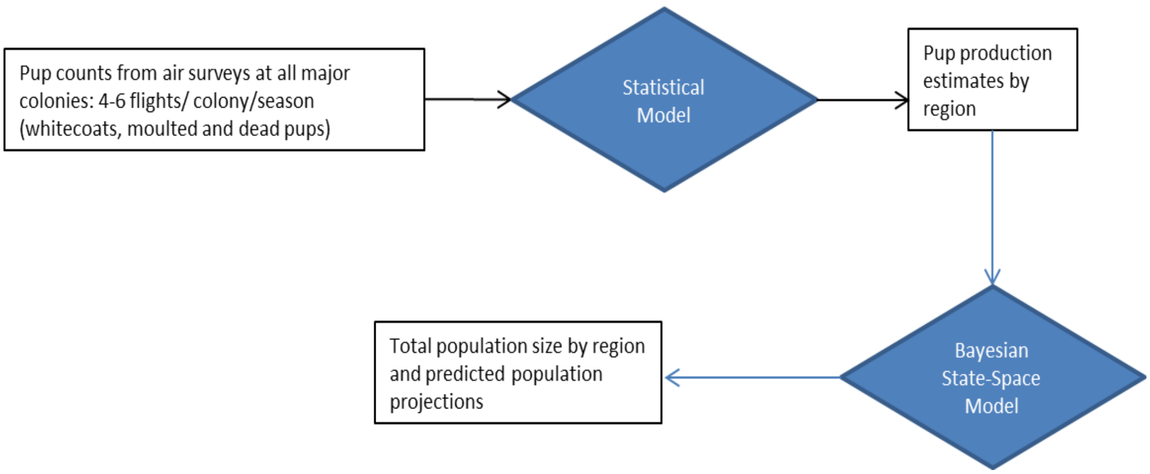

Figure 3. ***Schematic diagram of steps involved in estimating total population size from pup counts*** *(see also SCOS BP-09/02, SCOS BP-10/02).*

Using appropriate estimates of fecundity rates and both pup and non-pup survival rates we can convert pup production estimates into estimates of total population size. The estimate of the total population alive at the start of the breeding season depends critically on the estimates of these rates. We use a Bayesian state-space population dynamics model to estimate these rates.

Until the late 1990s all the regional populations grew exponentially, implying that the demographic parameters were, on average, constant over the period of data collection. Thus, estimates of the demographic parameters were available from a simple population model fitted to the entire pup production time series.

Some combination of reductions in the reproductive rate or the survival rates of pups, juveniles and adults (SCOS-BP 09/02, 10/02 and 11/02) has resulted in reduced population growth rates in the Northern and Western Isles. Fitting the model of grey seal population dynamics with density dependence acting through either fecundity or pup survival showed that the time series of pup production estimates did not contain sufficient information to allow us to quantify the relative contributions of these factors (SCOS-BP 06/07, 09/02). In 2010 and 2011, we incorporated additional information 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 10/04 and 11/06). Inclusion of the independent estimate allowed us to reject the models that assumed density dependent effects operated through fecundity and all estimates are therefore based on a model incorporating density dependent pup survival.

In 2012, SCOS discussed the priors on the model input parameters in some detail, following re-examination of the data being used and the differences made to the population estimates by changing a number of them to less informative priors (SCOS-BP 12/01 and SCOS-BP 12/02). In 2014 SCOS decided to use the results from a model run using these revised priors (SCOS-BP 12/02) and incorporated a prior based on a distribution for the ratio of males to females in the population (see SCOS-BP 14/02 for details) and the independent estimate of total population size from the summer surveys. Work on updating the priors is continuing. A re-analysis of all the combined data available from pup tagging studies (hat tags, phone tags and GPS/GSM tags) is underway. Preliminary results suggest that there are no significant sex-specific differences in first year pup survival. Updated estimates of adult female survival from the long term studies at the Isle of May reinforce the view that the fitted estimates in the population model are unrealistic. Re-analysis using constrained priors on survival is underway.

In 2014 SCOS adopted a set of revised priors, including a different prior on adult sex ratio, to generate the grey seal population estimates. In the absence of new pup counts the same analyses based on the same revised priors were used this year to predict beyond the last data point (2012) to give estimates of population size in 2013 and 2014. One small change in the data was that the total population size estimate was adjusted to account for the fact that the population model is based only on regularly monitored breeding colonies (approx. 94% of the total population). The model produced unreasonably high adult survival values of more than 0.99, so it was re-run with a prior on survival constrained to a more reasonable range of 0.8 to 0.97. Posterior mean adult survival with this revised prior was 0.95 (SD 0.03). The estimated total grey seal population associated with all regularly monitored colonies in 2014 was therefore 105,200 (95% CI 87,000-128,800) for the model incorporating density dependent pup survival, using the revised priors and including the independent estimate (details of this analysis are given in SCOS-BP 14/02 and SCOS-BP 15/02). A comprehensive survey of data available from the less frequently monitored colonies was presented in SCOS BP 11/01 and updated in 2014 (SCOS-BP 14/02). Total pup production at these sites was estimated to be approximately 5,670. The total population associated with these sites was then estimated using the average ratio of 2012 pup production to 2014 population size estimate for all annually monitored sites. Confidence intervals were estimated by assuming that they were proportionally similar to the pup survival model confidence intervals. This produced a population estimate for these sites of 11,600 (approximate 95% CI 9,600 to 14,200). Combining this with the annually monitored sites gives an estimated 2014 UK grey seal population of 116,800 (approximate 95% CI 96,600-143,000).

The fit of the model to the pup production estimates has been poor in some regions in recent years. Whilst the model accurately captures some aspects of the observed trends in pup production in some regions, the estimated adult survival rate from the model was very high and the maximum pup survival rate was very low. This suggests some other parameters, such as inter-annual variation in fecundity or survival senescence could be causing a mismatch between the estimates from the model and the pup production data. In addition, the selection of which parameter estimates are fitted and which are fixed in the pup production model may have a significant effect on the pup production estimates. The effect of this selection process on the estimates is being investigated (SCOS-BP 15/03).

Population trends
Model selection criteria suggest that density dependence is acting mainly on pup survival (see SCOS-BP 09/02). The independent population estimate is consistent with this conclusion. This also implies that the overall population should closely track the pup production estimates when experiencing density dependent control as well as during exponential growth. The model estimated that total population sizes for the annually monitored colonies have increased by approximately 0.7% p.a. (SCOS-BP 15/02) between 2012 and 2014. All of this is due to a continuing 3% p.a. increase in the North Sea; Orkney and the Hebrides are effectively stationary, increasing at <0.2% p.a. since 2010.

In the southern North Sea the rates of increase in pup production since 2010 (>22% p.a.) suggests that there must be some immigration from colonies further north.

UK grey seal population in a world context

The UK grey seal population represents approximately 39% 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 3). 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 39% of the total all age population.

Table 3. ***Relative sizes and status of grey seal populations****. Pup production estimates are generally used because of the uncertainty in overall population estimates*

Region	Pup Production	Year	Possible population trend[2]
UK	**57,000**	2012	Increasing
Ireland	2,100	2012[1]	Increasing
Wadden Sea	600	2014[2]	Increasing
Norway	1,300	2008[3]	Increasing
Russia	800	1994	Unknown
Iceland	1,200	2002	Declining
Baltic	4,700	2007[4,5]	Increasing
Europe excluding UK	**10,700**		Increasing
Canada - Sable Island	62,000	2010[6]	Increasing
Canada - Gulf St Lawrence + Eastern Shore Canada	14,200	2010[7]	Declining
USA	2,600	2008[8]	Increasing
WORLD TOTAL	**146,500**		Increasing

[1]Ó Cadhla, O., Keena, T., Strong, D., Duck, C. and Hiby, L. 2013. Monitoring of the breeding population of grey seals in Ireland, 2009 - 2012. Irish Wildlife Manuals, No. 74. National Parks and Wildlife Service, Department of the Arts, Heritage and the Gaeltacht, Dublin, Ireland.

[2]Brasseur, S., Borchardt, T., Czeck, R., Jensen, L.F., Galatius, A., Ramdohr, S., Siebert, U., Teilmann, J., 2012. Aerial surveys of Grey Seals in the Wadden Sea in the season of 2011-2012 - Increase in Wadden Sea grey seals continued in 2012. Trilateral Seal Expert Group.

[3]Øigård, T.A., Frie, A.K., Nilssen, K.T., Hammill, M.O., 2012. Modelling the abundance of grey seals (*Halichoerus grypus*) along the Norwegian coast. ICES Journal of Marine Science: Journal du Conseil, 69(8) 1436-1447.

[4]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.

[5]Baltic pup production estimate based on mark recapture estimate of total population size and an assumed multiplier of 4.7 HELCOM fact sheets (www.HELCOM.fi) & http://www.rktl.fi/english/news/baltic_grey_seal.html

[6] Bowen, W.D., den Heyer, C., McMillan, J.I., & Hammill, M.O. (2011). Pup production at Scotian Shelf grey seal (*Halichoerus grypus*) colonies in 2010. DFO Canadian Science Advisory Secretariat Research Document 2011/066.

[7] Thomas, L., Hammill, M.O. & Bowen, W.D. (2011). Estimated size of the Northwest Atlantic grey seal population 1977-2010 Canadian Science Advisory Secretariat: Research Document 2011/17, pp27.

[8]NOAA (2009) http://www.nefsc.noaa.gov/publications/tm/tm219/184_GRSE.pdf

Current status of British harbour seals

Harbour seals are counted while they are on land during their August moult, giving a minimum estimate of population size. Not all areas are counted every year but the aim is to cover the UK coast every 5 years.

Combining the most recent counts (2007-2014) gives a total of 29,109 animals in the UK. Scaling this by the estimated proportion hauled out produced an estimated total population for the UK in 2014 of 40,414 (approximate 95% CI 33,106, 55029).

Harbour seal counts were stable or increasing until around 2000 when declines were seen in Shetland (which declined by 30% between 2000-2009), Orkney (down 78% between 2000-2013) and the Firth of Tay (down 96% between 2000-2014). However, other regions have been stable (west coast of Highland region and the Outer Hebrides). Counts along the English east coast were very similar to those reported for 2013.

The most recent minimum population estimates by region are given in Table 4.

Table 4. ***UK harbour seal counts.***

Location	Most recent count (2007-2014)
England	4,806
Wales	0[1]
Scotland	23,355[2]
Northern Ireland	948
Total UK	**29,109**

[1] There are no systematic surveys for harbour seals in Wales

[2] Compiled from most recent surveys, see Table 5 for dates and details

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 15/04. It is impractical to survey the whole coastline every year, but SMRU aims to survey the entire coast across 5 consecutive years. However, in response to the observed declines around the UK the survey effort has been increased. The majority of the English and Scottish east coast populations are surveyed annually.

Harbour seals spend the largest proportion of their time on land during the moult and they are therefore visible to be counted in the surveys at this time. Most regions are surveyed by a method using thermographic aerial photography to identify seals along the coastline. However, 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 the time of year, the state of the tide and the 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.

The most recent counts of harbour seals by region are given in Table 5 and Figure 4. These are minimum estimates of the British harbour seal population. Results of surveys conducted in 2014 are described in more detail in SCOS-BP 15/04. It has not been possible to conduct a synoptic survey of the entire UK coast in any one year. Data from different years have therefore been grouped into recent, previous and earlier counts to illustrate, and allow comparison of, the general trends across regions.

Combining the most recent counts (2007-2014) at all sites, approximately 29,109 harbour seals were counted in the UK: 80% in Scotland; 17% in England; 3% in Northern Ireland (Table 5). Including the 3,500 seals counted in the Republic of Ireland produces a total count of ~32,600 harbour seals for the British Isless (i.e. the UK and Ireland).

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 English east coast populations were most affected. Counts from 2002 to 2008 did not indicate a recovery in The Wash population following the epidemic. From 2008 to 2010 the counts increased by around 40%. Since then numbers have been relatively stable and the 2014 count was very similar to that reported in 2013.

A breeding season aerial survey of the harbour seal population along the east Anglian coast was carried out in June 2014 and the results are given in SCOS-BP 15/05. The pup production in this region has continued to increase with the 2014 count being the highest recorded. The ratio of pups to total population was also extremely high, suggesting a large increase in apparent fecundity over the last 14 years.

Table 5. ***The most recent August counts of harbour seals at haul-out sites in Britain and Ireland by seal management unit compared with two previous periods, in 1996 and 1997 and between 2000 and 2006.***

Seal Management Unit / Country		Harbour seal counts					
		2007-2014		2000-2006		1996-1997	
1 Southwest Scotland		834	(2007)	623	(2005)	929	(1996)
2 West Scotland	a	13,878	(2007-2009; 2013-2014)	11,702	(2000; 2005)	8,811	(1996-1997)
2a West Scotland - South		6,339	(2007; 2009; 2014)	7,037	(2000; 2005)	5,651	(1996)
2b West Scotland - Central		6,424	(2014)	3,956	(2005)	2,700	(1996)
2c West Scotland - North		1,115	(2013; 2014)	709	(2005)	460	(1996-1997)
3 Western Isles		2,739	(2011)	1,981	(2003; 2006)	2,820	(1996)
4 North Coast & Orkney		1,938	(2013)	4,384	(2005-2006)	8,787	(1997)
4a North Coast		73	(2013)	146	(2005-2006)	265	(1997)
4b Orkney		1,865	(2013)	4,238	(2006)	8,522	(1997)
5 Shetland		3,039	(2009)	3,038	(2006)	5,994	(1997)
6 Moray Firth		733	(2008; 2011; 2013-2014)	1,028	(2005-2006)	1,409	(1997)
7 East Scotland		194	(2007; 2013-2014)	667	(2005-2006)	764	(1997)
SCOTLAND TOTAL		**23,355**	(2007-2009; 2011; 2013-2014)	**23,423**	(2000; 2003; 2005-2006)	**29,514**	(1996-1997)
8 Northeast England	b	90	(2008; 2014)	* 62	(2005-2006)	* 54	(1997)
9 Southeast England	c	4,681	(2014)	2,964	(2005-2006)	3,222	(1995; 1997)
10 West England & Wales	d	35	(estimate)	20	(estimate)	15	(estimate)
ENGLAND & WALES TOTAL		**4,806**	(2008; 2014)	**3,046**	(2005-2006)	**3,291**	(1995; 1997)
BRITAIN TOTAL		**28,161**	(2007-2009; 2011; 2013-2014)	**26,469**	(2000; 2003; 2005-2006)	**32,805**	(1995-1997)
NORTHERN IRELAND TOTAL	e	**948**	(2011)	**1,176**	(2002; 2006)		
UK TOTAL		**29,109**	(2007-2009; 2011; 2013-2014)	**27,646**	(2000; 2002-2003; 2005-2006)		
REPUBLIC OF IRELAND TOTAL	f	**3,489**	(2011-2012)	**2,955**	(2003)		
BRITAIN & IRELAND TOTAL		**32,598**	(2007-2009; 2011-2014)	**30,601**	(2000; 2002-2003; 2005-2006)		

SOURCES - Most counts were obtained from aerial surveys conducted by SMRU and were funded by Scottish Natural Heritage (SNH) and the Natural Environment Research Council (NERC). Exceptions are:

[a] Parts of the West Scotland survey in 2009 funded by Scottish Power and Marine Scotland.

[b] The Tees data collected and provided by the Industry Nature Conservation Association (Woods, 2014). The 2008 survey from Coquet Island to Berwick funded by the Department of Energy and Climate Change (DECC, previously DTI).

[c] Essex & Kent data for 2014 collected and provided by the Zoological Society London (Barker, 2015).

[d] No dedicated harbour seal surveys in this management unit and only sparse info available. Estimates compiled from counts shared by other organisations (Chichester Harbour Conservancy) or found in various reports & on websites (Boyle, 2012; Hilbrebirdobs.blogspot.co.uk, 2012, 2013; Sayer, 2010a, 2010b, 2011; Sayer *et al.*, 2012; Westcott, 2002). Apparent increases may partly be due to increased reporting and improved species identification.

[e] Surveys carried out by SMRU and funded by Northern Ireland Environment Agency (NIEA) in 2002 & 2011 (Duck, 2006; Duck & Morris, 2012) and Marine Current Turbines Ltd in 2006-2008 & 2010 (SMRU Ltd, 2010).

[f] Surveys carried out by SMRU and funded by the National Parks & Wildlife Service (Cronin *et al*., 2004; Duck & Morris, 2013a, 2013b).

*Northumberland coast south of Farne Islands not surveyed in 2005 & 1997, but no harbour seal sites known here.

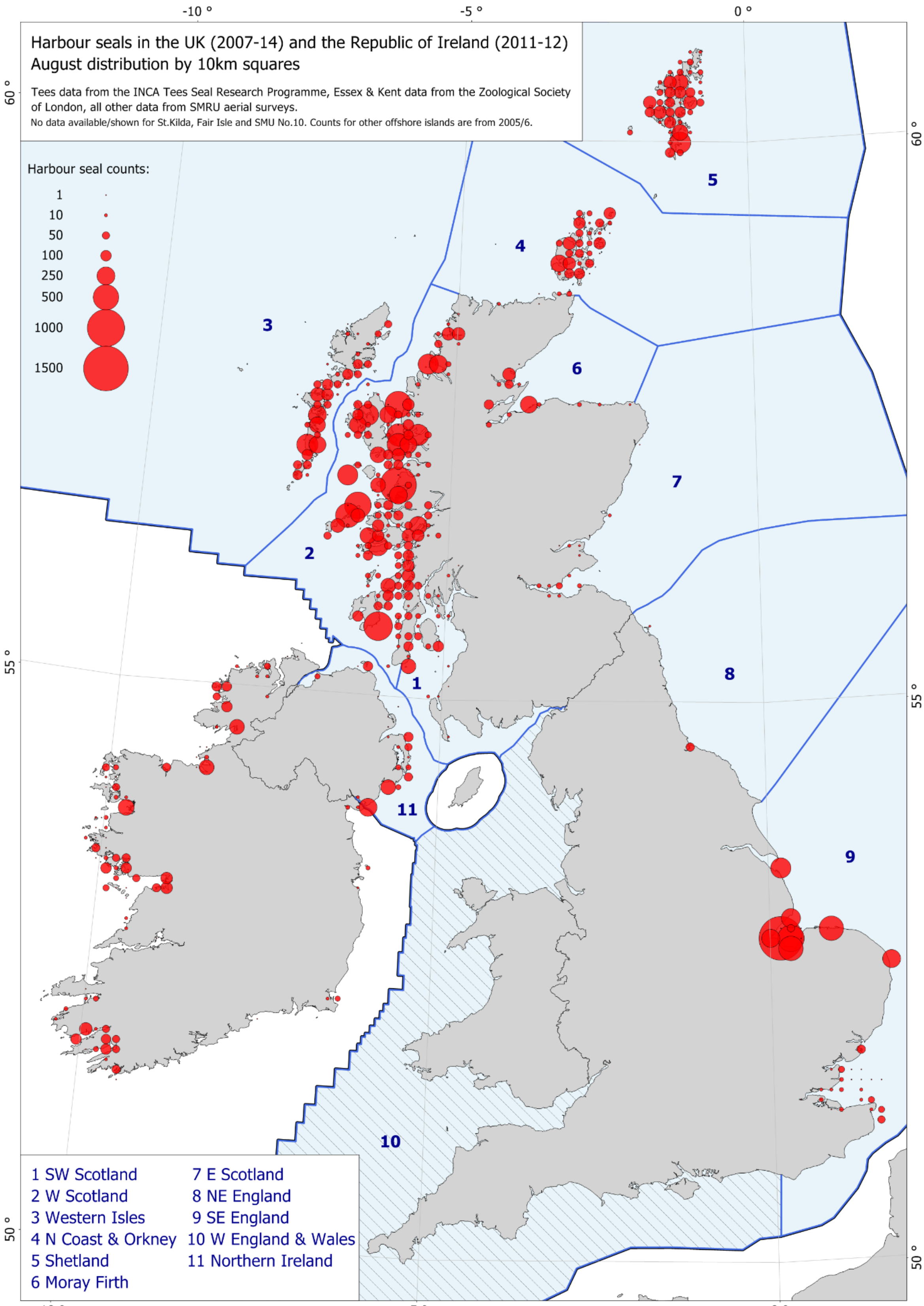

Figure 4. ***August distribution of harbour seals around the British Isles****.* *Very small numbers of harbour seals (<50) are anecdotally but increasingly reported for the West England & Wales management unit, but are not included on this map*

Population trends

As reported in SCOS 2008 to 2014, there have been general declines in counts of harbour seals in several regions around Scotland. Details are given in SCOS-BP 15/04. The recent trends are shown in Figure 5.

In August 2014, SMRU surveyed a large section of the Scottish west coast between Ullapool and the Firth of Lorn. The West Scotland harbour seal count increased by 39% between 2008 and 2014, equivalent to an average annual increase of 5.7%. Most of this increase occurred in the West Scotland – Central and West Scotland – North management regions (Table 5)

In the Moray Firth there is considerable variability in the August total counts for the entire region. The most recent count was the lowest recorded since the late 1980s. However, counts at Culbin Sands (between Findhorn and Nairn), have increased recently and 236 animals were counted in August 2014 (SCOS-BP 15/04).

The 2014 harbour seal moult count for the Firth of Tay and Eden Estuary Special Area of Conservation (SAC) (29) was 42% lower than the 2013 count of 50 (SCOS-BP 15/04). The 2014 count is a new all-time low for this harbour seal SAC and represents only 5% of the mean from counts between 1990 and 2002 (641). Harbour seals in this area are of sufficient concern that Marine Scotland has not issued any licences to shoot harbour seals within the East Scotland Management Area since 2010. Ongoing work funded by Marine Scotland is investigating the potential reasons for the differences in trends in different areas.

The combined counts for the Southeast England management unit in 2014 (4,681) was very similar to the 2013 count (4,504). Although the Southeast England population has returned to its pre-2002 epidemic levels, it is still lagging behind the rapid recovery of the harbour seal population in the Wadden Sea where counts have increased from 10,800 in 2003 to 26,788 in 2013, equivalent to an average annual growth rate of 9.5% over the last ten years.

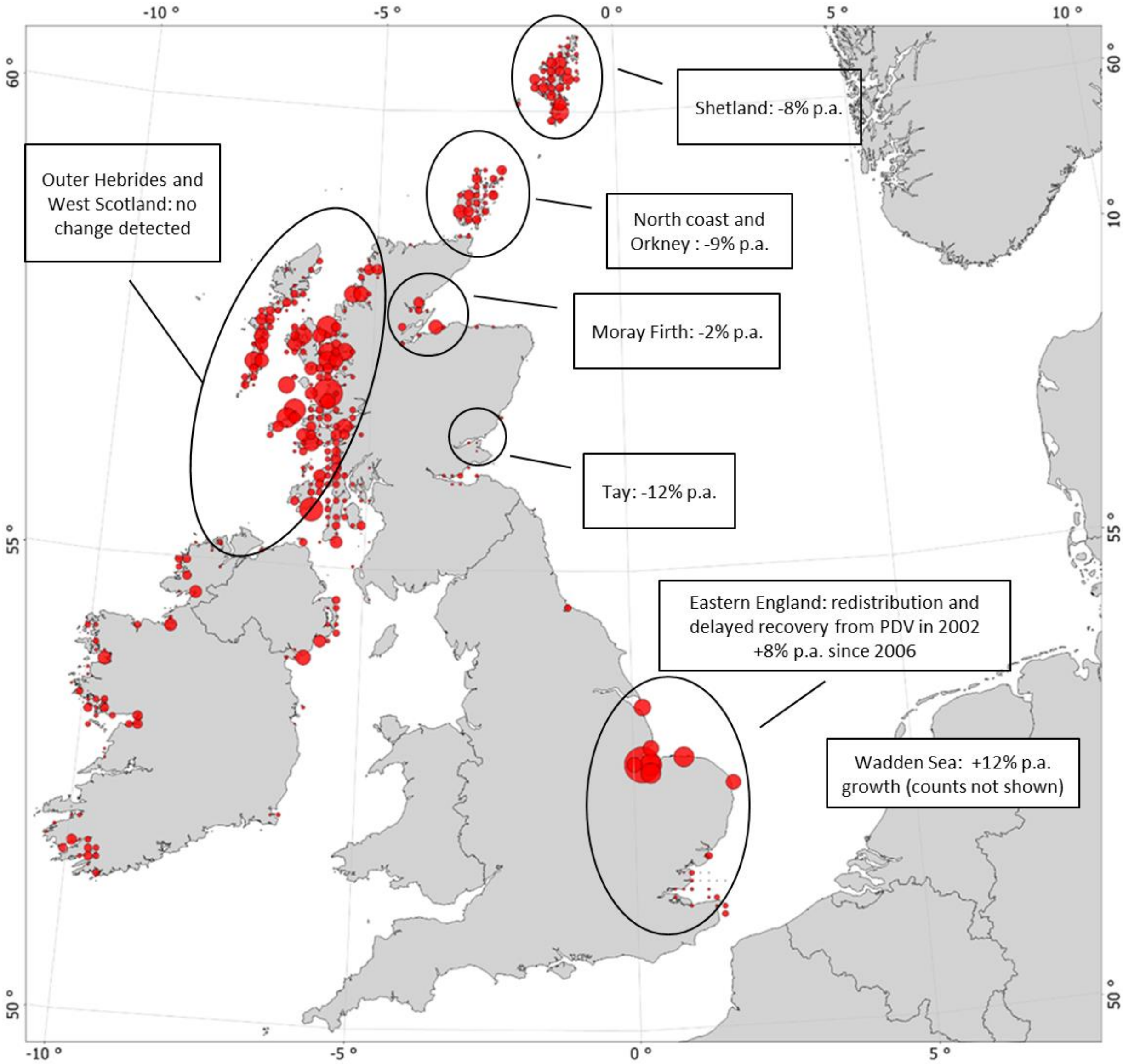

Figure 5. ***Recent trends in numbers of harbour seals counted in different parts of the UK****. Trends and annual changes are estimated between 1988 and 2013 using generalised linear models following the methods of Lonergan et al., (2007)*[4]

UK harbour seal populations in a European context

The UK harbour seal population represents approximately 30% of the eastern Atlantic sub-species of harbour seal (Table 6). The declines in Scotland and coincident dramatic increases in the Wadden Sea mean that the relative importance of the UK population is declining.

[4] Lonergan, M., Duck, C.D., Thompson D., Mackey, B.L., Cunningham, L. & Boyd I.L. (2007). Using sparse survey data to investigate the declining abundance of British harbour seals. Journal of Zoology, 271(3), 261-269.

Table 6. ***Size and status of European populations of harbour seals.*** *Data are counts of seals hauled out during the moult.*

Region	Number of seals counted[1]	Years when latest data was obtained
Scotland	**23,400**	2007-2014
England	**4,800**	2014
Northern Ireland	**900**	2011
UK	**29,100**	
Ireland	3,500	2011-12
Wadden Sea-Germany	16,100	2014
Wadden Sea-NL	7,100	2014
Wadden Sea-Denmark	3,400	2014
Lijmfjorden	1,400	2013
Kattegat	9,500	2013
Skagerrak	2,600	2007
Baltic	1,500	2013
Norway	7,100	2013
Iceland	11,000	2011
Barents Sea	1,900	2010
Europe excluding UK	**65,100**	
Total	**94,200**	

[1]Counts rounded to the nearest 100. They are minimum estimates of population size as they do not account for proportion at sea and in many cases are amalgamations of several surveys.

Data sources: ICES Report of the Working Group on Marine Mammal Ecology 2014; Desportes,G., Bjorge,A., Aqqalu, R-A and Waring,G.T. (2010) Harbour seals in the North Atlantic and the Baltic. NAMMCO Scientific publications Volume 8; Nilssen K, 2011. Seals – Grey and harbour seals. In: Agnalt A-L, Fossum P, Hauge M, Mangor-Jensen A, Ottersen G, Røttingen I,Sundet JH, and Sunnset BH. (eds). Havforskningsrapporten 2011. Fisken og havet, 2011(1).; Härkönen,H. and Isakson,E. 2010. Status of the harbor seal (*Phoca vitulina*) in the Baltic Proper. NAMMCO Sci Pub 8:71-76.; Olsen MT, Andersen SM, Teilmann J, Dietz R, Edren SMC, Linnet A, and Härkönen T. 2010. Status of the harbour seal (*Phoca vitulina*) in Southern Scandinavia. NAMMCO Sci Publ 8: 77-94.; Galatius A, Brasseur, S, Czeck R et al, 2014,,Aeiral surveys of harbour seals in the Wadden Sea in 2014, http://www.waddensea-secretariat.org; Härkönen T, Galatius A, Bräeger S, et al HELCOM Core indicator of biodiversity Population growth rate, abundance and distribution of marine mammals, HELCOM 2013, www.helcom.fi; http://www.fisheries.is/main-species/marine-mammals/stock-status/; http://www.nefsc.noaa.gov/publications/tm/tm213/pdfs/F2009HASE.pdf http://www.nammco.no/webcronize/images/Nammco/976.pdf, Nilssen K and Bjørge A 2014. Seals – grey and harbor seals. In: Bakketeig IE, Gjøsæter H, Hauge M, Sunnset BH and Toft KØ (eds). Havforskningsrapporten 2014. Fisken og havet, 2014(1).

2. **What is latest information about the population structure, including survival, fecundity and age structure of grey and common seals in UK and European waters? Is there any new evidence of populations or sub-populations specific to local areas?**	MS Q2; Defra Q2; NRW Q2

Grey seals

There is evidence for regional differences in grey seal demographics but information on vital rates would improve our ability to provide advice on population status. This includes the requirement for a time series of fecundity and survival rates on a regional basis.

The only contemporary data that we have on fecundity and adult survival has been estimated for adult females at the two breeding colonies which constitute the long term studies (see survival and fecundity rates below)

Age and sex structure

When the population was growing at a constant (i.e. exponential) rate, the female population size was directly proportional to the pup production. 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.

Survival and fecundity 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. An integrated analysis of mark-recapture data (for more details see SCOS-BP 14/04) suggest that fecundity differed between sites with a general estimate of 0.77 (0.750, 0.792 95% Bayesian credible intervals) for North Rona and 0.86 (0.835, 0.882 95% Bayesian credible intervals) for the Isle of May. These estimates are lower than previous estimates for UK grey seals of 0.94 for the Farne Islands but are comparable to the estimate of 0.83 for the Hebrides[5].

Adult survival (averaged over all years) at the Isle of May was not related to mass and was estimated to be generally high 0.926 (95% Bayesian credible interval 0.792, 0.977). Further analysis comparing a range of mark recapture models based on permanent marks, estimated apparent survival probabilities of between 0.92-0.94 (for details see SCOS-BP 15/06). At North Rona annual survival rates were estimated to be 0.936 (95% Bayesian credible interval 0.904, 0.961). For more details see SCOS-BP 14/04.

Regional differences in grey seal demographics and genetics

The difference in population trends between regions for UK grey seals suggests underlying regional differences in demographics. 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[6] and within Scotland, there are significant differences between grey seals breeding on the Isle of May and on North Rona[7].

[5] Boyd, I.L. (1985). Pregnancy and ovulation rates in grey seals (*Halichoerus grypus*) on the British coast. Journal of Zoology 205(A), 265-272.

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

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

Harbour seals

Knowledge of UK harbour seal demographic parameters (i.e. vital rates) is limited and therefore inferences about the population dynamics rely largely on count data from moulting surveys. Information on vital rates would improve our ability to provide advice on population status.

Age and sex structure
The absence of any extensive 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. Although seals found dead during the PDV epidemics in 1988 and 2002 were aged, these were clearly biased samples that cannot be used to generate population age structures.

Survival and fecundity rates
Survival estimates among adult UK harbour seals from photo-ID studies carried out in NE Scotland have been published[8,9]. This resulted in estimates of 0.95 (95% CI 0.91-0.97) for females and 0.92 (0.83-0.96) for males.

A population model for the Moray Firth harbour seals has been developed to investigate the sensitivity of the population to changes in various vital rates. The model suggests that even small changes in the survival of adult females could result in a decline in the population. Further details of the model and the potential impact of various covariates are given in SCOS-BP 15/07.

A study investigating survival in first year harbour seal pups using telemetry tags was carried out by SMRU in Orkney and on Lismore in 2007. Survival was not significantly different between the two regions and expected survival to 200 days was very low at only 0.3[10].

Genetics
Genetic data from a study directed toward resolving patterns of population structure of harbour seals from around the UK and adjacent European sites has recently been more thoroughly analysed. The results suggest four main genetic groupings: 1) a north-western group from Northern Ireland round the Scottish northwest coast and including the Outer Hebrides and the Pentland Firth; 2) a northeastern cluster from Shetland and Orkney south to the Tay and Eden estuaries; 3) a south-eastern cluster consisted of sites from Chichester harbour to the Wash, including sites in Normandy and the Dutch Wadden Sea; 4) seals from Norway.

These genetic data are in overall agreement with the existing management areas for UK harbour seals was observed. However, the genetic data suggest harbour seals in the Pentland Firth might be considered more as part of the north western grouping rather than with the Orkney population in the northeastern cluster.

[8]Cordes, L.S. & Thompson, P.M. (2014). Mark-recapture modelling accounting for state uncertainty provides concurrent estimates of survival and fecundity in a protected harbor seal population. Marine Mammal Science 30(2): 691-705.
[9]Mackey, B.L., Durban, J.W., Middlemas, S.J. & Thompson, P.M. (2008). A Bayesian estimate of harbour seal survival using sparse photo-identification data. Journal of Zoology, 274: 18-27
[10]Hanson, N., Thompson, D., Duck, C., Moss, S. & Lonergan, M. (2013). Pup mortality in a rapidly declining harbour seal (*Phoca vitulina*) population. PLoS One, 8: e80727.

Seal population trends.

3. How many year's data and what scale of change in numbers of seals counted are required to be able to say that a population is showing a significant upward or downward trend?	MS Q3; Defra Q3

The ability to detect a significant trend in population abundance depends to some extent on the observation uncertainty associated with the count data, the environmental conditions and the available budget. Thus monitoring effectiveness (ability to detect trends) and efficiency (ability to do this at low cost) are key considerations in determining the temporal and spatial scale required, as is the management objective being considered.

Maximizing the probability of detecting a trend in abundance if it is present requires survey data to be collected at the appropriate temporal and spatial scales. For many species it is particularly important to be able to detect a negative trend in abundance so that appropriate conservation measures can be implemented. However, this can be difficult as the power to detect a decline decreases as populations become smaller[11]. Whilst it is important to carry out a power analysis before the start of a survey programme, the decisions about how regular to survey are often modified due to financial constraints. Indeed Thomas (2009)[12] reported that very small trends in population abundance, such as 1% per year, are not detectable in any reasonable time-span. Conversely, retrospective power calculations can be controversial as different analyses can yield different results and thus the goal of the analysis needs to be clearly stated.[13]

For the UK seal populations, a simple linear trend power analysis has been carried out in relation to the use of these data for assessing 'good environmental status' under the Marine Strategy Framework Directive[14].

For harbour seals, the results were reported at SCOS 2014 (SCOS-BP-14/06). For those management regions where populations have been annually monitored (i.e. a single moult count each year) for the past 18 years the minimum detectable trend at a significance level of α= 0.05 and a power level of 0.8 was a decline of 10% over the 18 year survey period, at a survey coefficient of variation (CV) of 0.05. If the survey CV was 0.15 then the minimum detectable decline was 28% and 36% if the CV was 0.2. The magnitude of the detectable declines therefore increased substantially in regions where surveys have been much sparser. If the acceptable significance level is increased to α=0.2 (considered to be a pragmatic approach to conservation) then the minimum detectable trends were a 9%, 27% and 37% decline for the three survey CV levels respectively. Clearly this shows that even with a long, annual survey time series significant declines of less than 10% are difficult to detect.

Data from surveys in southern Scandinavia[15] found that within-season and between-year variances influenced the ability to detect a trend over a 6-year period. Overall, power was doubled when carrying out annual compared to biennial surveys and increased substantially when carrying out replicate surveys during the annual moult. The gain in power increased steeply up to three annual replicates.

[11]Taylor, B.L. & Gerrodette, T. (1993). The Uses of Statistical Power in Conservation Biology: The Vaquita and Northern Spotted Owl Conservation Biology. 7: 489-500.
[12]Thomas, L. (2009). Potential Use of Joint Cetacean Protocol Data for Determining Changes in Species' Range and Abundance: Exploratory Analysis of Southern Irish Sea Data. Available at: http://jncc.defra.gov.uk/pdf/JCP_Prelim_Analysis.pdf.
[13]Thomas, L. (1997). Retrospective Power Analysis. Conservation Biology 11:276-280
[14]Gerrodette, T. (1993). Trends - Software for a Power Analysis of Linear-Regression. Wildlife Society B 21:515-516
[15]Teilmann, J., Rigét, F. & Harkonen, T. (2010). Optimizing survey design for Scandinavian harbour seals: population trend as an ecological quality element. ICES Journal of Marine Science: Journal du Conseil 67:952–958.

For grey seals, again simple power analyses were conducted at the 2014 ICES Working Group on Marine Mammal Ecology Meeting[16] to investigate the rate of decline in grey seal relative abundance that could be detected from the SMRU biennial grey seal pup surveys. The surveys generate estimates of total pup production with a CV of about 0.1, now every two years. The probability of making a Type-I error was set at $\alpha = 0.05$. The probability of making a Type-II error was set at $\beta = 0.20$; equivalent to a power of 80%. The minimum detectable rate of decline per year for biennial surveys was 10% per annum over a six year period and 3% per annum over a 12 year period.

<u>Harbour/common seal population</u>

4. Is the existing harbour seal decline recorded in several local areas around Scotland continuing or not and what is the position in other areas?	MS Q4

The status of the local harbour seal population varies around the UK. Details of surveys carried out and the counts obtained are given above in answer to Question 1 and in SCOS-BP 15/04.

5. In the light of the latest reports, should the Scottish Government consider additional conservation measures to protect vulnerable local harbour seal populations in any additional areas to those already covered by sea conservation areas or should it consider removing existing conservation measure in any areas?	MS Q5

The measures to protect vulnerable harbour seal populations should remain in place.

The dramatic decline in the population of harbour seals in the Firth of Tay and Eden Estuary SAC is a clear cause for continued concern. In addition, a further decline was seen in Orkney (see SCOS Advice 2014). The potential biological removal (PBR) is calculated for each region for each year (SCOS-BP 15/08) and the recovery factor is reviewed annually based on the latest survey data.

Conservation orders are currently in place for the Outer Hebrides, Northern Isles and down the east coast as far as the border.

[16]ICES (2014). Report of the Working Group on Marine Mammal Ecology WGMME), 10-13 March, 2014, Woods Hole Massachusetts, International Council for the Exploration of the Sea, ICES CM 2014/ACOM: 27. 232 pp.

6. What is the latest understanding of the causes of the recent decline in common/harbour seals?	MS Q6

Potential causes of the decline in Scottish harbour seals in some regions include interactions with grey seals (both indirect, such as competition for resources and habitat, and direct, such as predation) and exposure to toxins from harmful algae. Funding granted to SMRU from Scottish Government has enabled the integrated research project suggested by the Harbour Seal Decline workshop (see SCOS 2014) to be undertaken. Study site identification and initial data collection has now started. Reports on the findings from this study will be presented to SCOS annually.

The Sea Mammal Research Unit has been funded by Scottish Government to investigate the causes of these declines. Although these have not yet been identified, various factors can now be ruled out as **primary** causes for the decrease in numbers. Some of these may be involved secondarily (such as changes in body condition and secondary infection) and the causes of the decline may not be the same in all regions.

Infectious disease (viral, bacterial, fungal, parasitic, protozoal) - Data from live captures, rehabilitation centres and dead stranded seals indicate that infectious diseases are not causing higher levels of mortality. Phocine Distemper Virus is no longer circulating and there have been no reports of sick seals on haulout sites.

Nutritional stress - Data from live captures, rehabilitation centres and strandings indicate that harbour seals in areas of decline are not in poor body condition and are not showing signs of starvation or nutritional stress.

Legal shooting - Introduction of the Moray Firth seal management plan and Marine Scotland (2010) Act have markedly reduced levels of shooting. The seal licensing system is ensuring that declining populations are protected.

Fisheries bycatch - Data from the bycatch observer programme and strandings indicate that harbour seals are not being caught in nets. There are no gillnet fisheries in the regions of decline.

Pollution - Levels of persistent organic pollutants are very low in areas of decline, well below any thresholds that have been identified as causing adverse health effects.

Loss of habitat - Data from aerial surveys and telemetry studies indicate that foraging, moulting and breeding sites have not been lost.

Dispersal and emigration - Data from telemetry studies indicate no permanent dispersal or emigration within or away from Scotland. Genetic structure studies also show that harbour seals on the west compared to the east coast remain distinctly genetically different, suggesting no recent regional movement.

Entanglement in marine debris - Data from stranded seals and from faecal samples from haul-out sites indicate that entanglement in marine debris or ingestion of plastics is probably not a major issue for UK seals.

Trauma (accidental killing) -The hypothesis that interactions with vessels cause spiral seal trauma has been superceded by the observation that these injuries can be caused by grey seals.

Current research

Current research, funded by Scottish Government, is now focussing on estimating the survival and birth rates for harbour seals at sites within the seal management regions that show contrasting population trajectories (such as Orkney compared to West Scotland).

These results will be combined with information on potential drivers of population change that have not been excluded as factors affecting harbour seal survival and birth rates. These include:

- prey quality and availability
- increasing grey seal population size and the potential for competition between the two seal species, including any evidence of mortality caused by grey seals
- the occurrence and exposure of seals to toxins from harmful algae

Dead seals that wash ashore and can be recovered will be examined post mortem by the veterinary pathologists from the Scottish Marine Animal Strandings Scheme.

Unusual seal mortalities

7. What is the latest understanding of the causes of the recent unusual seal mortalities including seal predation and of their potential impact on wider populations of both grey and harbour seals?	MS Q7; Defra Q7;
What is the latest understanding and how confident are we that ducted propellers are no longer the likely cause of 'corkscrew' seal injuries in all age classes? What is the scale and distribution of these impacts and the likely population effects?	NRW Q4

The latest understanding of the cause of the recent unusual spiral seal mortalities is that this is likely to be due to predation by male grey seals rather than ducted propellers. A study funded by Scottish Government is being carried out by SMRU to determine whether collisions with vessels remain a plausible explanation.

A series of observations of predation by adult male grey seals on harbour seals at Helgoland in Germany and grey seal pups at locations in Scotland and Wales indicate that such activity may be more widespread and frequent than had previously been recognised.

A detailed discussion of these events and the re-analysis of necropsy reports are presented in Onoufriou et al. (submitted)[17]. Briefly, the detailed examination of injuries resulting from observed predation events on grey seal pups in Scotland has confirmed that male grey seals can and do inflict wounds that have previously been identified as corkscrew injuries and assigned to anthropogenic causes. Observations in Germany confirm the same wounds were seen on harbour seals preyed on by at least two different grey seal males at Helgoland. These observed events included all age classes of harbour seals from pups to adult females. A re-examination of necropsy records from both grey and harbour seal carcasses in the UK suggests that all of the corkscrew seal deaths confirmed by detailed necropsy could have been due to such predation. This does not entirely discount the possibility of other causes such as propeller injuries, but the absence of any direct observations of such events and absence of suitable vessels to account for some cases calls that hypothesis into question. Conversely, predation by grey seals is a confirmed cause that could explain all recorded cases.

In the last year there have been observations of male grey seals killing grey seal pups in Orkney, the Firth of Forth and Wales[17,18]. There are anecdotal reports of previous observations of grey seals eating harbour seals in East Anglia and killing harbour seal pups in Orkney as well as the detailed observations from Germany.

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[17] Onoufriou J., Thompson D., Bishop A. & Brownlow A. (submitted). Grey seal (*Halichoerus grypus*) cannibalism may indicate the cause of spiral lacerations in seals. Proceedings of the Royal Society B.
[18] Boyle, D. (2011). Grey Seal Breeding Census: Skomer Island 2011. Countryside Council for Wales Regional Report CCW/WW/11/1.

Over recent years corkscrew injury has been a major cause of death in the relatively small number of harbour seals recorded by SMASS. Given the observed rates of predation due to individual male seals it is clear that single predatory grey seals can cause substantial mortality in local seal populations. The number of predatory seals is unknown, but if the corkscrew injuries are due to predation, the temporal and spatial pattern suggests that at least six individuals are currently active around UK and German coasts.

The concentrations of such carcass strandings suggest that only a small number of grey seal males could inflict substantial mortality on local harbour seal populations. If the 2010 Norfolk events were due to one predatory male, it could account for removing at least 28 breeding female harbour seals and additionally caused the deaths of their pups in one breeding season. The observations in Norfolk and Scotland show that the majority of harbour seal casualties have been adult females. A relatively small number of such predators targeting adult females would be capable of causing declines even in large harbour seal populations.

Seal Licensing and PBRs

8. What, if any, changes are suggested in the Permitted/Potential Biological Removals (PBRs) for use in relation to the seal licence system (see Annex II for details)?	MS Q8; Defra Q7

This question has been addressed in relation to the details given in the Appendix to the question (see Annex II for these details). The response to all the points raised can be found in the document entitled 'PBR question Appendix 2015'.

The provisional regional PBR values for Scottish seals for 2016 are given in SCOS-BP 15/09.

Seals and Marine Renewables

9. What is the current state of knowledge of interactions actual or potential between seals and marine renewable devices and possible mitigation measures?	MS 10; Defra Q10

Since reporting in 2014 (see SCOS Advice 2014), there are a number updates on the interactions between seals and marine renewable devices (wind, wave, and tide).

Wind

GPS/GSM tags were deployed on 24 harbour seals in the Washand the behaviour of these animals was monitored during the construction of a wind farm The results from this study, suggest that seals were not excluded from the vicinity of the windfarm during the construction phase[19]. Analysis of the at sea locations of individual seals during pile driving showed that the closest distance of each seal to pile driving varied from 4.7 to 40.5 km. Pile driving data and acoustic propagation models, were combined with seal movement and dive data to predict possible auditory damage in each seal. This comparison suggests that half of the seals exceeded published auditory damage thresholds[20]. However, these results must be viewed as preliminary because there are a number of key

[19] Hastie G.D., Russell D.J.F., McConnell, B.J., Moss, S., Thompson, D. & Janik, V.M. (2015). Sound exposure in harbour seals during the installation of an offshore wind farm: predictions of auditory damage. Journal of Applied Ecology, 52:631-640.

uncertainties associated with the model. The biological implications of this are currently unclear and the study provides no information to assess the possible effects on individual survival or fecundity. Further analyses of changes in the at-sea distribution of seals during pile driving are ongoing and are due for reporting late 2015.

Current evidence suggests there is no large scale displacement of animals from operational wind farms. Using telemetry tags, harbour and grey seals were observed within an operational windfarm with no apparent differences in behaviour compared to control areas[20]. Similarly, a proportion of harbour seals tagged in the Thames (8/10 tagged seals) and The Wash (7/22 tagged seals) in 2012, entered operational windfarms suggesting animals are not completely displaced from them. Further, some individual seals spend prolonged periods at the individual turbine foundations, probably because of foraging opportunities through an artificial reef effect[21].

Mitigation

Recommended operational protocols to minimise the likelihood of harm to seals during pile driving operations were published by the JNCC in 2010[22]. These suggest that monitoring should be carried out for at least 20 minutes before piling commences. As seals do not regularly vocalise the monitoring must rely on visual observations. If any marine mammals are detected within 500m of the pile, the start should be delayed until at least 20 minutes have elapsed with no further detections. They also recommend that piling should not start in poor visibility and that a soft start should be incorporated in the piling schedule.

The use of bubble curtains to attenuate the noise from piling has been tested but has produced variable results and was not recommended by JNCC. However, in trials with small scale pile driving operations the use of bubble curtains reduced peak to peak noise levels by >14dB and reduced avoidance behaviour of captive porpoises[23]. The attenuation achieved was estimated to reduce the predicted range at which a source would cause temporary threshold shift (TTS) by around 75%.

The use of ADDs as potential measures to mitigate the effects of pile driving on seals has been tested during a series of controlled exposure experiments with tagged harbour seals[24.] All seals tested out to a range of 1km showed an identifiable change in behaviour. However, not all responses resulted in straight forward movement away from the sound source and responses varied depending on the particular circumstances of the experiment and probably the motivation and status of the subjects. Further work will be required to confirm the effectiveness of this approach.

Wave

Data on the interactions between seals and wave energy devices remain lacking and no commercial scale developments are planned to date.

Tidal

The only direct information on interactions between seals and tidal stream energy devices (turbines) remains that collected in Strangford Narrows in Northern Ireland where a long term study of seal populations and seal foraging movements has been carried out during the development and deployment stage of SeaGen, a large twin rotor tidal turbine.

Telemetry data shows harbour seals used Strangford Narrows throughout periods of turbine operation and SeaGen is not an overt barrier to their movements. Analysis of all of the tagged seals

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[20] McConnell, B., Lonergan, M.E. & Dietz, R. (2012). Interactions between seals and offshore wind farms. Marine Estate Research Report, The Crown Estate, 41pp.

[21] Russell, D.J.F., Brasseur, S.M.J.M., Thompson, D., Janik, V.J., Aarts, G., McClintock, B.T., Matthiopoloulos, J., Moss, S.E.M. & McConnell, B. (2014). Marine mammals trace anthropogenic structures at sea. Current Biology 24:R638-R639

[22] https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/50006/jncc-pprotocol.pdf

[23]Lonergan, M.E., Sparling, C.E., & McConnell., B. (submitted). Behaviour of harbour seals (*Phoca vitulina*) around an operational tidal turbine. Plos One.

[24] Gordon, J., Blight, C., Bryant, E., & Thompson, D. (2015). Tests of acoustic signals for aversive sound mitigation with harbour seals . Sea Mammal Research Unit, University of St Andrews, Report to Scottish Government, no. MR 8.1 , St Andrews , 35 pp. http://www.smru.st-andrews.ac.uk/documents/scotgov/MR8-1_ADD_mitigation_VF2.pdf

showed no statistically significant change during operation and non-operation of SeaGen; however, this was likely to be partly due to high inter-individual variation in transit rates. Further investigation of the effect of operation and non-operation on seals that transited the Narrows frequently showed that they did transit less during operation. The biological significance of this is unclear and the study provides no information to assess the possible cumulative effects of multiple devices.

In 2006 (pre- turbine installation), the majority of the transits occurred in the middle of the channel, in 2008 (during turbine installation), the peak in locations occurred on the east side of the channel whereas in 2010 (operational) there was a distinct bimodal distribution with peaks in transits at approximately 250m either side of the turbine location. The variability also made 52% of the comparisons between individual distributions appear significant at the 5% level. However, there was a great deal of variation between the individuals within each year, and the grouped test shows no significant difference between 2006 and 2010 ($p>0.1$)[23].

A series of acoustic playbacks of tidal turbine sounds were carried out as part of the NERC funded RESPONSE project. A programme of land based visual observations of harbour seal activity during signal playbacks (simulated turbine signal based on SeaGen) plus equivalent control signals were made in a narrow, tidally energetic channel on the west coast of Scotland (Kyle Rhea: 57°14'8.10"N, 5°39'15.25"W). Furthermore, the behaviour of ten individual seals was monitored through swimming tracks of high resolution UHF/GPS telemetry tagged seals were collected in conjunction with the playback trials. Results of this study showed that there was no significant difference in the numbers of seals sighted within the channel between playback and silent control periods. Further analyses of the individual responses of the tagged seals are currently ongoing and are due for reporting late 2015.

Mitigation

For tidal turbines, the most effective mitigation for reducing collision risk would be to consider this risk at an early turbine design stage and include engineering mitigation measures through early design modifications (e.g. rotor speed reductions). Work is currently being carried out at SMRU to assess the physical damage inflicted upon a seal when struck by a turbine blade in a series of collision impact tests; this was carried out on seal carcasses using a simulated turbine blade attached to the keel of a jet drive boat, driven over the carcasses at known speeds (adjusted displacement speeds varied from 2.07 to 5.67 ms^{-1}). Post-trial radiographs of each seal showed no discernible evidence of skeletal damage; cranial, abdominal and pelvic bones remained intact. Carcasses were necropsied and again no indications of damage to visceral organs were apparent. These results suggest that collisions with the tips of tidal turbines at these speeds are unlikely to produce serious or fatal injuries in grey seals.

In terms of operational mitigation, the only mitigation method that has proven effective for tidal turbines at this stage is the shutdown protocol at Strangford Lough; this requires observers to monitor the outputs of a series of active sonar systems on the turbine and effect an automated shutdown if a target thought to be a marine mammal approaches within a pre-defined mitigation zone. However, this is clearly effort intensive and expensive and therefore not a viable option; automated sonar detection systems are currently being developed and may prove an effective alternative in the future.

Alternative operational mitigation measures that have the potential to reduce the risk of collisions include the use of ADDs to deter seals from approaching turbines. However, given that behavioural responses by animals are likely to be highly context specific and will depend on factors such as age class, motivation of the animal to remain in the area, and prior exposure history, it is perhaps not surprising that reports of the effectiveness of ADDs are mixed. The use of ADDs was summarised for SCOS 2013.

10. What progress is being made in understanding how seals behave around tidal turbine devices, including diving behaviour and about what might be an appropriate avoidance rate to be applied in collision risk modelling?	MS Q11; Defra Q11

See question 9 above.

Further, as part of the Scottish Government project "Scottish Government Demonstration Strategy: Trialling methods for tracking the fine scale underwater movements of marine mammals in areas of marine renewable energy development (Project USA/010/14)", field trials to test a seabed mounted platform equipped with high frequency multibeam imaging sonars that aim provide high-resolution 3D movements of seals, are due to take place in August 2015. Initial results are due for reporting early 2016. This will provide a means of understanding how seals behave around tidal turbine devices once operational turbines are available, currently expected to be in early 2016.

Seals and salmon netting stations

11. What is the current state of knowledge of interactions between seals and salmon netting stations and possible mitigation measures and what are the priority areas for research in terms of practical non-lethal options?	MS Q12; Defra Q12

SMRU have tested the use of acoustic deterrent devices at salmon bagnet stations with evidence that they are effective under some conditions. In addition net modifications are potentially an effective means of limiting seal predation, primarily by preventing whole fish being removed from the fish court, and further modifications seem promising in terms of reducing predation in the outer parts of the net still further.

Seals reduce the profitability of bag net fisheries by damaging fish that have been caught in the bagnets and by removing whole fish from nets, so that catch per unit effort is reduced. Furthermore, it seems likely that overall seal predation on salmon may be increased over natural background rates due to the availability of salmon in bagnets, notwithstanding the fact that such predation may involve only a small number of individual seals. We have not yet tried to quantify the likely or potential scale of this netting-associated predation on wild salmon stocks.

Seal depredation at salmon bagnets has been studied extensively in Sweden and Finland as well as Scotland. Seal depredation and damage rates to Baltic salmon bagnets has been quantified[25,26,27] while the impacts of seal depredation at bagnets on Baltic salmon stocks has also been addressed[28]. Trials in the Baltic have shown that, as in Scotland, 'rogue' or specialist seals (all males) were responsible for most damage[29]. Limiting the effects of seal damage to Baltic bagnets has been addressed in four ways. Firstly, some success has been achieved by the use of acoustic deterrence[30]

[25] Fjälling, A. (2005). The estimation of hidden seal-inflicted losses in the Baltic Sea set-trap salmon fisheries. ICES Journal of Marine Science, 62, 1630–1635.
[26] Lunneryd, S. G., & Westerberg, H. (1997). By-catch of, and gear danages by, grey seal *(Halichoerus grypus)* in Swedish waters. ICES paper No. CM 1997/Q:11
[27] Suuronen, P., Siira, A., Kauppinen, T., Riikonen, R., Lehtonen, E., & Harjunpää, H. (2006). Reduction of seal-induced catch and gear damage by modification of trap-net design: Design principles for a seal-safe trap-net. Fisheries Research, 79(1-2), 129–138.
[28] Kauppinen, T., Siira, A., & Suuronen, P. (2005). Temporal and regional patterns in seal-induced catch and gear damage in the coastal trap-net fishery in the northern Baltic Sea: effect of netting material on damage. Fisheries Research, 73(1-2), 99–109.
[29] Königson, S., Fjälling, A., Berglind, M., & Lunneryd, S. (2013). Male gray seals specialize in raiding salmon traps. Fisheries Research, 148, 117–123.
[30] Fjalling, A., Wahlberg, M., & Westerberg, H. (2006). Acoustic harassment devices reduce seal interaction in the Baltic salmon-trap, net

as in Scotland[31]. Modifications to seal bag net design in the Baltic to limit seal damage have been addressed: by changing the netting materials used[27,28] by investigating ways to limit seal access to the fish court[27,32,33] and by redesigning the fish court as a twin walled cylinder that can be raised to the surface on inflatable pontoons[34] while the latter has also been modified to allow depredating seals to be caught and removed[35]. Work by SMRU in Scotland has augmented or developed several of these initiatives from the Baltic.

Controlled trials of a Lofitech ADD during 2009 and 2010 found that although the ADD did not completely exclude seals, the ADD was an effective seal deterrent at one site with significantly fewer seals sighted at the net and significantly more salmon landed per unit effort during ADD 'on' treatments compared to 'off'. Subsequent fisher-led trials, which did not involve formal control periods, initially seemed to be effective with apparently low seal damage rates during 2011 and 2012. However, in 2013 the number of reported seal sightings increased and seal damaged fish were landed during ADD 'on' periods, leading to the suggestion of a resurgent seal problem, although CPUE was still higher when the ADD was 'on' than during 'off' periods.

In 2014 an observer accompanied the fishery over the six weeks of the fishery. As in the preceding three years there were no formal controls, with 'off' periods being determined by operational factors or fisher decisions. CPUE remained considerably higher in the 'on' periods, while seal presence was also lower during 'on' periods compared with 'off' periods. Nevertheless at least two identified male grey seals were seen at the net repeatedly while the device was 'on', and were also observed with salmon in their mouths. An Airmar device was deployed at the end of the season, but only for a few days, during which salmon CPUE increased and seal presence decreased. This device is being trialled again in 2015, and was also the subject of a controlled trial at a second site.

Trials at the second site in 2013 and 2014 using an Airmar ADD have not so far shown any significant difference in salmon CPUE between ADD 'on' and 'off' periods. Although this is perhaps not surprising given the relatively low level of seal activity at this site during ADD 'off' periods. Nevertheless, fewer seals were observed near the nets when the ADD was on, a result also supported by underwater video recordings inside the net, which demonstrated a large reduction in the number of instances of a seal entering the net when the ADD was on. Damaged fish rates were also lower when the ADD was on.

A review of the use of ADDs in controlling seal damage to capture fisheries more widely was also supplied to the Scottish Government in 2014[36].

Various modifications to salmon bagnet design have also been tested at both the experimental sites in the Moray Firth, including the addition of steel bars to the entrance to the fish court, and changes to the width of the entrance. Adding steel bars appears to prevent seal entering the fish court to remove whole fish, resulting in a much higher catch per unit effort. But this advantage is partly offset by the fact that the steel bars also inhibit the entrance of the fish, which are then more vulnerable to being attacked just outside the fish court. Evidence from 2014 indicated that a wider fish court entrance (190mm) could still keep seals out while significantly reducing the time taken for fish to enter the fish court, increasing CPUE again by reducing seal depredation immediately outside the fish court.

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fishery. ICES Journal of Marine Science, 63(9), 1751–1758.

[31] R.N. Harris, C.M. Harris, C.D. Duck & Boyd, I.L. (2014). The effectiveness of a seal scarer at a wild salmon net fishery. ICES Journal of Marine Science 71 (7) 1913-1920.

[32] Björnstad, G. (2014). Obstacles to prevent grey seals *(Halichoerus grypus)* from entering static fishing gear. MSc, University of Lund, Campus Helsingborg.

[33] Westerberg, H., & Stenstrom, J. (1997). Towards an efficient seal protection of salmon trap nets ICES Paper No. CM 1997/Q:12.

[34] Hemmingsson, M., Fjälling, A., & Lunneryd, S.-G. (2008). The pontoon trap: Description and function of a seal-safe trap-net. *Fisheries Research*, *93*(3), 357–359.

[35] Lehtonen, E., & Suuronen, P. (2010). Live-capture of grey seals in a modified salmon trap. *Fisheries Research*, *102*(1-2), 214–216.

[36] Coram, A., Gordon, J., Thompson, D. & Northridge, S. (2014). Evaluating and assessing the relative effectiveness of non-lethal measures, including Acoustic Deterrent Devices, on marine mammals. Report to Scottish Government, Sea Mammal Research Unit, University of St Andrews, St Andrews.

Overall we can say that salmon depredation at bag nets has a significant economic impact on these small scale fishing operations, and also increases overall mortality on wild salmon stocks. ADDs are effective at reducing seal presence at or around nets, and have been shown in one site to maintain an increased salmon CPUE over several years. They are not 100% effective and some seals appear willing or able to ignore them. They also carry a risk of damaging hearing in marine mammals and displacing non-target species such as dolphins and porpoises (see Q15). Furthermore, they require vigilance and technical support during deployment to maintain them in good working order, which appears difficult at some sites.

Modifications to the nets themselves are potentially an effective means of limiting seal predation at nets, primarily by preventing whole fish being removed from the fish court, and further modifications seem promising in terms of reducing predation in the outer parts of the net still further. Optimising the fish court entrance and net design to hasten the entry of the fish while excluding seals seems like a promising approach. Further work in this area, including trials of the pontoon net developed in the Baltic or similar double skin nets to limit seal damage to captured fish held in the fish court would be a useful approach.

Coram et al., 2014[36] discussed the potential use of Conditioned Taste Aversion (CTA). This is a process by which an animal "learns" to avoid food which has made it ill in the past. Once an animal has been made ill by eating poisoned or tainted food it will usually exhibit disgust and may vomit when it encounters that food again. Typically, this aversion is learnt after a single trial and persists for months or years and the method has been used effectively on a range of predators. A particularly attractive aspect of CTA is that it is not associated with a particular location and the aversion applies to the food type wherever and in whatever context it is encountered.
Trials with captive sea lions showed that CTA can be elicited and can cause seals to avoid eating highly preferred prey items even when no alternative food was provided. Trials with fur seals at salmon farms in Tasmania were also apparently successful but not followed up. To date there have been no trials of CTA on grey or harbour seals.

Reactions of captive grey and harbour seals to low voltage electric fields in sea water has been tested at SMRU[37]. Seals could be prevented from entering a feeding station by the application of short (0.2 - 1.0 msec duration), low voltage (24 to 36V) DC pulses to electrodes either side of a 30cm wide entrance. It seems likely that such fields could be used to prevent seals entering the inner courts of bag nets. However, further research would be required to assess the effect of the field on fish passage into the net.

[37] Milne, R., Lines, J., Moss S. & Thompson D. (2013) Behavioural responses of seals to pulsed, low-voltage electric fields in sea water (preliminary tests). Report commissioned by SARF and produced by the Sea Mammal Research Unit, University of St Andrews, St Andrews http://www.sarf.org.uk/cms-assets/documents/124766-207045.sarf071-revised.pdf

Seals and river fisheries

12. What is the current state of knowledge about potential non-lethal options for deterring seals from entering and/or transiting up river systems?	MS Q13 ; Defra Q13

ADDs have been successfully trialled to limit the passage of seals up salmon rivers but there are concerns related to how they are deployed and maintained.

At present we are aware that ADDs have been deployed by several District Salmon Fishery Boards to try to prevent seals from swimming up salmon rivers. Details are provided in annual research reports by SMRU to Marine Scotland, (the most recent of which can be found at smru.st-andrews.ac.uk/documents/scotgov/SSI_seals_and_salmon_VF1.pdf (See Annex 1)). Collaboration with Dee DSFB personnel, who have installed two Lofitech seal scarers in the river Dee, has shown that seals are still able to swim upriver past these devices. This work has highlighted the difficulties in using ADDs in salmon rivers, in particular maintaining sound head position / orientation and delivering an adequate power supply, which are possibly the two most difficult and critical issues. A very similar series of events occurred in the North Esk; seals were found to have swum upriver past a Lofitech device, which was found to have power supply problems and a misplaced sound head on the river bed.

Once seals have learned to bypass ADDs within rivers, other measures need to be adopted. One approach has been to attempt to 'sweep' seals back to the sea using a boat fitted with an ADD, a method that proved successful in the Kyle of Sutherland. SMRU have provided the loan of an ADD to the Dee DSFB for this purpose and it has been used successfully by the fishery board to return seals downriver.

Other than lethal removal, and acoustic sweeping, it would be possible though difficult to capture seals alive and return them to the sea, in the hope that they would not then return past a properly functioning ADD (or changing to an ADD that utilises different sound characteristics), though there are obvious risks to this approach. A better understanding of how seals are utilising rivers and a method to detect their presence might enable a triggered response to their presence, using ADDs or even electric field gradients to prevent them moving up river. It may be worth noting that considerable research has been devoted to trying to deter pinnipeds from predating wild salmonids in several western US rivers, with mixed success. In such instances, which mainly involve California Sea Lions, the use of ADDs has not been effective[38], but physical exclusion and trapping and removing animals have been more successful.

[38] Stansell, R.J., Gibbons, K.M. & Nagy, W. T. (2010). Evaluation of pinniped predation on adult salmonids and other fish in the Bonneville Dam tailrace, 2008-2010. US Army Corps of Engineers, Bonneville Lock and Dam, Cascade Locks, OR. October 14, 2010.

Seals and fish farms

13. What is the current state of knowledge of interactions between seals and fin fish farms and possible mitigation measures and what are the priority areas for research in terms of practical non-lethal options?	MS Q14; Defra Q14

No new work has been completed to understand interactions between seals and fin fish farms since the last SCOS report, where recent trials of a 'startle response' ADD were reported.

A review evaluating the effectiveness of acoustic deterrents and other-non lethal measures to mitigate marine mammal conflicts especially with fish farms was published by Marine Scotland in late 2014[36]. Suggestions for further research into resolving conflicts between seals and fish farms included:

1) Improving baseline data on factors associated with greatest levels of seal damage
2) Experimental or analytical approaches to quantify efficacy of existing mitigation measures
3) Exploration of factors that may or may not make anti-predator nets effective in Scotland
4) Examination of unintended environmental consequences of the use of Acoustic Deterrent Devices on
    a. the hearing of target species (seals)
    b. the disturbance and consequent ecological consequences for non-target species notably harbour porpoises
5) Further work on electric field deterrents and / or conditioned taste aversion

A startle response ADD device[39], marketed by Genuswave[40], has been found to significantly decrease seal predation on a farm without habituation effects over a one year period. It has also been used successfully to reduce acute seal attacks at several farms on the West coast and in Orkney and Shetland. Trials of the commonly used Terecos ADD also suggested little or no effect on the detection rate of porpoise vocalisations[41]. Other flexible systems with signals tailored to particular target species are being developed, for example the FaunaGuard system, developed by Van Oord and SEAMARCO has been tested on a wide range of species including fish, turtles and porpoises.

Work funded by SARF to help understand how seals are able to take salmon from nets is still ongoing. Results are not available yet but will be presented to SCOS 2016. This work is now mainly focused on examining seal behaviour in relation to nets and quantifying forces they are able to generate for a food reward in SMRU's captive facility.

[39] Gotz, T. & Janik, V.M. (2014) Target-specific acoustic predator deterrence in the marine environment. Animal Conservation, 18(1), 102-111.
[40] Note: The University of St Andrews has a commercial interest in this device
[41] Northridge, S., Coram, A. & Gordon, J. (2013). Investigations on seal depredation at Scottish fish farms. Edinburgh: Scottish Government. 79pp.

Use of acoustic deterrents

14. What is the latest understanding of the relative effectiveness of existing models of acoustic deterrents for preventing seal predation at fisheries or fish farms (including locations with or without a high level of cetacean presence) and for avoiding the possibility of seal collisions with tidal energy devices?	MS Q15 ; Defra Q15

To our knowledge, no research specifically to compare the relative effectiveness of different acoustic deterrent devices has been carried out.

Apart from the trial conducted by Janik and Gotz[42] (2013: reported at SCOS 2014), which showed a reduction in seal depredation after a 'startle response' ADD was deployed at three farm sites, we are not aware of any independent studies on the effectiveness of ADDs at farm sites.

SMRU have shown that the Lofitech device does increase salmon CPUE, reduce the proportion of damaged fish, and also reduces the frequency of seal sightings, at salmon bag net stations. Preliminary data also suggest that the Airmar device can also reduce the frequency of seal visits to bag nets as well as the proportion of damaged fish in the nets. Previous work has also shown the effectiveness of the Lofitech in reducing seal ingress up salmon rivers.

The Lofitech device has also been tested as a potential longer range deterrent to act as part of a mitigation method for avoiding damage to seals from pile driving and other potentially harmful anthropogenic activity. In a series of at-sea behavioural response trials with telemetry tagged harbour seals, the Lofitech ADD caused avoidance behaviour at ranges up to 1km[43].

It is important to note throughout, however, that the use of ADDs bears the risk of damaging hearing in seals and other marine mammals[36]. They can also deter cetaceans from an area. This is to be considered especially when using several ADDs for example on bag nets within a small area. Harbour seals can experience compromised hearing when spending as little as 3 min within 10 m of a high-powered ADD. This effect is reversible, but will have a more permanent effect on hearing if this threshold is exceeded repeatedly. Effects on cetaceans occur more easily. For example, temporary effects on porpoise hearing can occur at ranges of 89 to 345 m when spending 3 min within that range. Permanent threshold shift* is predicted to occur in porpoises when spending between 4 and 21 hours within 76 to 345 m of an ADD (depending on whether Lofitech or Airmar is used and on the selected duty cycle). Effects on killer whales occur at even lower exposures. However, effects in cetaceans are likely mediated by a deterrence effect on some species. This effect has been most dramatic when using a Lofitech device with harbour porpoises avoiding an area of at least 7.5 km around the device. This kind of habitat avoidance can be problematic if devices were used around Scotland on a large scale. The presence of cetaceans will not alter the efficacy of the device, though clearly where disturbance of cetaceans is a concern, then one of the 'cetacean friendly' ADDs may be a preferred management option (see comments on ADD effects on cetaceans in Question 15 below).

There are as yet no direct tests of the effectiveness of ADDs at preventing seal collisions with tidal turbines. The potential use of ADDs at tidal turbine sites and relevant knowledge gaps was discussed by Coram *et al*.[36]

[42]Götz, T. & Janik, V.M. (2013). Acoustic deterrent devices to prevent pinniped depredation: Efficiency, conservation concerns and possible solutions. Marine Ecology Progress Series 492, 285–302.

[43]Gordon, J., Blight, C., Bryant, E., & Thompson, D. (2015). Tests of acoustic signals for aversive sound mitigation with harbour seals. Sea Mammal Research Unit, University of St Andrews, Report to Scottish Government, no. MR 8.1, St Andrews, 35pp

* a permanent reduction in hearing sensitivity at particular frequencies

15. Is it possible to provide specific recommendations about which models of acoustic deterrents might be more effective in the situations outline above?	MS Q16; Defra Q16

SCOS does not recommend any specific devices as there has not been any experimental work to test their respective efficacy at fisheries or fish farms.

SCOS recommends that a standardised testing protocol should be developed to assess the relative effectiveness of different ADDs. Protocols should address deterrence of seals and non-target species. To avoid cetacean disturbance, certain devices have been designed to have a lower impact on cetaceans and could be used where there are concerns about disturbance to porpoises as has been shown for high frequency, high amplitude devices.

In a recent report to Marine Scotland, recommendations for research on ADDs in relation to disturbance and its ecological consequences for porpoises were identified. These could involve controlled experimental exposure of porpoises to the full suite of ADDs currently available to be able to make robust comparisons regarding disturbance, while also looking at porpoise densities at sites with and without active ADDs.

Bycatch

16. What are the latest annual estimates of seal mortality from bycatch in SW Britain, i.e. the West of England and Wales management unit?	NRW Q3

Estimates of seal bycatch are reported annually to the European Commission in the UK "Report on the implementation of Council Regulation (EC) No 812/2004", which is produced annually by SMRU under contract to Defra and the Scottish Government. Seal bycatch estimates for static net fisheries are included in the Annex to that report[44] by ICES subdivision, but should be treated with caution as several caveats apply, and point estimates only are given.

Roughly 340 (grey) seals might be expected to have been killed in static net fishing operations by UK vessels in the Irish Sea, Celtic Sea, Bristol Channel and Western English Channel in 2014 (total from rows highlighted in Table 7. Seal bycatch is known to occur in several other fisheries operated by countries in the same area, but robust and current estimates of total seal bycatch are unavailable[45,46].

[44] Northridge, S., Kingston, A. & Thomas, L. (2015). Annual report on the implementation of Council Regulation (EC) No 812/2004 during 2014. Available at: http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&ProjectID=18535

[45] Cosgrove, R., Cronin, M., Reid, D., Gosch, M., Sheridan, M., Chopin N. & Jessopp, M. (2013). Seal depredation and bycatch in set net fisheries in Irish waters. Fisheries Resource Series Vol. 10 (2013). ISSN 1649-5357 ISBN 1-903412-48-X ISBN 978-1-903412-48-0

[46] Berrow S.D., O'Neill, M. & Brogan, D. (1998). Discarding practices and marine mammal by-catch in the Celtic sea herring fishery. Biology and Environment Proceedings of the Royal Irish Academy, 98B, 1-8.

*Table 7. **Estimates of seal bycatch by vessels size class and subdivision** (Reproduced Table A2.10 from Northridge et al. 2015[44]) for UK static net (gillnet) fisheries in 2014. Subdivisions of relevance to the question (SW Britain) are shaded.*

Subdivision[47]	Vessel size		Totals
	Under 12m sector	**Over 12m sector**	
IVA	0	24	24
IVB	6	2	8
IVC	32	0	32
VIB	0	7	7
VIIA Irish Sea	4	0	4
VIID English Channel	82	0	82
VIIE English Channel	114	10	124
VIIF Bristol Channel	94	3	97
VIIG Celtic Sea	1	9	10
VIIH Celtic Sea	1	12	13
VIII	0	5	5
VIIJ Celtic Sea	0	11	11
Totals	**335**	**82**	**417**

Marine Strategy Framework Directive

17. Building on the work SCOS has already undertaken on Marine Strategy Framework Directive (MSFD) indicators can you provide the latest available data for the UK and where possible other countries in the region to feed into the IA2017 by August 2016 (initial draft due Nov 2015): M3 – seal abundance and distribution; M5 – Grey seal pup production?	Defra Q17

The latest available data from the UK were used to perform a preliminary assessment of MSFD indicators M-3 and M-5 describing changes in grey seal and harbour seal population abundance and distribution and the results are given in SCOS-BP 15/09. It was necessary to arbitrarily subdivide UK Assessment Units into smaller subareas to calculate distribution metrics for harbour seals. The distribution metrics showed no catastrophic contraction or shift in distribution has occurred for either grey or harbour seals in any Assessment Unit.

Simple models were fitted to count data and 95% confidence intervals of the specified metrics were calculated from bootstrap resamples of the data to provide estimates of the uncertainty surrounding each metric. In some cases, wide confidence intervals that include target values indicate that

[47]See http://ices.dk/marine-data/maps/Pages/default.aspx for map of ICES Subdivisions

confidence in the assessment is low. Targets that use both rolling and stationary baselines are presented and give added information about (nonlinear) population trends. The distribution metrics showed no catastrophic contraction or shift in distribution has occurred for either grey or harbour seals in any Assessment Unit. These simple metrics – with added information about uncertainty and number of surveys – should prove applicable to other European datasets, as well as being understandable and useful to policy-makers. Further details of the targets and results from each Assessment Unit are presented in SCOS-BP-15/10.

Disturbance

18. What recent research is there on the impacts to seals from visual disturbance (anthropogenic activity) and the recommended distances to maintain away from seals to avoid disturbance? Apart from underwater noise, what are the other main sources of disturbance for grey seals that are cause for concern and is there any evidence that these have adverse effects on grey seal populations? If there is an impact, are there new approaches (mitigation) to reducing the impact of such disturbance on seals?	Defra

SMRU carried out a regular series of controlled disturbance trials at harbour seal haulout sites in the Sound of Islay. Details of this work are presented in SCOS-BP 15/10. Repeated disturbance by boats did not cause seals to abandon haulout sites and seals either hauled out again or went on typical foraging trips after disturbance events. Similar studies in Danish waters produced broadly similar results. Grey seals at several haulout and breeding sites have habituated to close approaches by pedestrians, low flying fighter jets and loud noises from nearby bombing ranges.

Controlled disturbance trials involved direct approaches by boat to groups of hauled out seals. In each case the seals were chased into the water. Trials were repeated at roughly 3 day intervals over the summer. This represented quite severe disturbance to those sites. Behaviour of individual seals was recorded using GPS/GSM telemetry tags and haulout patterns were documented by time lapse photography of important haul out sites. Results suggest that even such intense boat based disturbance did not cause seals to reduce their use of the disturbed sites. Transition rates to other sites were no higher than after undisturbed haulout events and numbers using the disturbed sites quickly returned to pre-disturbance numbers.

Such observations suggest that boat based disturbance may not be a major problem for harbour seal haulout sites. The study did not produce information on the level of boat activity needed to cause disturbance as each approach continued until all seals had been flushed off the haul-out site. The study did not address the effects of terrestrial disturbance which may cause a different and potentially more profound response.

A similar study on harbour seals at Anholt, Denmark, investigated sporadic disturbances of hauled out from pedestrians, boats, low-flying aeroplanes and grey seals. Pedestrian disturbances caused significantly longer-lasting trips at sea than undisturbed trips. Disturbed and undisturbed seals used the same areas at sea, suggesting that these represent normal foraging areas. Seals consistently returned to the same haul-out site, even when subjected to repeated disturbances[48].

We are not aware of any similar studies of the direct effects of disturbance to grey seals. Grey seals are susceptible to disturbance on haulout sites: human activity on land and close approaches by boats,

[48] Andersen, S. M., Teilmann, J., Dietz, R., Schmidt, N. M., & Miller, L. A. (2012). Behavioural responses of harbour seals to human-induced disturbances. Aquatic Conservation - Marine and Freshwater Ecosystems, 22, 113-121.

particularly kayaks are known to cause seals to move off haul-outs in some cases. The consequent behaviour of disturbed grey seals has not been studied.

In several locations grey seals have become relatively habituated to close approaches by boats and boat based grey seal watching industries have developed. The largest grey seal haulout sites in the UK are within the RAF bombing range at Donna Nook and grey seals continue to use these sites during extremely loud bombing and gunnery practice on the range. This same site is also one of the largest grey seal breeding sites in the UK. Again, the breeding grey seals at Donna Nook appear to be habituated to and are not affected by the near-by bombing activity.

Seals using this and other mainland breeding sites in eastern England have also been shown to habituate to relatively intense human disturbance on the colonies, e.g. at Donna Nook large numbers of tourists observe breeding grey seals at close range. However, at other relatively undisturbed sites grey seals are apparently susceptible to disturbance during the breeding season and on non-breeding haulout sites. SCOS are unaware of any new approaches to reducing or mitigating the effects of such disturbance.

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, Marine Coastal and Access Act 2009 and the Marine (Scotland) Act 2010.

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

Dr M. Hammill (Chair)	Maurice Lamontagne Institute, Canada;
Dr A. Hall	Sea Mammal Research Unit, University of St Andrews;
Dr F. Daunt	Centre for Ecology and Hydrology, Edinburgh;
Dr J. Forcada	British Antarctic Survey, Cambridge;
Dr K. Brookes	Marine Scotland, Science, Aberdeen;
Dr J. Teilmann	Aarhus University, Denmark;
Dr C. Lynam	Centre for Environment Fisheries and Aquaculture Science, Lowestoft;
Professor P. Thompson	Institute of Biological and Environmental Sciences, University of Aberdeen;
Dr D. Mason (Secretary)	Natural Environment Research Council, Swindon Office.

ANNEX II

Dear Mrs Mason

MARINE (SCOTLAND) ACT 2010 (CONSEQUENTIAL PROVISIONS) ORDER 2010: ANNUAL REVIEW OF MANAGEMENT ADVICE

Thank you for your letter of 12 March concerning the next meeting of the Special Committee on Seals on 3 and 4 September 2015 and asking whether the Scottish Government has any specific questions on which it would welcome the Committee's scientific advice.

It would be very helpful if the Committee could provide a general update on seal populations and respond to some more specific questions on particular issues as set out below.

We have, as usual, structured our request for advice from the Committee in two broad categories. The first comprises a shorter than usual list of standard questions seeking a update on some of the key information regularly provided by the Committee in previous years:-

1. **What are the latest estimates of the number of seals in Scottish waters?**
2. **What is the latest information about the population structure, including survival and age structure, of grey and common/harbour seals in European and Scottish waters? Is there any new evidence of populations or sub-populations specific to local areas?**

Specific questions about improving seal management:-

Seal Population Trends

3. How many year's data and what scale of change in numbers of seals counted are required to be able to say that a population is showing a significant upward or downward trend?

Harbour/Common Seal Population

4. Is the existing harbour seal decline recorded in several local areas around Scotland continuing or not and what is the position in other areas?
5. 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 to those already covered by seal conservation areas or should it consider removing existing conservation measures in any areas?
6. What is the latest understanding of the causes of the recent decline in common/harbour seals?

Unusual Seal Mortalities

7. What is the latest understanding of the causes of the recent unusual seal mortalities, including seal predation, and of their potential impact on wider populations of both grey and harbour seals?

Seal Licensing and PBRs

8. What, if any, changes are suggested in the Permitted/Potential Biological Removals (PBRs) for use in relation to the seal licence system?

9. The Annex attached sets out a number of specific questions in relation to the use of PBR beyond the usual seal licensing application.

Seals and Marine Renewables

10. What is the current state of knowledge of interactions actual or potential between seals and marine renewable devices and possible mitigation measures?

11. What progress is being made in understanding how seals behave around tidal turbine devices, including diving behaviour, and about what might be an appropriate avoidance rate to be applied in collision risk modelling?

Seals and Salmon Netting Stations

12. What is the current state of knowledge of interactions between seals and salmon netting stations and possible mitigation measures and what are the priority areas for research in terms of practical non-lethal options?

Seals and River Fisheries

13. What is the current state of knowledge about potential non-lethal options for deterring seals from entering and/or transiting up river systems?

Seals and Fish Farms

14. What is the current state of knowledge of interactions between seals and fin fish farms and possible mitigation measures and what are the priority areas for research in terms of practical non-lethal options?

Use of Acoustic Deterrents

15. What is the latest understanding of the relative effectiveness of existing models of acoustic deterrents for preventing seal predation at fisheries or fish farms (including locations with or without a high level of cetacean presence) and for avoiding the possibility of seal collisions with tidal energy devices?

16. Is it possible to provide specific recommendations about which models of acoustic deterrents might be more effective in the situations outlined above?

As in previous years, it is our intention to publish a link to the advice provided by the Committee on the Scottish Government web-site. We will liaise about the timing of that in due course.

I also enclose the information requested on licences issued by the Scottish Government during 2014 under The Marine (Scotland) Act 2010. This information can be found on the Scottish Government web-site through the following link (see Tables 1, 2a and 2b):-

http://www.gov.scot/Topics/marine/Licensing/SealLicensing/2011/2014

I am copying this letter to Defra colleagues for information.

Yours sincerely

IAN WALKER

Marine Environment

Annex

Scottish Government questions and additional information relating to request for advice on PBR

<u>Request for advice</u>

We would like to understand SCOS's opinion on the issues detailed in the attached supporting information and whether they give rise to sufficient concern regarding the use of PBR, such that they would recommend an alternative framework that might make fuller use of best available evidence?

Marine Scotland currently uses PBR for both determining numbers of seals that can be licensed to be shot and for marine renewable assessments. Taking account of the points raised in the supporting information (particularly section A), does SCOS consider that PBR is suitable for both of these applications, and if not, do they consider that a single assessment framework accounting for both sets of licensable activities is required?

Given that the development and application of any framework would require close collaboration between policy makers and scientists, would SCOS consider it had a role in working with policy makers to deliver an alternative framework? If so, do SCOS have any recommendations at this stage on how best to make that happen?

In considering your response, please note that Marine Scotland do not request a definitive and final view from SCOS on all the issues raised. We anticipate that doing so would take considerable time and may require new work. A response that provides indicative consideration with respect to the overall purpose and aim of this request would be most useful at this stage.

<u>Supporting information</u>

<u>Background and purpose</u>

The Scottish Government uses advice based on PBR to determine the annual numbers of grey and harbour seals that can be removed from the populations through shooting. There are a number of other potential pressures upon seal populations that the Scottish Government wishes to assess with respect to regulation of the population consequences and the appropriate magnitude of effects. For example, the potential impacts of marine renewable energy, and of port developments also need to be appropriately regulated. The overall purpose of this question is to initiate fuller consideration of what constitutes the best available evidence and most suitable techniques for addressing potential impacts to seal populations. The aim is to consider the strengths and weaknesses of PBR and the potential for other approaches.

<u>Issues raised in association with PBR</u>

A list of concerns have been raised on several occasions about the utility of PBR for undertaking cumulative impact assessments. These issues are detailed in sections A to C below and SCOS are invited to comment on these, or any other issues associated with use of PBR. It would be particularly helpful if SCOS were able to provide an indication of whether or not these issues have reasonable foundation, whether anything could be done in the context of applying PBR to address them, and to also consider if they provide good reason to consider alternative approaches to assessment frameworks.

There are three main areas that these issues can be separated into:

A. the concept of using PBR as a framework for both determining annual numbers of shooting licences and for licensing renewable developments,

B. the underlying principles of PBR, and

C. the implementation of PBR as it is currently used by the Scottish Government under the advice of SCOS.

A. Issues relating to the concept of using PBR for licensing purposes:

1. Appropriate consideration of uncertainty associated with effects is a key aspect of any assessment. There are clearly very different levels of certainty associated with shooting seals and modelling collision risk. An issue is how best to account for these varying levels of uncertainty, and whether or not PBR provides the most useful threshold setting tool in light of the uncertainties associated with effects. In particular, PBR does not allow for nuances such as a probability of death.
2. PBR is recalculated annually, based on latest population estimates. While this is appropriate for use in the iterative management of effects for issuing licences to shoot seals, it may be less suitable for a robust assessment of the effects to a population over the 20-25 year operational life span of a renewable energy development for the purposes of licensing decisions. In these situations, Marine Scotland must carry out an assessment that is competent of considering impacts throughout the lifetime of the project under scrutiny, and any consent granted would be difficult to revoke at a later stage. Is it reasonable to use PBR to assess effects occurring over forthcoming decades when it was originally designed for assessing effects annually?
3. PBR assumes that the effect upon a population is via adult survival rates and that maximum productivity is achieved when the carrying capacity is reduced below a certain level. In practice, certain effects (e.g. noise) may impact productivity rates. A recent example is the application of PBR for seabird responses to wind farms, which included effects of displacement. Does PBR have any role to play in assessment of non-lethal effects, or in situations where an impact may have both lethal and non-lethal effects?

B. Issues relating to the underlying principles of PBR:

1. The density dependent response assumed in the PBR model assumes that maximum net productivity level (MNPL) is at half of carrying capacity. Whilst the density dependent response function assumed in the PBR model was clearly carefully selected based on sensitivity testing (against the objectives of PBR as stated in Wade 1998) the issue remains that we may reasonably expect species with varying life history traits to respond differently to perturbations and that the PBR model's simplification of density dependent response cannot be as readily tested with empirical data compared to other approaches.
2. The PBR model assumes a fixed carrying capacity over a period of decades (up to 100 years). Populations of species, such as seals, respond to a number of spatio-temporal effects that give rise to legitimate questions about the suitability of regulating effects over local areas based on an assumption that the population can recover to a fixed level that will remain constant over decades. The effects of climate change on prey, or the potential for harbour seal populations to respond to inter-specific competition with grey seals could reasonably be considered as examples of effects that are ultimately changing the capacity of the local environment to support a population at the current level. If it is only other anthropogenic effects that require regulation by society, then the issue arises of whether PBR provides the most suitable model, since we might assume that carrying capacity would change over time.
3. Populations' vital rates can reasonably be expected to respond to perturbations in a stochastic manner. Resource availability varies and species with differing life history traits are more or less likely to be able to respond over specified time periods. Point B1 above raised this issue with respect to the assumed population level at which MNPL occurs, but it may also be an issue with respect to the rate (or range of rates) at which populations may respond to change. As with point B2, is there a concern that embedding a simplification into assessments may give rise to the assessment framework acting as a barrier to progressive improvement in our understanding of how populations respond to perturbation?

4. Productivity, and Rmax values, will similarly be expected to have spatial and temporal variation. This leads to questions about the applicability of published references to local circumstances.
5. The PBR equation uses a denominator of 2. Is this because MNPL is half of carrying capacity? Or because MNPL is based on the proportion of the population that is female? If species have an MNPL that is a different percentage of carrying capacity, or an unbalanced sex ratio, should this ideally be reflected in the denominator? The objectives that PBR are intended to achieve, relate to assumptions about how the level of the starting population compares to carrying capacity, a period of recovery after carrying capacity and a population level that would be achieved with respect to carrying capacity at the end of the recovery period. Each of these factors may differ from the circumstances associated with a cumulative impact assessment (e.g. it may not be considered useful to assume any, or the same, period of recovery). The PBR objectives were specifically developed to meet the statutory requirements of the US Marine Mammals Protection Act. Do SCOS have any comments to make with respect to the usefulness/appropriateness of managing populations to the PBR's objectives under other statutory frameworks?

C. Implementation of PBR as it is currently used by the Scottish Government under the advice of SCOS

1. The minPop value to be used in PBR is the 20th percentile of the most recent population estimate. It is our understanding that the purpose is to add a conservative measure given the uncertainty associated with the size of many marine mammal populations (especially cetaceans). Firstly, is this approach taken by SCOS or are average values used? Secondly, if average values are used, has the sensitivity been tested, and is there a risk that the assumptions of PBR are violated by using the most robust estimates of population size? Thirdly, if we consider that there is a relatively small spread of uncertainty associated with seal population estimates, is there a risk of the assumptions of PBR being violated even when the 20th percentile is used?
2. F values. What criteria do SCOS use to propose new values? How are the criteria classified to arrive at decisions? How consistent is application of these criteria with other users of PBR? Could the decision making associated with F values be made more transparent and objective?

<u>Reference</u>

Wade, P.R., (1998) Calculating limits to the allowable human-caused mortality of cetaceans

Annexes

Questions from Defra

Dear Mrs Mason

CONSERVATION OF SEALS ACT 1970: ANNUAL REVIEW OF MANAGEMENT ADVICE

Thank you for your email letter of 12 March 2015, asking if Defra has any specific questions on which it wishes to receive scientific advice.

With reference to today's Scottish Government response to the same request, the following are the same standard questions seeking a general update on information regularly provided by the Committee in previous years but relating to seals in English waters, rather than Scottish waters on the understanding that each devolved administration would ask similar questions so that a UK wide picture would be provided in the annual SCOS report.

Seal populations in English waters

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

2. What is the latest information about the population structure, including survival and age structure, of grey and common/harbour seals in English waters and is there any new evidence of populations or sub-populations specific to local areas?

Specific questions about improving seal management:- again, for the purposes of consistency, we ask the same questions (bar questions 4 to 6, which are specific to harbour/common seal declines in Scottish waters) as Scottish Government has in its response to the same request. Question 17 is a Defra specific question

Seal Population Trends

3. How many year's data and what scale of change in numbers of seals counted are required to be able to say that a population is showing a significant upward or downward trend?

Unusual Seal Mortalities

7. What is the latest understanding of the causes of the recent unusual seal mortalities, including seal predation, and of their potential impact on wider populations of both grey and harbour seals?

Seal Licensing and PBRs

8. What, if any, changes are suggested in the Permitted/Potential Biological Removals (PBRs) for use in relation to the seal licence system?

9. The Annex attached sets out a number of specific questions in relation to the use of PBR beyond the usual seal licensing application.

Seals and Marine Renewables

10. What is the current state of knowledge of interactions actual or potential between seals and marine renewable devices and possible mitigation measures?

11. What progress is being made in understanding how seals behave around tidal turbine devices, including diving behaviour, and about what might be an appropriate avoidance rate to be applied in collision risk modelling?

Seals and Salmon Netting Stations

12. What is the current state of knowledge of interactions between seals and salmon netting stations and possible mitigation measures and what are the priority areas for research in terms of practical non-lethal options?

Seals and River Fisheries

13. What is the current state of knowledge about potential non-lethal options for deterring seals from entering and/or transiting up river systems?

Seals and Fish Farms

14. What is the current state of knowledge of interactions between seals and fin fish farms and possible mitigation measures and what are the priority areas for research in terms of practical non-lethal options?

Use of Acoustic Deterrents

15. What is the latest understanding of the relative effectiveness of existing models of acoustic deterrents for preventing seal predation at fisheries or fish farms (including locations with or without a high level of cetacean presence) and for avoiding the possibility of seal collisions with tidal energy devices?

16. Is it possible to provide specific recommendations about which models of acoustic deterrents might be more effective in the situations outlined above?

Additional Defra specific question

Marine Strategy Framework Directive (MSFD) indicators

The UK has agreed, under its obligations to the OSPAR Commission, to lead on the delivery of assessments of seal populations for the OSPAR Intermediate Assessment in 2017 (IA2017).

17. Building on the work SCOS has already undertaken on Marine Strategy Framework Directive (MSFD) indicators (under Q51, 2014), can you provide the latest available data for the UK and where possible other countries in the region, to feed into the IA2017 by August 2016 (initial draft due Nov 2015): M3 – Seal abundance and distribution; M5 – Grey seal pup production?

I hope this satisfies your requirements. If you have any queries about this letter please contact me.

Yours sincerely

Simon Liebert

Wildlife Management Policy Officer

Annexes

Questions from Natural Resources, Wales

Dear Mrs Mason

CONSERVATION OF SEALS ACT (1970): ANNUAL REVIEW OF MANAGEMENT ADVICE

Thank you for your email to ask if Natural Resources Wales (NRW) has any specific questions on which it wishes to receive scientific advice.

It would be very helpful if the Committee could provide an update on the following:

1. What are the latest estimates of the number of grey seals in UK and its management units (MU) e.g. the West England and Wales MU?
2. What is the latest information on survival estimates, age-structure, and fecundity of grey seals in European, UK and Welsh waters?
3. What are the latest annual estimates of seal mortality from bycatch in SW Britain i.e. the West England and Wales management unit?
4. What is the latest understanding and how confident are we that ducted propellers are no longer the likely cause of 'corkscrew' injuries in all age classes? What is the scale and distribution of these impacts and the likely population effects?

Many thanks for your consideration, it is very much appreciated.

Sincerely,

Dr Tom Stringell

Senior Marine Mammal Ecologist

ANNEX III

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 Briefing papers should represent a record of work that can be carried forward to future meetings of SCOS.

<u>List of Briefing Papers</u>

15/01 Grey seal pup production in Britain in 2014 – a progress report. Duck, C. and Morris. C.

15/02 Estimating the size of the UK grey seal population between 1984 and 2014. Thomas, L.

15/03 Review of parameters of grey seal pup production model. Russell, D., Duck, C., Morris C. and Thompson, D.

15/04 The status of UK harbour seal populations in 2014, including summer counts of grey seals. Duck, C., Morris C. and Thompson, D.

15/05 Preliminary report on the distribution and abundance of harbour seals (*Phoca vitulina*) during the 2014 breeding season in The Wash. Thompson, D.

15/06 Updating adult female grey seal survival estimates at the Isle of May. Pomeroy, P., Jesus, A., Moss, S., Ramp, C., and Smout S.

15/07 Harbour seal population modelling: The Moray Firth. Smout, S., Caillat, M., Thompson, P., Cordes, L., Mackey, B., Duck, C., Thompson, D. and Matthiopoulos, J.

15/08 Provisional regional PBR values for Scottish seals in 2016. Thompson, D., Morris, C. and Duck, C.

15/09 Report on UK contribution to Marine Strategy Framework Directive seal indicators. Hanson, N. and Hall, A.

Grey seal pup production in Britain in 2014
A progress report

Callan D. Duck and Chris D. Morris

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Abstract

This is a simple progress report and is provided for information only.

In the 2014 grey seal breeding season, SMRU successfully surveyed the 69 main grey seal breeding colonies in Scotland. Grey seal pups born at four colonies in England were ground-counted by staff from the National Trust, Lincolnshire Wildlife Trust and Natural England.

By the end of July, 196 of 335 aerial colony surveys have been counted, including all surveys of colonies in the Inner Hebrides.

Using the standard model run (0.9 for proportion of moulters classified and 31.5 days for mean time to leave), pup production at the Inner Hebrides colonies was estimated to be **4,054**, slightly lower than the 2012 estimate of 4,088.

At the four English North Sea colonies, pup production in 2014 was **6,627** compared with 4,963 in 2012 and 5,539 in 2013. There was a massive increase in the number of pups born at Blakeney Point (2,425 pups born in 2014 compared with 1,560 in 2013) which is now the biggest grey seal breeding colony in England, overtaking Donna Nook (1,799 pups) for the first time.

Introduction

Grey seals breed at traditional colonies, with females frequently returning to the same colony to breed in successive years (Pomeroy et al. 2001). Some females return to breed at the colony at which they were born. Habitual use by grey seals of specific breeding colonies, combined with knowledge of the location of those colonies, provides opportunity for the numbers of pups born at the colonies to be monitored.

While grey seals breed all around the UK coast, most (approximately 85%) breed at colonies in Scotland (Figure 1). Other main breeding colonies are along the east coast of England, in south-west England and in Wales. Most colonies in Scotland and east England are on remote coasts or remote off-lying islands. Breeding colonies in south-west England and in Wales are either at the foot of steep cliffs or in caves and are therefore extremely difficult to monitor.

Until 2010, SMRU conducted annual aerial surveys of the major grey seal breeding colonies in Scotland to determine the number of pups born (Duck & Mackey 2005, Duck & Morris 2013). Reductions in funding, combined with increasing aerial survey costs, have resulted in SMRU moving from monitoring the main Scottish grey seal breeding colonies annually to a biennial survey regime. The first year with no survey was 2011. The number of pups born at colonies along the east coast of England is monitored annually by counting on the ground 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 Natural England count pups born at Horsey/Winterton, on the east Norfolk coast. Scottish Natural Heritage (SNH) staff ground counted grey seal pups born in Shetland and on South Ronaldsay in Orkney.

In 2012, SMRU replaced the film-based large-format Linhof AeroTechnika system used since 1985 with a new digital camera system, funded by NERC. Increased numbers of images acquired during a full aerial survey season (approx. 30,000 digital images compared with 6,000 frames) resulted in a delay in completing estimating pup production at all 60 Scottish colonies.

This Briefing Paper reports on progress with the 2014 survey of the grey seal breeding colonies in Scotland. It should be considered in conjunction with the Briefing Paper investigating the effect of altering the Time-to-Leave parameter (SCOS BP-15/03).

Materials and Methods

SMRU aerially surveys the main breeding colonies around Scotland. Pups born at colonies in England are counted from the ground annually by staff from the National Trust (Farne Islands and Blakeney Point), Lincolnshire Wildlife Trust (Donna Nook) and Natural England (Horsey/Winterton) and by SNH (Shetland).

The numbers of pups born (pup production) at the aerially surveyed colonies in Scotland is estimated from a series of 3 to 5 counts derived from aerial images, using a model of the birth process and the development of pups. The method used to obtain pup production estimates for 2014 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. However, investigation of the effect of changing the time-to-leave parameter is under investigation (SCOS BP-15/03).

Between four and five surveys of the main grey seal breeding colonies in Scotland were carried out between September and November 2014. Paired digital images were obtained from two Hasselblad H4D 40MP cameras mounted at opposing angles of 12 degrees from vertical in SMRU's modified Image Motion Compensating cradle (Figure 2). As previously, a series of transects were flown over each breeding colony, ensuring that all areas used by pups were photographed (Figures 3 and 4). Images were recorded directly onto hard drives, one for each camera. Hard drives were downloaded and backed up after each day's survey.

All images were first adjusted for brightness and sharpness using Hasselblad's image processing software, Phocus. Individual images were then stretched from rectangular to trapezoid to closely match the ground area covered by oblique photographs taken at an angle of 12 degrees (Figure 3). All perspective-corrected images covering one survey of a particular colony were then stitched together to create a single digital image of the entire colony up to 15GB in size. Images were stitched and exported as PSB files using Microsoft's Image Composite Editor V.1.4.4. In a few cases where the stitching software could not stitch all images, such as with images of areas with large differences in ground elevation, images were stitched or adjusted manually using Adobe Photoshop CS5. The final composites were then saved as LZW compressed TIFF files (large images were split if TIFF's 4GB maximum file size was exceeded) and imported into Manifold GIS 8.0 for counting. The imported images were compressed within Manifold to reduce file size without losing too much image detail. Separate layers were created for marking whitecoat, moulted and dead pups (Figures 5 and 6).

Previously, because there was a significant risk of misclassifying moulted pups as whitecoats, the pup production model used a fixed value of 50% for the proportion of correctly classified moulted pups. Pups spend a lot of time lying on their back or side and, depending on light conditions during a survey, it is possible to misclassify a moulted pup exposing its white belly as a whitecoat. Misclassification of a whitecoat as a moulted pup is considerably less likely.

The pup production model allows different misclassification proportions to be incorporated. In Shetland, where pups are counted from the tops of cliffs and misclassification of moulted pups is likely to be low, a correctly classified proportion of 90% was used (SCOS-BP-05/01).

The digital images were of sufficient quality to reduce misclassification, so a proportion of 90% was used as standard for all production estimates.

Results & Discussion

All the main grey seal breeding colonies were successfully surveyed between September and December 2014. Four or five surveys of all colonies in the Inner Hebrides, Outer Hebrides, the north coast of Scotland, Orkney, north-east mainland Scotland, and the Firth of Forth were completed. A late (sixth) survey of Fast Castle in the Firth of Forth was completed in December. For the first time using the digital camera system, a single survey of the four main breeding colonies on the east coast of England (Farne Islands, Donna Nook, Blakeney Point and Horsey/Winterton) was completed.

At the end of July, images from all five surveys of colonies in the Inner Hebrides have been counted as have images from three of five surveys in the Outer Hebrides, two of four or five surveys of Orkney, two of five surveys of the north-east Mainland colonies, three of four or five surveys of the Firth of Forth, and two of four surveys of colonies on the North coast of Scotland. In total, 196 colony surveys have been counted out of 335. A summary is provided in Table 1.

In the Inner Hebrides, grey seal pup production at 13 colonies was estimated to be 4,054. This estimate used the current standard parameters of 0.9 for the proportion of moulted pups correctly identified and 31.5 days for the mean time-to-leave. An investigation into the time-to-leave parameter is provided in a separate Briefing Paper (SCOS BP-15/03). The 2014 Inner Hebrides production estimate (4,054) was 34 lower than the 2012 estimate. Pup production trajectories for individual colonies in the Inner Hebrides are shown in Figure 7. Production slightly increased in seven colonies and slightly declined in five.

In England, 6,627 pups were born at the annually monitored colonies on the east coast (Figure 8). A big increase in pups born at Blakeney Point saw it become the biggest grey seal breeding colony in England, overtaking the Farne Islands and Donna Nook.

A full report on grey seal pup production in 2014 will be provided to SCOS in 2016.

References

Pomeroy PP, Twiss SD, Redman P (2001) Philopatry, site fidelity and local kin associations within grey seal breeding colonies. Ethology 10:899-919

Duck CD, Mackey BL (2005) Grey seal pup production in Britain in 2004. SCOS Briefing Paper 05/01. pp 22-33 in: Scientific advice on matters related to the scientific management of seal populations: 2005. Unpublished Report to the Special Committee on Seals: 2005. Url: http://www.smru.st-andrews.ac.uk/documents/SCOS_05_v2f.pdf

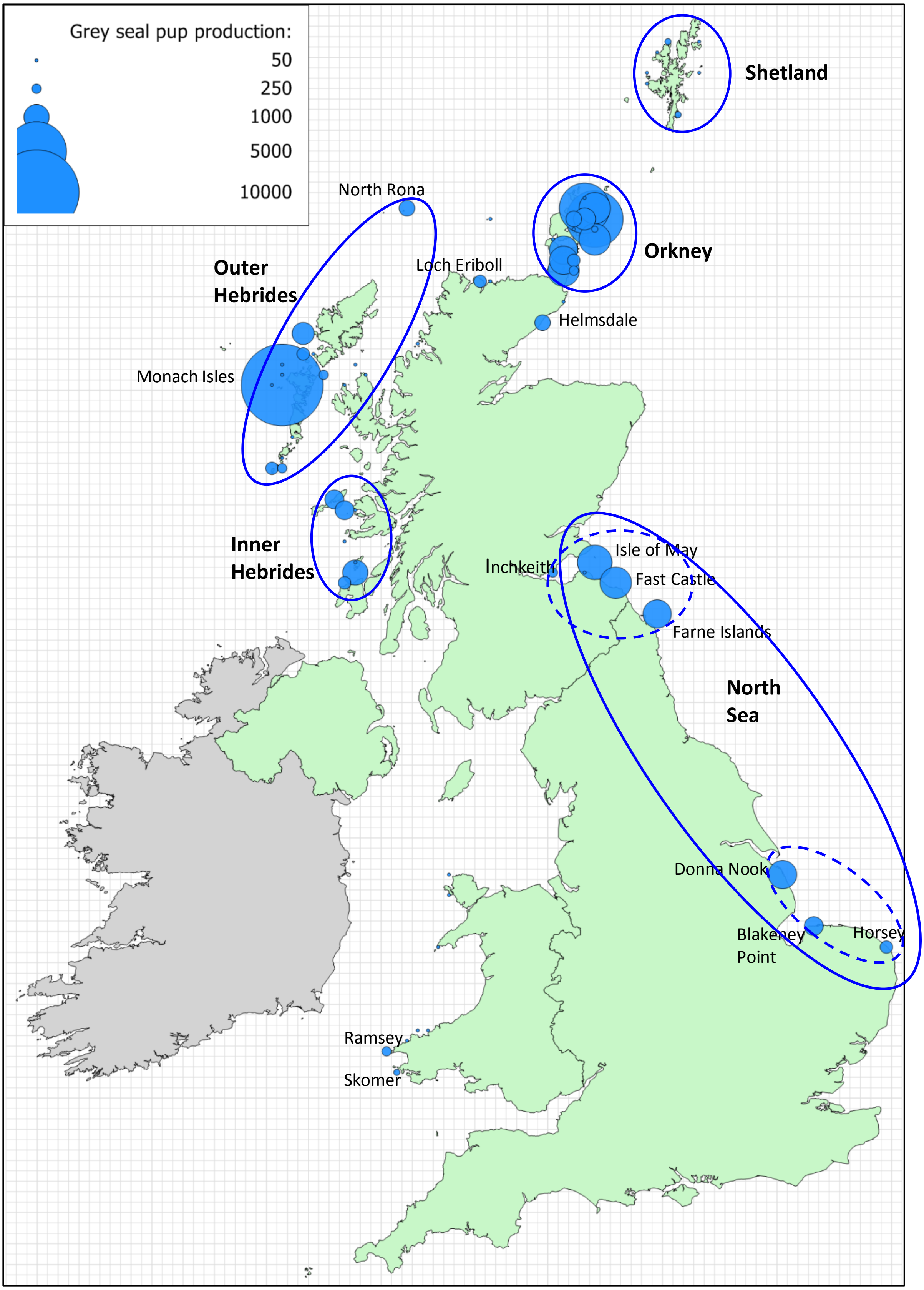
Duck CD, Morris CD (2013) Grey seal pup production in Britain in 2012: First use of a digital system. SCOS Briefing Paper 13/01. pp 71-88 in: Scientific advice on matters related to the scientific management of seal populations: 2013. Unpublished Report to the Special Committee on Seals: 2013. Url: http://www.smru.st-and.ac.uk/documents/1803.pdf

Russell DJF, Duck CD, Morris CD, Thompson D (2015) Review of parameters of the grey seal pup production model. SCOS Briefing Paper SCOS BP-15/03

Table 1. Progress of image processing and colony counts for 2014 breeding season.

Island Group	Number of colonies	Number of surveys	Images processed	Images stitched	surveys counted	Comments
Inner Hebrides	13	5	All	All	All	
Outer Hebrides	16	5	4 of 5	4 of 5	3 of 5 completed	
Orkney	29	4/5	2-3 of 4/5	2-3 of 4/5	2 of 4/5 completed	
North Mainland	2	4	4 of 4	2-3 of 4	2-3 of 4 completed	
North-east Mainland	5	5	3 of 5	2 of 5	2 of 5 completed	
Firth of Forth	4	4/5	2-3 of 4/5	3 of 4/5	2-3 of 4/5 completed	Last flight over Fast Castle only
East England	4	1	0	0	0 of 1 completed	1 survey in December

Figure 1. Pup production at the main grey seal breeding colonies in the UK in 2012 at a 10km resolution. Smaller numbers of grey seals will breed at locations other than those indicated here, including in caves.



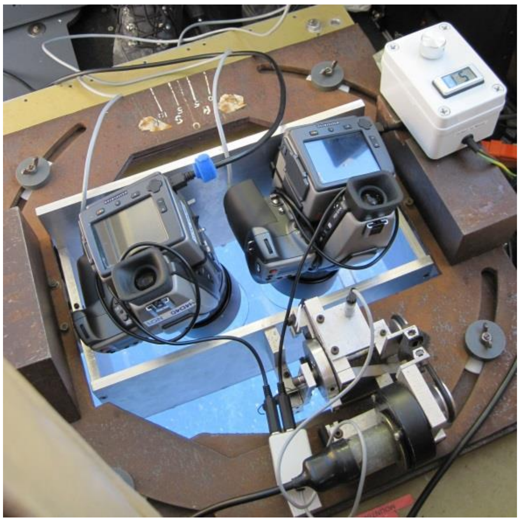


Figure 2. Two Hasselblad H4D-40 medium format cameras fitted in SMRU's Image Motion Compensation (IMC) mount. Each camera is set at an angle of 12 degrees to increase strip width. The cradle holding the cameras rocks backwards and forwards during photo runs. Rocking speed is set depending on the altitude and the ground speed of the aircraft. The camera shutters are automatically triggered and an image captured every time the cameras pass through the vertical position on each front-to-back pass. Images are saved directly to a computer as 60MB Hasselblad raw files and can be instantly viewed and checked using a small LED screen. The H4D-40 can take up to 40 frames per minute allowing for ground speeds of up to 140kts at 1100ft (providing 20% overlap between consecutive frames). The resulting ground sampling distance is approximately 2.5 cm/pix.

Figure 3. The individual footprints of each pair of photographs taken on a run over Eilean nan Ron, off Oronsay in the Inner Hebrides, flying at 1100 ft (red: left-hand camera; yellow: right-hand camera).

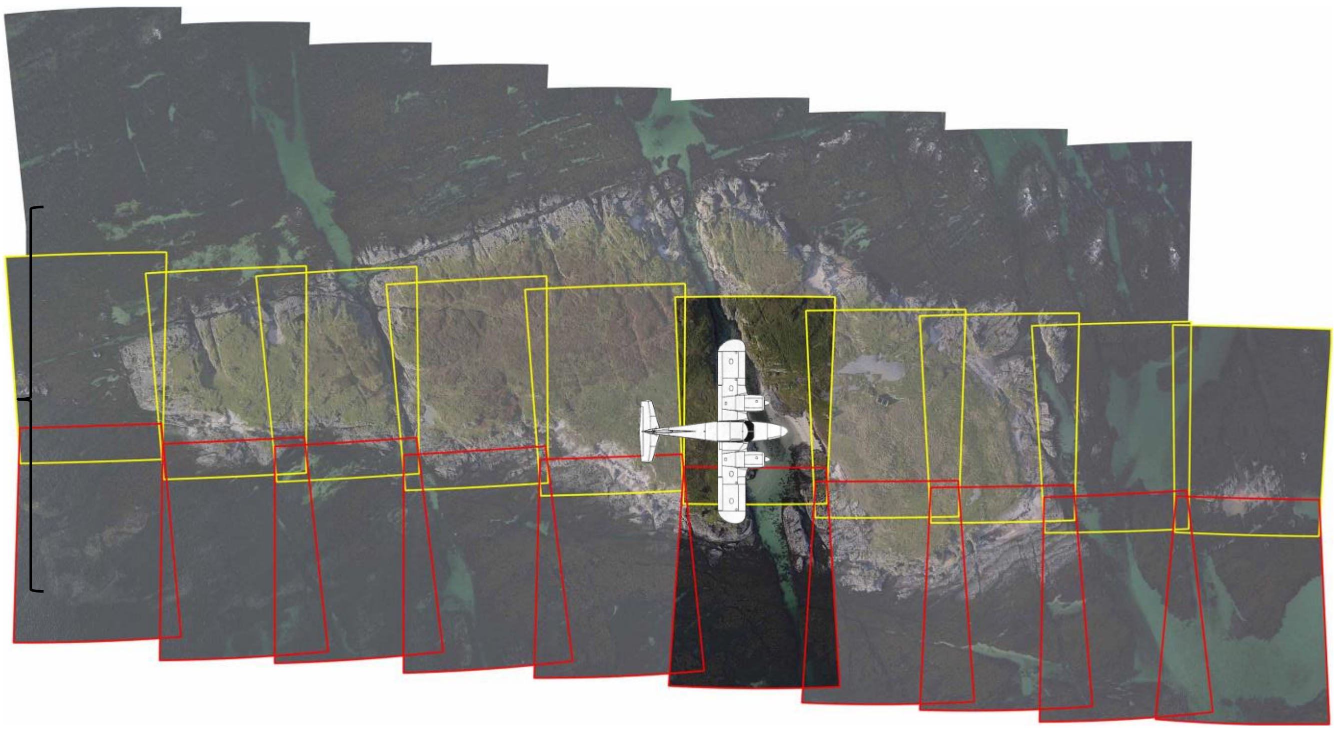


Figure 4. Survey runs and approximate camera trigger locations (yellow dots) for five colonies in the Monach Isles in the Outer Hebrides on 26 October 2012.

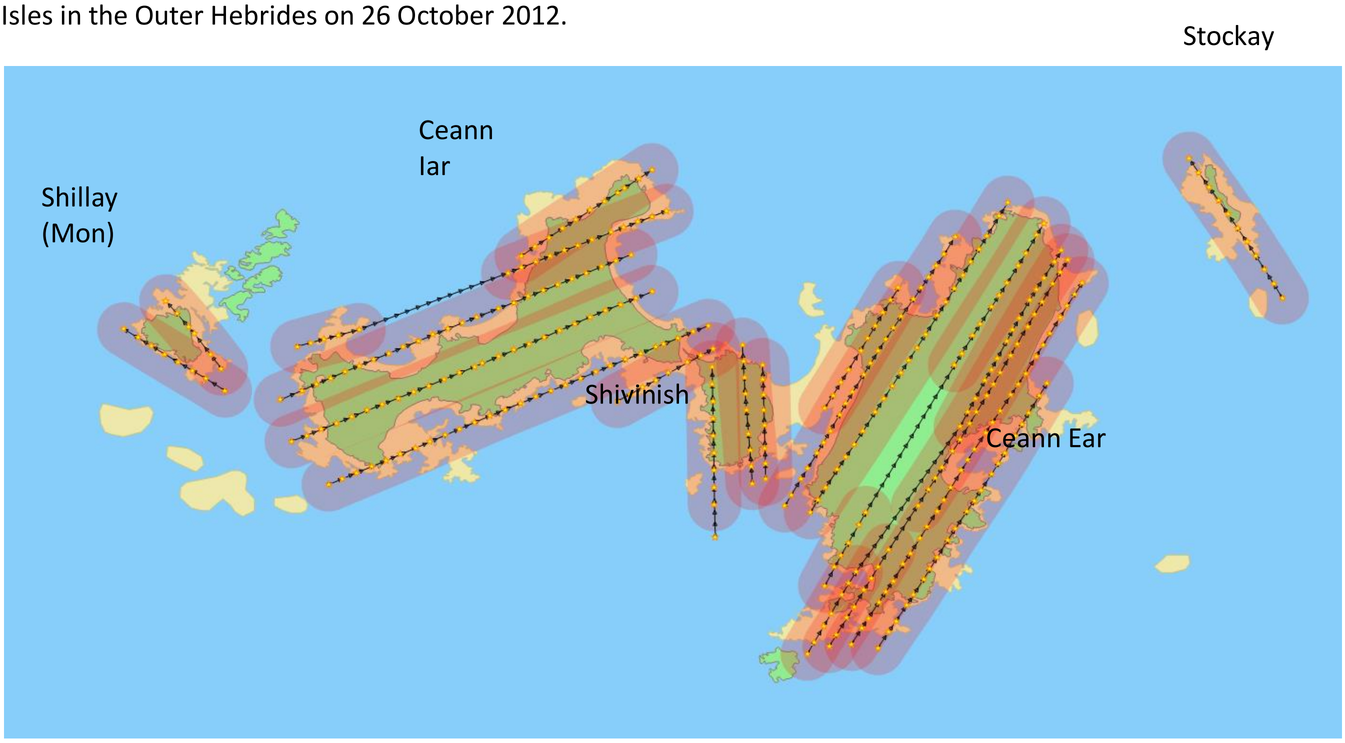


Figure 5. Ceann Iar, the second biggest of the Monach Isles in the Outer Hebrides, is the largest grey seal breeding colony in Europe (ca. 6,000 pups are born each year). This screenshot shows white-coated (white), moulted (blue) and dead pups (red) counted from approximately 200 stitched photographs taken on 7 October 2012. The composite image was stitched together and exported using Microsoft's Image Composite Editor v1.4.4®. The resulting 7.2 gigapixel PSB file (15 GB) was split into 30,000x30,000 pix TIFF tiles using Adobe Photoshop CS5®. These were then imported into Manifold GIS 8.0® for counting.

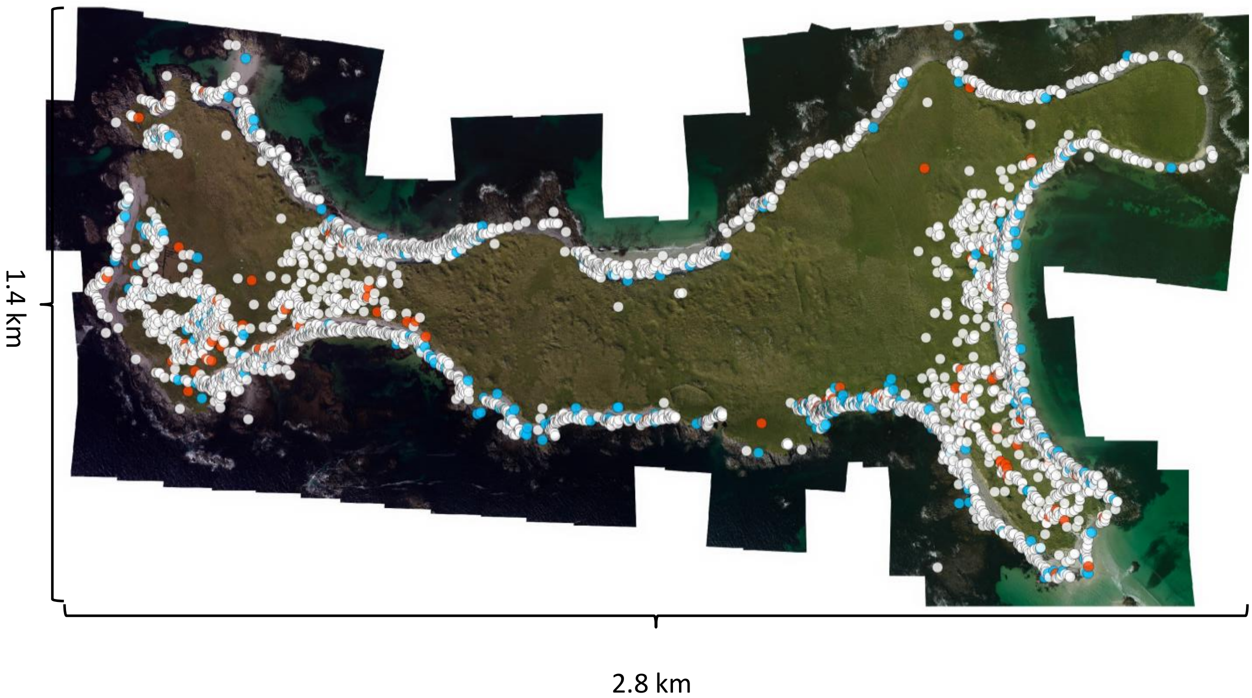


Figure 6. Manifold GIS 8.0® screenshot showing grey seal pups counted on Ceann Iar. Pups of each category (whitecoat, moulted, dead) are counted on a separate layer. The images are not currently geo-referenced but there is the potential for further processing, thus obtaining approximate coordinates for every pup counted.

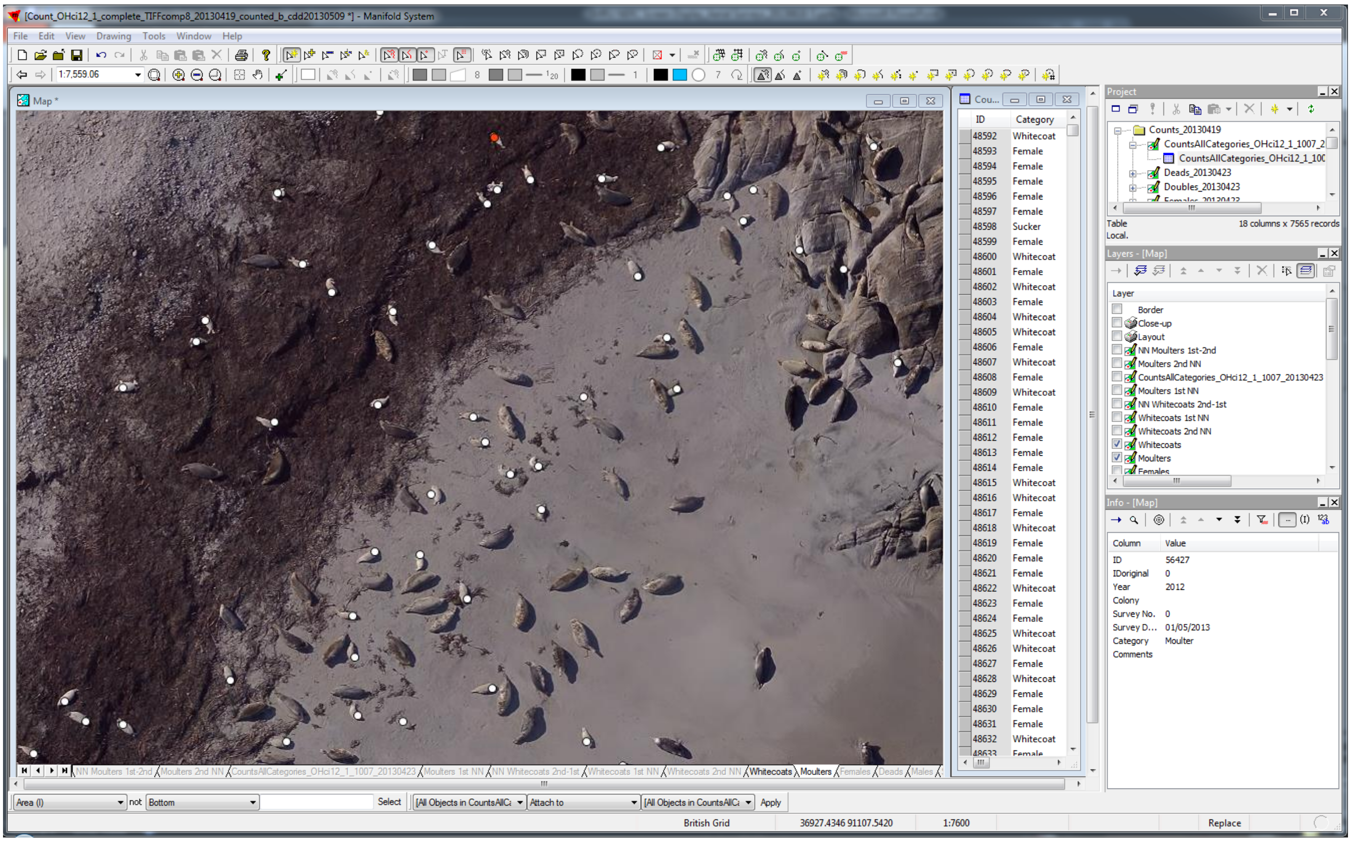


Figure 7. Grey seal pup production at 13 colonies in the Inner Hebrides, calculated using the standard Time to Leave of 31.5 days.

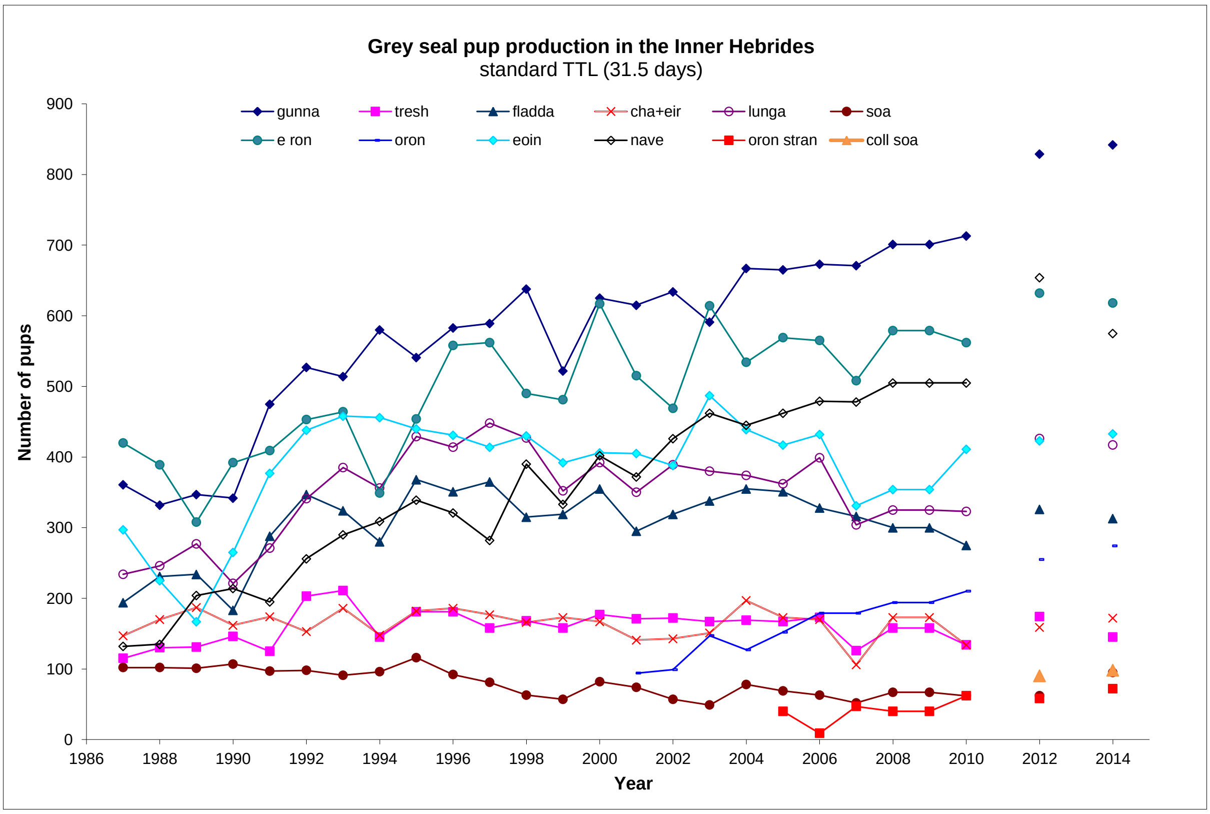
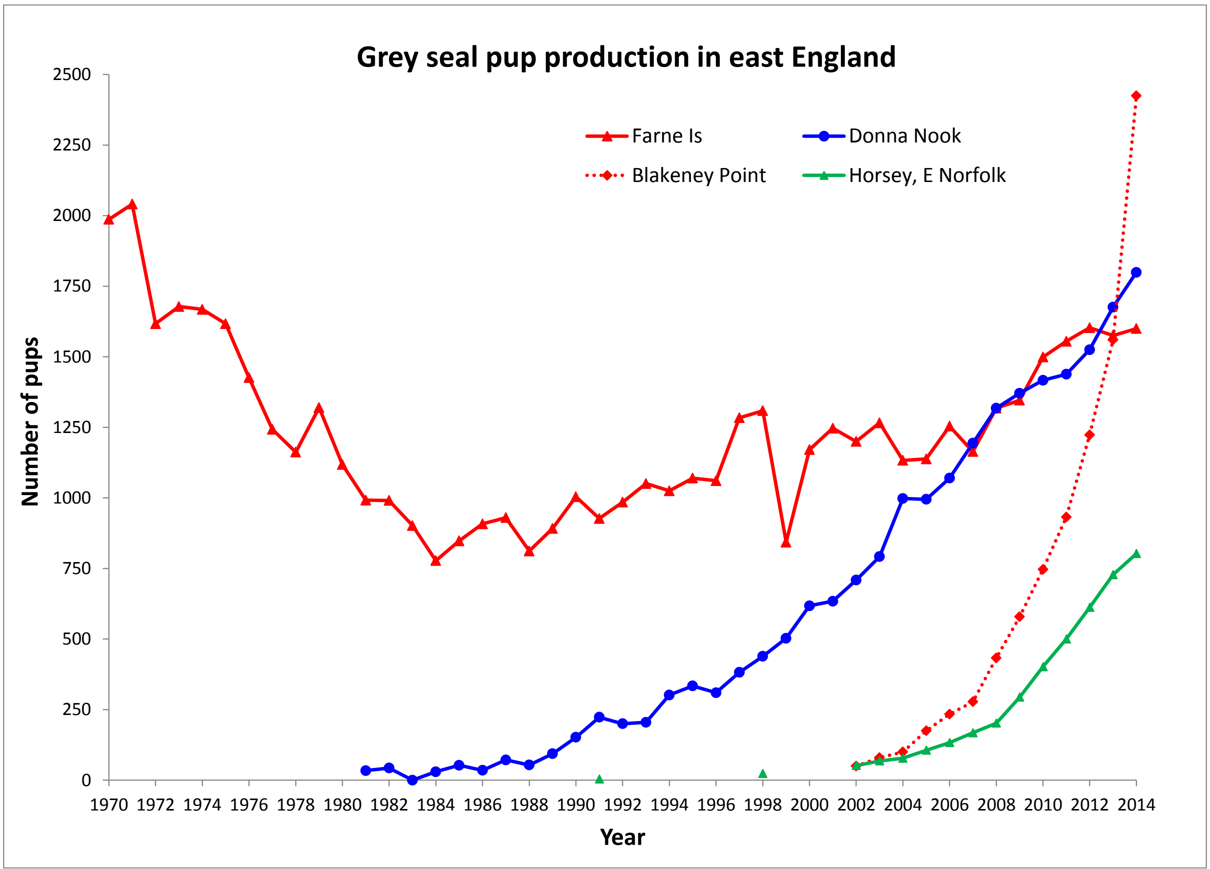


Figure 8. Grey seal pup production at four colonies in east England. A large increase in the number of pups born at Blakeney Point saw it become the biggest grey seal breeding colony in England, overtaking the Farne Islands and Donna Nook.



Estimating the size of the UK grey seal population between 1984 and 2014

Len Thomas

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Abstract

We fitted a Bayesian state-space model of British grey seal population dynamics to two sources of data: (1) regional estimates of pup production from 1984 to 2012, and (2) an independent estimate assumed to be of total population size just before the 2008 breeding season. The model allowed for density dependence in pup survival, using a flexible form for the density dependence function, and assumed no movement of recruiting females between regions. This model is identical to that used to provide last year's advice, and the same "revised" priors were used, including a prior on adult sex ratio. One small change in data was that the total population size estimate was adjusted to account for the fact that the population model is based only on regularly monitored breeding colonies (approx. 94% of the total population). We used the model to predict past the last data point (2012) to give estimates of population size in 2014. Estimated adult population size in 2014 was 95,200 (95% CI 76,400-127,500).

The model assumes constant adult (i.e., aged 6+) female survival. The prior distribution has support in the range (0.8, 1.0) with a prior mean of 0.91 (SD 0.05); the posterior mean is an implausibly high 0.99 (SD 0.01). We investigated the effect of constraining the prior to the range (0.8, 0.97). Posterior mean adult survival with this revised prior was 0.95 (SD 0.03); estimated population size with this revised prior was 105,200 (95% CI 87,000-128,800).

Female survival is currently assumed to be the same for all ages. We investigated the possible effect of including survival senescence, and concluded that adding it would make no practical difference to the modelled population dynamics.

Sex ratio is an important parameter in the model, scaling estimates of adult female population size from the population dynamics model to total population size. The current prior is highly informative (prior mean on ratio of total population:adult females 1.7 SD 0.02). We investigated the consequences of using a less informative prior suggested in a previous briefing paper (prior mean 1.2 SD 0.63). With this prior (and the revised prior on adult female survival), total population size was estimated to be much lower (88,600 with 95%CI 70,200-111,700), but the ratio of total population:adult females was an implausibly low 1.14 (SD 0.09).

Introduction

This paper presents estimates of British grey seal population size and related demographic parameters, using identical models and fitting methods to Thomas (2014, and previous years), but projecting forward two years from the last pup production estimate (2012) to estimate population size in 2014. Models are specified using a Bayesian state space framework with informative priors on demographic parameters, and fitted using a Monte Carlo particle filter. In past briefing papers, multiple models of the population dynamics have been fitted and compared, representing differing hypotheses about the demographic parameter subject to density dependent regulation. The model where density dependence affects pup survival was found to be better supported by the data than one where density dependence affects female fecundity (Thomas 2012); hence only the former is used here.

A revised set of priors were suggested by Lonergan (2012), based on updated information and discussions within the Sea Mammal Research Unit; these were used by Thomas (2012, 2013, 2014) to assess what difference these make to the population estimates compared to the priors previously used (note that a different prior on adult sex ratio was used in the 2014 analysis); the revised priors were adopted for use in the SCOS advice in 2014. We therefore use these revised priors here.

Two additional investigations are carried out: (1) the impact of using a revised prior on adult female survival, so that survival is constrained to maximum of 0.97; (2) the potential impact of including survival senescence.

Materials and Methods

Process model

The population dynamics model is described fully in Thomas and Harwood (2008) and papers cited therein (it is referred to there as the EDDSNM model), except that those models assumed a fixed adult sex ratio. The model was extended to allow estimation of adult sex ratio by Thomas (2012). In summary, the model tracks seal population numbers in 8 age and sex groups (pups, age 1-5 females, which do not pup, and age 6+ females, which may produce a single pup, and age 1+ males) 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 age 5 females between regions, but we assume no movement in the current model.) Age 1+ ("adult") males are not tracked explicitly, but instead are linked to the number of females by a sex ratio parameter. The model has 9 parameters: adult (i.e., age 1 and older) female survival, ϕ_a, maximum pup survival, $\phi_{j\max}$, one carrying capacity parameter-related parameter for each region, β_1 - β_4, a parameter, ρ, that dictates the shape of the density-dependent response, fecundity (i.e., probability that an age 6+ female will birth a pup), α, and adult sex ratio ω.

Data, observation models, and priors

One source of input data was the pup production estimates for 1984-2010 and 2012 from Duck (2014) covering the regularly surveyed colonies, aggregated into regions. These estimates were assumed to be normally distributed with mean equal to the true pup production in each region and year, and constant coefficient of variation (CV). This CV was estimated from an initial run of the model by Thomas (2014), and for the runs performed here was fixed to this value (10.5%).

The second source of input data was a single estimate of adult population size obtained by Lonergan et al. (2010) from summer haulout counts and telemetry data. We followed previous briefing papers in assuming the estimate was of population size just before the start of the 2008 breeding season, and by representing the uncertainty in the estimate (which Lonergan obtained via a nonparametric bootstrap) using a right-shifted gamma distribution. However, one important change is that we did not previously account for the fact that this adult population estimate covers the whole UK population of seals while the pup production model covers only the breeding colonies regularly surveyed – estimated to be 92.34% of total pup production in 2008 (Duck 2009). We therefore scaled the adult estimate to make it comparable with the pup production model outputs, from 88,300 (95% CI 75,400-105,700) to 81,530 (95% CI 69,650- 96,690).

Prior distributions for the process model parameters were the same as the "revised priors" used in Thomas (2014); these in turn are those suggested by Longeran (2012, Table 1), except for the prior on adult sex ratio, which was first suggested by Thomas (2014). We followed Thomas and Harwood (2005) 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).

In summary, the priors used here are identical to those used by Thomas (2014); the data were identical except the total population estimate was revised down by 7.66%.

Fitting method

The fitting method was identical to that of Thomas (2014), again using the particle filtering algorithm of Thomas and Harwood (2008). This involves simulating samples ("particles") from the prior distributions, projecting them forward in time according to the population model, and then resampling and/or reweighting them (i.e., "filtering") according to their likelihood given the data. An identical algorithm to that of Thomas and Harwood (2008) was used for the pup production data, and the additional adult data was included by reweighting the final output according to the likelihood of the estimated 2008 population size, as described by Thomas (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 1,750 replicate runs of 1,000,000 samples. A technique called 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). The rejection control threshold used was w_c=1000.

Additional investigation: revised prior on survival

The model population dynamics model assumes constant adult (i.e., aged 6+) female survival. The prior distribution was a scaled beta (0.8+0.2xbeta(1.6,1.2)) with support in the range (0.8, 1.0), which has a prior mean of 0.91 (SD 0.05). However, given this prior and the available data, Thomas (2014) obtained a posterior distribution that was implausibly high: the posterior mean was 0.99 (SD 0.01). At the request of SCOS, we therefore investigated the effect of an alternative prior, with the upper bound constrained to 0.97. A prior with support in the range (0.8, 0.97) and the same mean and SD (0.8+0.17xbeta(0.988,0.482)) had an implausible shape (Figure 1b); we therefore obtained a prior by scaling the previous one from the range (0.8,1.0) to (0.8,0.97), i.e., 0.8+0.17xbeta(1.6,1.2) (Figure 1c). This gave a prior with a mean of 0.90 (SD 0.04). We re-fit the model using this revised prior, using 3,000 replicate runs of 1,000,000 samples.

Additional investigation: effect of survival senescence on population dynamics

In the current population dynamics model, female survival is assumed to be the same for all ages. Age 6+ seals are modelled together, in an "absorbing" age class, which implies that some very old seals may be present in this age class. SCOS asked us to investigate the possible effect of including age senescence in the model. This could be done by expanding the number of age classes followed in the model (e.g., to 50) and by making survival a decreasing function of age. We undertook a preliminary investigation of the potential effect of survival senescence by parameterizing a plausible survival function and calculating the proportion of adults that would likely be in the senescent age classes.

The survival function was based on the Gompertz-Makeham function (Makeham 1860), which describes the instantaneous hazard of mortality at age x as

$$H(x) = \lambda + \nu \exp(\delta x) \quad (1)$$

where λ (>0) is a baseline mortality hazard, and ν (<0) and δ (>0) index how survival declines with age. The population model used here is a discrete-time (annual) model; therefore eqn. (1) was integrated to derive the cumulative probability of survival between two annual time points

$$\phi_x = \phi_{base} \exp\left[\frac{\nu}{\delta}(\exp[\delta x] - \exp[\delta(x+1)])\right] \quad (2)$$

where ϕ_x is survival from age x to age $x + 1$ and $\phi_{base} = \exp(\lambda)$ is the baseline survival for young animals.

A range of plausible values were used for baseline survival (ϕ_{base}): 0.97, 0.95 and 0.90. Values for the other two parameters (ν and δ) were derived by least-squares fitting to preliminary adult female survival estimates from Sable Island, Canada (Don Bowen, pers. comm.). Given the fitted functions, we calculated the relative proportion of adults expected in each age class, assuming a stable age structure.

Additional investigation: revised prior on sex ratio

Sex ratio is an important parameter in the model, scaling estimates of adult female population size from the population dynamics model to total population size. The current prior, introduced by Thomas (2014) is highly informative, with a prior mean on the ratio of total population:adult females of 1.7 and standard deviation of 0.02 – almost all of the prior mass lies between 1.66 and 1.76. Longeran (2012) suggested a much less informative prior, which also had a rather lower mean (prior mean 1.2 SD 0.63), and this prior was used by Thomas (2013). For the purposes of demonstration, we repeated the analysis using that less informative prior, coupled with the above 0.8-0.97 bounded prior on adult survival.

Results

Parameter and population estimates

Model fits to pup production estimates are shown in Figure 2, and the estimated adult population size is shown in Figure 3, together with the scaled independent estimate. Posterior parameter estimates are shown in Figure 4 and Table 1. As with Thomas (2014), the posterior mean adult survival is very high (0.99) when the pup production data and independent estimate are used, with the mode being even higher (Figure 4b); conversely maximum juvenile survival is very low (0.282).

Adult population size estimates by region for 2014 are given in Table 2; the posterior mean total population size was 99,500 (95% CI 81,500-124,100). Estimates for all years are given in Appendix 1.

Additional investigation: revised prior on survival

Model fits to the pup production estimates (not shown) were almost identical to those from the main analysis described above. Estimated population size was approximately 10% lower (Figure 5 and Table 2). The posterior distribution on adult population size was bounded at 0.97, with a mode close to 0.97 but a lower mean of 0.96 (Figure 6). Mean maximum juvenile survival was nearly 50% higher than in the main analysis (0.39) and fecundity was slightly higher (0.95).

Additional investigation: effect of survival senescence on population dynamics

The fitted age-specific survival function is shown in Figure 7 (assuming a baseline survival of 1.0). Survival is 90% of baseline at age 33 and drops to 10% of baseline by age 44. Life tables, showing the relative numbers of adults at each age given baseline survival of 0.97, 0.95 and 0.90, are shown in Figure 8. The proportion of the population that is older than 33 given these three baseline survival rates is 4%, 3% and 1%, respectively. Hence, even when baseline survival is high, very few adults in the population are estimated to be old enough to be exhibiting senescence.

Additional investigation: revised prior on sex ratio

The wider prior on sex ratio led to a much wider posterior on adult population size without the independent estimate (blue line in Figure 9); this in turn led to more weight being placed on the independent estimate when it was included in the analysis and hence the final population trajectory (red line in Figure 9) passing very close to the independent estimate. The revised sex ratio prior was also lower, meaning that less change was required to the demographic parameter estimates to accommodate the independent estimate (i.e., the top and bottom panels of Figure 10 are more similar than those of Figures 4 and 6). Although the posterior distributions of many parameters were sensible, the posterior on sex ratio was implausible: the ratio of total population size to adult females was estimated at 1.14 (SD 0.09)

Discussion

The population size estimated in this briefing paper is essentially a projection of the stochastic population dynamics model fit to data from 2012 and earlier. The posterior mean adult population size of 94,500 for 2013 is slightly lower than that for 2013 given by Thomas (2014) (which was 98,800), reflecting the amended independent of total population size used. In all other ways, the fit of the model to the data is nearly identical to that shown in Thomas (2014).

Thomas (2014) documents some inadequacies of the model fit: the fitted model does not capture all features of the pup production data, with clear runs of positive or negative residuals and particular lack of fit to the last data point; the implausible posterior estimate of adult and maximum pup survival also noted above. It is not surprising that high estimated adult survival and low maximum pup survival should occur in the same model, since the two estimates are strongly correlated (Thomas 2013). If the estimates are truly considered infeasible, then consideration should be given to revising the priors to restrict posteriors to feasible regions.

We are investigated a model where the prior range on adult population size, φ_a, is restricted to a maximum of 0.97. This produced a more realistic posterior mean φ_a, although the mode was close to the upper bound of the prior. The estimate of maximum pup survival was higher, as we would expect given the strong correlation between the two parameters; estimated fecundity was also slightly higher. Together, the change in parameter estimates produced a higher estimate of total population size from the population dynamics model (compare blue lines in Figures 3 and 5), which in turn led to a higher total population size estimate when the independent estimate was factored in (red line in Figure 5). Population size estimates using this revised prior on φ_a are approximately 10% higher than those using the previous prior.

Our initial analysis of the importance of senescence appears to show that ignoring it is unlikely to have any practical impact on the dynamics or parameter estimates of the population model. Even using a high baseline adult survival estimate, a very small proportion of the adult population are estimated to be at the age where senescence begins to have a significant effect on survival – assuming the preliminary information provided about survival senescence in Canadian grey seals applies also to UK grey seals.

Using a wider and lower prior on sex ratio had a significant impact on posterior parameter distributions and estimates of total population size. The lower prior mean meant that the estimate from the pup production and population dynamics model was closer to the independent estimate; the wider prior meant that the population dynamics-based estimate was given less weight relative to the independent estimate. Although the posterior on sex ratio was implausible (suggesting there are approximately 7 adult females per adult male), it does suggest that a re-examination of the current sex ratio prior may be helpful in bringing the output from the population dynamics model closer to the independent estimate.

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Table 1. Prior parameter distributions and summary of posterior distributions. (The two parameters of the gamma distribution specified here are shape and scale respectively.) Posterior summaries are all from analyses that use both 1984-2010 and 2012 pup production estimates, and the 2008 total population estimates.

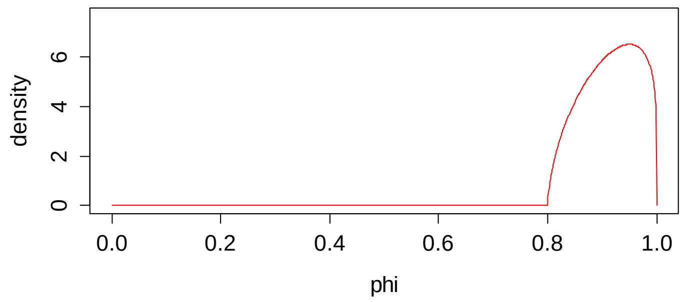
	Main analysis			Additional investigation on adult survival			Additional investigation on sex ratio		
Parameter	Prior distribution	Prior mean (SD)	Posterior mean (SD)	Prior distribution	Prior mean (SD)	Posterior mean (SD)	Prior distribution	Prior mean (SD)	Posterior mean (SD)
adult survival ϕ_a	0.8+0.2*Be(1.6,1.2)	0.91 (0.05)	0.99 (0.01)	0.8+0.17*Be(1.6,1.2)	0.90 (0.04)	0.96 (0.01)	0.8+0.17*Be(1.6,1.2)	0.90 (0.04)	0.95 (0.02)
pup survival ϕ_j	Be(2.87,1.78)	0.62 (0.20)	0.27 (0.05)	same as main analysis	0.62 (0.20)	0.37 (0.06)	same as main analysis	0.62 (0.20)	0.57 (0.11)
fecundity $\alpha_{\max}$	0.6+0.4*Be(2,1.5)	0.83 (0.09)	0.90 (0.05)	same as main analysis	0.83 (0.09)	0.95 (0.03)	same as main analysis	0.83 (0.09)	0.87 (0.08)
dens. dep. ρ	Ga(4,2.5)	10 (5)	6.12 (2.31)	same as main analysis	10 (5)	4.24 (0.9)	same as main analysis	10 (5)	3.47 (0.9)
NS carrying cap. χ_1	Ga(4,2500)	10000 (5000)	15800 (7540)	same as main analysis	10000 (5000)	13700 (4440)	same as main analysis	10000 (5000)	14400 (4150)
IH carrying cap. χ_2	Ga(4,1250)	5000 (2500)	3760 (448)	same as main analysis	5000 (2500)	4390 (213)	same as main analysis	5000 (2500)	3600 (295)
OH carrying cap. χ_3	Ga(4,3750)	15000 (7500)	13200 (1650)	same as main analysis	15000 (7500)	12300 (628)	same as main analysis	15000 (7500)	12700 (775)
Ork carrying cap. χ_4	Ga(4,10000)	40000 (20000)	23300 (3510)	same as main analysis	40000 (20000)	20800 (2270)	same as main analysis	40000 (20000)	23200 (3530)
observation CV ψ	Fixed	0.89 (0)	-	same as main analysis	0.89 (0)	-	same as main analysis	0.89 (0)	-
sex ratio ω	1.6+Ga(28.08, 3.70E-3)	1.7 (0.02)	1.7 (0.02)	same as main analysis	1.7 (0.02)	1.7 (0.02)	1+Ga(0.1,2)	1.2 (0.63)	1.14 (0.09)

Table 2. Estimated size, in thousands, of the British grey seal population at the start of the 2014 breeding season, derived from models fit to pup production data from 1984-2012 and the additional total population estimate from 2008, using the revised parameter priors. Numbers are posterior means with 95% credible intervals in brackets**.**

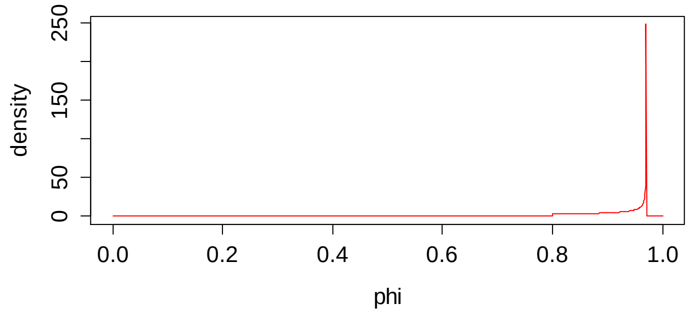
	Estimated population size in thousands (95% CI)		
	Main analysis	Additional investigation on adult survival	Additional investigation on sex ratio
North Sea	22.5 (16 29.7)	25.6 (18.6 33.5)	20.8 (15.1 27.9)
Inner Hebrides	6.9 (5.7 8.1)	7.6 (6.6 8.9)	6.2 (5.1 7.7)
Outer Hebrides	24.4 (20.9 29)	26.8 (23.9 30.8)	22 (18.3 26.4)
Orkney	41.4 (33.8 50.7)	45.2 (37.9 55.7)	39.6 (31.7 49.7)
Total	95.2 (76.4 117.5)	105.2 (87 128.8)	88.6 (70.2 111.7)

Figure 1. Prior distributions on adult female survival.

(a) Prior used in main analysis.



(b) Prior with same mean and standard deviation as (a), but constrained to a maximum of 0.97.



(c) Prior based on (a) scaled so that the maximum is 0.97 – this is the prior used in the additional investigation.

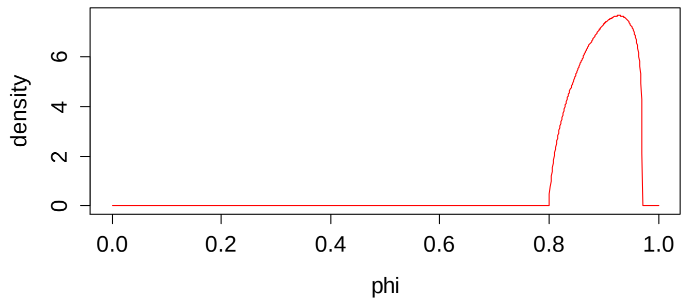


Figure 2. Posterior mean estimates of pup production (solid lines) and 95%CI (dashed lines) from the model of grey seal population dynamics, fit to pup production estimates from 1984-2012 (circles) and a total population estimate from 2008. Blue lines show the fit to pop production estimates alone; red lines show the fit to pup production estimates plus the total population estimate.

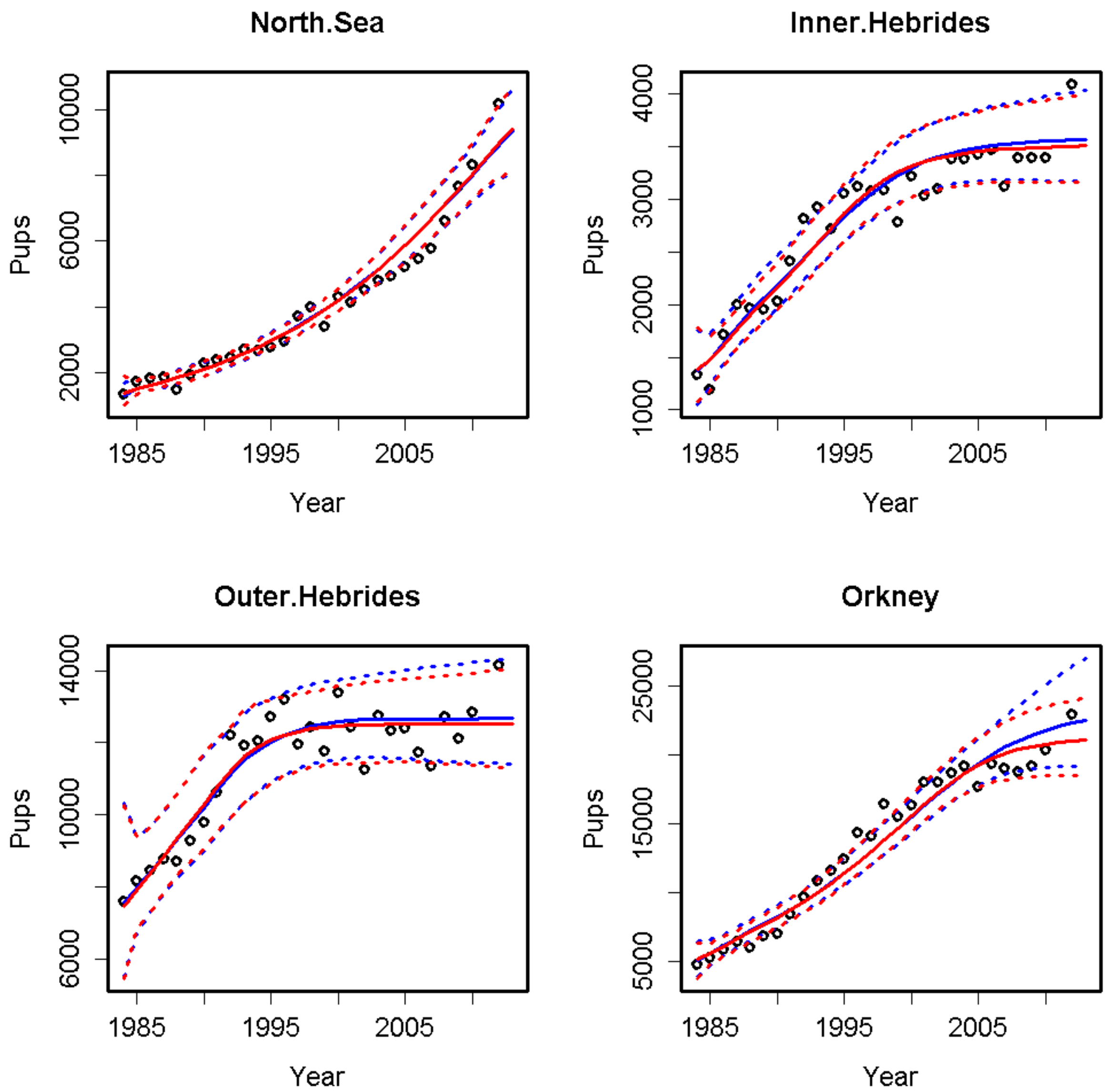


Figure 3. Posterior mean estimates (solid lines) and 95%CI (dashed lines) of total population size in 1984-2014 from the model of grey seal population dynamics, fit to pup production estimates from 1984-2012 and a total population estimate from 2008 (circle, with horizontal lines indicating 95% confidence interval on the estimate). Blue lines show the fit to pop production estimates alone; red lines show the fit to pup production estimates plus the total population estimate.

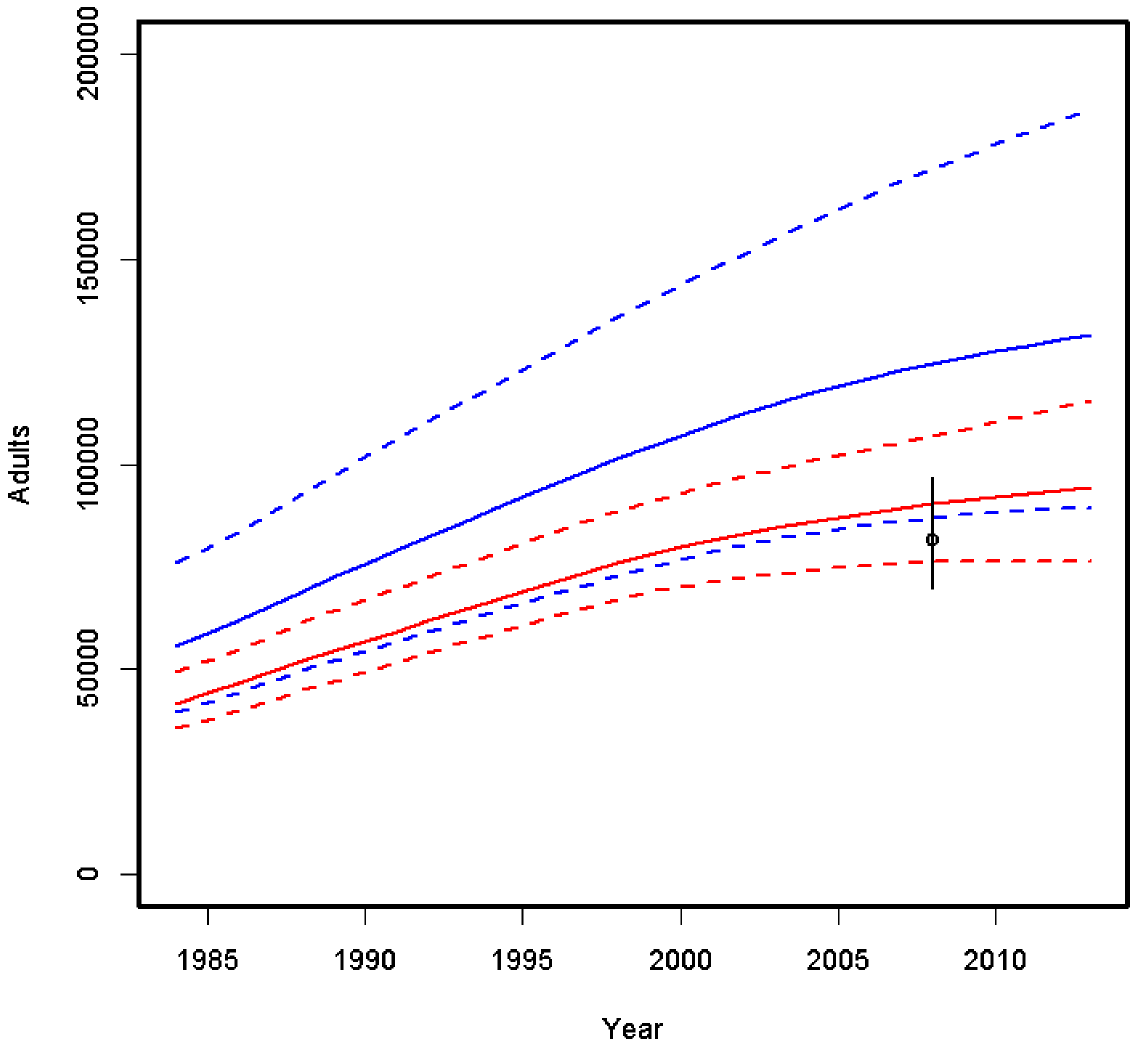
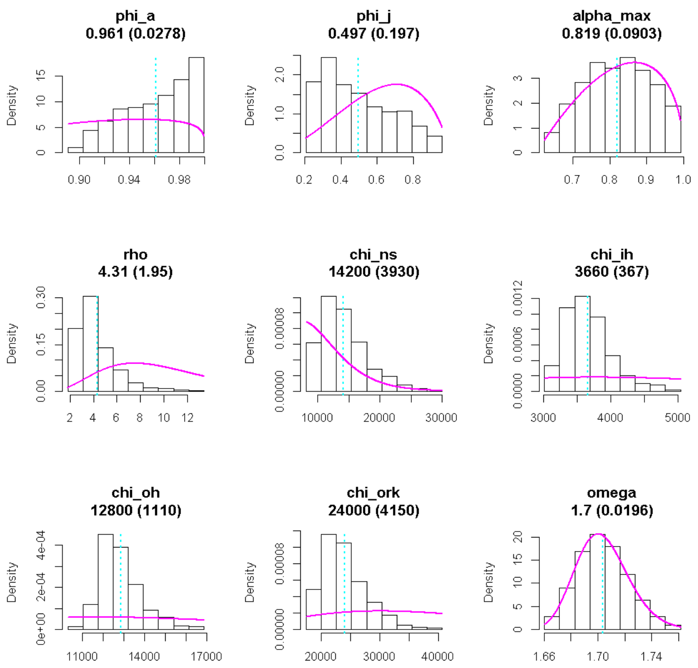


Figure 4. Posterior parameter distributions (histograms) and priors (solid lines) for the model of grey seal population dynamics, fit to pup production estimates from 1984-2012 and a total population estimate from 2008. The vertical line shows the posterior mean; its value is given in the title of each plot after the parameter name, with the associated standard error in parentheses.

(a) Pup production data alone



(b) Pup production data and 2008 population estimate

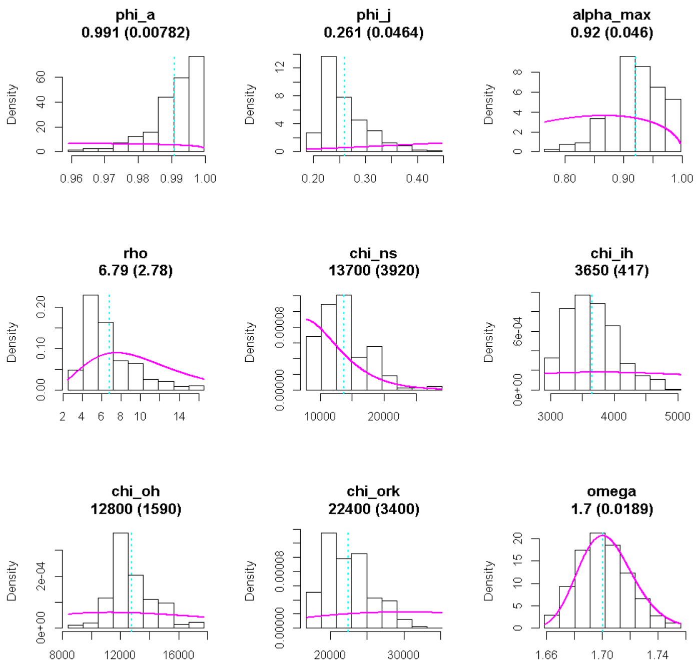


Figure 5. Posterior mean estimates (solid lines) and 95%CI (dashed lines) of total population size in 1984-2014 from the model of grey seal population dynamics, fit to pup production estimates from 1984-2012 and a total population estimate from 2008 (circle, with horizontal lines indicating 95% confidence interval on the estimate), and using a prior on adult survival constrained to have a maximum of 0.97. Blue lines show the fit to pop production estimates alone; red lines show the fit to pup production estimates plus the total population estimate.

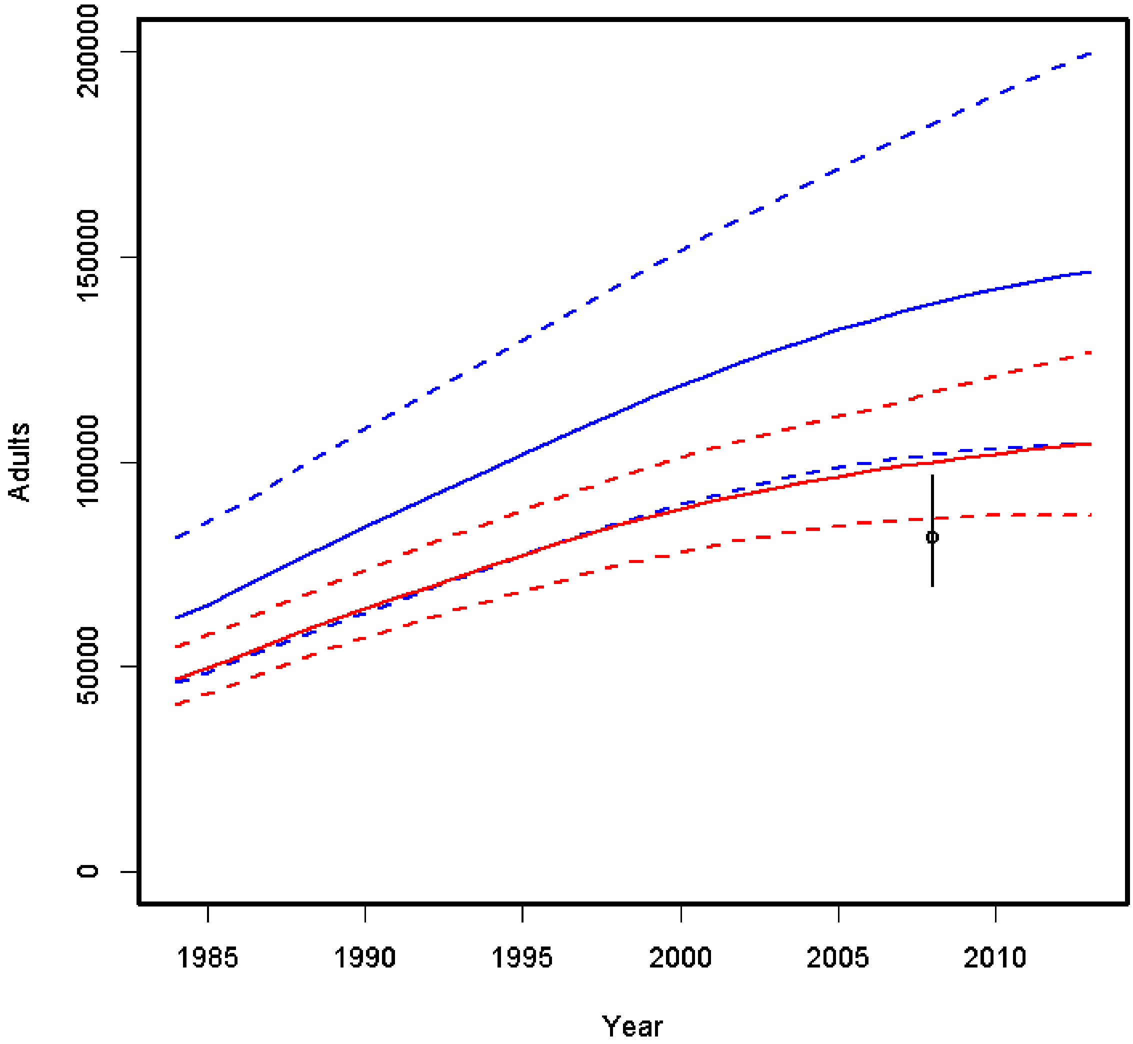
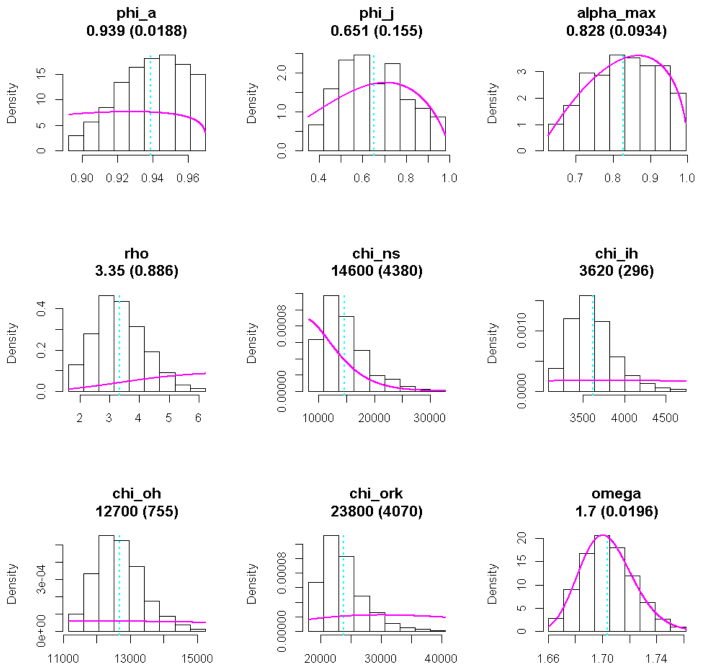


Figure 6. Posterior parameter distributions (histograms) and priors (solid lines) for the model of grey seal population dynamics, fit to pup production estimates from 1984-2012 and a total population estimate from 2008, and using a prior on adult survival constrained to have a maximum of 0.97. The vertical 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.

(a) Pup production data alone



(b) Pup production data and 2008 population estimate

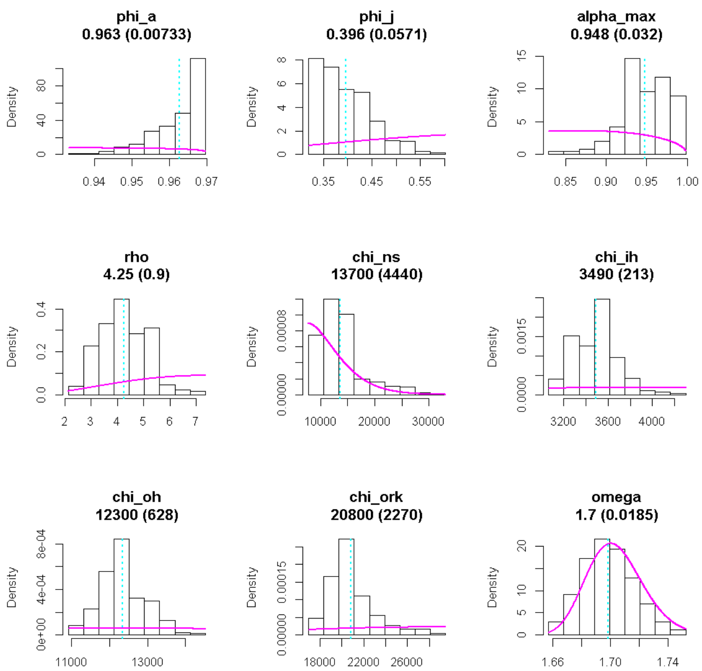


Figure 7. Age specific survival function used to investigate senescence, assuming baseline survival rate of 1.0.

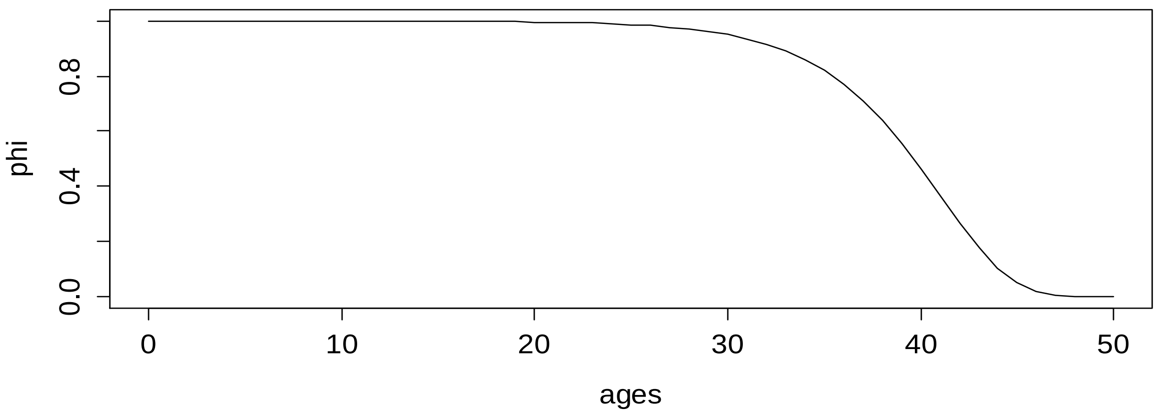


Figure 8. Life tables, showing the number of adults in at each age, relative to those aged 1 using the age-specific survival function in Figure 7, but assuming baseline adult survival rates of 0.97, 0.95 and 0.90.

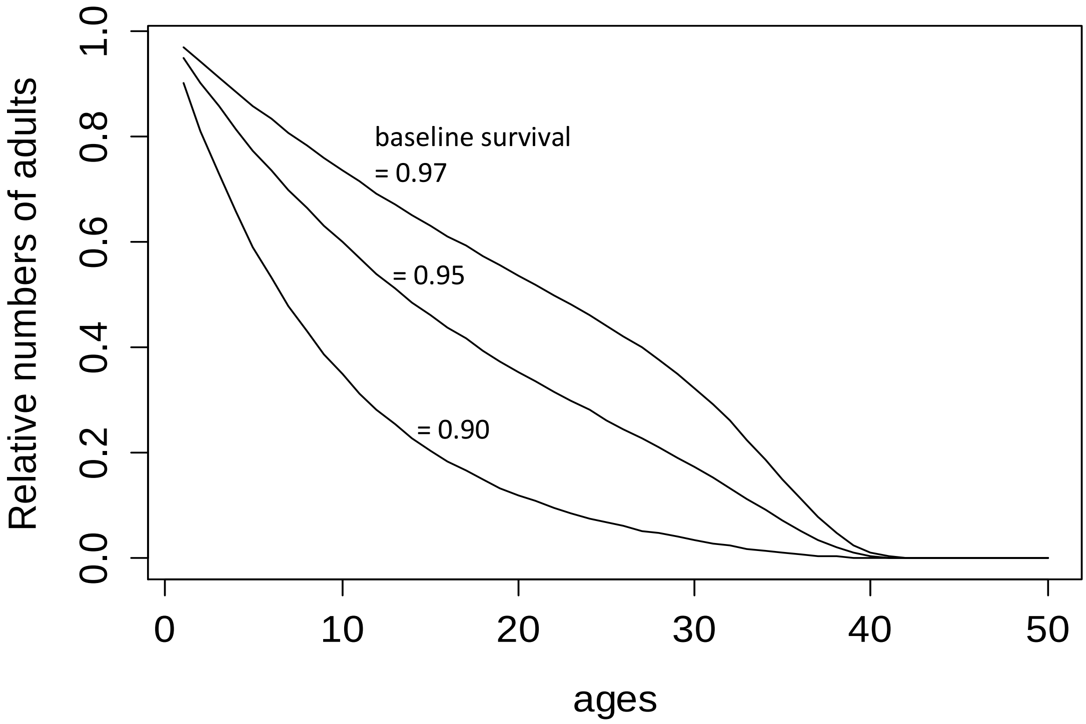


Figure 9. Posterior mean estimates (solid lines) and 95%CI (dashed lines) of total population size in 1984-2014 from the model of grey seal population dynamics, fit to pup production estimates from 1984-2012 and a total population estimate from 2008 (circle, with horizontal lines indicating 95% confidence interval on the estimate), and using a prior on adult survival constrained to have a maximum of 0.97 and a less informative prior on sex ratio. Blue lines show the fit to pop production estimates alone; red lines show the fit to pup production estimates plus the total population estimate.

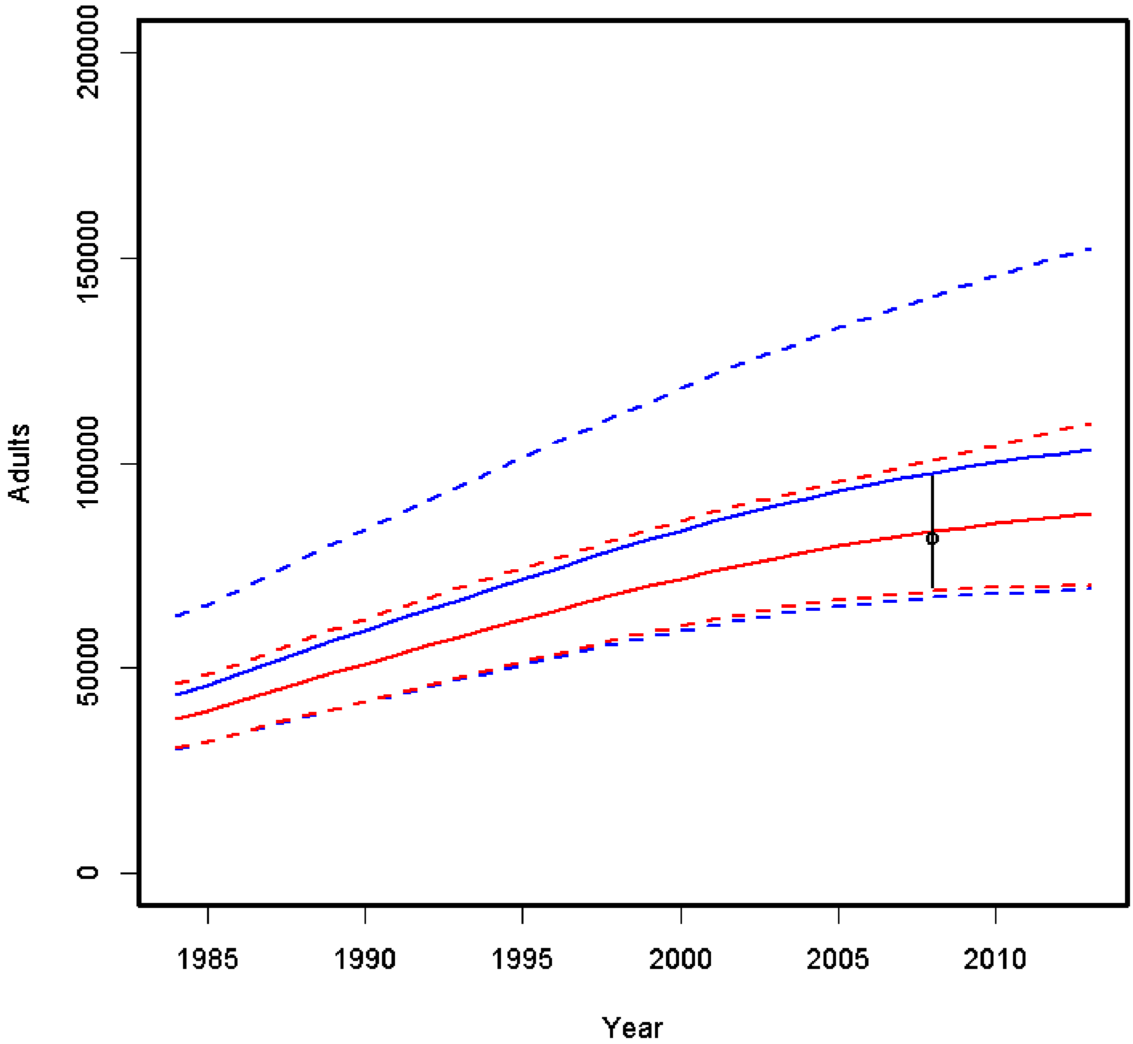
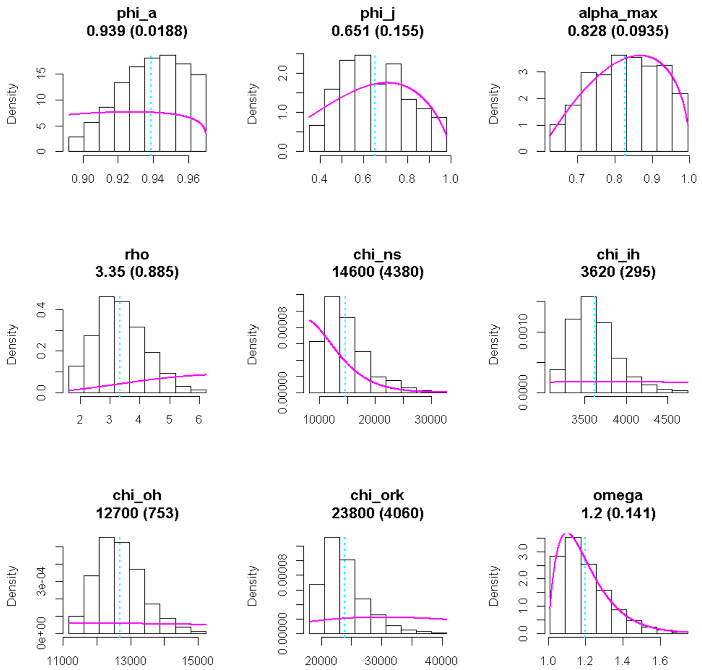
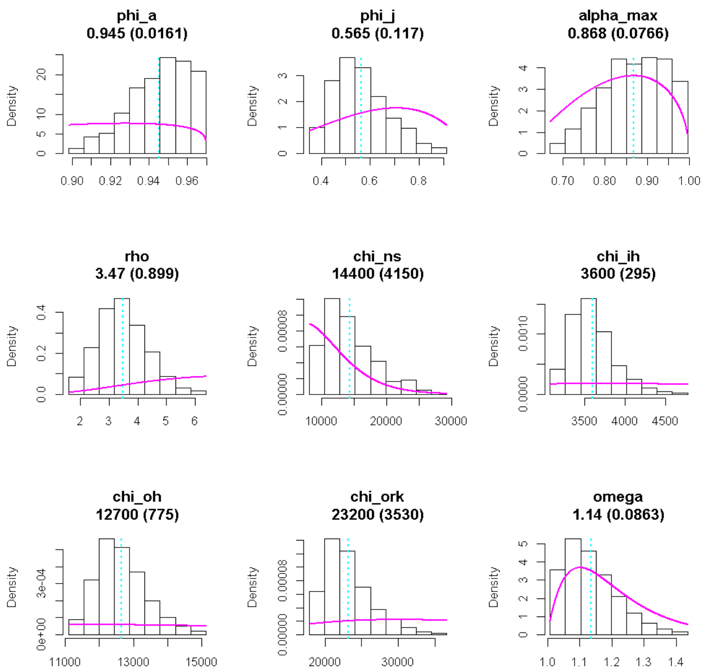


Figure 10. Posterior parameter distributions (histograms) and priors (solid lines) for the model of grey seal population dynamics, fit to pup production estimates from 1984-2012 and a total population estimate from 2008, and using a prior on adult survival constrained to have a maximum of 0.97. The vertical 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.

(a) Pup production data alone



(b) Pup production data and 2008 population estimate



Appendix 1.

Estimates of total population size, in thousands, at the beginning of each breeding season from 1984-2014, made using the 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% credible intervals in brackets.

Year	North Sea	Inner Hebrides	Outer Hebrides	Orkney	Total
1984	3.8 (3.3 4.5)	3.9 (3.4 4.7)	19.1 (16.1 22.8)	14.8 (12.7 17.4)	41.7 (35.6 49.4)
1985	4.1 (3.6 4.8)	4.2 (3.6 5)	20 (16.9 23.9)	15.8 (13.6 18.5)	44.1 (37.7 52.2)
1986	4.4 (3.9 5.1)	4.5 (3.9 5.3)	20.9 (17.8 24.8)	16.9 (14.5 19.8)	46.7 (40 55)
1987	4.7 (4.2 5.5)	4.7 (4.1 5.6)	21.8 (18.6 25.7)	18.1 (15.6 21.2)	49.3 (42.5 58)
1988	5 (4.5 5.9)	5 (4.4 5.9)	22.6 (19.3 26.8)	19.3 (16.7 22.7)	52 (44.8 61.3)
1989	5.4 (4.8 6.3)	5.3 (4.6 6.2)	23.1 (19.8 27.3)	20.7 (17.9 24.3)	54.5 (47.1 64.1)
1990	5.8 (5.1 6.8)	5.6 (4.9 6.6)	23.4 (20.2 27.6)	22.1 (19.2 26)	56.9 (49.4 66.9)
1991	6.2 (5.5 7.3)	5.8 (5.1 6.9)	23.7 (20.5 27.9)	23.6 (20.5 27.7)	59.3 (51.7 69.7)
1992	6.6 (5.9 7.8)	6 (5.3 7.1)	23.9 (20.8 28.2)	25.2 (22 29.5)	61.7 (53.9 72.5)
1993	7.1 (6.3 8.3)	6.2 (5.4 7.3)	24 (21 28.3)	26.8 (23.5 31.3)	64.1 (56.2 75.3)
1994	7.6 (6.7 8.9)	6.3 (5.6 7.5)	24.1 (21.1 28.4)	28.5 (25.1 33.2)	66.6 (58.5 78)
1995	8.1 (7.2 9.6)	6.4 (5.7 7.6)	24.2 (21.2 28.4)	30.2 (26.7 35.1)	69 (60.7 80.7)
1996	8.7 (7.7 10.2)	6.5 (5.7 7.7)	24.2 (21.3 28.4)	31.9 (28.3 37)	71.3 (63 83.4)
1997	9.3 (8.2 11)	6.6 (5.8 7.8)	24.3 (21.4 28.5)	33.5 (29.8 38.8)	73.6 (65.2 86.1)
1998	10 (8.8 11.7)	6.6 (5.8 7.9)	24.3 (21.4 28.5)	35 (31.1 40.4)	75.8 (67.1 88.5)
1999	10.6 (9.4 12.5)	6.7 (5.8 7.9)	24.3 (21.4 28.6)	36.3 (32.2 41.8)	77.9 (68.9 90.8)
2000	11.4 (10 13.4)	6.7 (5.8 8)	24.3 (21.4 28.6)	37.4 (33 43)	79.8 (70.3 93)
2001	12.2 (10.7 14.3)	6.7 (5.8 8)	24.3 (21.4 28.6)	38.3 (33.4 44.1)	81.5 (71.4 95.1)
2002	13 (11.4 15.3)	6.8 (5.8 8)	24.3 (21.4 28.7)	39 (33.7 45)	83 (72.4 97)
2003	13.8 (12.2 16.3)	6.8 (5.8 8)	24.3 (21.4 28.7)	39.5 (33.8 45.8)	84.4 (73.2 98.8)
2004	14.7 (12.9 17.3)	6.8 (5.8 8)	24.3 (21.4 28.7)	39.9 (33.9 46.5)	85.7 (74 100.6)
2005	15.6 (13.7 18.3)	6.8 (5.8 8)	24.3 (21.3 28.7)	40.3 (33.9 47.1)	87 (74.7 102.2)
2006	16.5 (14.4 19.4)	6.8 (5.8 8)	24.3 (21.3 28.8)	40.5 (33.9 47.6)	88.1 (75.4 103.8)
2007	17.4 (15.1 20.5)	6.8 (5.8 8.1)	24.3 (21.3 28.8)	40.7 (33.9 48.1)	89.2 (76 105.4)
2008	18.2 (15.5 21.6)	6.8 (5.8 8.1)	24.3 (21.2 28.8)	40.9 (33.9 48.6)	90.3 (76.4 107.1)
2009	19.1 (15.7 22.7)	6.8 (5.8 8.1)	24.3 (21.2 28.8)	41 (33.8 49)	91.3 (76.5 108.7)
2010	19.9 (15.9 23.9)	6.8 (5.8 8.1)	24.3 (21.1 28.9)	41.1 (33.8 49.4)	92.2 (76.6 110.3)
2011	20.6 (15.9 25.3)	6.8 (5.8 8.1)	24.3 (21.1 28.9)	41.2 (33.8 49.8)	93 (76.6 112.1)
2012	21.3 (16 26.8)	6.8 (5.7 8.1)	24.4 (21 28.9)	41.3 (33.8 50.1)	93.8 (76.5 113.9)
2013	22 (16 28.2)	6.8 (5.7 8.1)	24.4 (21 28.9)	41.4 (33.8 50.4)	94.5 (76.5 115.7)
2014	22.5 (16 29.7)	6.9 (5.7 8.1)	24.4 (20.9 29)	41.4 (33.8 50.7)	95.2 (76.4 117.5)

Appendix 2.

Estimates of total population size, in thousands, at the beginning of each breeding season from 1984-2014, made using the model of British grey seal population dynamics fit to pup production estimates and a total population estimate from 2008, and using a prior on adult survival constrained to have a maximum of 0.97. Numbers are posterior means followed by 95% credible intervals in brackets.

Year	North Sea	Inner Hebrides	Outer Hebrides	Orkney	Total
1984	4.3 (3.7 4.9)	4.4 (3.8 5.2)	21.3 (18.7 25.1)	16.8 (14.6 19.5)	46.9 (40.8 54.7)
1985	4.6 (4 5.2)	4.7 (4 5.5)	22.4 (19.6 26.3)	18 (15.8 20.8)	49.6 (43.4 57.8)
1986	4.9 (4.3 5.6)	5 (4.3 5.7)	23.5 (20.7 27.6)	19.2 (16.9 21.9)	52.5 (46.3 60.9)
1987	5.3 (4.7 6)	5.3 (4.6 6.1)	24.5 (21.8 28.6)	20.5 (18.1 23.3)	55.5 (49.2 64.1)
1988	5.7 (5.1 6.4)	5.6 (4.9 6.4)	25.4 (22.7 29.7)	21.9 (19.4 24.9)	58.6 (52 67.5)
1989	6.1 (5.4 6.9)	5.9 (5.2 6.8)	26 (23.3 30.3)	23.4 (20.8 26.6)	61.3 (54.7 70.6)
1990	6.5 (5.8 7.4)	6.1 (5.4 7.1)	26.5 (23.7 30.8)	25 (22.2 28.4)	64.1 (57.1 73.7)
1991	7 (6.2 7.9)	6.4 (5.6 7.4)	26.9 (24 31.2)	26.6 (23.6 30.3)	66.8 (59.4 76.7)
1992	7.4 (6.7 8.5)	6.6 (5.8 7.7)	27.1 (24.2 31.4)	28.3 (25.1 32.1)	69.5 (61.8 79.7)
1993	7.9 (7.2 9.1)	6.9 (6 7.9)	27.3 (24.3 31.5)	30 (26.6 34)	72.1 (64.1 82.6)
1994	8.5 (7.7 9.8)	7.1 (6.2 8.2)	27.4 (24.4 31.5)	31.8 (28.1 36)	74.7 (66.3 85.4)
1995	9.1 (8.2 10.4)	7.2 (6.3 8.4)	27.4 (24.5 31.5)	33.5 (29.6 37.9)	77.3 (68.6 88.2)
1996	9.7 (8.8 11.2)	7.4 (6.4 8.5)	27.4 (24.5 31.5)	35.2 (31 39.8)	79.7 (70.7 91)
1997	10.4 (9.4 12)	7.5 (6.5 8.6)	27.4 (24.5 31.4)	36.9 (32.4 41.7)	82.1 (72.7 93.7)
1998	11.1 (10 12.8)	7.5 (6.6 8.7)	27.3 (24.4 31.3)	38.4 (33.6 43.5)	84.4 (74.6 96.3)
1999	11.9 (10.7 13.6)	7.6 (6.6 8.8)	27.2 (24.4 31.2)	39.9 (34.8 45.1)	86.6 (76.5 98.7)
2000	12.7 (11.4 14.6)	7.6 (6.6 8.8)	27.1 (24.3 31.1)	41.1 (35.8 46.6)	88.6 (78.1 101.1)
2001	13.5 (12.1 15.5)	7.6 (6.6 8.8)	27.1 (24.3 31)	42.2 (36.6 47.9)	90.5 (79.7 103.3)
2002	14.4 (12.9 16.6)	7.7 (6.7 8.8)	27 (24.2 31)	43.1 (37.3 49)	92.2 (81.1 105.3)
2003	15.3 (13.7 17.6)	7.7 (6.7 8.8)	26.9 (24.1 30.9)	43.8 (37.9 49.9)	93.8 (82.4 107.3)
2004	16.3 (14.5 18.8)	7.7 (6.6 8.8)	26.9 (24 30.9)	44.4 (38.3 50.7)	95.2 (83.5 109.2)
2005	17.3 (15.3 20)	7.6 (6.6 8.8)	26.8 (24 30.8)	44.8 (38.4 51.4)	96.5 (84.4 111.1)
2006	18.3 (16 21.3)	7.6 (6.6 8.8)	26.8 (23.9 30.8)	45.1 (38.6 52.1)	97.8 (85.2 113)
2007	19.3 (16.6 22.6)	7.6 (6.6 8.8)	26.8 (23.9 30.8)	45.2 (38.7 52.7)	98.9 (85.8 114.9)
2008	20.3 (17.2 24.1)	7.6 (6.6 8.8)	26.8 (23.9 30.8)	45.3 (38.6 53.2)	100 (86.3 116.9)
2009	21.2 (17.6 25.6)	7.6 (6.6 8.8)	26.7 (23.9 30.8)	45.4 (38.6 53.8)	101 (86.6 119)
2010	22.2 (17.9 27.2)	7.6 (6.6 8.8)	26.7 (23.9 30.8)	45.4 (38.5 54.2)	101.9 (86.9 121.1)
2011	23.1 (18.2 28.8)	7.6 (6.6 8.8)	26.7 (23.9 30.8)	45.4 (38.3 54.7)	102.8 (87 123.1)
2012	24 (18.4 30.4)	7.6 (6.6 8.9)	26.7 (23.9 30.8)	45.3 (38.1 55)	103.7 (87 125.1)
2013	24.9 (18.5 32)	7.6 (6.6 8.9)	26.7 (23.9 30.8)	45.3 (38 55.4)	104.5 (87 127)
2014	25.6 (18.6 33.5)	7.6 (6.6 8.9)	26.8 (23.9 30.8)	45.2 (37.9 55.7)	105.2 (87 128.8)

Review of parameters of grey seal pup production model

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Abstract

Counts of grey seal pups from aerial surveys are used within a pup production model to provide estimates of pup production for 60 breeding colonies around Scotland. These estimates are then used within the grey seal population model, along with priors on life history parameters and an independent population estimate, to provide estimates of population size and life history parameters. The pup production model uses two key parameters to fit pup production curves to the aerial survey data: time to moult (TTM) and time to leave (TTL). The parameter values currently used in these models are from work on the Isle of May in the 1980s; TTM: 23 days, TTL 31.5 days. Here, on reviewing the current evidence from the literature and unpublished data on TTM and TTL, we found that TTL varies between colonies and that mean TTL was likely to be higher than 31.5 days. For pup count data (mostly 2008 data), the pup production models were rerun using two values for TTL (extracted from the literature) and also allowing the model itself to estimate colony-specific TTLs - this is only possible if five or more pup counts are conducted at a colony in one breeding season. The weighted (by maximum pup count) mean TTL was estimated to be 37.2 days but TTL varied between colonies. There was no significant pattern by region but TTL in Orkney was generally higher than elsewhere. Allowing the model to estimate TTL, pup production in 2008 was between 10.6 and 11.5% lower than previously estimated (using the standard TTL of 31.5 days). Our limited evidence from two regions across two years, suggests this percentage difference would be similar across years. These results have important implications for the population model. We discuss the further work required on this issue, both for historic pup count data and for future count data, recognising that it may not be possible to conduct the five flights required to estimate TTL for each colony, each survey year.

Introduction

Grey seal pups born at the majority of colonies are counted by aerial survey except in Eastern England where ground counts are conducted. The aerial survey counts are inputted into a pup production model to provide estimates of pup production. These estimates are then used within a population model, along with priors on life history parameters and an independent population estimate, to provide estimates of population size and life history parameters. The pup production uses two key parameters to fit pup production curves to the aerial survey data: time taken to fully moult (TTM) and time to leave the breeding colony (TTL). The parameter values currently used in these models are from work at two sites with differing habitat (one a low lying inland habitat and the other a cliff-backed beach) on the Isle of May; TTM: 23 days, TTL 31.5 days (Wyile 1988). Here, using unpublished data and available literature, we review current evidence surrounding TTM and TTL.

Time to Moult

TTM moult is defined here as the age at which a pup is fully moulted (<5% lanugo). The TTM observed by Wyile (1988) was 21 days (n=92) but some pups had left before completing their moult or were not fully moulted by the end of the field season. A model accounting for these pups estimated TTM at 23 days (SD = 5). Wyile (1988) noted that this TTM value may still be an underestimate. Although there was no significance difference in the estimated age that males and females started moulting, TTM was significantly lower in males; Wyile (1988) speculated this was a result of males growing faster and growth rate being associated with TTM. There was a small effect of year on TTM but no effect of habitat (the two sites) on TTM. Comparisons with other studies need to be made with care because in some studies moulting age (age at start of moult) is considered rather than age at which moult is completed (TTM).

More recently (2010), the mean TTM observed on the Isle of May (Bennett unpublished data) was 23.14 days (n=23). This is reassuringly close to the findings of Wyile (1988). Both studies were conducted on the Isle of May so no spatial variation would be represented. Data are available from the long-term North Rona study but this study is focussed on lactation. While time to start of moult was recorded, the time to completion of moult was often not known unless it occurred during lactation. We have not used these data here but they could be combined with other data (i.e. time spent moulting) to generate estimates of TTM. The mean estimate of TTM from Sable Island (Bowen et al. 2003) was 22.25 (n=49). The closeness of the means from the limited number of studies provides no evidence that the current TTM value of 23 days is inappropriate, but this should be reviewed as more data are collected.

Time to leave (TTL)

Wyile (1988) apportioned TTL into two categories: voluntary (vTTL) and involuntary (inTTL). vTTL was defined as departures on or after 10 days of age and had an observed mean of 24.84 days (n=129). Removing starvelings, mean vTTL was observed to be 25.22 days (n=117). The observed mean was an underestimate of vTTL because 28% of the study pups were still present when the field study ended and of these, 69% of those pups which remained were older than the average vTTL. It should be noted that the 10 day threshold for vTTL is arbitrary. In reality, leaving prior to moult is unlikely to be voluntary. Most of the pups that are washed off before they are moulted, presumably haul out elsewhere (and thus are available to be counted) or die. The literature suggests that the proportion of involuntary departures will depend on both the topography on the breeding site and the weather conditions. Indeed, Wylie found that the proportion of pups that disappeared before moulting was significantly higher on the cliff-backed beach site compared to the inland site. The observed TTL (combined vTTL and inTTL) for the Isle of May was 23.09 (n=141) but for reasons noted above, this was an underestimate. Taking into account those pups which had not left by the end of the study, TTL was estimated to be 31.5 days (SD=7). It was significantly different for males and females, being later in males. Although there were significant differences between the two study sites within year, the direction of these differences differed with year. TTL appeared to decrease as the season progressed but inclusion of this change in the pup production model did not improve estimates of pup production. The estimated TTL did not vary between the cliff-backed beach and the inland site. Wyile (1988) noted that 31.5 days was likely to be an underestimate of the true mean TTL on these sites.

Due to the paucity of data and literature on TTL, we also examined data and studies on its constituent parts: the duration of lactation and the duration of the post weaning fast (PWF). Evidence suggests that lactation duration does not affect leaving age so lactation duration (Noren et al. 2008) and PWF duration can be considered both independently and combined. Although there is considerable variation in lactation duration, the mean is reasonably consistent across studies both in the UK (Pomeroy & Fedak 1999, Bennett unpublished data; Pomeroy unpublished data) and at Sable Island (Boness & James 1979, Noren et al. 2008, Lang, Iverson & Bowen 2009, 2011). The mean seems to be between 17 and 19 days, although it does appear plastic as it is shorter for ice-breeders (Baker, Barrette & Hammill 1995).

Estimates of PWF duration and leaving age from the UK are limited and appear variable. Furthermore, most of these consider pups which have already weaned and for the most part exclude involuntary departures so in some places their estimates are likely to be maximums. A study on North Rona (Reilly 1991) estimated a mean PWF duration of 16 days (SE=0.3) which would lead to a leaving age of 34 days but the result was based on only 8 pups and was probably an underestimate as the study ended prior to all animals leaving. A later study on the Isle of May reported a mean leaving age of 40.59 days (n=46, Bennett et al. 2010, Bennett unpublished data). This estimate may be biased upwards as all animals were penned and were released from the pens at a mean age of 34.83 days (range 26 - 48 days). However, the bias may be minimal as only 8 of the 46 individuals left the day they were released. In addition, another study on the Isle of May conducted by Robinson (unpublished data) in 2010 showed that 7 of 8 individuals had not left the island at the last capture attempt which was 13-15 days post weaning. Assuming 18 day lactation duration, this would result in a mean minimum leaving age of 31.75 (n=8). Although these animals were penned, all were released within 7 days of post weaning and thus by age 25 days, and all were recaptured four days later so it is unlikely the penning affected PWF duration or leaving age. There are also hat tag data from the Farne Islands – it shows a very short PWF duration (mean=10, SD=5) which would result in an estimated leaving age of only 28 days. However, age was estimated and so may be inaccurate. The study with the highest sample size and focussed on PWF duration is from Canada and estimates a mean PWF duration of 21 days and leaving age of 40 days (SE=1.1, Noren et al. 2008). It is likely that PWF duration is dependent on habitat and is likely to be quite plastic (Jenssen et al. 2010).

The current 31.5 day estimate for TTL used in the pup production model is close to the minimum estimated value from the literature and unpublished data. To test the sensitivity of the pup production estimate to this value we have rerun the model using values of 34 days (Reilly 1991) and 40 days (Noren et al. 2008; maximum reported mean). Recognising that TTL is likely to depend on weather and topography we have also estimated TTL in colonies for which five or more counts were conducted. We reran the pup production model for 2008, a year for which most colonies were counted at least five times. To compare between years we also ran the model and estimated TTL for 2012 in Orkney and 2014 in Inner Hebrides.

Materials and Methods

The pup production model was rerun under four scenarios: (i) the standard model – TTL set at 31.5 days, (ii) TTL set at 34 days, (iii) TTL set at 40 days, (iv) TTL estimated within the pup production model.

We used the resulting pup production estimates and TTL estimates (under scenario iv) to investigate the following questions:

1. **How do estimates of TTL from the pup production vary between colonies, regions and years?**
2. **How do estimates of pup production compare to the maximum count?**
3. **How do estimates of pup production vary with the four scenarios?**
4. **How do estimates of TTL and pup production change when colonies are combined?**

Results and Discussion

1. **How do estimates of TTL from the pup production vary between colonies, regions and years?**

In 2008, TTL could be estimated for all colonies (n) in Firth of Forth (n=3), Inner Hebrides (n=11), Outer Hebrides (n=14) and Orkney (n=26). These estimates were all based on five or more flights, with the exception of one colony in the Inner Hebrides for which TTL estimates were based on only four flights so may have been unreliable; this colony was excluded from investigations of estimates TTL. There was considerable variation in estimated TTL between colonies, especially in small colonies (indicated by maximum pup count; Figure 1). It is unclear whether this is a result of unreliable estimates of TTL in small colonies or whether larger counts simply integrate out the variation in sub-sites around the colonies. For three colonies in 2008 (one each in Firth of Forth, Outer Hebrides and Orkney), TTL was estimated to be 50, which was the maximum value allowed to be estimated by the model.

There was no significance difference in TTL between regions (Figure 2; ANOVA: $F_{3,47} = 1.9$, $P > 0.1$). Regional and overall mean TTLs were lower and more similar if weighted by an index of colony size (maximum count; Table 1). Nevertheless colonies in Orkney show a consistently high TTL which, due the large number of pups born there, raised the overall weighted mean to 37.2 days.

Figure 1. Estimated time to leave (TTL) shown with the maximum pup count for each colony.

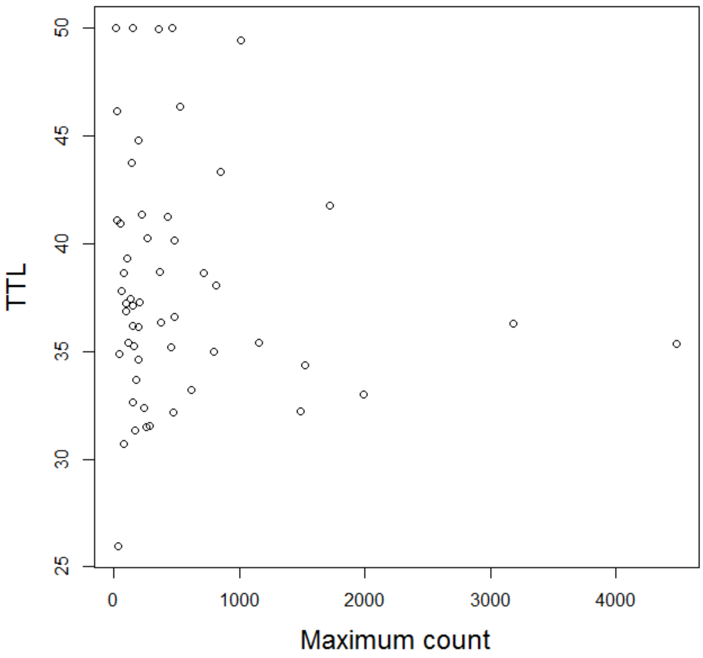


Figure 2. Estimated TTL by region in 2008. The lines represent the TTL for scenarios i-iii.

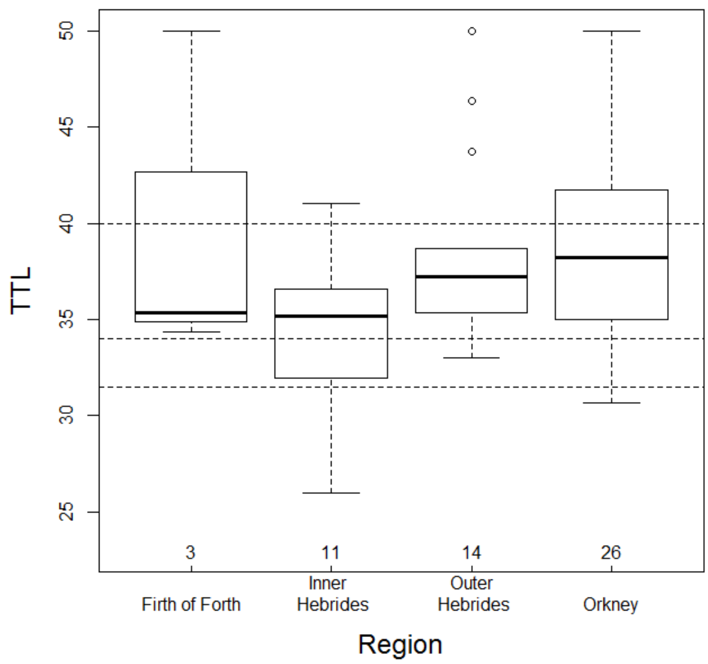


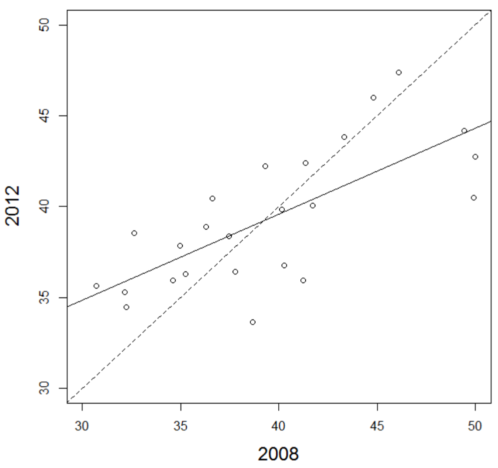
Table 1. TTL by region in 2008, estimated from the pup production model.

Region	All colonies			Colonies with estimate TTL of 50 removed		
	median	mean	weighted mean	median	mean	weighted mean
Firth of Forth	35.4	39.9	35.6	34.9	34.9	34.8
Inner Hebrides	35.2	34.6	34.0	35.2	34.6	34.0
Outer Hebrides	37.2	38.5	35.9	37.2	37.6	35.8
Orkney	38.2	39.1	38.9	37.8	38.6	38.5
All	37.0	38.0	37.2	36.6	37.3	36.9

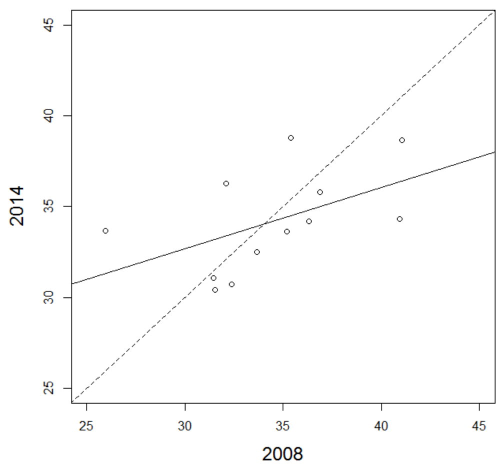
In addition to the estimates in 2008, TTL could be estimated in Orkney and Inner Hebrides in 2012 and 2014, respectively. This allowed the year to year consistency of estimated TTLs to be examined. In Orkney there was a strong relationship between TTL estimated for colonies in 2008 and in 2012 (linear model: $F_{1,22} = 23.4$, $P < 0.0001$, adjusted $R^2 = 0.49$). The relationship was weaker and non-significant at the 5% level in the Inner Hebrides (linear model: $F_{1,10}=3.4$, $P < 0.1$, adjusted $R^2 = 0.18$). In both regions the best fit relationship between the colony TTLs between years was not one to one (Figure 3). However, it should be noted, that this comparison is between a film survey (2008) and digital surveys (2012, 2014) which may have affected the reliability and thus comparability of TTL estimates.

Figure 3. The relationship between TTL estimated for two years in (a) Orkney and (b) Inner Hebrides with a line of best fit. The 45 degree line (one to one relationship) is shown as a dashed line.

(a)



(b)



2. **How do estimates of pup production compare to the maximum count?**

As expected, the ratio of maximum pup counts to pup production varied greatly with colony and also TTL scenario (Table 2). It also varied between years; for example in the Inner Hebrides, it was 0.82 and 0.73 of the total estimated pup production under scenario (i), in 2008 and 2014 respectively. Thus maximum count should only be used as an index of pup production with caution.

Table 2. The ratio between estimated pup production and maximum count in 2008 under different TTL scenarios. This includes the ratio of the total max count to the total estimated pup production under each scenario.

Region	Maximum count	Ratio to max count to pup production estimate under each TTL scenario			
		(i)31.5	(ii) 34	(iii) 40	(iv) estimated TTL
Firth of Forth	2,830	0.89	0.91	1.00	0.93
Inner Hebrides	2,703	0.82	0.86	0.97	0.85
Outer Hebrides	9,767	0.80	0.85	0.95	0.87
Orkney	14,435	0.78	0.86	0.94	0.92
Total	29,735	0.80	0.86	0.96	0.90

3. How do estimates of pup production vary with the four scenarios?

Estimating TTL produces a pup production estimate which is 11.5% lower than the standard estimate of pup production (scenario (i); Table 3). However, this disparity only applies to pup production estimates from air surveys and would not affect estimates from ground counted colonies (3,271 pups in 2008, c. 8%, of the estimated pup production for annually monitored colonies). If we take these into account then pup production of the annually monitored colonies (using estimated TTL) would be 10.6% lower than the standard estimate in 2008.

Please note that this calculation does not consider colonies for which pup production estimates in 2008 had to be extrapolated from previous years' counts or for which less than five surveys were conducted; these colonies produced an estimated 5,291 pups in 2008 (c. 11.5% of the total estimated pup production). The majority of the counts on which extrapolations were based, were conducted by aerial survey and thus if the data were available, the new estimate (using estimated TTL) would likely represent a similar percentage of the standard estimates as in aerial surveyed colonies considered here. Thus the overall pup production in the UK in 2008, using estimated TTL would likely have been between 10.6 and 11.5% lower than the current estimate of 45,943 pups.

Table 3. Estimates of pup production in 2008 under different TTL scenarios.

Region	Pup production estimates				Estimates as percentage of the production using TTL of 31.5 days		
	(i) 31.5	(ii) 34	(iii) 40	(iv) estimated	(ii) 34	(iii) 40	(iv) estimated
Firth of Forth	3,195	3,099	2,816	3,044	97.0	88.1	95.3
Inner Hebrides	3,315	3,126	2,777	3,174	94.3	83.8	95.7
Outer Hebrides	12,253	11,531	10,232	11,164	94.1	83.5	91.1
Orkney	18,618	16,869	15,276	15,708	90.6	82.0	84.4
Total	37,381	34,625	31,101	33,090	92.6	83.2	88.5

It is likely that in other years there would be a similar percentage difference between the pup production using estimated TTL and the standard TTL of 31.5 days, as there was in 2008. In regions for which we have analysed data for multiple years (Orkney and Inner Hebrides), the percentage difference between the pup production estimated using estimated TTL and the standard TTL of 31.5

days in the two years considered was within 1% of each other. The mean regional TTL estimates in the two years were within half a day of each other.

4. How do estimates of TTL and pup production change when colonies are combined?

The definition of a colony is not always straightforward and can comprise multiple habitats (e.g. inland and cliff-backed beaches on the Isle of May). Combining colonies which have different TTL may generate bimodal pup production curves. In two colonies, in both the Inner Hebrides (2008 and 2014) and Orkney (2008 and 2012), TTL and pup production was estimated both for the individual two adjacent colonies and combined. We used this limited data to investigate the potential biases associated with how colonies are defined. We found that despite large differences in the mean TTL between the two colonies (between 3 and 9 days), the percentage difference in total estimates for the two colonies (when modelled separately and together) were small (between 2 and 3%).

Conclusions and further work

The evidence (both from literature, unpublished data, and the estimated TTLs from the pup production model) suggests that the current value of TTL used in the pup production model (31.5 days) is too low. Using more appropriate values suggests that pup production in 2008 was between 83.2 and 92.6% of the current estimate for colonies considered here (Table 3). The literature also suggests the TTL varies with habitat and weather conditions, and our current study suggests that it varies between colonies and year. Thus it may be more accurate to estimate TTL within the pup production model. Doing this, the estimate of pup production for colonies considered here would be 88.5% of the current estimate for 2008.

However, estimating TTL is not straightforward. It may be biased for very small colonies and would be influenced by the presence of pups which have already left another colony and hauled out on the colony being counted. Furthermore, the historic pup count data often did not encompass five flights, so it will not be possible to provide estimates of pup production based on colony estimates of TTL for the historic pup data. The results from this study indicate that TTL for each colony varies from year to year and thus the values from one year may not be appropriate for other years. Regional TTLs may be much less variable between years. The limited evidence suggests that combining colonies with different TTL values do not have a large effect on the overall population estimate. It may also reduce the impact on the estimated TTLs and pup production of pups leaving one colony and hauling out at another. Thus we propose that the 2008 data (for which there were five counts for the majority of the colonies) are used to conduct a sensitivity analysis. We will fit the pup production model on a regional level using the estimated regional TTLs and compare the results with TTLs estimated on a colony level. To examine the reliability of the TTL estimates, we will also compare estimates of pup production when six counts were conducted and use degraded data (five counts) to examine how this affects estimated pup production. In addition, we will further investigate the consistency of TTL estimates between years within one type of photography (film or digital) to check the results above for which we compared years of film and digital photography. For future counts, the digital images are of higher quality and thus we will investigate whether additional categories to white coat and moulted pups can be used. This additional data may mean that TTL could be estimated with fewer than five flights. The current pup production model created by Lex Hiby is coded in FORTRAN and it is not straightforward to manipulate. In light of the development of both statistics and software available, we may want alter and update its structure. Thus we will rewrite the model in an appropriate language, in order to conduct the testing described above.

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The status of UK harbour seal populations in 2014, including summer counts of grey seals

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Abstract

In August 2014, during the harbour seal moult, the Sea Mammal Research Unit (SMRU) surveys in Scotland covered a large section of the Scottish west coast, between Ullapool and the Firth of Lorn. Part of the Moray Firth and the Firth of Tay and Eden Estuary were surveyed along with some of the more distant off-lying Scottish islands. The SMRU surveys in England covered the coast of Lincolnshire, Norfolk and Suffolk. The Tees Seal Research Programme kindly provided information on seal numbers in the Tees Estuary (Woods, 2014). Data from surveys carried out in the Thames Estuary, by the Zoological Society of London, are included in the total for England. Grey seals are counted during harbour seal surveys although during the summer months, grey seal counts can vary more than harbour seal counts.

From August surveys carried out between 2007 and 2014, the minimum number of harbour seals counted in Scotland was **23,355** and in England & Wales **4,806** making a total count for Great Britain of **28,161** (Table 1). Including **948** harbour seals counted in Northern Ireland in 2011, the UK harbour seal total count for this period was **29,109**.

From August surveys carried out between 2007 and 2014, the minimum number of grey seals counted in Scotland was **20,449** and in England & Wales **9,708** making a total count for Great Britain of **30,157** (Table 2). Including **468** grey seals counted in Northern Ireland in 2011, the UK grey seal total count for this period was **30,625**.

In the annually surveyed part of the Moray Firth (Helmsdale to Findhorn), the moult count was the lowest ever recorded for this area. The severe decline in the Firth of Tay & Eden Estuary harbour seal SAC continued, with the 2014 moult count (29) being the lowest recorded to date, 42% lower than the 2013 count (50). This new count suggests that only 6% of the average population counted between 1990 and 2002 currently remain within this harbour seal SAC. No additional declines have been identified in other parts of the UK, for which new data is available (i.e. east coast of England, W Scotland), where populations seem to be stable or possibly even increasing. Surveys planned for August 2015 will hopefully complete the current round-Scotland survey.

Introduction

Most surveys of harbour seals are carried out in August, during their annual moult. At this time of their annual cycle, harbour seals tend to spend longer at haul-out sites and the greatest and most consistent counts of seals are found ashore. During a survey, however, there will be a number of seals at sea which will not be 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, not actual 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, the Moray Firth and in east and southeast England, between Lincolnshire and Kent (Figure 1). Only very small, dispersed groups are found on the south and west coast of England or in Wales.

Since 1988, SMRU's surveys of harbour seals around the Scottish coast have been carried out on an approximately five-yearly cycle, with the exception of the Moray Firth (between Helmsdale and

Findhorn) and the Firth of Tay & Eden Estuary SAC which have been surveyed annually since 2002. Surveys carried out in 2006, revealed significant declines in harbour seal numbers in Shetland, Orkney and elsewhere on the UK coast (Lonergan *et al*. 2007). Between 2007 and 2009, SMRU surveyed the entire Scottish coast including a repeat survey of some parts of Strathclyde and Orkney. In 2010, Orkney was surveyed again to determine whether previously observed declines continued. A new round-Scotland survey started in 2011 and is due for completion in 2015. A complete survey of Northern Ireland and the Republic of Ireland was carried out in 2011 and 2012.

In England, the Lincolnshire and Norfolk coast holds approximately 90% of the English harbour seal population and is usually surveyed twice annually during the August moult. Since 2004, additional breeding season surveys (in early July) of harbour seals in The Wash (which lies within the August survey area) were undertaken for Natural England. The Suffolk, Essex and Kent coasts were last surveyed by SMRU during the breeding season in 2011 and during the moult in August 2014 by the Thames Harbour Seal Conservation Project, run by the Zoological Society of London.

Methods

Seals hauling out on rocky or seaweed covered shores are well camouflaged and difficult to detect. Surveys of these coastlines in Scotland are carried out 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, since 2007, oblique photographs were obtained using a hand-held camera equipped with an image-stabilised zoom lens. Both harbour and grey seals were digitally photographed and the images used to classify group composition. The grey seal counts from these images have previously been used to inform the models used to estimate the total grey seal population size (Lonergan *et al*. 2011, SCOS BP 10/4).

Surveys of the estuarine haul-out sites on the east coast of Scotland and England were by fixed-wing aircraft using hand-held oblique photography. On sandbanks, where seals are relatively easily located, this survey method is highly cost-effective.

To maximise the counts of seals on shore and to minimise the effects of environmental variables, surveys are restricted to within two hours before and two hours 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 because seals will increasingly abandon their haul-out sites and return into the water, and because the thermal imager cannot 'see' through rain.

In southeast England, from Suffolk to Kent, the Thames Harbour Seal Conservation Project coordinated August surveys by air, from boat and from land on three days in August 2014 (Barker, 2015).

Results and Discussion

1. Minimum population size estimate for harbour seals in the UK

The overall distribution of harbour seals around the British Isles from August surveys carried out between 2007 and 2014 is shown in Figure 1. For ease of viewing at this scale, counts have been aggregated by 10km squares.

The most recent minimum harbour seal population estimates (i.e. counts between 2007 and 2014) for UK seal management units (SMUs) are provided in Table 1 and are compared with two previous periods (2000 to 2006 and 1996 to 1997). Estimates for Ireland are also given for the two most recent periods.

Mean values were used for any areas where repeat counts were available (primarily in eastern England and occasionally the Moray Firth).

The most recent minimum estimate of the number of harbour seals in Scotland, obtained from counts carried out between 2007 and 2014, is **23,355** (Table 1). This is virtually the same as the 2000-2006 count (23,423) and 20% lower than the 1996-1997 count (29,514; Table 1). Since 2001, harbour seal counts have declined in Shetland, Orkney and along the north and east coasts of Scotland (Lonergan *et al*., 2007; Duck & Morris, 2014) while counts in the West Scotland Seal Management Area appear to have increased.

The most recent minimum estimate for England & Wales, obtained from surveys carried out mainly in 2014, is **4,806** (Table 1). This is 58% higher than the 2005-2006 count and 46% higher than the 1996-1997 count (which includes some data from 1995).

The 2011 count for Northern Ireland of **948** was 25% lower than the previous complete count in 2002 (1,267).

The sum of all the most recent counts carried out between 2007 and 2014 gives a UK total count of **29,109** harbour seals (Table 1).

1.1. Grey seals in the UK counted during August harbour seal surveys

Grey seals are counted in all harbour seal surveys but, because grey seal counts are significantly more variable than harbour seal counts in August, they have not previously been fully reported. In conjunction with grey seal telemetry data, the grey seal summer counts from 2007 and 2008 have been used to calculate an independent estimate of the size of the grey seal population (Lonergan *et al*. 2011). August grey seal counts will similarly be used in future.

The overall UK and Ireland distribution of grey seals from August harbour seal surveys carried out between 2007 and 2014 is shown in Figure 2. For ease of viewing at this scale, counts have been aggregated by 10km squares. The most recent estimate of the number of grey seals in Scotland, obtained from August counts carried out between 2007 and 2014 is **20,449** (Table 2). This is very similar (2% lower) to the total Scotland count of 20,813 from August surveys between 2000 and 2006.

There were **8,408** grey seals counted in eastern England in 2008 to 2014 and combined with an estimate of **1,300** in West England & Wales and the 2011 count of **468** in Northern Ireland (Table 2), the most recent UK total count of grey seals in August is **30,625**.

2. Harbour seals in Scotland

The survey area for August 2014 comprised the west coast of Scotland from Ullapool to the Firth of Lorn. Details of the survey can be found in the Scottish Natural Heritage (SNH) Commissioned Report (Duck & Morris, 2015).

Figure 3 shows when each part of the Scottish coast was last surveyed between 2007 and 2014. Areas surveyed in 2014 are in black; areas in red were last surveyed in 2007 and most urgently require updating.

The most up to date distribution of harbour seals in Scotland, from surveys between 2007 and 2014, is shown in Figure 4. The trends in counts of harbour seals in different Seal Management Areas in Scotland, from surveys carried out between 1996 and 2014 are shown in Figure 6. Harbour seal counts from the most recent surveys and from two previous survey periods (2000 to 2006 and 1996 to 1997) are in Table 1.

2.1 West Scotland

The current count of harbour seals in the large West Scotland Management Area is **13,878** from surveys carried out in 2009, 2013 and 2014 compared with 9,972 from the previous survey carried out in 2007 and 2008, 11,702 from surveys between 2000 and 2005 and 8,811 from surveys in 1996 and 1997 (Table 1). The West Scotland harbour seal count increased by 39% between 2008 and 2014, equivalent to an average annual increase of 5.7%.

2.1.1 West Scotland - North

Most of West Scotland - North was surveyed in August 2013 (Duck & Morris, 2014), only a small section from the head of Loch Broom to Rubha Reidh was surveyed in 2014. A total of **1,115** harbour seals were counted in 2013 and 2014 compared with 692 in 2008 (Table 3). This represents an overall increase of 61 or an average annual increase of 8.3% and is in marked contrast to the declines in harbour seals numbers observed in Orkney and the North Coast, in Shetland and on the East Coast.

2.1.2 West Scotland - Central

All of West Scotland - Central was surveyed in August 2014. A total of **6,424** harbour seals were counted compared with 4,004 counted in 2007 and 2008 (Table 3). This represents an overall increase of 60% or an average annual increase of 8.2% very similar to that observed in West Scotland - North. The highest count of harbour seals was recorded in 13 of the 16 subregions that comprise West Scotland - Central.

2.1.3 West Scotland - South

Only the northern part of West Scotland - South was surveyed, from Ardnamurchan Point to the Firth of Lorn, opposite Scarba. In this area, a total of **4,230** harbour seals were counted in 2014 compared with 3,031 in 2007 (Table 3). This represents an overall increase of 40% over seven years or an average annual increase of 4.9%.

2.2 Moray Firth

Detailed breeding and moulting season ground-counts of harbour seals in inner subarea of the Moray Firth (from Loch Fleet to Ardersier) were collected annually by Aberdeen University's Lighthouse Field Station between 1988 and 2005. These ground-based counts are shown in Figure 8 (moulting season counts) and Figure 9 (breeding season counts, excluding pups). SMRU's aerial survey counts for the same areas are included, together with counts from adjacent haul-out sites which lie to the north-east of Loch Fleet and to the east of Ardersier (harbour seals: Table 3, Figure 7; grey seals: Table 4). A detailed view of the part of the Moray Firth surveyed by SMRU, together with the August counts of harbour and grey seals in 2014, is shown in Figure 10.

2.2.1 Moray Firth – harbour seal moult season counts (August)

SMRU's August aerial surveys of harbour seals in the Moray Firth started in August 1992 and the counts are shown in Table 3 with the trends in different parts of the Moray Firth in Figure 8. The counts represent a combination of both thermal imaging and fixed-wing surveys of the area. Between the mid-1990s and 2007, counts indicated a decline in the Moray Firth harbour seal population. This may, at least in part, have been due to a bounty system for seals which operated in the area at the time (Thompson *et al*., 2007; Matthiopoulos *et al*., 2014). There is considerable variability in the August total counts for the entire Moray Firth although there seems to have been a decline over the past three years.

There have been some obvious changes in harbour seal distribution within smaller parts of the Moray Firth. Following a significant decline between 1992 (662) and 2002 (220), harbour seal numbers within the Dornoch Firth and Morrich Mor SAC now appear to be continuing to decline with the 2014 count (111) the lowest recorded. A decline has also been observed in the Beauly Firth where over 200 harbour seals were regularly counted in the 1990s, but only between 30 and 60 counted since 2011. In contrast, harbour seal numbers in Loch Fleet have increased since the 1990s, with 156 counted in August 2014 being the highest count recorded during SMRU's August aerial surveys. The most noticeable increase in recent years, however, was at Culbin Sands between Findhorn and Nairn. Up to 2009, harbour seal counts at Culbin rarely reached double figures, whereas in August 2014, 236 were counted. Harbour seals have recently started to haul out more regularly at a site by Milton in the Inverness Firth.

Causes for these changes have not been identified, but it is possible that the ever changing sandbank system in the Beauly Firth has become less suitable for seals to haul out compared with other available sites in the near vicinity.

2.2.2 Moray Firth – harbour seal breeding season counts (June & July)

During the 2014 breeding season, SMRU completed four aerial surveys of harbour seals in the Moray Firth between 20th June and 11th July. The mean number of adults counted during these surveys, with standard errors, is shown in Figure 10. Following a long period of decline in breeding season haul-out group size from 1993 to 2007 and an increase in 2009 and 2010, numbers have declined over the last three years. As during the moult, this is partly due to a significant reduction in seals using the Beauly Firth which used to be the main pupping site in the Moray Firth. Whereas the maximum pup count in 2010 was 172, it was never higher than 10 in 2013. While the mean count for the 2014 Moray Firth breeding season surveys, between Helmsdale and Findhorn, was 680, almost the same as the 2013 count of 693 (1.5% lower), the 2014 mean count between Loch Fleet and Ardersier was 429 compared with 511 in 2013 (16% lower).

2.3 Firth of Tay & Eden Estuary Special Area of Conservation (SAC)

The Firth of Tay and Eden Estuary SAC is shown in Figure 12 with the distribution and numbers of harbour seals counted during the August 2014 survey.

The 2014 harbour seal moult count for the SAC (29) was 42% lower than the 2013 count of 50 (Figure 14; Table 5). The 2014 count is a new all-time low for this harbour seal SAC and represents only 5% of the mean of counts between 1990 and 2002 (641). Harbour seals in this area are of sufficient concern that Marine Scotland has not issued any licences to shoot harbour seals within the East Scotland Management Area since 2010.

The numbers of grey seals counted in the Firth of Tay & Eden Estuary SAC during harbour seal moult surveys are in Table 6.

3. Harbour seal surveys in England & Wales

3.1. England & Wales – harbour seal moult season counts (August)

The coast of England and Wales has been divided into three Management Units (Figure 1). In Northeast England, small numbers of harbour seals are found at Holy Island and in the Tees Estuary. The 2014 count for Northeast England was 90, a combined count from 2008 (Holy Island) and 2014 (Tees Estuary; Woods, 2014). Harbour seals in the Tees Estuary are monitored by the Industry Nature Conservation Association (INCA). The very slow increase in numbers seems to be continuing, with the August 2014 mean count of 81 being the highest since recording began in 1988 (Woods, 2014). The number of pups born in the Tees Estuary is low, but has been increasing slowly over the last ten years (19 pups born in 2014; Woods, 2014).

The great majority of English harbour seals are found in Southeast England (Figure 1). In 1988, the previously increasing numbers of harbour seals in The Wash declined by approximately 50% as a result of the phocine distemper virus (PDV) epidemic. Following the epidemic, from 1989, the area has been surveyed once or twice annually in the first half of August (Table 7, Figure 14). After recovering to 1988 levels by 2001, the population was hit by another PDV outbreak in 2002. It was reduced by around 20% but recovered to pre-epidemic levels by 2012.

One aerial survey of harbour seals was carried out by SMRU in Lincolnshire and Norfolk during August 2014 (Table 7). The 2014 count for this area from Donna Nook to Scroby Sands (4,192) was slightly higher than the 2013 count (4,022), and almost identical to the 2012 count (4,189). The Zoological Society of London surveyed the wider Thames area between Hamford Water (in Essex) and Goodwin Sands (off the Kent coast) and counted 489 harbour seals (Barker, 2015), the highest count recorded for this area.

The combined counts for the Southeast England Management Unit (Flamborough Head to Newhaven) in 2014 (4,681) was very similar to the 2013 count (4,504; Tables 1 and 7). Although the Southeast England population has returned to its pre-2002 epidemic levels, it is still lagging behind the rapid recovery of the harbour seal population in the Wadden Sea where counts increased from 10,800 in 2003 to 26,788 in 2013 (Reijnders *et al*., 2003; Trilateral Seal Expert Group, 2013), equivalent to an average annual growth rate of 9.5% over the last ten years. There was a slight decline in the Wadden Sea count in 2014 (26,576; Trilateral Seal Expert Group, 2014).

No dedicated harbour seal surveys are routinely carried out in the West England & Wales Management Unit. Estimates given in Table 1 are derived from compiling information from various different sources listed in the Table.

3.2. England & Wales – harbour seal breeding season counts (June & July)

The only regular harbour seal breeding season surveys in England & Wales are the annual SMRU aerial surveys around The Wash. A single survey conducted around the expected peak date (30 June 2014) produced a count of 1,802 pups and 4,020 older seals (1+ age classes) compared with1,308 pups and 3,345 older seals in 2013 and 1,496 pups and 3,551 older seals in 2012. Estimated peak pup counts have increased at an average rate of 9% p.a. since 2003 although there is considerable variation about the fitted exponential (R^2=0.8).

4. UK harbour seal surveys in 2015

4.1 Harbour seal surveys in 2015 – breeding season

Only two of five planned breeding season fixed-wing surveys were carried out in the Moray Firth in June and July 2015 due to adverse weather conditions and unserviceable aircraft. The survey results will be presented to SCOS in 2016.

Four breeding season fixed-wing survey were carried out around The Wash in June and July 2015. The results will be presented to SCOS in 2016.

4.2 Harbour seal surveys in 2015 – moult season

In Scotland in 2015, the remainder of the Scottish coast (not surveyed since 2009) will be surveyed, weather permitting. The area to be covered includes West Scotland from Craobh Haven to the tip of the Mull of Kintyre, all of Southwest Scotland (Firth of Clyde and Solway Firth), part of Southeast Scotland from the Farne Islands to Aberlady Bay, and Shetland.

As in previous years, a single fixed-wing survey will be carried out during August in the Moray Firth (between Helmsdale and Findhorn) as well as in the Firth of Tay & Eden Estuary SAC.

In Southeast England SMRU intends to carry out two August surveys of the coast between Donna Nook and Scroby Sands. In addition, the Zoological Society of London intends to carry out two surveys of the Essex and Kent coasts.

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Table 1. The most recent August counts of harbour seals at haul-out sites in Britain and Ireland by Seal Management Unit compared with two previous periods, in 1996 and 1997 and between 2000 and 2006. The grey bars represent a histogram of the counts by region to indicate the relative proportion each contributes to the total count.

Seal Management Unit / Country		Harbour seal counts 2007-2014		2000-2006		1996-1997	
1 Southwest Scotland		834	(2007)	623	(2005)	929	(1996)
2 West Scotland	a	13,878	(2007-2009; 2013-2014)	11,702	(2000; 2005)	8,811	(1996-1997)
2a West Scotland - South		6,339	(2007; 2009; 2014)	7,037	(2000; 2005)	5,651	(1996)
2b West Scotland - Central		6,424	(2014)	3,956	(2005)	2,700	(1996)
2c West Scotland - North		1,115	(2013; 2014)	709	(2005)	460	(1996-1997)
3 Western Isles		2,739	(2008; 2011)	1,981	(2003; 2006)	2,820	(1996)
4 North Coast & Orkney		1,938	(2013)	4,384	(2005-2006)	8,787	(1997)
4a North Coast		73	(2013)	146	(2005-2006)	265	(1997)
4b Orkney		1,865	(2013)	4,238	(2006)	8,522	(1997)
5 Shetland		3,039	(2009)	3,038	(2006)	5,994	(1997)
6 Moray Firth		733	(2008; 2011; 2013-2014)	1,028	(2005-2006)	1,409	(1997)
7 East Scotland		194	(2007; 2013-2014)	667	(2005-2006)	764	(1997)
SCOTLAND TOTAL		**23,355**	(2007-2009; 2011; 2013-2014)	**23,423**	(2000; 2003; 2005-2006)	**29,514**	(1996-1997)
8 Northeast England	b	90	(2008; 2014)	* 62	(2005-2006)	* 54	(1997)
9 Southeast England	c	4,681	(2014)	2,964	(2005-2006)	3,222	(1995; 1997)
10 West England & Wales	d	35	(estimate)	20	(estimate)	15	(estimate)
ENGLAND & WALES TOTAL		**4,806**	(2008; 2014)	**3,046**	(2005-2006)	**3,291**	(1995; 1997)
BRITAIN TOTAL		**28,161**	(2007-2009; 2011; 2013-2014)	**26,469**	(2000; 2003; 2005-2006)	**32,805**	(1995-1997)
NORTHERN IRELAND TOTAL	e	**948**	(2011)	**1,176**	(2002; 2006)		
UK TOTAL		**29,109**	(2007-2009; 2011; 2013-2014)	**27,646**	(2000; 2002-2003; 2005-2006)		
REPUBLIC OF IRELAND TOTAL	f	**3,489**	(2011-2012)	**2,955**	(2003)		
BRITAIN & IRELAND TOTAL		**32,598**	(2007-2009; 2011-2014)	**30,601**	(2000; 2002-2003; 2005-2006)		

SOURCES - Most counts were obtained from aerial surveys conducted by SMRU and were funded by Scottish Natural Heritage (SNH) and the Natural Environment Research Council (NERC). Exceptions are:

a Parts of the West Scotland survey in 2009 funded by Scottish Power and Marine Scotland.

b The Tees data collected and provided by the Industry Nature Conservation Association (Woods, 2014). The 2008 survey from Coquet Island to Berwick funded by the Department of Energy and Climate Change (DECC, previously DTI).

c Essex & Kent data for 2014 collected and provided by the Zoological Society London (Barker, 2015).

d No dedicated harbour seal surveys in this management unit and only sparse info available. Estimates compiled from counts shared by other organisations (Chichester Harbour Conservancy) or found in various reports & on websites (Boyle, 2012; Hilbrebirdobs.blogspot.co.uk, 2012, 2013; Sayer, 2010a, 2010b, 2011; Sayer *et al.,* 2012; Westcott, 2002). Apparent increases may partly be due to increased reporting and improved species identification.

e Surveys carried out by SMRU and funded by Northern Ireland Environment Agency (NIEA) in 2002 & 2011 (Duck, 2006; Duck & Morris, 2012) and Marine Current Turbines Ltd in 2006-2008 & 2010 (SMRU Ltd, 2010).

f Surveys carried out by SMRU and funded by the National Parks & Wildlife Service (Cronin *et al*., 2004; Duck & Morris, 2013a, 2013b).

*Northumberland coast south of Farne Islands not surveyed in 2005 & 1997, but no harbour seal sites known here.

Table 2. The most recent August counts of grey seals at haul-out sites in Britain and Ireland by Seal Management Unit compared with two previous periods. Grey seal summer counts are known to be more variable than harbour seal summer counts. Caution is therefore advised when interpreting these numbers. The grey bars represent a histogram of the counts by region to indicate the relative proportion each contributes to the total count.

Seal Management Unit / Country	Grey seal counts					
	2007-2014		2000-2006		1996-1997	
1 Southwest Scotland	374	(2007)	206	(2005)	75	(1996)
2 West Scotland [a]	4,095	(2007-2009; 2013-2014)	2,383	(2000; 2005)	3,435	(1996-1997)
2a West Scotland - South	2,649	(2007; 2009; 2014)	1,771	(2000; 2005)	2,125	(1996)
2b West Scotland - Central	1,056	(2014)	361	(2005)	931	(1996)
2c West Scotland - North	390	(2013; 2014)	251	(2005)	379	(1996-1997)
3 Western Isles *	3,743	(2008; 2011)	3,528	(2003; 2006)	4,062	(1996)
4 North Coast & Orkney	8,035	(2013)	10,155	(2005-2006)	9,427	(1997)
4a North Coast	195	(2013)	576	(2005-2006)	597	(1997)
4b Orkney	7,840	(2013)	9,579	(2006)	8,830	(1997)
5 Shetland	1,536	(2009)	1,371	(2006)	1,724	(1997)
6 Moray Firth	532	(2008; 2011; 2013-2014)	1,272	(2005-2006)	551	(1997)
7 East Scotland	2,134	(2007; 2013-2014)	1,898	(2005-2006)	2,328	(1997)
SCOTLAND TOTAL	**20,449**	(2007-2009; 2011; 2013-2014)	**20,813**	(2000; 2003; 2005-2006)	**21,602**	(1996-1997)
8 Northeast England [b]	2,345	(2008; 2014)	† 1,100	(2005-2006)		
9 Southeast England [c]	6,063	(2014)	2,266	(2005-2006)		
10 West England & Wales [d]	1,300	(estimate)	1,150	(estimate)		
ENGLAND & WALES TOTAL	**9,708**	(2008; 2014)	**4,516**	(2005-2006)		
BRITAIN TOTAL	**30,157**	(2007-2009; 2011; 2013-2014)	**25,329**	(2000; 2003; 2005-2006)		
NORTHERN IRELAND TOTAL [e]	**468**	(2011)	**275**	(2002; 2006)		
UK TOTAL	**30,625**	(2007-2009; 2011; 2013-2014)	**25,605**	(2000; 2002-2003; 2005-2006)		
REPUBLIC OF IRELAND TOTAL [f]	**2,964**	(2011-2012)	**1,309**	(2003)		
BRITAIN & IRELAND TOTAL	**33,589**	(2007-2009; 2011-2014)	**26,914**	(2000; 2002-2003; 2005-2006)		

SOURCES - Most counts were obtained from aerial surveys conducted by SMRU and were funded by Scottish Natural Heritage (SNH) and the Natural Environment Research Council (NERC). Exceptions are:

a Parts of the West Scotland survey in 2009 funded by Scottish Power and Marine Scotland.

b The Tees data collected and provided by the Industry Nature Conservation Association (Woods, 2014). The 2008 survey from Coquet Island to Berwick funded by the Department of Energy and Climate Change (DECC, previously DTI).

c Essex & Kent data for 2014 collected and provided by the Zoological Society London (Barker, 2015).

d No SMRU surveys in this management unit but some data available. Estimates compiled from counts shared by other organisations (Natural Resources Wales, RSPB) or found in various reports & on websites (Boyle, 2012; Büche & Stubbings, 2014; Hilbrebirdobs.blogspot.co.uk, 2012, 2013; Leeney *et al*., 2010; Sayer, 2010b, 2011, 2012a, 2012b; Sayer *et al*., 2012; Westcott, 2002, 2009; Westcott & Stringell, 2004). Apparent increases may partly be due to increased reporting.

e Surveys carried out by SMRU and funded by Northern Ireland Environment Agency (NIEA) in 2002 & 2011 (Duck, 2006; Duck & Morris, 2012) and Marine Current Turbines Ltd in 2006-2008 & 2010 (SMRU Ltd, 2010).

f Surveys carried out by SMRU and funded by the National Parks & Wildlife Service (Cronin et al., 2004; Duck & Morris, 2013a, 2013b).

* During the 2011 survey, warm weather probably kept hundreds of grey seals from hauling out at the Monach Isles. Therefore the 2011 count for the Monach Isles has been replaced with the 2008 count .

† Northumberland coast south of Farne Islands not surveyed in 2005, so count may be incomplete.

Table 3. August counts of harbour seals in the Moray Firth, 1992-2014. Mean value if more than one count in any year; red = lowest count, green = highest count. Data are from aerial surveys by the Sea Mammal Research Unit. Since 2006, all surveys incorporated hand-held oblique digital photography. See Figure 10 for a map showing the 2014 distribution of seals in the Moray Firth and Figure 7 for a histogram of these data.

	Area	1992	1993	1994	1997	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
	Survey method	fw	fw	fw	ti	fw	fw &ti	fw	2fw	2fw &1ti	fw &ti	fw &ti	fw &ti	fw	fw	ti	fw	fw	fw
(MEAN) COUNTS	Duncansby Head to Helmsdale		2		1					1			1						
	Helmsdale to Brora		92		193		188			113	150	54	73	19	101	87	102	70	1
	Loch Fleet		16		27	33	59	56	64	71	80	83	82	65	114	113	133	135	156
	Dornoch Firth (SAC)	662		542	593	405	220	290	231	191	257	144	145	166	219	208	157	143	111
	Cromarty Firth	41		95	95	38	42	113	88	106	106	102	90	90	140	101	144	63	100
	Beauly Firth (incl. Milton)	220		203	219	204	66	151	178	127	176	146	150	85	140	57	60	30	37
	Ardersier (incl. Eathie)			221	234	191	110	205	202	210	197	154	145	277	362	195	183	199	28
	Culbin & Findhorn			58	46	111	144	167	49	93	58	79	92	73	123	163	254	218	260
	Burghead to Fraserburgh			0	1					3		0				29		39	
TOTALS	Dornoch Firth to Ardersier			1,061	1,141	838	438	759	699	634	736	546	530	618	861	561	544	435	276
	Loch Fleet to Ardersier				1,168	871	497	815	763	705	816	629	612	683	975	674	677	570	432
	Loch Fleet to Findhorn				1,214	982	641	982	812	798	874	708	704	756	1,098	837	931	788	692
	Helmsdale to Findhorn				1,407		829			911	1,024	762	777	775	1,199	924	1,033	858	693
	Moray Firth SMA *				1,409		831			915	1,028	763	778	776	1,200	954	1,063	898	733

* For years where only the main area was surveyed (i.e. Helmsdale to Findhorn), the most recent counts for the outlying areas are used to give a total for the Moray Firth Seal Management Area.
fw, fixed-wing survey; ti, thermal imager helicopter survey; SMA, Seal Management Area.

Table 4. August counts of grey seals in the Moray Firth, 1992-2014. Mean value if more than one count in any year; red = lowest count, green = highest count per area. Data are from aerial surveys by the Sea Mammal Research Unit. Since 2006, all surveys were by hand-held oblique digital photography. See Figure 10 for a map showing the 2014 distribution of seals in the Moray Firth.

	Area	1992	1993	1994	1997	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
	Survey method	fw	fw	fw	ti	fw	fw &ti	fw	2fw	2fw &1ti	fw &ti	fw &ti	fw &ti	fw	fw	ti	fw	fw	fw
(MEAN) COUNTS	Duncansby Head to Helmsdale *		33		0					59			9			15			
	Helmsdale to Brora				3		6			111	102	52	449	72	635	156	316	81	27
	Loch Fleet		0		0	0	0	0	0	0	1	3	1	0	7	7	20	18	7
	Dornoch Firth (SAC)	233		903	456	121	321	79	473	431	748	516	523	819	717	679	74	604	127
	Cromarty Firth	9		0	0	0	0	0	0	0	1	0	0	0	1	2	1	3	1
	Beauly Firth (incl. Milton)	8		2	3	8	0	0	0	0	3	4	0	0	2	3	1	5	2
	Ardersier (incl. Eathie)			36	24	85	0	3	44	55	142	74	142	94	297	74	24	109	2
	Culbin & Findhorn			0	0	0	0	10	0	11	11	28	75	58	58	179	121	218	93
	Burghead to Fraserburgh			30	65					205		61				18		258	
	Dornoch Firth to Ardersier			941	483	214	321	82	517	486	894	594	665	913	1,017	758	100	721	132
TOTALS	Loch Fleet to Ardersier				483	214	321	82	517	486	895	597	666	913	1,024	765	120	739	139
	Loch Fleet to Findhorn				483	214	321	92	517	497	906	625	741	971	1,082	944	241	957	232
	Helmsdale to Findhorn				486		327			608	1,008	677	1,190	1,043	1,717	1,100	557	1,038	259
	Moray Firth SMA †				551		392			872	1,272	797	1,260	1,113	1,787	1,133	590	1,311	532

* In 2011, Duncansby Head to Wick was not surveyed. Therefore the 15 grey seals given for the northern most area in 2011 include 7 counted in 2008.

† For years where only the main area was surveyed (i.e. Helmsdale to Findhorn), the most recent counts for the outlying areas are used to give a total for the Moray Firth Seal Management Area.

fw, fixed-wing survey; ti, thermal imager helicopter survey; SMA, Seal Management Area.

Table 5. August counts of harbour seals in the Firth of Tay & Eden Estuary harbour seal SAC, 1990-2014. Mean value if more than one count in any year; red = lowest count, green = highest count per area. Data are from aerial surveys by the Sea Mammal Research Unit. Since 2006, all surveys were by hand-held oblique digital photography. See Figure 11 for a map showing the 2014 distribution of harbour seals in the SAC and Figure 12 for a histogram of these data.

	Area	1990	1991	1992	1994	1997	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
	Survey method	1fw	1fw	1fw	1fw	1ti	1fw	1fw	1fw	1fw	2fw ,1ti	1fw	1fw ,1ti	2fw	1fw	1fw	1fw	1fw	1ti	1fw
(MEAN) COUNTS	Upper Tay	27	73	148	89	113	115	51	83	134	91	91	63	49	45	41	16	40	36	21
	Broughty Ferry	77	83	97	64	35	52	0	90	55	51	31	27	13	28	15	18	16	3	0
	Buddon Ness	13	86	72	53	0	113	109	142	66	25	96	64	27	8	23	11	8	10	1
	Abertay & Tentsmuir	319	428	456	289	262	153	167	53	126	63	34	31	50	8	9	0	5	0	0
	Eden Estuary	31	0	0	80	223	267	341	93	78	105	90	90	83	22	36	32	19	1	7
	SAC total	467	670	773	575	633	700	668	461	459	335	342	275	222	111	124	77	88	50	29

Table 6. August counts of grey seals in the Firth of Tay & Eden Estuary harbour seal SAC, 1990-2014. Mean value if more than one count in any year; red = lowest count, green = highest count per area. Data are from aerial surveys by the Sea Mammal Research Unit. Since 2006, all surveys were by hand-held oblique digital photography. See Figure 11 for a map showing the 2014 distribution of seals in the SAC and Figure 13 for a histogram of these data.

	Area	1990	1991	1992	1994	1997	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
	Survey method	1fw	1fw	1fw	1fw	1ti	1fw	1fw	1fw	1fw	2fw ,1ti	1fw	1fw ,1ti	2fw	1fw	1fw	1fw	1fw	1ti	1fw
(MEAN) COUNTS	Upper Tay	0	0	18	20	61	64	78	50		42	22	27	41	55	98	16	39	127	62
	Broughty Ferry	0	3	0	9	0	0	0	16		0	8	1	4	0	0	2	3	0	2
	Buddon Ness	0	0	1	104	0	101	0	33		11	25	85	4	0	12	22	13	18	0
	Abertay & Tentsmuir	912	1,546	1,191	1,335	1,820	2,088	1,490	1,560		763	1,267	1,375	442	395	1,406	1,265	1,111	323	531
	Eden Estuary	0	0	16	0	10	0	25	4		27	57	31	17	0	39	17	36	14	39
	SAC total	912	1,549	1,226	1,468	1,891	2,253	1,593	1,663		843	1,379	1,519	508	450	1,555	1,322	1,202	482	634

fw, fixed-wing survey; ti, thermal imager helicopter survey; SAC, Special Area of Conservation

Table 6. August counts of harbour seals on the English east coast, 1988-2014. In years where more than one survey was carried out, values are means with number of surveys in parentheses. Blank grey cells mean 'no survey was carried out'.

	Northeast England		**Southeast England**				
Year	N'umberland	The Tees	Donna Nook	The Wash	Blakeney Point	Scroby Sands	Essex & Kent
1988			173	3,053	701		
1989		16 (31)	126	1,549 (2)	307		
1990		23 (31)	57	1,543	73		
1991		24 (31)		1,398 (2)			
1992		27 (31)	32 (2)	1,671 (2)	217		
1993		30 (31)	88	1,884	267		
1994	13	35	103 (2)	2,005 (2)	196	61	
1995		33 (31)	115	2,084 (2)	415 (2)	49	130
1996		42 (31)	162	2,151	372	51	
1997	12	42 (31)	251 (2)	2,466 (2)	311 (2)	65 (2)	
1998		41 (31)	248 (2)	2,374 (2)	637 (2)	52	
1999		36 (31)	304 (2)	2,392 (2)	659 (2)	72 (2)	
2000	10	59 (31)	390 (2)	2,779 (2)	895	47 (2)	
2001		59 (31)	233	3,194	772	75	
2002		52 (31)	341	2,977 (2)	489 (2)		
2003		38 (31)	231	2,513 (2)	399	38	180
2004		40 (31)	294 (2)	2,147 (2)	646 (2)	57 (2)	
2005	17	50 (31)	421 (2)	1,946 (2)	709 (2)	56 (2)	101
2006		45 (31)	299	1,695	719	71	
2007	7	43 (31)	214	2,162	550		
2008	9	41 (31)	191 (2)	2,011 (2)	581 (2)	81 (2)	319
2009		49 (31)	267 (2)	2,829 (2)	372	165 (2)	
2010		53 (31)	176 (2)	2,586 (2)	391	201 (2)	379
2011		57 (31)	205	2,894	349	119	
2012		63 (31)	192 (2)	3,372 (2)	409	216	
2013		74 (31)	396	3,174	304	148	482
2014		81 (31)	353	3,086	468	285	489

SOURCE - Counts from SMRU aerial surveys using a fixed-wing aircraft funded by NERC except where stated otherwise:
Northumberland - One complete survey in 2008 (funded by DECC (previously DTI). Helicopter surveys with thermal imager from Farne Islands to Scottish border in 1997, 2005 & 2007. Fixed-wing surveys of Holy Island only in 1994 & 2000.
The Tees - Ground counts by Industry Nature Conservation Agency (Woods, 2014). Single SMRU fixed-wing count in 1994.
Southeast England - All SMRU aerial surveys, except for Essex & Kent 2013: data from surveys (aerial/by boat/from land) carried out by the Zoological Society of London (Barker, 2015). The 130 for 1995 are an estimate based on a partial SMRU aerial survey.

Table 7. August counts of grey seals on the English east coast, 1995-2014. In years where more than one survey was carried out, values are means with number of surveys in parentheses. Blank grey cells mean 'no survey was carried out'.

	Northeast England		Southeast England				
Year	N'umberland	The Tees	Donna Nook	The Wash	Blakeney Point	Scroby Sands	Essex & Kent
1995		10	123	66 (2)	18 (2)	32	
1996		11	119	60	11	46	
1997	603	10	289 (2)	49 (2)	45 (2)	34 (2)	
1998		11	174 (2)	53 (2)	33 (2)	23	
1999		12	317 (2)	57 (2)	14 (2)	89 (2)	
2000	568	11	390	40 (2)	17	40 (2)	
2001		11	214	111	30	70	
2002		12	291	75 (2)	11 (2)		
2003		11	232 (2)	58 (2)	18	36	96
2004		13	609 (2)	30 (2)	10 (2)	93 (2)	
2005	1,092	12 (31)	927 (2)	49 (2)	86 (2)	106 (2)	
2006		8 (31)	1,789	52	142	187	
2007	1,907	8 (31)	1,834	42			
2008	2,338	12 (31)	2,068 (2)	68 (2)	375 (2)	137 (2)	160
2009		12 (31)	1,329 (2)	118 (2)	22	157 (2)	
2010		14 (31)	2,188 (2)	240 (2)	49 (2)	292 (2)	393
2011		14 (31)	1,930	142	300	323	
2012		18 (31)	4,978	258 (2)	65		
2013		16 (31)	3,474	219	63	219	203
2014		16 (31)	4,437	223	445	509	449

SOURCE - Counts from SMRU aerial surveys using a fixed-wing aircraft funded by NERC except where stated otherwise:
Northumberland - One complete survey in 2008 (funded by DECC (previously DTI). Helicopter surveys with thermal imager from Farne Islands to Scottish border in 1997, 2005 & 2007. Fixed-wing surveys of Holy Island only in 1994 & 2000.
The Tees - Ground counts by Industry Nature Conservation Agency (Woods, 2014). For years prior to 2005, only monthly maximums are available for grey seals. For these years, the given values are estimates calculated using the mean relationship of mean to maximum counts from 2005-2013.
Southeast England - All SMRU aerial surveys, except for Essex & Kent 2013: data from surveys (aerial/by boat/from land) carried out by the Zoological Society of London (Barker, 2015).

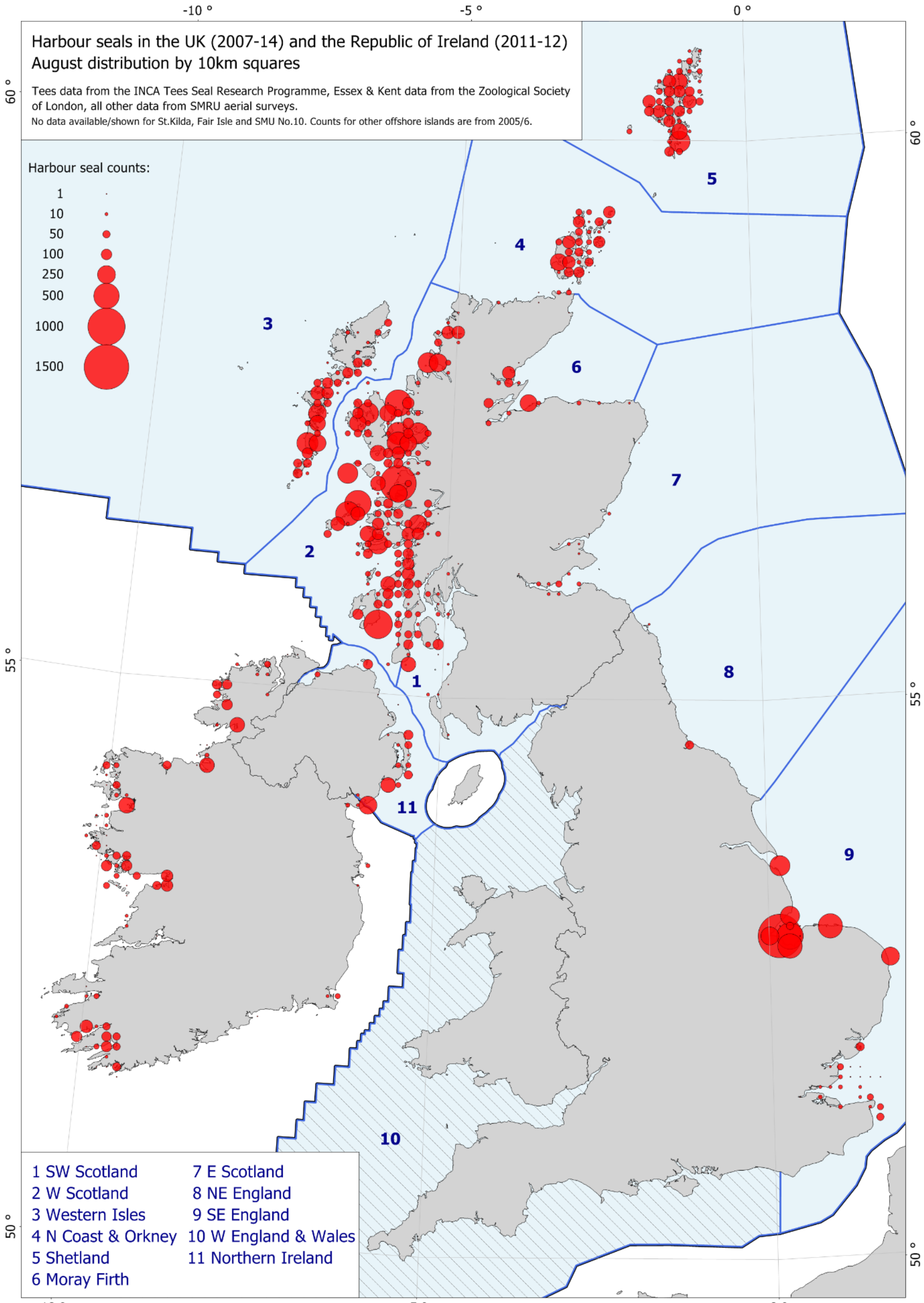


Figure 1. August distribution of harbour seals around the British Isles. Very small numbers of harbour seals (<50) are anecdotally but increasingly reported for the West England & Wales management unit, but are not included on this map.

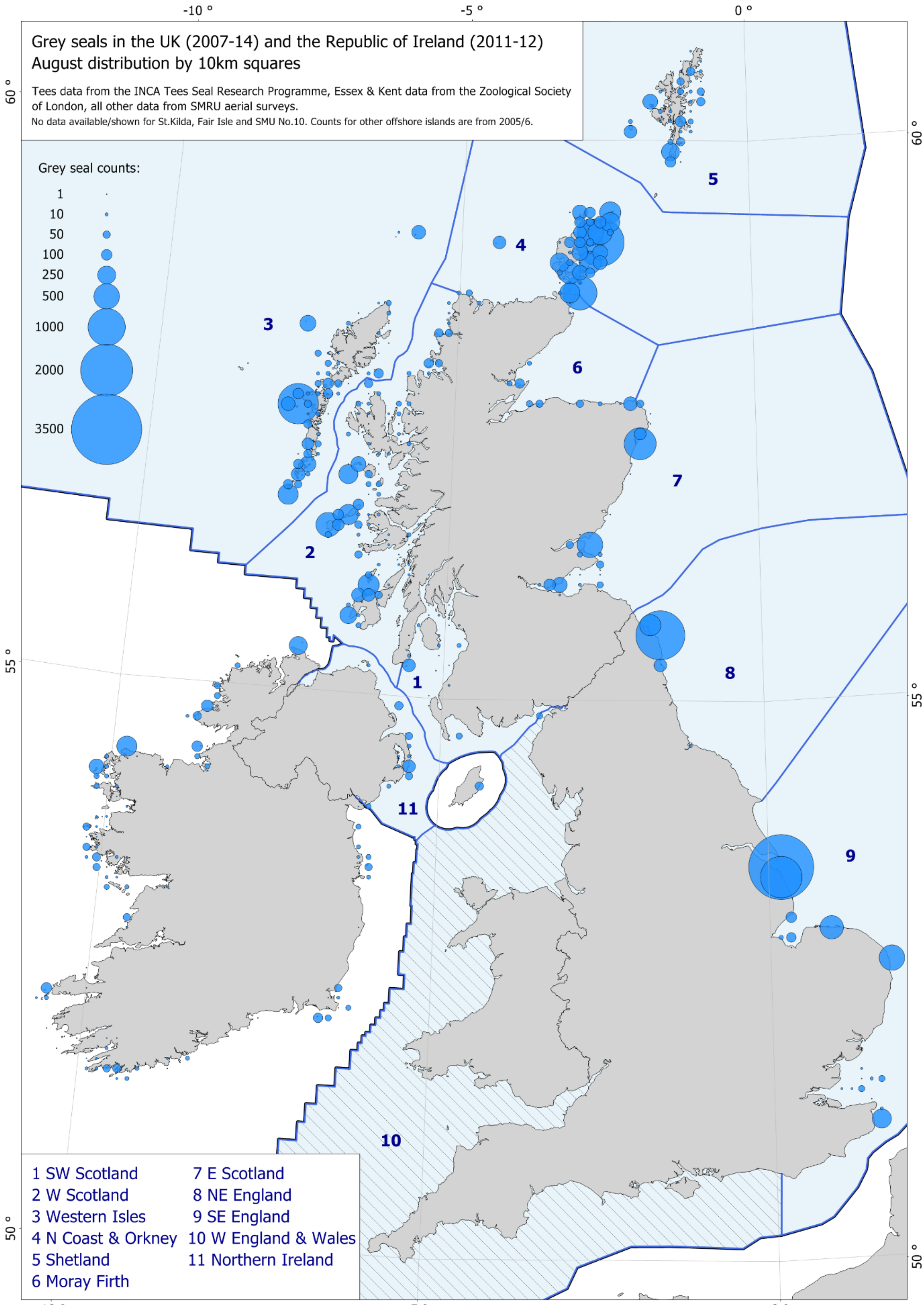


Figure 2. August distribution of grey seals around the British Isles. Only few August counts are available for grey seals in the West England & Wales management unit. Current estimates would add approximately 1,300 animals for this unit, but these are not included on this map.

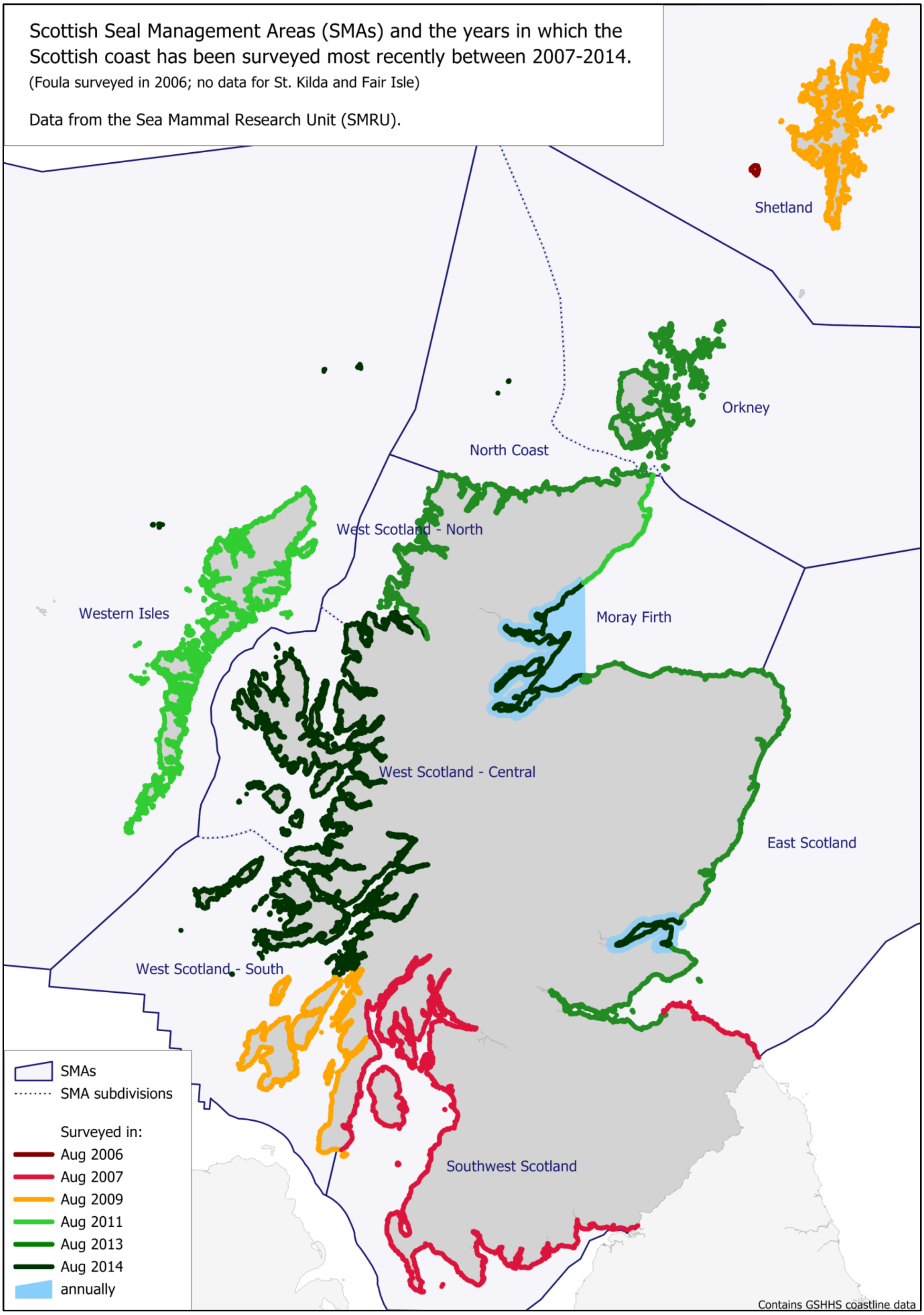


Figure 3. Years in which different parts of Scotland were surveyed most recently by helicopter using a thermal imaging camera. Most areas were surveyed between 2007 and 2014. Foula, off Shetland, was last surveyed in 2006. The enclosed areas of the Firth of Tay and the Moray Firth (between Findhorn and Helmsdale) are surveyed every year, usually by fixed-wing aircraft.

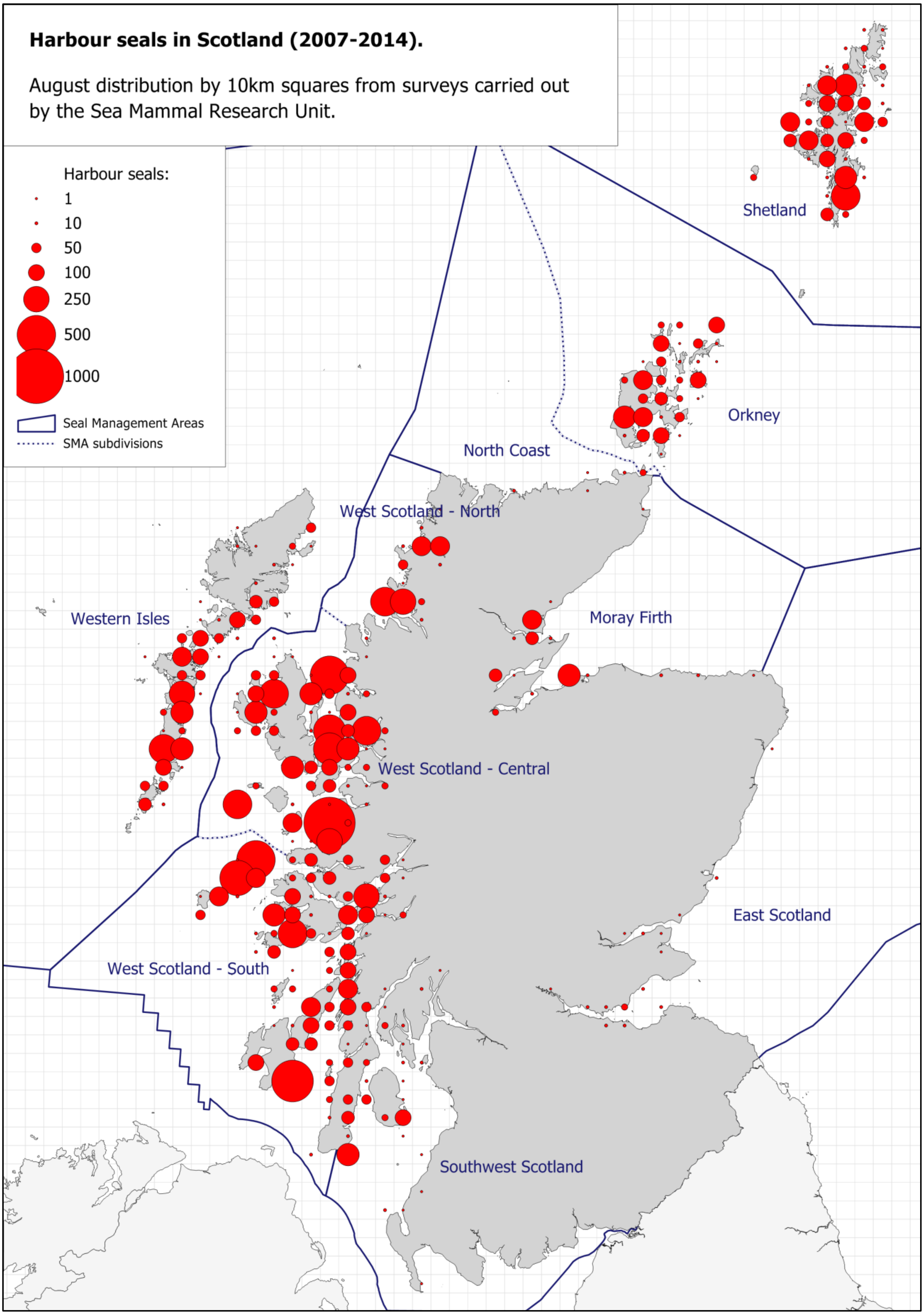


Figure 4. August distribution of harbour seals in Scotland. All areas were surveyed by helicopter using a thermal imaging camera, except for the Moray Firth area between Helmsdale and Findhorn, which was surveyed by fixed-wing aircraft without a thermal imager.

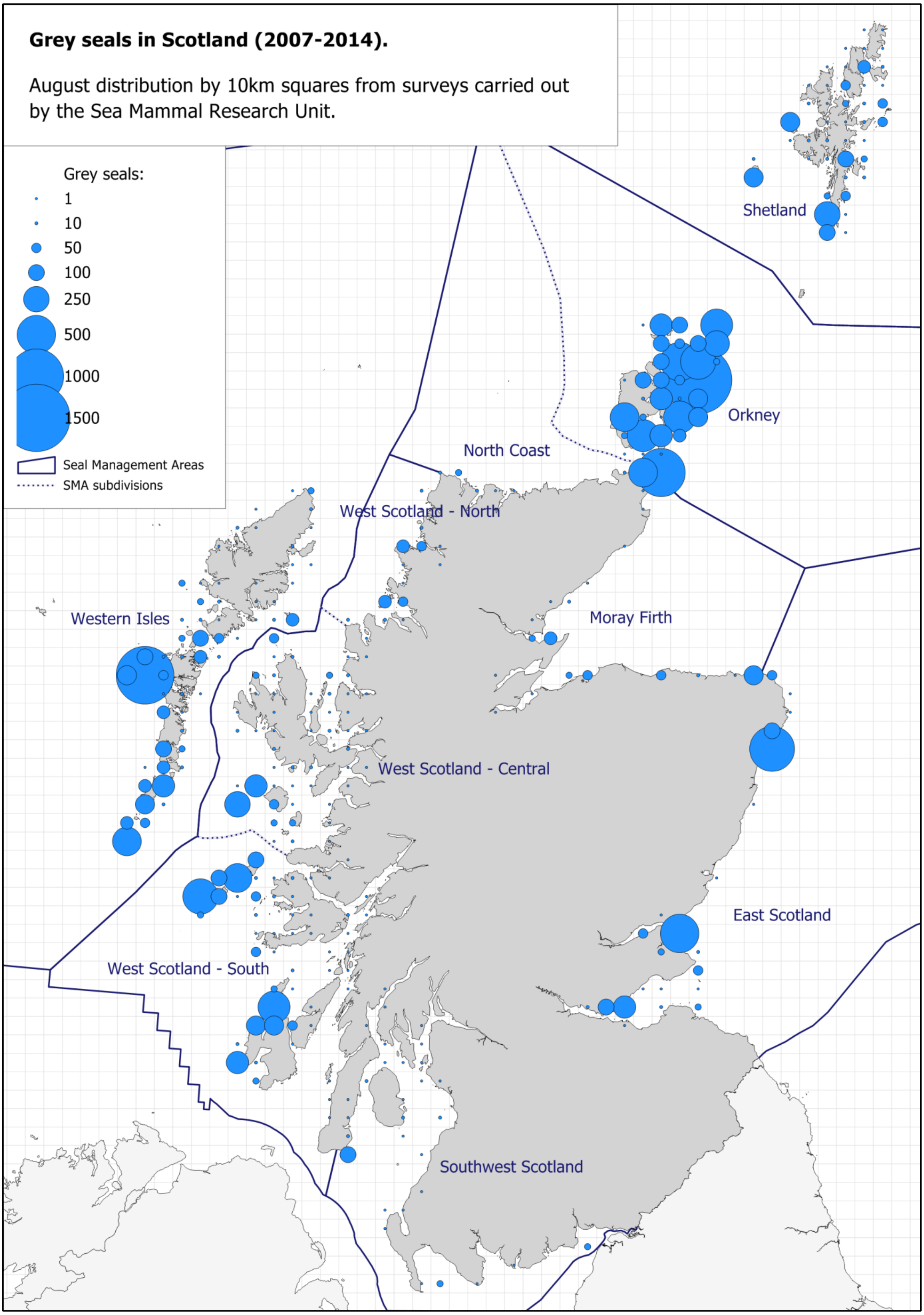


Figure 5. August distribution of harbour seals in Scotland. All areas were surveyed by helicopter using a thermal imaging camera, except for the Moray Firth area between Helmsdale and Findhorn, which was surveyed by fixed-wing aircraft without a thermal imager.

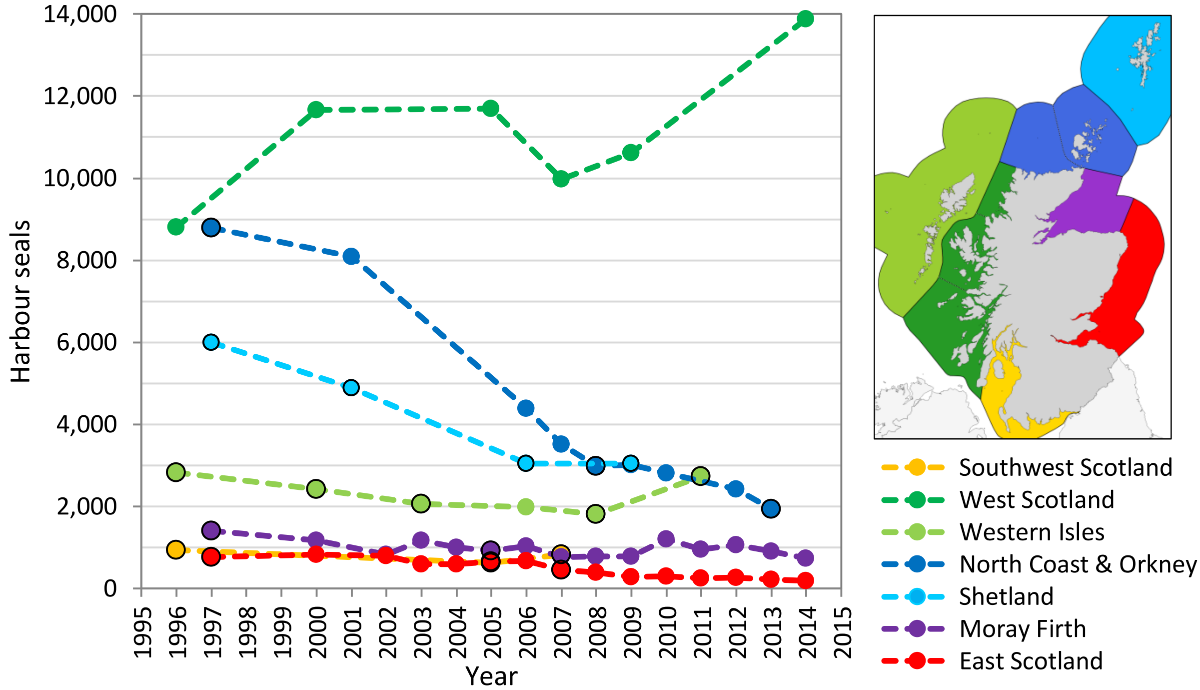


Figure 6. August counts of harbour seals in Scottish Seal Management Areas, 1996-2014. Data from the Sea Mammal Research Unit. Note that because these data points represent counts of harbour seals distributed over large areas, individual data points may not be from surveys from only one year. Points are only shown for years in which a significant part of the SMA was surveyed. Points with a black outline are counts obtained in a single year.

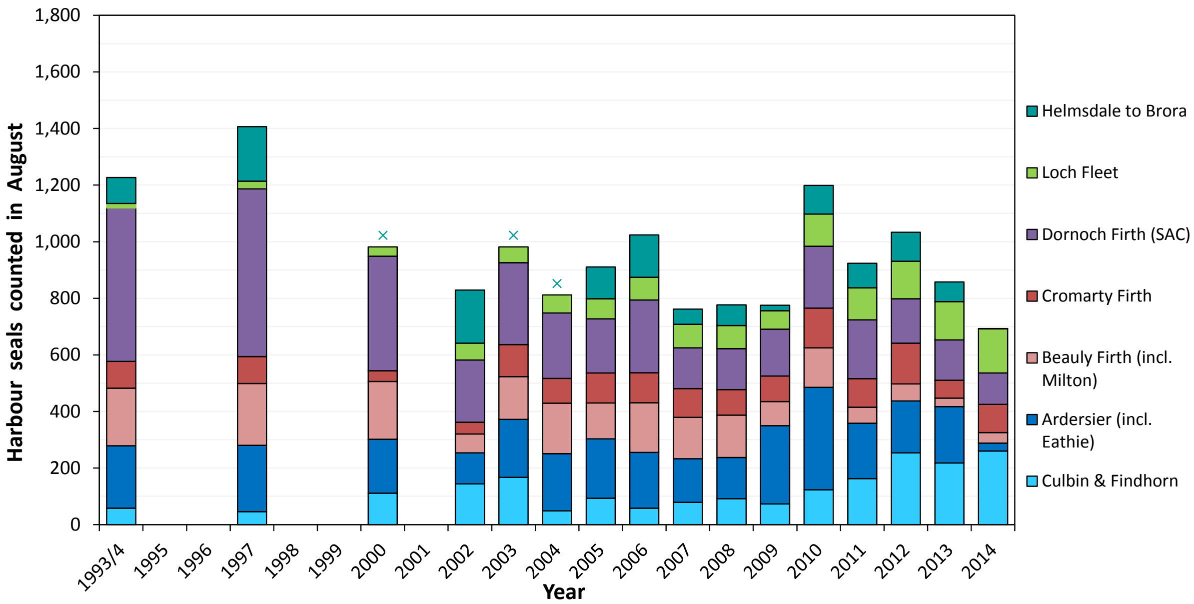


Figure 7. August counts of harbour seals in different areas of the Moray Firth, 1994-2014. Data are from the Sea Mammal Research Unit. x: Helmsdale to Brora not surveyed in 2000, 2003 or 2004.

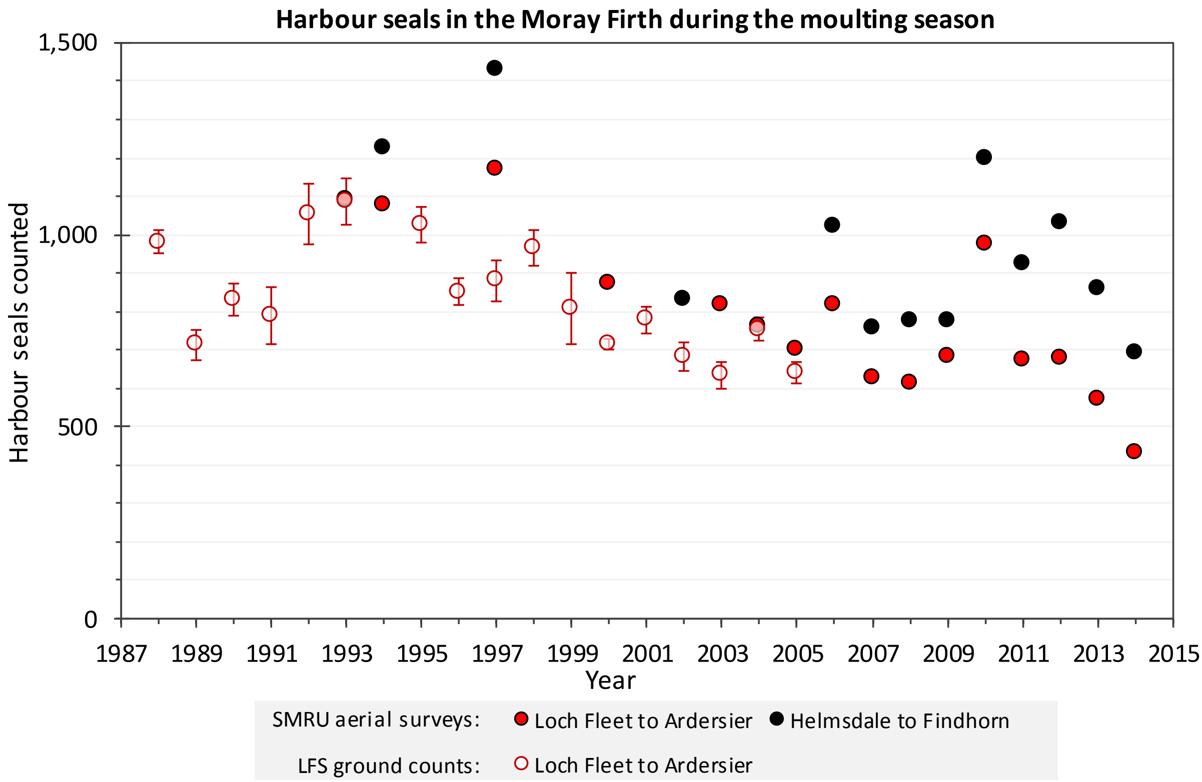


Figure 8. Counts of harbour seals in the Moray Firth during the moult season (August), 1988-2014. Plotted values are means ±SE where available. LFS = Lighthouse Field Station (University of Aberdeen).

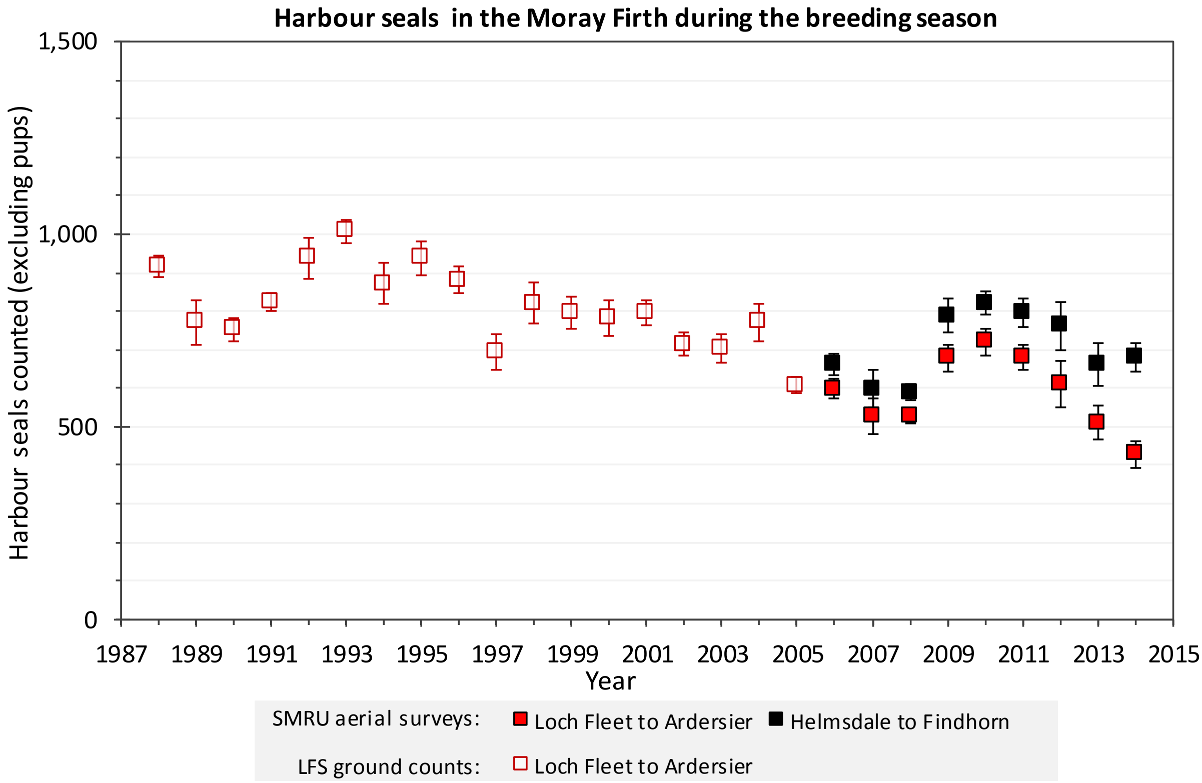


Figure 9. Counts of harbour seals in the Moray Firth during the breeding season (June & July), 1988-2014. Plotted values are means ±SE. LFS = Lighthouse Field Station (University of Aberdeen).

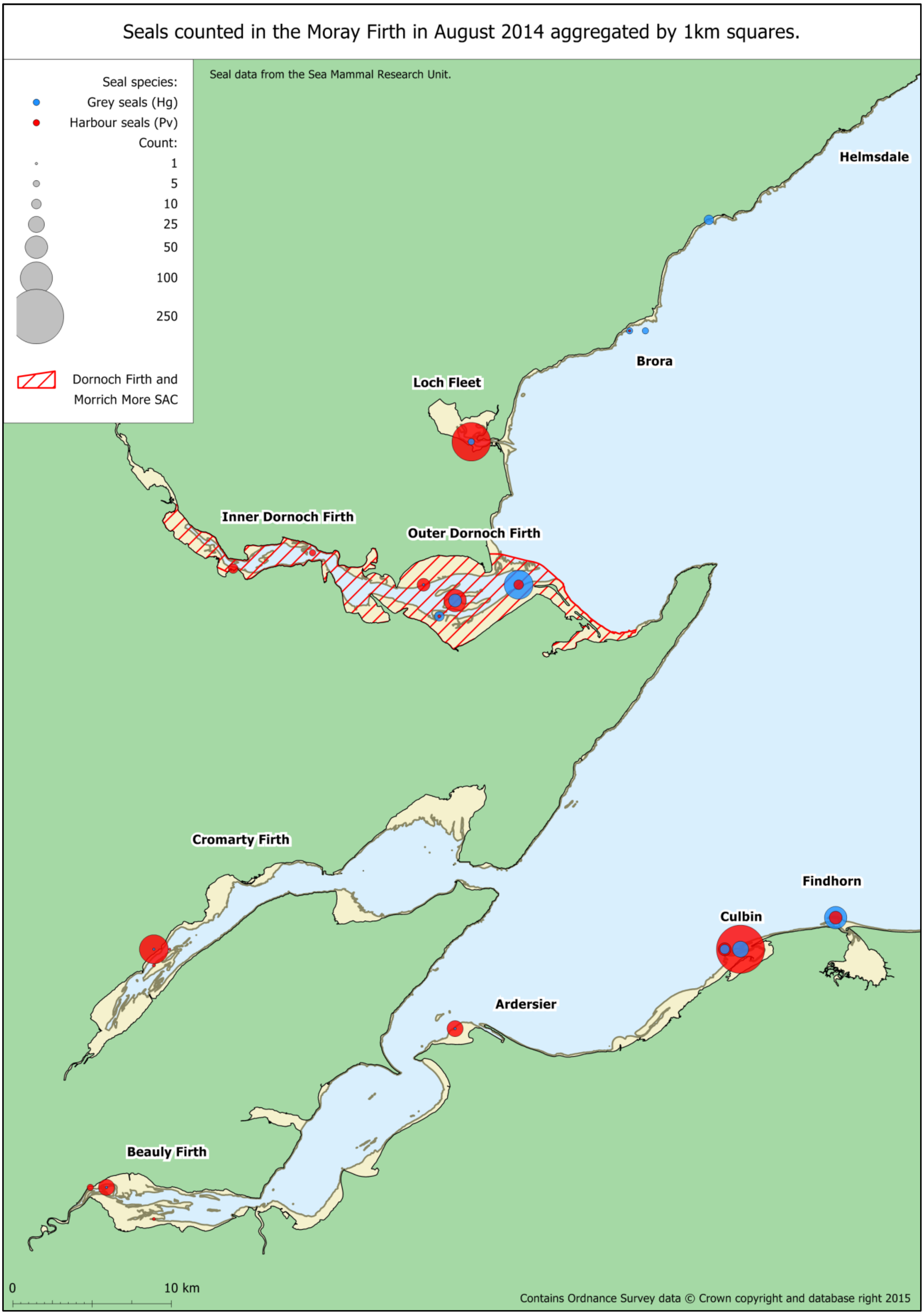


Figure 10. Distribution of harbour and grey seals in the annually surveyed part of the Moray Firth, between Findhorn and Helmsdale, from an aerial survey carried out on 21st August 2014.

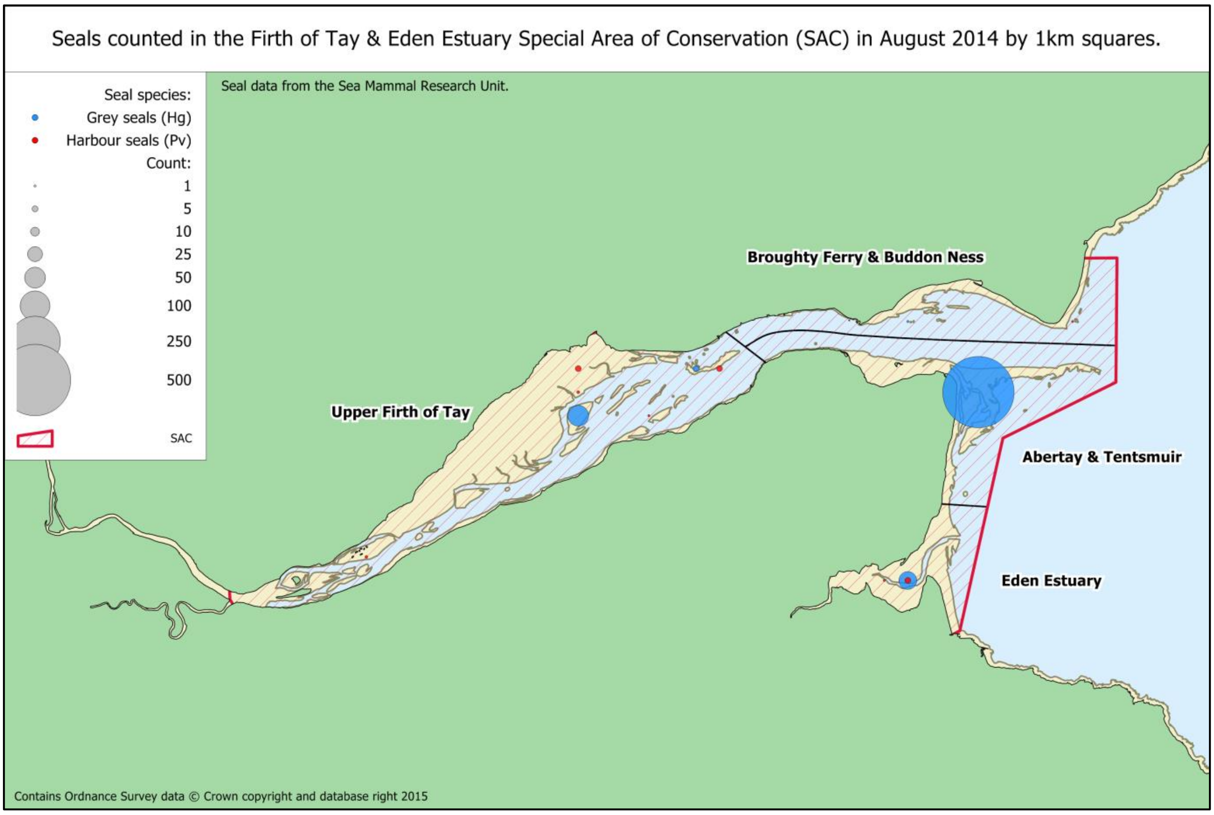


Figure 11. Distribution of harbour seals in the Firth of Tay & Eden Estuary SAC on 20th August 2014.

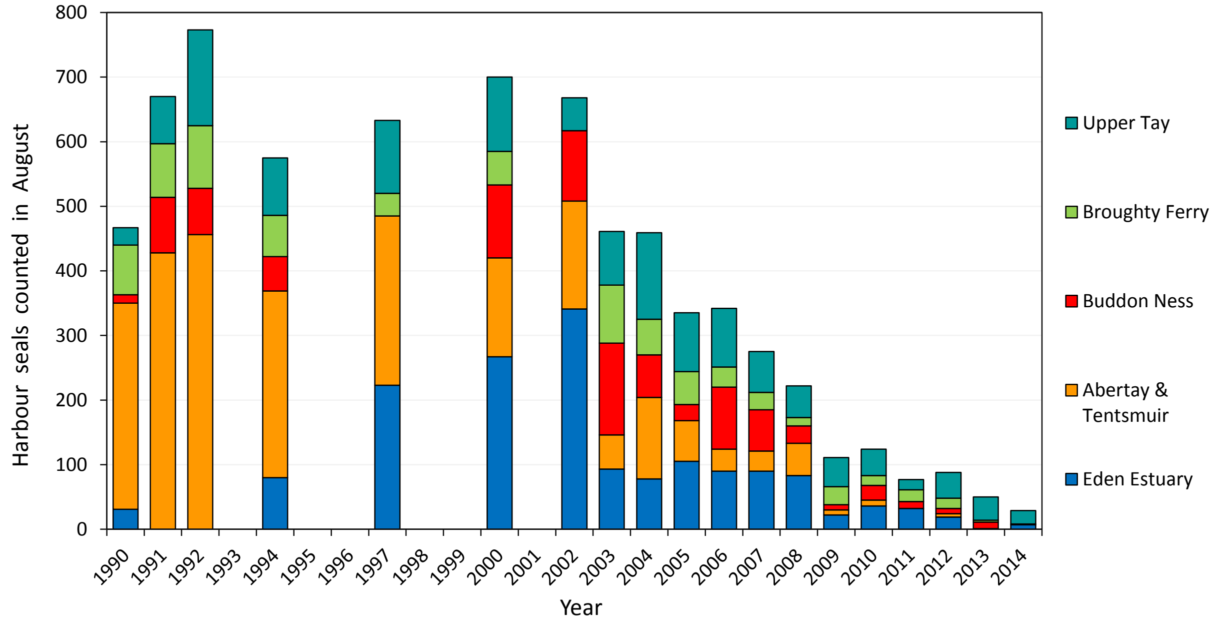


Figure 12. August counts of harbour seals in different areas of the Firth of Tay & Eden Estuary SAC, 1990-2014. Data are from the Sea Mammal Research Unit.

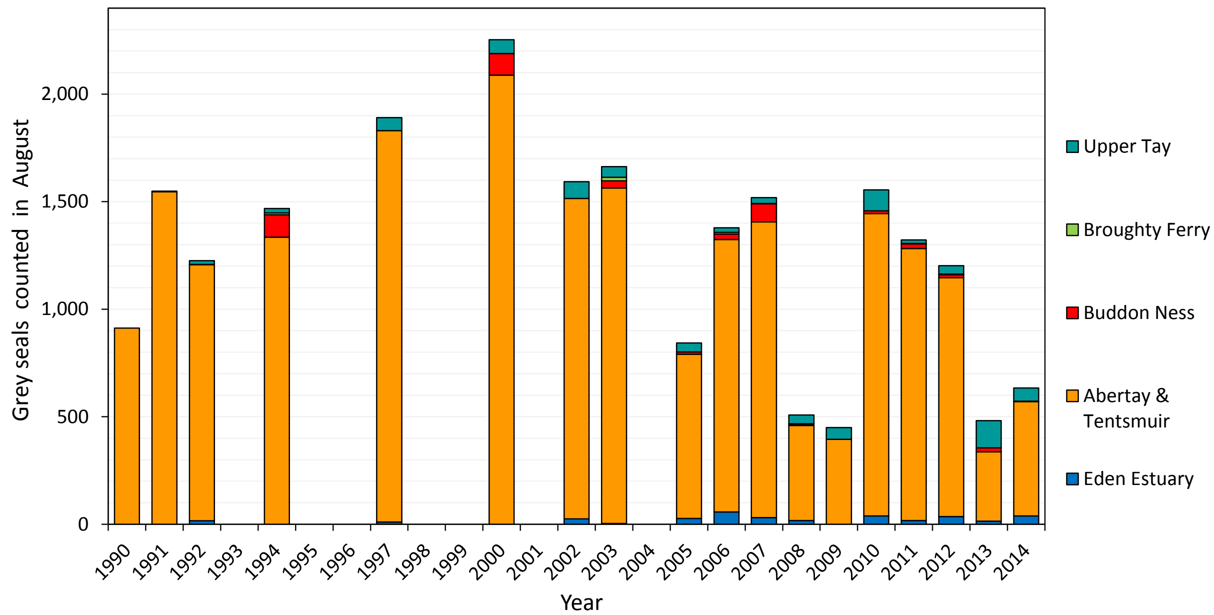
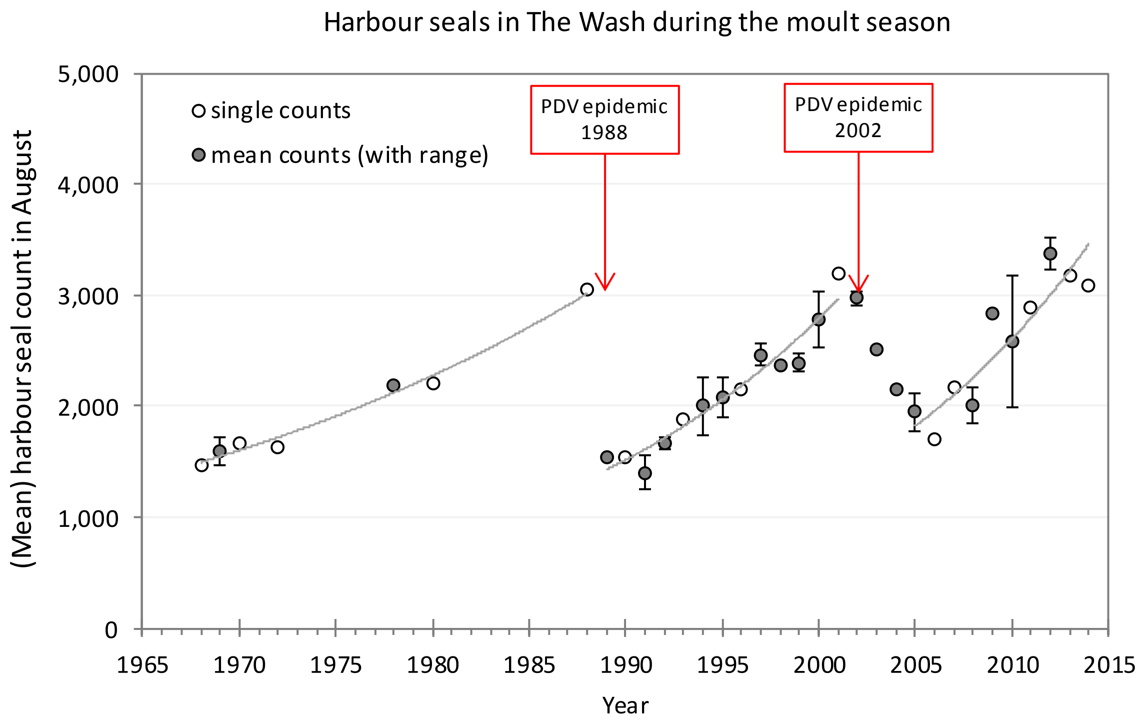


Figure 13. August counts of grey seals in different areas of the Firth of Tay & Eden Estuary SAC, 1990-2014. Data are from the Sea Mammal Research Unit.



NOTE - vertical bars indicate the range of the counts used to calculate the mean.

Figure 14. Counts of harbour seals during the August moult season in The Wash, 1967-2014. Vertical bars indicate the range of the counts used to calculate the mean (where more than one survey was carried out).

Preliminary report on the distribution and abundance of harbour seals (*Phoca vitulina*) during the 2014 breeding season in The Wash

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Abstract

This report presents preliminary results of a breeding season aerial survey of the harbour seal population along the English east coast between Donna Nook in Lincolnshire and Scroby Sands off the Suffolk coast on 30th June 2014.

Results suggest that:

- The pup production in the Wash has continued to increase with the 2014 pup count of **1,802** being the highest recorded: 37% higher than in 2013 and 22% greater than the previous high in 2012.
- Pup production has increased and having grown at around 9% p.a. since surveys began in 2001.
- The ratio of pups to total population was extremely high in 2014 maintaining the previously noted increase. The ratio was 3.4 times higher in 2014 than in 2001 suggesting a large increase in apparent fecundity over that period. No significant change has been observed over the past 5 years.

Introduction

The Wash is the largest estuary in England, and holds the majority of the English harbour seal (*Phoca vitulina*) population (Vaughan, 1978). This population has been monitored since the 1960s, using counts of animals hauled out as indices of population size. The initial impetus for monitoring this population was to investigate the effects of intensive pup hunting. When this hunt ceased in 1973 the monitoring program was reduced

In the summer of 1988 an epidemic of phocine distemper virus (PDV) spread through the European harbour seal population. More than 18000 seal carcasses were washed ashore over a 5 month period, many of them in areas with high levels of human activity (Dietz, Heide-Jorgensen & Härkönen, 1989). Mortality in the worst affected populations, in the Kattegat-Skagerrak, was estimated to be around 60% (Heide-Jorgensen & Härkönen, 1992). After the end of 1988, no more cases of the disease were observed until the summer of 2002, when another epidemic broke out (Harding et al. 2002). Mortality in the European population during the 2002 epidemic was 47%, similar to that seen in 1988 (Härkönen et al. 2006). However, on the English East coast the mortality rate estimated from pre and post epidemic air survey counts was much lower, approximately 22% (Thompson, Lonergan & Duck, 2005). The pre-epidemic population in 2002 was similar in size to the pre-epidemic population in 1988 and the disease hit the English population at the same time of year, so to date there is no clear explanation for the lower mortality rate.

In general, harbour seal population monitoring programmes have been designed to track and detect medium to long-term changes in population size. As it is difficult to estimate absolute abundance, monitoring programmes have usually been directed towards obtaining indices of population size. If consistent, such time series are sufficient to describe a populations' dynamics and have been used to track the long-term status of the English harbour seal population. However, these indices are based on the numbers of individuals observed hauled out, so their utility depends on this being constant over time and unaffected by any changes in population density or structure.

Counts are usually carried out during the annual moult, when the highest and most stable numbers of seals haulout. Unfortunately such counts do not provide a sensitive index of current population health. It is generally accepted that breeding success is a more sensitive index. The breeding season is also the time when disturbance of seal haul-out groups is likely to have direct effects. For example, disturbance of mother/pup pairs will lead to temporary separation which may have direct effects on pup survival, especially if the disturbance is repeated.

Most of the UK harbour seal population breeds on rocky shore habitats, where identifying and counting pups is both difficult and expensive. However, on the English east coast harbour seals breed on open sand banks where pups are relatively easy to observe and count. As a first step towards improving the monitoring program (to increase its sensitivity to short term changes), we identified a need for a baseline survey to map the distribution of breeding harbour seals. In June 2001 Fenland District Council commissioned Sea Mammal Research Unit to conduct an aerial survey of the entire breeding population in the Wash. Since 2004 Natural England have commissioned single annual breeding season surveys to develop a time series of pup counts as an adjunct to the annual moult surveys to obtain a more sensitive index of current status as well as to monitor the distribution of breeding seals. These counts are conducted at the end of June or beginning of July when the peak counts are expected. In 2008 and 2010 additional funds were provided to obtain a time series of counts within single breeding seasons to define the parameters of the pupping curve. In addition to confirming the date of the peak number of pups ashore and available to be counted, these results can provide an estimate of the ratio between peak pup counts and pup production and provide an indication of the likely error on estimates of pup production.

Routine annual moult surveys cover the coast from Donna Nook in Lincolnshire to Scroby Sands off Great Yarmouth in Suffolk. There are known to be smaller groups of seals at various sites along the Essex and the north and east Kent coasts. These sites have been surveyed sporadically during the moult since 2002. In 2011 the Wash pup survey was extended to cover all sites between Scroby Sands and the Goodwin Sands off eastern Kent.

Historical data

One or two complete surveys of the Wash were carried out during the moult, in the first half of August in each year from 1988 to present. The results, combined with counts at the same time of year from the period 1968-1982 are shown in Figure 1. The counts increased between the late 1960s and 1988, at an average of 3.4% p.a. (R^2=0.62, $p<0.0001$). The 1988 count was obtained approximately one week before the first reports of sick and dead seals being washed up on the UK coast. The number hauling out fell by approximately 50% between 1988 and 1989, coincident with the PDV epidemic. After 1989 the number increased again, at an average of 5.9% p.a. (R^2=0.77, $p<<0.0001$). The post epidemic rate of increase was significantly higher than the pre epidemic rate (t=2.87, d.f.=20, $p<0.01$, comparison of regression coefficients for small samples with unequal residual variances).

Post epidemic counts were also obtained at the other major east coast haul-outs outside the Wash, at Blakeney (45km east) and Donna Nook (40km north). At both sites the counts fell after 1988, reaching a minimum in 1990 (Figure 2). Between 1990 and 2001 Blakeney counts increased by an average of 14.4% pa. (R^2=0.47, $p<0.01$), and Donna Nook counts by 18% pa (R^2=0.35, $p<0.03$). The total for all three east coast sites increased at an average rate of 7.2% pa. (R^2=0.87, $p<<0.0001$, Figure 2).

In 2002 there was another outbreak of PDV. The timing of the epidemic and the population size were similar to 1988. The population in the Wash declined by an estimated 22% based on results of surveys in 2003 and on a fitted population growth model (Thompson, Duck & Lonergan, 2005). There appears to have been a continued decline or at least a failure to recover in the moult counts for the English east coast population in the three or four years following the 2002 epidemic. Overall, the combined count during the moult for the English East coast population in 2006 was 12% lower than the mean count in 2005. Since 2006 the counts have increased such that by 2010 and 2011 the counts were similar to the pre-epidemic counts. This apparent lack of recovery or continued decline immediately after the epidemic contrasts with the rapid recovery of the Wadden Sea population that has been increasing at around 12% p.a. since 2002. The initial failure to recover from the 2002 epidemic is unexplained but is similar to the apparent lack of recovery in the years immediately following the 1988 PDV epidemic.

Previous breeding season surveys 2004 to 2013

Based on a preliminary assumption that the peak number of pups would be encountered at the end of June or beginning of July we have surveyed the breeding population between 27th June and 4th July in each year from 2004 to 2013. In addition in both 2008 and 2010 we carried out four additional surveys between 12th June and 13th July to establish the form of the pups ashore curve. Surveys were carried out over the period 1.5 hours before to 2 hours after low water. All tidal sand banks and all creeks accessible to seals were examined visually. Small groups were counted by eye and all groups of more than 10 animals were photographed using either colour reversal film in a vertically mounted 5"X4" format, image motion compensated camera in 2004 & 2005 or with a hand held digital SLR camera since. The equipment and techniques are described in detail in Hiby, Thompson & Ward (1987) and Thompson et al. (2005). Photographs were processed and all seals were identified to species. Harbour seals were then classified as either pups or 1+ age class. No attempt was made to further differentiate the 1+ age class.

2014 Survey Results

In 2014 we surveyed the entire coast and offshore banks from Donna Nook in Lincolnshire to Blakeney Point in Norfolk on 30th June. A total of 1,802 pups and 4,020 older seals (1+ age classes) were counted in the Wash. No pups were observed at Donna Nook, but in contrast to previous surveys a total of 29 pups were seen at Blakeney Point. This count compares with the previous highest peak counts of 1,469 pups and 3,345 older seals (1+ age classes) during the 2012 breeding season survey and 1,308 pups and 3,345 older seals (1+ age classes) during the 2013 breeding season survey. The pups in the Wash were distributed over approximately 50 separate haulout groups (Figure 3), although the number of sites is to some extent a function of the arbitrary division or pooling of groups. Figure 1 shows the distribution of haulout sites in the Wash. Figure 4 shows the flight path during the 2012 breeding season survey. The same survey route was taken in 2014. The tracks in combination with the photographs and the observers' knowledge of locations of seals on the beach have been used to confirm the positions of all the sites. Figure 5 shows the counts of pups at each site obtained during the 2014 breeding season survey. Table 2 presents the data for 2012 to 2014.

The 2014 survey produced the highest pup count ever in the Wash. This was 22% greater than the previous highest count in 2012. Figure 6 suggests that the trend in the counts can still be approximated by an exponential increase at an annual rate of increase of 9% p.a. since 2001. Despite the large inter-annual increase, inclusion of the 2014 count had little effect on the estimated growth trajectory.

The evolving time series indicates that there was no evidence of a major decline in pup production after the 2002 PDV epidemic. This continued increase in pup production contrasts with the apparent decrease in the moult counts between 2003 and 2007 (Figure 1). The moult count appears to be increasing over the past 6 years. The different trajectories of the pup counts and the independent index of population size represented by the moult count means that the apparent productivity or apparent population fecundity has changed over the period. An index of productivity or population fecundity, i.e. the maximum pup count in each year divided by the moult count or counts in that year shows a major increase from approximately 0.25 at the start of the series in 2001 to 2005 up to an average of 0.45 since 2006.

Discussion

The 2014 breeding season survey confirms the continued upward trend in pup production of the Wash harbour seal population. At present we do not have a direct conversion from peak count to pup production, but there is no reason to suspect a systematic change in that ratio. Therefore the observed 9% p.a. increase in pup count should be a reliable indication of the rate of increase in pup production.

The recent low intensity pup survey effort has produced two interesting results that highlight the advantage of a two pronged approach to seal monitoring. Although there was a well-documented decline of over 20% in the population as a result of the 2002 PDV epidemic there was no apparent decrease in pup production between the pre and post epidemic counts. There are several potential explanations for the lack of a decline. If there was differential mortality, the number of adult females lost to the epidemic may have been small. Alternatively any decrease in adult female population could have been masked by variations in fecundity. Alternative scenarios involving temporary immigration are thought to be less likely.

The most recent data suggest that the apparently dramatic step change in pup production between 2005 and 2006 may have simply been part of a continuing increasing trend. Despite large inter-annual variation the increase has been maintained through to 2014 with pup production increasing by approximately 9% p.a. since 2004. Although the moult counts in the Wash continued to decline after the 2002 epidemic they had clearly stabilised around 2005 to 2007and have increased rapidly since then. Interestingly, although the moult counts in recent years, 2010 to 2014 have been similar to the 2001 count, the estimated peak pup count in 2014 was 3.2 times greater than in 2001 and the number of 1+age class animals counted in the breeding season was more than double the 2001 estimate. If the moult count is a consistent index of the total population size then the apparent fecundity of the Wash population has increased by more than a factor of 3 since 2001.

The fact that pup production varies much more than the moult population index and more rapidly than could be accounted for by changes in adult female numbers, means that there must be wide fluctuations in fecundity and or short term immigration and emigration. At present we do not have information on pregnancy rates in any UK harbour seal population. Telemetry data from both the English and Dutch populations suggests that there is limited movement between the two areas is unlikely to be sufficient to account for these changes. However, to date the telemetry data has been primarily targeted on ….

The observed large increase in pup production in the absence of an equivalent increase in the moult counts is unexplained at present. It could be generated in various ways:

1. Immigration of a large number of adult females. The absence of any substantial populations on the east coast means that the source of seals would have to be either the Wadden Sea or the Scottish East coast. Data on seal movements and recent genetic studies suggest that immigration from Scotland is unlikely and that movement between the English and European populations is unlikely to be frequent enough to explain these changes.
2. A continual increase in fecundity. This seems unlikely given the scale of the increase since 2005, although rapid changes in both directions may suggest wide variation in fecundity rates.

At present we have no information to allow us to differentiate clearly between these options and it is likely that a combination of some or all could be operating. However, in each case the explanation would represent a major change in harbour seal demographics.

The results of the 2001 pup survey suggested that there had been a significant shift in the spatial distribution of breeding seals over the preceding 30 years. The 2004 and 2005 distribution was similar to the 2001 distribution, suggesting that there has been a real shift in distribution with a much higher proportion of pups being found in the south eastern corner of the Wash. At present we do not know why this distributional change is occurring but the results through to 2014 indicate that the relative importance of the SE corner of the Wash is still increasing.

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Table 1. Counts of harbour seal pups and 1+ age classes in the Wash from 2001 to 2014.

Year	2001	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Pups	548	613	651	1054	984	994	1130	1432	1106	1469	1308	1802
1+ age classes	1802	1766	1699	2381	2253	2009	2523	3702	3283	3561	3345	4020

Table 2. Counts of harbour seal pups and 1+ ages at haulout sites in the Wash, 2012-2014.

			30/06/2014		6/07/2013		1/07/2012	
			harbour	seals	harbour	seals	harbour	seals
site name	lat	long	1+ages	pups	1+ages	pups	1+ages	pups
Inner & Outer Knock	53.082	0.364	105	32	94	33	80	20
Inner Dogs Head	53.036	0.376	30	6	35	6	25	4
Friskney	53.034	0.309	69	38	62	29	72	34
Friskney Middle	52.997	0.225	27	15	40	25	56	18
Friskney South	52.953	0.119	40	30	32	16	56	25
Long Sand N/E End	53.019	0.334					76	30
Long Sand Middle	53.005	0.297	122	37	105	29	11	1
Ants	52.978	0.264	2	1	9	4		
Rodger	52.963	0.217	9	2			8	1
NW total			404	161	377	142	384	133
Black Buoy	52.924	0.117	69	4	14	3	85	17
Boston Channel	52.900	0.029	103	35	112	35	76	34
Herring Shoal	52.904	0.064	94	6	62	3	48	0
Toft East	52.932	0.153	30	7	21	1	36	4
Toft West	52.920	0.133	46	5	36	14	14	8
Mare Tail	52.917	0.152	169	92	23	11	37	12
Main End	52.907	0.193	7	5	6	1	16	6
Gat End	52.912	0.203						
Gat Sand	52.935	0.198	86	16	71	15	79	17
SW total			604	170	345	83	391	98

			30/06/2014		6/07/2013		01/07/2012	
			harbour	seals	harbour	seals	harbour	seals
site name	lat	long	1+ages	pups	1+ages	pups	1+ages	pups
Puff	52.899	0.121	3	2	109	55	31	5
Kenzies Creek	52.900	0.106	13	7	18	12	10	5
Fleet Haven Marsh	52.877	0.152					201	64
Fleet Haven Middle	52.884	0.157	342	154	255	115	186	69
Fleet Haven Lower	52.909	0.157					17	2
Fleet Haven Mouth	52.922	0.158	9	6	60	29	48	20
Evans Creek	52.878	0.169	120	55	62	29	108	54
Dawesmere Creek	52.859	0.191	64	37	86	30	118	63

Creeks total			551	261	590	270	719	282
OWMK 1	52.875	0.233	103	37	9	5	4	3
OWMK 2	52.867	0.250	3	1	31	17	4	2
Nene Channel 1(or pooled)	52.875	0.220	11	6	48	16	75	47
Nene Channel 2	52.867	0.216	119	62	81	19	92	63
Nene Channel 3 Barge	52.860	0.214	49	39	18	8	27	14
Nene Channel 4	52.845	0.206	19	21	33	15	6	2
Nene Channel 5	52.827	0.219						
IWMK	52.852	0.235	55	21	52	37	38	16
Scalmans Sled	52.857	0.258	180	126	269	139	257	189
Breast Sand	52.828	0.275	264	111	92	36	114	30
Thief West	52.878	0.273	45	13	38	12	32	7
Thief East	52.878	0.273	5	2	7	2	0	0
Seal Sand (West)/Black Shore	52.875	0.312	90	38	113	57	48	12
Seal sand (East)	52.881	0.352	232	80	168	49	204	64
Seal Sand/Daseleys	52.882	0.351					0	0
Hull Sand	52.840	0.307	530	221	404	116	507	262
Bull Dog Sand	52.866	0.378	122	65	89	39	47	21
Pandora	52.862	0.355	267	107	179	71	154	25
Black Guard	52.883	0.372	5	1			8	0
Old Bell	52.900	0.372	4	0				
Stylemans Middle	52.887	0.380	96	60	34	8	4	0
Pie Corner	52.834	0.327	94	46	12	5	112	60
Lynn Channel	52.810	0.367	162	153	339	157	294	137
Sunk Sand	52.975	0.493	6	0	17	5	40	2
East total			2461	1210	2033	813	2067	956
Wash Total			**4020**	**1802**	**3345**	**1308**	**3561**	**1469**

Figure 1. Aerial survey counts of harbour seals in the Wash during the annual moult in August for the period 1968 to 2013. Dramatic declines in 1988 and 2002 were the result of epidemics of Phocine Distemper Virus. Fitted lines are exponential growth curves (growth rates given in text) with a 2nd order polynomial for post-2002 counts for illustration.

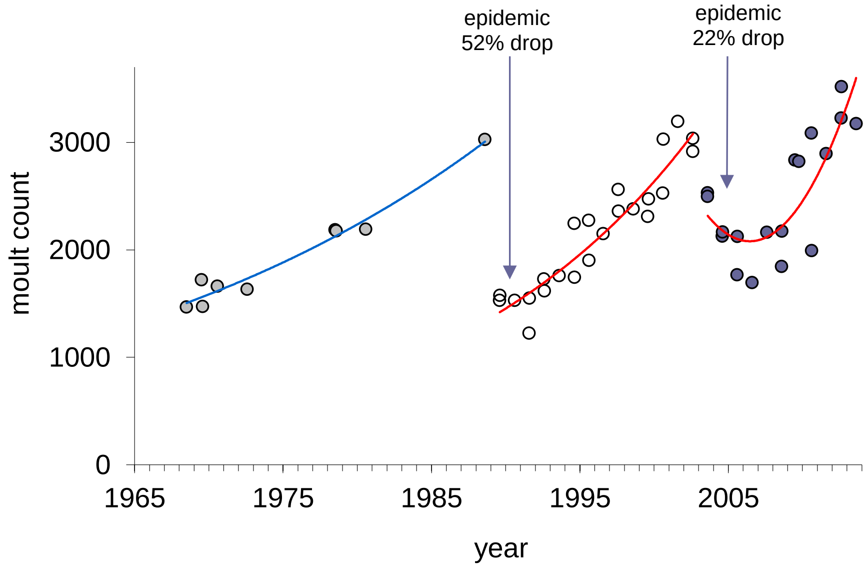


Figure 2. Aerial survey counts of harbour seals at major sites in East Anglia during recovery from the 1988 and 2002 PDV epidemics. There were no significant changes between 2003 and 2013, the fitted polynomial is included simply for illustration.

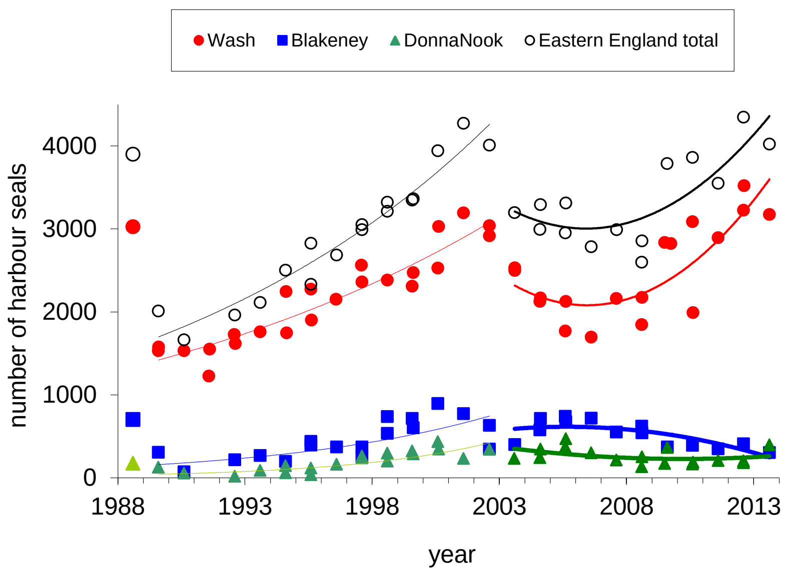


Figure 3. Locations of seal haulout sites during the pupping season in the Wash. Numbers correspond to counts in Table 2

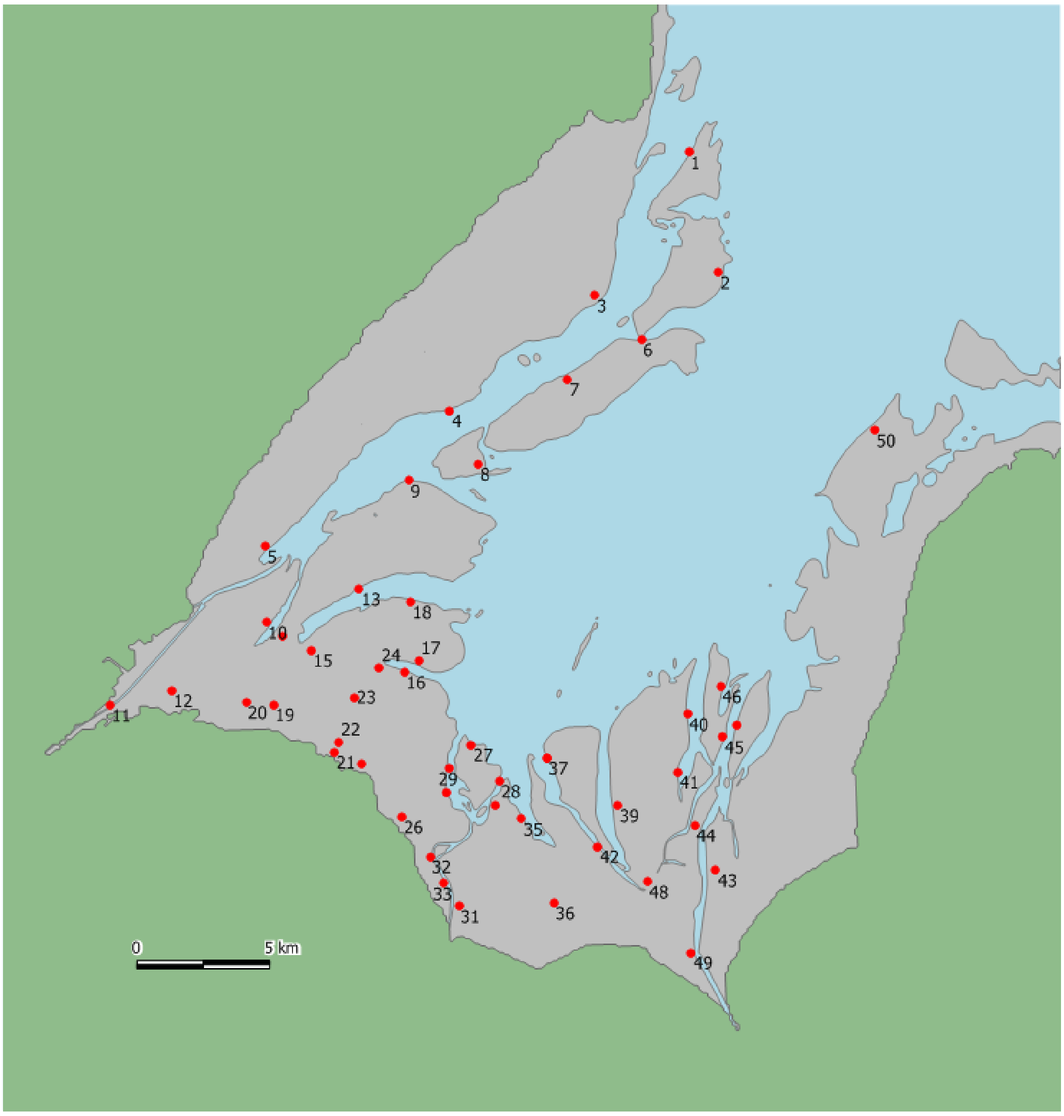


Figure 4. Survey flight path over the Wash during the breeding season survey. The lat long positions of the groups can be derived from a combination of the positions of the tight turns and our observations of the location of seals within the turn in terms of position on the bank. Dark sections of the track indicate positions at which individual photographs were taken.

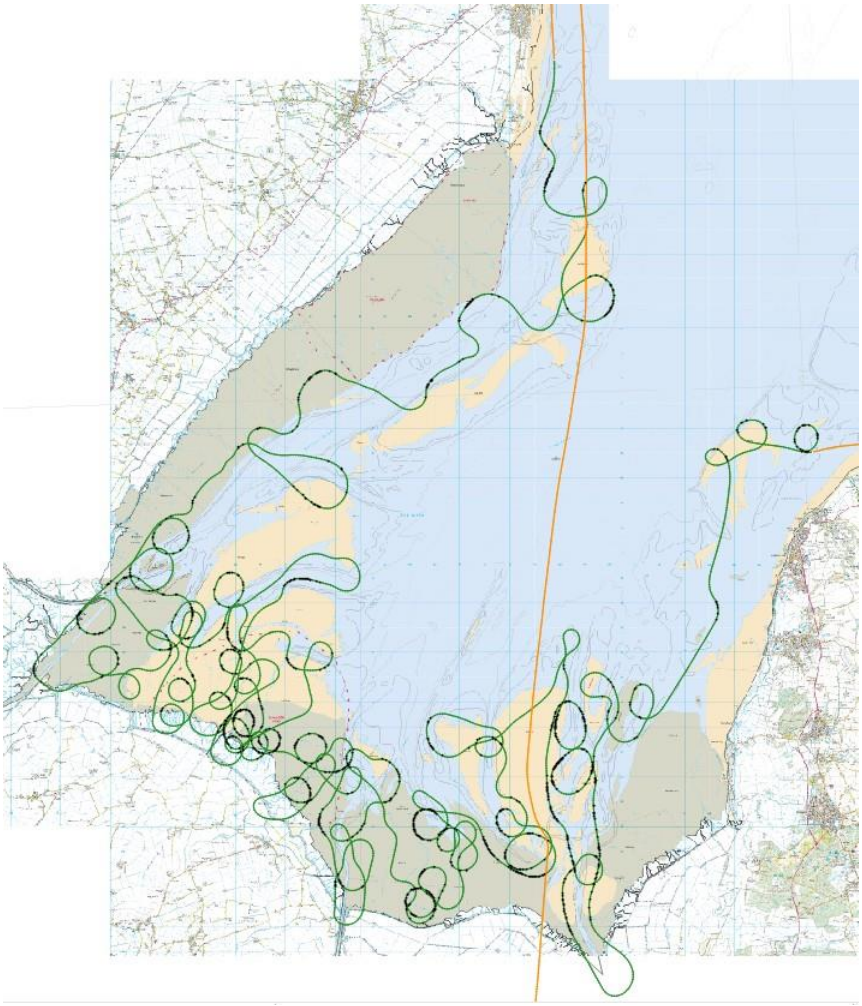


Figure 5. Distribution of pups in the Wash in 2014. Numbers of pups are represented by the areas of the circles on each site. Locations given to nearest 50m. Names of haulout sites together with latitudes and longitudes and numbers of seals at each site are given in Table 2.

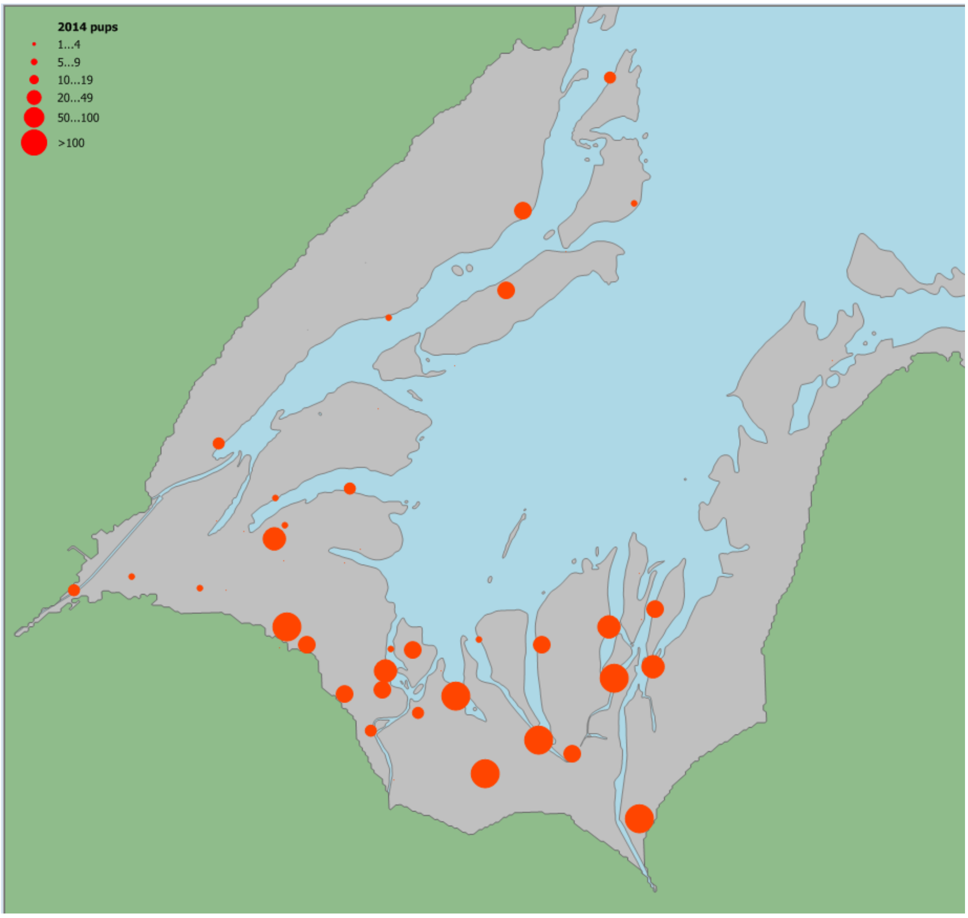


Figure 6. Maximum counts of pups in the Wash between 2001 and 2014. The fitted line is a simple exponential. Pup counts have increased at an average rate of approximately 9% p.a.

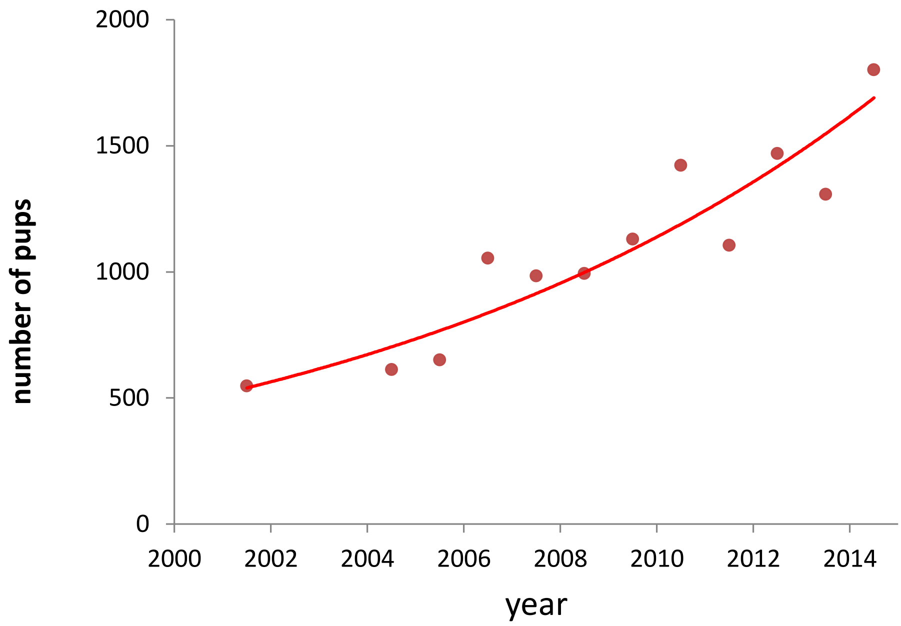
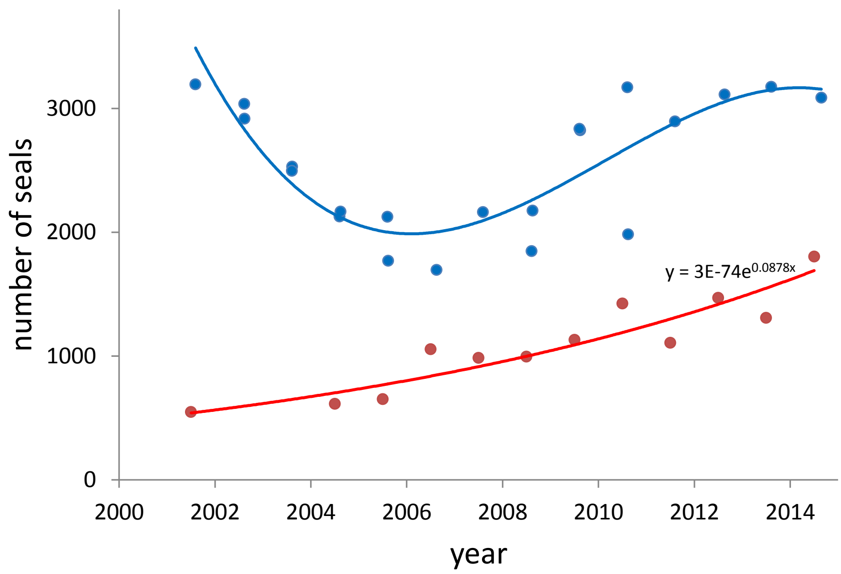


Figure 7.Maximum counts of pups in The Wash between 2001 and 2014 alongside the annual moult count over the same period. , An index of fecundity, derived as the peak pup count (an index of productivity) divided by the moult count (an index of population size).has increased over the period of surveys. The fitted line is a simple exponential through the pup counts and a cubic polynomial through the moult counts for illustration only



Updating adult female grey seal survival estimates at the Isle of May

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Abstract

This paper presents the most recent analyses of capture-mark-recapture (CMR) data from long term individual-based studies of grey seals at the Isle of May (IoM), Scotland, to provide updated apparent survival estimates. Results from different models, unbiased by tag loss, estimated apparent survival probabilities of between 0.92-0.94. There was some indication of preferential movement of seals from southern to northern pupping sites on the island.

Introduction

Estimates of adult female survival inform the grey seal population model (Thomas 2014; Thomas & Harwood 2009). Recent model formulations have pushed model-derived estimates of survival towards around 0.99 (Thomas 2014). Although the practical implications of such a high survival rate are biologically unlikely, previous empirical estimates put apparent female survival at the IoM at around 0.975, much higher than the 0.89 suggested at North Rona, but each consistent with pup production estimates and trends at their respective colonies (Smout et al. 2011a,b). Pup production at the IoM in 1977 was estimated at around 30, it had exceeded 2100 by 2000 and thereafter has stabilized around 2000+, with the associated rapid growth of the extended mainland colony between Cockburnspath and Eyemouth to the south. The increase in pup production on the IoM was associated with an expansion of areas used for pupping, particularly in the southern part of the island (Pomeroy et al. 2000, Twiss et al. 2001).

Practical considerations can have significant effects on the parameter estimates obtained, including duration of the study in relation to the lifespan of the subjects, marking method, assumptions about their availability to be resighted and aspects of the resighting schedule. The IoM data set is not homogenous in a number of ways, particularly for marking type. Therefore our analyses explore the potential biases and artifacts which may be present in such data. Smout et al.'s (2011a, b) estimates for apparent survival at the IoM were restricted to tag and brand resights up to 2006. Here, we extend the dataset to include animals identified by pelage markings up to and including 2010 for the first time, and also use resights of tagged and branded animals up to and including 2014.

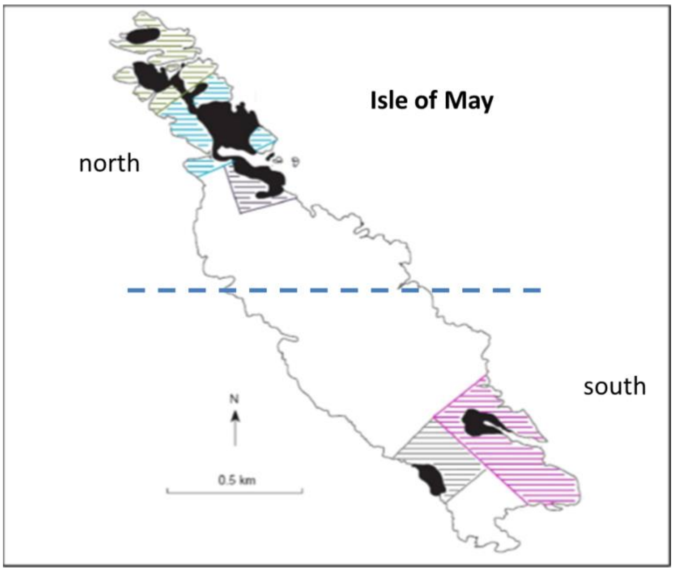
We tested the following hypotheses: (1) apparent survival probability and recapture probability are constant for adult female grey seals at the Isle of May; (2) recapture probability and apparent survival estimates are the same for seals with each type of identification mark; (3) given that seals survive, recapture probabilities and movement probabilities between north and south are the same over time at the IoM colony.

Materials and Methods

Study area

The Isle of May, 56°11'N 2°3' W, situated 8 km offshore in the Firth of Forth, Scotland, is a small island less than 2 km long and less than 0.5 km wide (Figure 1). Although the annual grey seal pup production on the island is one of the highest in the North Sea (Hiby et al. 1990, Duck & Morris 2008), much of its topography is unsuitable for breeding seals (Pomeroy et. al 2000). Up to the late 1990s, most births occurred on the northern part of the island, possibly due to its low lying topography and ease of access to the water, although the small rocky beach at Pilgrim's Haven was also regularly in use (Pomeroy et. al 2000, Figure 1). Subsequently, the southern part of the island around Kirkhaven and Kaimes has become heavily populated. For the purpose of this study all records of pupping locations on the Isle of May were categorized as either North or South (Figure 1).

Figure 1. Isle of May, Firth of Forth, showing main areas used as pupping sites (shaded) by grey seals and division of island into north and south pupping site categories.



Animal marking

Briefly, mothers are identified initially by branding, tagging or photographing. Branding of individuals started in 1987 and stopped in 2000; flipper tagging has continued from 1988 (Pomeroy et al. 2010). Both techniques require capture and handling of the animals, carried out under the U.K. Home Office License. Hot iron brands consisting of a letter-number combination were applied to each flank after anaesthetising the seal. Brands are quick, long-lasting, extremely visible and a reliable way of identifying an individual, even when the condition of the brand is already considered of low quality (Harwood et al. 1976, Schwarz et al. 1997, Pomeroy et al. 1999). Individually numbered plastic tags applied to interdigital hind flipper webs are subject to loss after application and are not easy to read, as they comprise five digits which wear. Ingrained dirt or impaired view may render them temporarily or permanently illegible even if the tag is retained (Pomeroy et al. 1999). Seals were flipper tagged either as breeding adults or as pups in associated work.

Pelage identification uses natural, stable pelage patterns to identify individual females. It began on the IoM in the late 1990s and continues, with the advantage that animals do not need to be handled (although some may be). The ExtractCompare semi-automated photo-ID system allows comparison of extracted pelage patterns from multiple parts of the body (Hiby et al. 2012).

Following Smout (2011a,b) multiple-marked animals were classed according to the "priority sighting" model where the most obvious marking type was used (brand>pelage> tag).

Presence, breeding status and location of recognizable animals was recorded daily by surveys within the colony, collected during daylight hours. The work in the colony was restricted to minimize disturbances. Photo-id resights were generated *post-hoc*.

Breeding site fidelity

Breeding site fidelity of adult females at the Isle of May was examined using fieldwork observations and *post hoc* resights from the SMRUPHOT photo-ID database. For this study, only the females with pupping site location (identifiable as a location in the north or south of the island where the mother was seen with her pup) were included in the database to evaluate individuals' site fidelity within the island. Overall, this dataset consists of records from 800 mothers.

CMR analyses

Program MARK (White and Burnham 1999) is a widely used programme for analyzing CMR data using standard models. Program UCARE (Choquet et. al 2005) deals specifically with checking the fit of data with assumptions of models.

CMR Models used

The standard Cormack Jolly-Seber model (Lebreton et al. 1992) in MARK is used to test our first two hypotheses (Table 1 and Table 2). This model allows for year-specific estimates of apparent survival (Φ) and recapture (p) probabilities in an open population. However, CJS models assume homogeneous survival and recapture probabilities among animals of the same type, and that sampling duration is negligible compared to intervals between samples.

Table 1. Hypothesis 1: apparent survival probability and recapture probability are constant for adult female grey seals at the Isle of May

CJS model, apparent survival Φ and recapture probability p can be time constant (.) or time specific (t). Where time specific estimates are indicated, we give arithmetic means for convenience.

Seals marked by brands B, T, or pelage P.

Data set		**All marks B, T, P; to 2014**			**All marks B,T,P; to 2010**			**Marks B, T; to 2014**			**Marks B, P; to 2010**		
Variables		best model	estimate	95%CI	best model	estimate	95%CI	best model	estimate	95%CI	best model	estimate	95%CI
Model CJS	Φ survival probability	(t)	$\bar{x}$ = 0.91		(.)	0.93	0.92-0.94	(.)	0.92	0.91-0.94	(.)	0.92	0.91-0.94
	p recapture probability	(t)	$\bar{x}$ = 0.41		(t)	$\bar{x}$ = 0.38		(.)	0.55	0.53-0.58	(t)	$\bar{x}$ = 0.38	

Table 1a. CJS model outputs

(a)			(b)			(c)			(d)		
Best model	**Φ(t).p(t)**		Best model	**Φ(t).p(t)**		Best model	**Φ(t).p(t)**		Best model	**Φ(t).p(t)**	
∑ GOF test	χ2	1057.29	∑ GOF test	χ2	1037.32	∑ GOF test	χ2	329.1	∑ GOF test	χ2	959.1045
	df	145		df	120		df	106		df	111
	ĉ	7.292		ĉ	8.644		ĉ	3.105		ĉ	8.641
Test 2CT	χ2	447.11	Test 2CT	χ2	435.2	Test 2CT	χ2	136.96	Test 2CT	χ2	430.94
	df	25		df	21		df	25		df	21
	p	0		p	0		p	0		p	0
Test 3SR	χ2	254.44	Test 3SR	χ2	278.3	Test 3SR	χ2	67.42	Test 3SR	χ2	225.3809
	df	25		df	21		df	23		df	19
	p	0		p	0		p	0		p	0
Best model (ĉ adjusted)	**Φ(t).p(t)**		Best model (ĉ adjusted)	**Φ(.).p(t)**		Best model (ĉ adjusted)	**Φ(.).p(.)**		Best model (ĉ adjusted)	**Φ(.).p(t)**	

Table 2. Hypothesis 2: apparent survival probability and recapture probability are constant for all mark types on adult female grey seals at the Isle of May

CJS model, apparent survival Φ and recapture probability p can be time constant (.),time specific (t), mark type specific (g) or both time and mark type specific (g*t).

Seals marked by brands B, tags T, or pelage P

Data set		**All marks B, T, P; to 2014**			**All marks B,T,P; to 2010**			**Marks B, T; to 2014**			**Marks B, P; to 2010**		
Variables		best model	estimate	95%CI	best model	estimate	95%CI	best model	estimate	95%CI	best model	estimate	95%CI
Model CJS	Φ survival probability	(g)	T: 0.87	0.83-0.89	(g)	T: 0.85	0.80-0.89	(g)	T: 0.87	0.84-0.90	(.)	0.94	0.93-0.95
			B: 0.95	0.94-0.96		B: 0.95	0.93-0.96		B: 0.95	0.94-0.96			
			P: 0.94	0.92-0.95		P: 0.94	0.92-0.95						
	p recapture probability	(g*t)	T: $\bar{x}$ = 0.25		(g*t)	T: $\bar{x}$ = 0.22		(g*t)	T: $\bar{x}$ = 0.26		(g*t)	B: $\bar{x}$ = 0.63	
			B $\bar{x}$ =0.63			B $\bar{x}$ =0.54			B: $\bar{x}$ = 0.63			P: $\bar{x}$ = 0.26	
			P: $\bar{x}$ =0.23			P: $\bar{x}$ =0.26							

Table 2b. CJS model outputs

(a)				(b)				(c)				(d)		
Best model	**Φ(t).p(g*t)**			Best model	**Φ(t).p(g*t)**			Best model	**Φ(g).p(g*t)**			Best model	**Φ(t).p(g*t)**	
∑ GOF test	χ2	641.39		∑ GOF test	χ2	575.61		∑ GOF test	χ2	232.52		∑ GOF test	χ2	520.1219
	df	205			df	163			df	136			df	124
	ĉ	3.129			ĉ	3.531			ĉ	1.710			ĉ	4.195
Best model (ĉ adjusted)	**Φ(g).p(g*t)**			Best model (ĉ adjusted)	**Φ(g).p(g*t)**			Best model (ĉ adjusted)	**Φ(g).p(g*t)**			Best model (ĉ adjusted)	**Φ(.).p(g*t)**	

The multi-state model is a generalization of the basic CJS formulation, taking account of potential movements between states (locations or life stages). In the multi-state Arnason-Schwarz model, probability of survival is replaced by a probability of survival-movement which assumes that survival from time *i* to *i*+1 does not depend on the state at *i*+1. Here, we use the multi-state model to examine the conditional survival S (the probability of survival from *i* to *i*+1 given that an animal is in state A at time *i)* and recapture probabilities (p) in both locations (North and South) as well as calculate the conditional probability that an individual in state A at time *i* moves to state B at time *i*+1 given that the individual is alive at *i*+1. This is given as (Ψ) in each direction.

Each parameter of each model was run as time-dependent or constant. Thus, multiple models were generated to account for all the combinations of these parameters. Model selection relied on the likelihood criterion Akaike Information Criterion (AIC) (Akaike 1981). The AIC analyzes the fit of the model, penalizing according to the number of parameters used during the process. The model with AIC closest to zero is classified as the most parsimonious. AICc incorporates a small sample bias adjustment, useful if the model has many parameters in relation to data. Model selection was determined by comparison of AICc values (delta AICc). Models with a delta AICc between 4 and 7 indicated a difference in support for the models and those with a difference of more than 10 indicated strong evidence of differences between models (Burnham & Anderson 1998). The best fitting model was used to report parameter estimates of interest.

The program U-CARE was then used to assess goodness-of-fit (GOF) of the model to the data, a statistical test that calculates the over-dispersion estimation factor (Lebreton et al. 1992), or variance inflation factor (Burnham et. al 1992), c-hat. U-CARE considers 4 tests (3.SR, 3.SM, 2.CT and 2.CL), each examining different characteristics of the data. The 3.SR and 3.SM tests are used to detect the prevalence of transient animals (single encounters, due to permanent emigration or mortality). Tests 2.CT and 2.CL determine if there were 'trap-dependent' or 'trap-shy' animals in the dataset which can be used as a proxy for the level of capture heterogeneity (Pradel 1993).

Each of the main models was tested using a Pearson $\chi 2$ test and G2 test in U-CARE. If estimations produce unexpectedly low or high test values, models are likely to be unreliable. In these cases, UCARE applies a Fisher exact test. The $\chi 2$ is then calculated again and the p-value alters accordingly. The outcome of the 4 tests are pooled together to generate the goodness of fit (GOF) result and to calculate c-hat. The c-hat is then adjusted in the generated models in MARK, since MARK automatically gives a c-hat of 1.0 (assuming perfect fit) when fitting models. Adjusting c-hat renders AICc unusable because of over-dispersion, therefore AICc value is replaced by the Quasi Akaike Information Criterion (QAICc) for assessing the models.

Our dataset comprises animals recognized using different mark types and over different periods. The full dataset includes 86 branded, 158 tagged and 556 pelage-id seals, marked at different times during the study. Tags and brands have been resighted by observers every year of the study. Photo-ID records have been processed, compared and completed for 2 separate periods, 1998-2001 and 2007-2010, using extracts from 9167 photos.

To examine the effects of data heterogeneity, each hypothesis was analyzed using (a) all mark types (800 seals) over all years for which data was available; (b) all mark types up to 2010, i.e. all mark types restricted to years when all types were monitored, (771 individuals); (c) tagged and branded animals in all years, i.e. ignoring pelage animals (244 individuals); and (d) photo-ID and branded animals up to 2010, considering only animals and years in which any effects of tag loss would be negligible (652 individuals). Comparing across these data groups indicates the influence of omission or inclusion of extra data years and/or mark types.

Results

Hypothesis 1: Cormack Jolly Seber (CJS)

Table 1 presents the parameter estimates for the best model in each of a, b, c, d data groups.

Using all data (a), both apparent survival and recapture are time dependent. Using only years to 2010 (b) the best model gives a common φ, 0.93 (95% CI 0.92-0.94) but time dependent recaptures. If pelage id seals are ignored (c) the best model gives a common φ of 0.92 (95% CI 0.91-0.94) and common p of 0.55. Using only brands and pelage records to 2010, the best model gives a common φ of 0.92 (95% CI 0.91-0.94) and time dependent p.

Hypothesis 2: CJS by mark type

Table 2 presents the parameter estimates for the best model in each of a, b, c, d data groups.

Using all data (a) the best model gives constant mark-specific estimates of apparent survival [Φ T=0.87 (0.83-0.89), B=0.95 (0.94-0.96), P=0.94 (0.92-0.95)] and time-dependent recapture (p) probabilities for each mark type. The variation in time dependent recapture probabilities is partly explainable by features of the dataset and sampling regime (see legend to Figure 2).

Using only years to 2010 (b) the best model gives similar results: constant mark-specific estimates of apparent survival [Φ T=0.85 (0.80-0.89), B=0.95 (0.93-0.96), P=0.94 (0.92-0.95)] and time-dependent recapture (p) probabilities for each mark type.

If pelage id seals are ignored (c) the best model gives constant mark-specific estimates of apparent survival [Φ T=0.87 (0.84-0.89), B=0.95 (0.94-0.96)] and time-dependent recapture (p) probabilities in each mark type.

Using only brands and pelage records to 2010, the best model gives a common φ of 0.94 (95% CI 0.93-0.95) and time dependent p for each mark type.

Hypothesis 3: Multi-state model

Table 3 presents the parameter estimates for the best model in each of a, b, c, d data groups.

Using all data (a), the best model gives time dependent survival in each N and S parts of the island. Recapture probabilities were constant in N at 0.50, but time dependent in S. Seals were as likely to move pupping sites from N:S as S:N (p=0.11, 0.12 respectively).

Using only years to 2010 (b) the best model gives constant but different survival estimates for N and S areas
[S N=0.97 (95% CI 0.95-0.98), S=0.90 (0.88-0.92). Recapture probability was constant in N at 0.47 (95% CI 0.44-0.49) but time-dependent in S (Figure 3). Movement probabilities were time dependent from N to S (Figure 4), but constant at 0.11 (95% CI 0.10-0.13) from S to N.

If pelage id seals are ignored (c) the best model gives constant and similar survival estimates for N and S areas [S N=0.92 (95% CI 0.91-0.94), S=0.92 (0.88-0.95). Recapture probability was constant in N at 0.54 (95% CI 0.51-0.57) and constant in S at 0.58 with a wider confidence interval (95% CI 0.50-0.66). Movement probabilities for NS and SN were constant with time and different: NS= 0.04 (95% CI 0.03-0.05) and SN= 0.13 (95% CI 0.10-0.18).

Using only brands and pelage records to 2010, the best model found constant S=0.93 (95% CI 0.91-0.94) and S=0.95(0.93-0.97) in N and S respectively. Recapture probability was constant in N at 0.54 but time dependent in S (Figure 5). Seals were as likely to move pupping sites from N:S as S:N (p=0.11, 0.12).

Table 3. Hypothesis 3: survival probability given resighting, the recapture probability and the probability of movement between N and S is the same for the north (N) and the south (S) of the IoM. Multi-state model
Survival S and recapture probabilities p can be time constant (.) or time specific (t). Seals marked by brands B, tags T, or pelage P. Movement probability is designated by Ψ.

Data set		**All marks B, T, P; to 2014**			**All marks B,T,P; to 2010**			**Marks B, T; to 2014**			**Marks B, P; to 2010**		
Variables		best model	estimate	95%CI	best model	estimate	95%CI	best model	estimate	95%CI	best model	estimate	95%CI
Model: multi-state	S survival probability given resighting	N: (t)	$\bar{x}$ = 0.90		N: (.)	0.97	0.95-0.98	N: (.)	0.92	0.91-0.94	N: (.)	0.93	0.91-0.94
		S: (t)	$\bar{x}$ = 0.91		S: (.)	0.90	0.88-0.92	S: (.)	0.92	0.88-0.95	S: (.)	0.95	0.93-0.97
	p recapture probability	N: (.)	0.50	0.47-0.54	N: (.)	0.47	0.44-0.49	N: (.)	0.54	0.51-0.57	N: (.)	0.54	0.50-0.58
		S: (t)	$\bar{x}$ =0.22		S:(t)	$\bar{x}$ =0.35		S: (.)	0.58	0.50-0.66	S: (t)	$\bar{x}$ =0.15	
	Ψ movement probability	NS: (.)	0.11	0.09-0.14	NS:(t)	$\bar{x}$ =0.11		NS: (.)	0.04	0.03-0.05	NS: (.)	0.11	0.09-0.14
		SN: (.)	0.12	0.10-0.14	NS:(.)	0.11	0.10-0.13	SN: (.)	0.13	0.10-0.18	SN: (.)	0.12	0.10-0.14

Table 3a. Multi-state model outputs

(a)			(b)		
Best model	**SA(t).SB(.).pA(t).pB(t).ΦA-B(t)ΦB-A(t)**		Best model	**SA(.).SB(.).pA(t).pB(t).ΦA-B(.)ΦB-A(.)**	
∑ GOF tests 3G and M	χ2	661.02	∑ GOF tests 3G and M	χ2	622.333
	df	165		df	140
	ĉ	4.006		ĉ	4.445
Best model (ĉ adjusted)	**SA(t).SB(t).pA(.).pB(t).ΦA-B(.)ΦB-A(.)**		Best model (ĉ adjusted)	**SA(.).SB(.).pA(.).pB(t).ΦA-B(t)ΦB-A(.)**	
(c)			(d)		
Best model	**SA(.).SB(.).pA(t).pB(.).ΦA-B(t)ΦB-A(t)**		Best model	**SA(t).SB(.).pA(t).pB(t).ΦA-B(t)ΦB-A(t)**	
∑ GOF tests 3G and M	χ2	190.735	∑ GOF tests 3G and M	χ2	635.285
	df	76		df	130
	ĉ	2.510		ĉ	4.887
Best model (ĉ adjusted)	**SA(.).SB(.).pA(.).pB(.).ΦA-B(.)ΦB-A(.)**		Best model (ĉ adjusted)	**SA(.).SB(.).pA(.).pB(t).ΦA-B(.)ΦB-A(.)**	

Figure 2. CJS Model 2 output: all data, as in Table 2 (a). Time dependent recapture probabilities for each mark type, tag, brand and pelage (photo-id). Low tag recapture from 1991-1997 explained by absence of tag-only animals in marked population; low pelage recaptures in early 2000s and post-2010 due to unprocessed pictures.

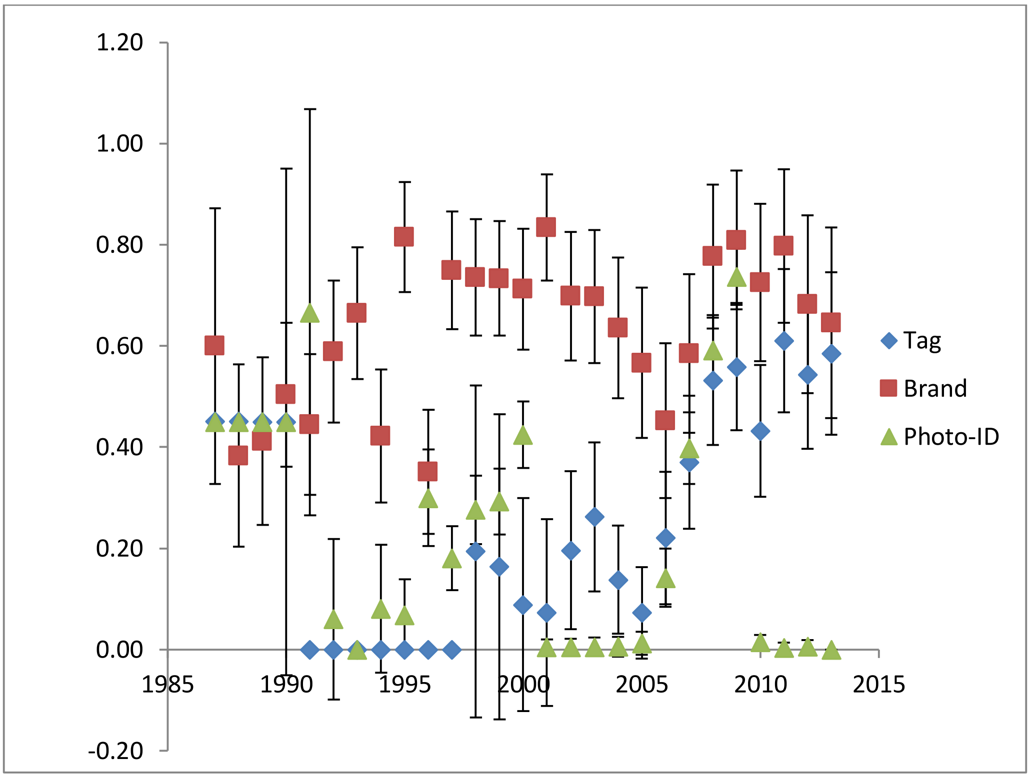


Figure 3. Model 3 multi-state. All mark types to 2010 as in Table 3 (b). Time -dependent recapture probabilities for seals in the south of the Isle of May.

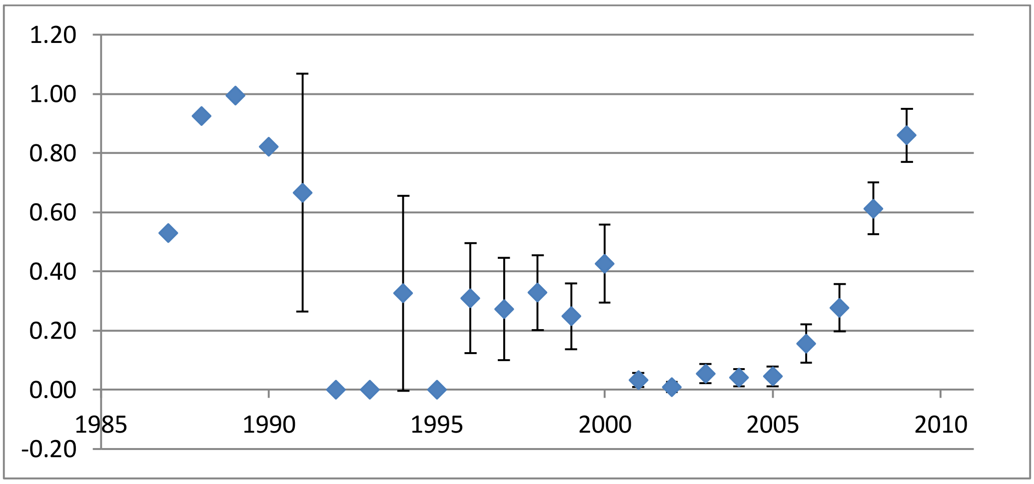


Figure 4. Model 3 multi-state. All mark types to 2010 as in Table 3 (b). Time- dependent movement probabilities Ψ from N to S.

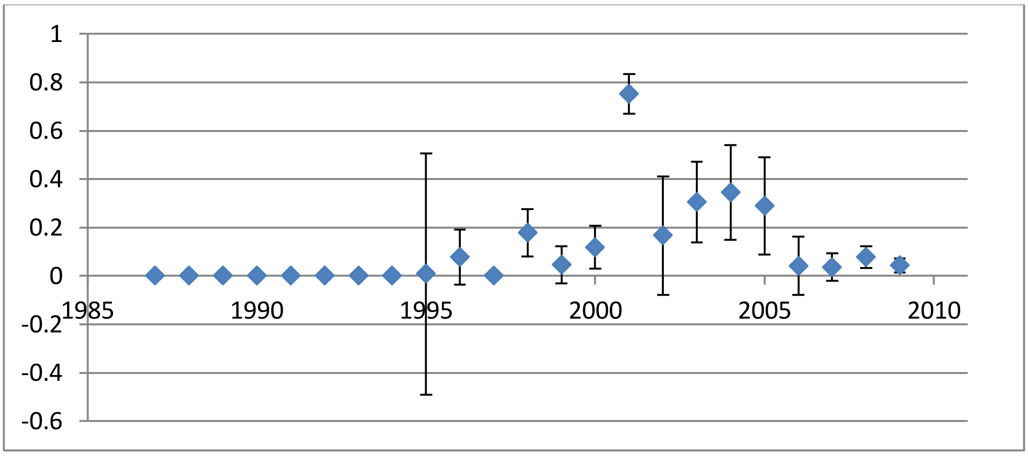
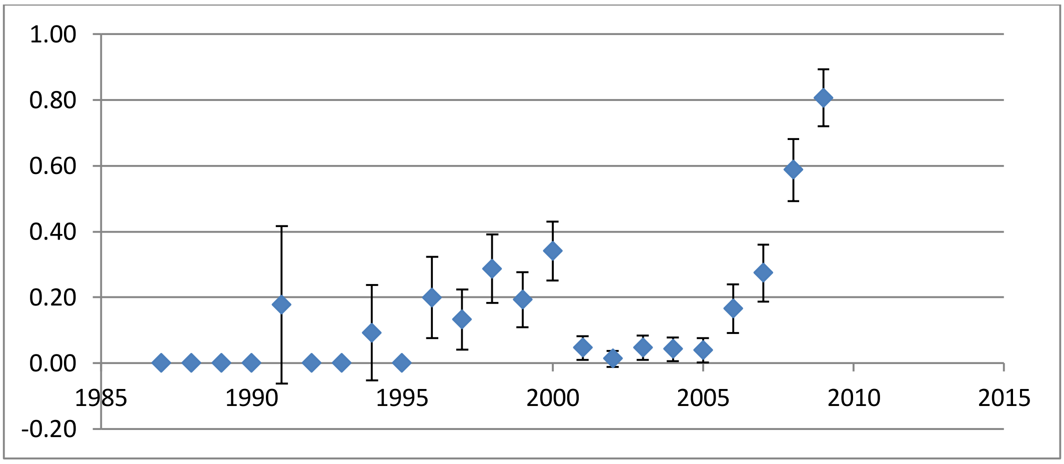


Figure 5. Model 3 multi-state. Brands and pelage to 2010 as in Table 3 (d). Time- dependent recapture probabilities in south of Isle of May.



Discussion

Updated apparent survival estimates for the most complete resighting dataset of grey seals breeding at the IoM to 2014 gave lower values than derived previously (typically 0.93 vs 0.975, Smout et al. 2011b). However the current dataset adds up to 9 years' resights to tagged and branded seals and includes pelage-ID seals at the Isle of May for the first time. Thus not only does the current dataset comprise many more animals than Smout et al.'s data but it also includes a large number of non-handled animals. One potential explanation for our new lower survival estimate may be that many study animals up to 2006 were probably in their prime breeding years.

Smout et al.'s (2011) models adjusted for tag loss explicitly, which we do not. The effects of tag loss may be seen by comparing the estimates derived for tags and brands with those using only pelage and brand data. In CJS models (Table 2), tagged animals had the lowest apparent survival rates (0.85) compared to other mark types, and also had low recapture probabilities. Apparent survival rates for brands and pelage animals (0.94) were not subject to tag loss, but the low resight probabilities for the pelage-ID seals suggests either poor coverage of the animals, or failure to make matches, possibly because of patterns obscured by mud, low light, or poor photographs. In the multi-state models (Table 3), comparison of survival estimates for all marks to 2010 with brands and pelage to 2010 (groups (b) and (d)) show that the effect of including tagged animals is not the same in the north as in the south. In the north, removing tagged animals caused the survival rate to fall from 0.97 to 0.93, while in the south the estimate rose from 0.90 to 0.95. Taken along with movement parameters and the fact that very few branded animals move between N and S (see below) this might suggest that tagged animals are more likely to stay in the north and move from the south.

Small differences in annual survival rate have large consequences for longevity. For example, after 10 years experiencing survival rates of 0.95 and 0.93 there will be 599 and 484 survivors respectively from 1000 animals, and after 20 years there are 358 and 234 (Appendix Table 1).

The extent to which site fidelity occurs in grey seal breeding colonies can be very different. Breeding sites are believed to be chosen according to the absence of predators and the easy access to the sea (Bartholomew 1970). "Traditional" breeding colonies have been used since historical times. When a certain population increases, each colony may expand in area (Hiby et al. 1990, Pomeroy et al. 2000) and/or new colonies may form (e.g. Duck & Morris 2011). Grey seals in the UK have done both. The factors that contribute to expansion of a breeding colony are still poorly understood, however physical characteristics of sites are important (Twiss et al. 2000).

Newly occupied breeding sites in an established colony are likely to be peripheral, and occupied by primiparous or less competitive females. New sites are likely to have a higher cost in accessing water, either energetically or in interactions with conspecifics and are probably less suitable for seals, likely leading to an increase in pup mortality (Pomeroy et al. 2000). Therefore it seems reasonable to expect that, once a female has been pupping at a given site, it would be riskier to move elsewhere and be a newcomer there than stay and use experience and familiarity to make the best of the known site. Similarly, only primiparous or less competitive females might be expected to move from the traditional sites on the north of the IoM to the south, although conditions in the south changed rapidly as new areas became full. Animals moving from the south might find only experienced animals in the north. The most striking feature of the movement parameters in our models is the difference between NS movement probabilities for brands and tags to 2014 (0.04) compared to 0.11 for brands and pelage animals. It is not clear whether this difference reflects a greater tendency for pelage ID animals to move south than tagged animals, or if behaviour changed substantially over the last few years. In all models, the probability of movements S:N was around 0.12.

The spatial accuracy of site fidelity at the Isle of May colony for tagged and branded animals before the south was used heavily was more extreme than that on N Rona (Pomeroy et al. 2000). This contrasts with our estimate here of 0.12 probability of movement between N and S on the colony. In addition, movements within and between years to different breeding colonies are known to occur (e.g. Isle of May to the Farne Islands: Pomeroy et. al 2000). Smout et al. (2011b) found no support for transient animals in the Isle of May data to 2006, but no pelage identified animals were considered. Some naturally marked animals may well behave differently to the captured and handled group.

Resighting known animals in a seal colony is a challenge. This is not academic – repeated measures on known animals are required for long term life history studies and recognizing individuals in the field is a fundamental requirement. To some extent breeding site fidelity and stability of birth date assist recaptures. However, the balance of marked animals in the colony has changed over time. That resighting probabilities are so high in recent years for tagged animals particularly is testament to the effort expended in collecting these data. Processing of the accumulated photo backlog, particularly from 2011-2014 is critically important now as branded animals die out, and has the potential to reveal how effective it is in this colony but also extends the possibilities of picking up movements between adjacent and other N Sea colonies also being photographed.

We conclude that our new estimates of adult female survival rates centered around 0.93, and that these should be robust to mark loss. The probability of interannual movements of seals between N and S was no greater than 0.12 and there was some evidence of directional movement according to mark type.

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Appendices

Appendix Table 1. The number of surviving adult seals from a group of 1000 experiencing different survival rates.

		survival rate				
	years	0.97	0.95	0.93	0.91	0.89
number alive after n years	10	737	599	484	389	312
	15	633	463	337	243	174
	20	544	358	234	152	97

Appendix 1

CMR data format

The format and attributes of the input data in MARK and U-CARE are very similar. The first columns must correspond to each occasion, in this case 28 years, in which "1" (if present and alive) or "0" (if absent or not encountered) are displayed. In case of more complex models, "1" can be replaced by occasions ("A": North, "B": South) or dead-live encounter data ("10": alive and recorded, "01": encountered dead, "02": not encountered but known alive because is recorded in later occasions). The data format of this last example is commonly called "LDLD" (Live-Dead) and is coded accordingly for MARK. For U-CARE both "10" and "02" are replaced by "1", while "01" is "2" and GOF test is performed using multi-state analyzes.

Appendix 2

SMRUPHOT is a large catalogue of grey seal pictures. Taken mostly of adult females, it contains pictures and their associated metadata and extracted 2d pelage patterns taken from a 3d model of a grey seal. ExtractCompare is the software which allows image handling, pattern extraction and semi-automated matching procedure in which only the top ranking potential matches are offered for manual inspection and confirmation. The process is based on the photo-identification method, matching the animals with similar pelage patterns and assigning each individual a numerical identification. The process of matching photos is critical and identification is only accepted when there is a high confidence value on the matching (Karlsson et al. 2005, Hiby et al 2012).

Interpretation of the pelage data is complex. The IoM photo dataset currently comprises photos processed mainly from 1998-2001 and 2007-2010 inclusive. In the early part of the study, fewer pictures were taken, thus sampling effort is most definitely uneven between the two periods covered and almost absent in the 5 year interval between. Hiby et al. 2012 note that the chance of missing matches is 33% when 2 photos of the same aspect of the same seal are compared, but this falls rapidly as more images are acquired. In the current photo dataset, there are 654 unique ids that have been seen 2 or more times, but restricting this to 2007-2010, there are 381 individuals sighted at least twice. With annual pup production on the island currently around 2100, this means that the majority of the animals using the colony are unknown.

Appendix 3

The Barker model (Barker 1997, 1999) is an extension of Burnham's (1993) live-dead model to the case where live re-sightings are reported during the open period between live recapture occasions. Our data does not contain dead recoveries, but with these set to zero, this model effectively assesses 'fidelity' (the likelihood of returning to the colony, F and F') in successive occasions within their encounter history.

Results (Appendix Table 2). Tagged and branded animals show a high probability of returning in successive years (0.90, 95%CI 0.85-0.93), compared to a lower estimate for brands and pelage animals (0.69 95% CI 0.65-0.73). For these two groups the likelihood of returning after a year's absence was 0.33 (95%CI 0.26-0.41) and time variable.

Appendix Table 2. Barker model: Temporary absence: seals will return to the island after an absence with the same probability as if they were there in successive years. Survival S and recapture probabilities p can be time constant (.) or time specific (t). Seals marked by brands B, tags T, or pelage P. F denotes probability of successive presences, F' denotes probability of presence following an absence.

		All marks B, T, P; to 2014			**All marks B,T,P; to 2010**			**Marks B, T; to 2014**			**Marks B, P; to 2010**		
Variables		best model	estimate	95%CI	best model	estimate	95%CI	best model	estimate	95%CI	best model	estimate	95%CI
Model: Barker	S survival probability given resighting	(t)	$\bar{x}$ =0.91		(.)	0.92	0.91-0.93	(.)	0.91	0.89-0.92	(.)	0.93	0.92-0.94
	p recapture probability	(.)	0.71	0.66-0.75	(.)	0.81	0.76-0.85	(.)	0.75	0.70-0.79	(.)	0.84	0.79-0.88
	F probability of successive presences	(.)	0.80	0.75-0.84	(.)	0.70	0.66-0.74	(.)	0.90	0.85-0.93	(.)	0.69	0.65-0.73
	F' probability of presence after absence	(.)	0.26	0.23-0.30	(t)	$\bar{x}$ =0.30		(.)	0.33	0.26-0.41	(t)	$\bar{x}$ =0.29	

Harbour seal population modelling: the Moray Firth

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Abstract

In parts of Scotland, harbour seals (*Phoca vitulina*) have been in decline for more than 10 years, suggesting an urgent need to better understand the underlying causes. Building on earlier work, behavioural, demographic and population data from a population in part of the Moray Firth (north-east Scotland) was used to fit an age-structured population model in order to estimate vital rates and changes in these rates over time. A Bayesian hidden process approach facilitated detailed modelling of observation errors e.g. allowing for the behaviour of animals to influence the probability of observing them. The effects of removals due to shooting were included. Forecasts from the model suggest a slow population recovery in the near future. Of the demographic rates, the fecundity rate seems to vary most rapidly, suggesting this parameter is particularly sensitive to short-term environmental changes. Simulations suggest the study population is sensitive to changes in the survival of adult females: a small number of extra removals will result in overall population decline. The possible impact of covariates on vital rates was also investigated including prey, environmental indices, and biological variables such as grey seal population density and concentration of biotoxins. Evidence of an effect was found for two of these: (a) grey seal (*Halichoerus grypus*) abundance (affecting pup survival) (b) sandeel abundance (affecting fecundity). With the grey seal abundance covariate included in the model the projected near-future population trend was different to the one in the baseline model, with a decreasing population arising from a reduced pup survival rate, which was linked to a projected continuing increase in the overall grey seal population size.

Introduction

The goal of this work was to develop methodology to evaluate and test different hypotheses about the causes of change in harbour seal populations (Cordes & Thompson 2013) based on the analysis of data from a single well-studied location, the northern Moray Firth (Figure 1). The count data used in this study are exceptional due to the long time series and the number of repeated surveys per year (Table 1). All the other time-series count data at Scottish sites are collected during the moult, with very few or no breeding season counts.

Figure 1. Map of the Moray Firth area showing the location of haul-out sites in the study area (▪) which lies north-east of the dashed line and includes the Dornoch Firth SAC

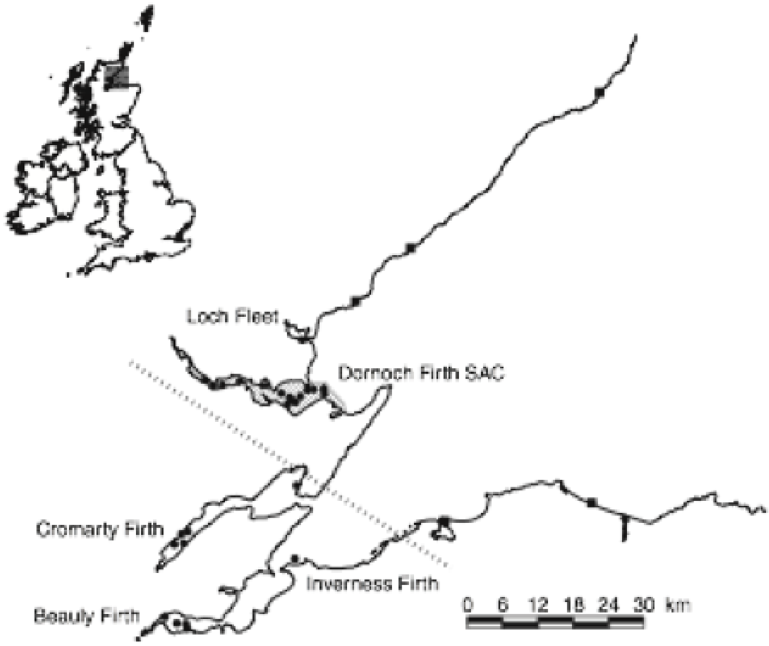


Table 1. Summary of the data (counts and covariates) used.

		1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012
	Survey types	**Ground**																		**Ground and aerial**	**Aerial**					
Harbour Seal counts	No of Breeding Surveys	2	6	2	0	2	8	4	5	5	5	5	5	5	5	5	5	2	4	10	4	5	4	5	4	4
	No of Moult Surveys	1	1	1	2	2	10	0	2	2	0	3	2	5	3	2	2	2	2	2	2	2	1	1	1	1
Grey Seals	No of surveys																			5	4	5	4	5	4	4
Harbour Seals	Independent estimate						1																1			
	Herring																									
	Cod																									
	Sprat																									
	Sandeels																									
	SST																									
	NOA																									
	Saxitoxin																									
	D.A																									

Based on previously published work, an age-structured population model was developed to estimate harbour seal demographic rates such as fecundity and survival for different age classes. Count data and two independent estimates of population size, based on telemetry and capture-recapture photo-ID studies were used to fit the model along with historical records of shooting of seals in the area. A Bayesian hidden process approach facilitated detailed modelling of observation errors e.g. allowing for the haul-out behavior of animals to influence the probability of observing them. Using this baseline model, three objectives were addressed.

Using simulations, the likely trends in the population in future years was estimated. The sensitivity of the population to different scenarios of fecundity, survival or seal management was explored.

The possible effects of covariates which might impact vital rates was examined: the model was fitted again to the seal count data and independent estimates, but also to covariate data including prey abundance, climate indices, and the local population size of grey seals.

To explore the potential to fit similar models at sites where fewer data are available, the baseline model was re-fitted using subsets of the original data. Results were then compared with those obtained using the full data set.

Methods

An age-structured, discrete time population model (the baseline model) was fitted to harbour seal count data collected during aerial and ground-based surveys of animals hauled-out during the breeding and moulting seasons in Loch Fleet between 1988 and 2012 (see Table 1). Records of shooting in the area were also included. Two estimates of total population size based on photo-ID capture-recapture studies were used as additional data points when fitting the model (these are referred to as 'independent estimates'). A Bayesian hidden process method implemented using WinBUGS was used to estimate age and sex-specific survival rates and female fecundity, with informative priors based on independent studies (Cordes & Thompson 2014). The resulting baseline model was structured as described in Matthiopoulos et al. (2013) with the following modifications:

The original two-peak seasonal model of haul-out probability was changed to a step-function model that includes two different estimates of haul-out probability, one for breeding and one for moult and specific to each age class and sex (Cordes & Thompson 2015). Haul-out probability can vary from site to site (Huber et al. 2001, Lonergan et al. 2013) and at numerous sites the magnitude of the haul-out probability is not known. Therefore different assumptions for these values were explored to determine whether they have a strong effect on conclusions about overall trends in demographic rates. The results of the baseline model were found to be robust to these changes in haul-out probability: consistent demographic trends were detected over a range of plausible haul-out probabilities that were tested.

To represent stochasticity in the observation process, a daily variability representing the 'noise' around each average haul-out probability was added to the constant baseline probabilities. This daily variability could be due to anthropogenic disturbances prior or during the survey, or environmental conditions e.g. weather, food availability.

The new model allowed that some animals counted at moult (a small proportion) might be pups born in the same year.

The time-dependence in survival and fecundity was simplified, reducing the number of parameters in the model. Vital rates became linear functions of time, with a density-dependent term.

Harbour seal usage of different areas can change over time (Cordes et al 2011). The division of the local harbour seal population between the main haul-outs in Loch Fleet and other haul-outs that are only included in the aerial surveys (e.g. at Brora) was explicitly modelled.

Objective 1

The model was then used to obtain projections of future population size up to 2020. For 'future' years i.e. those beyond the years for which data were available, vital rates (survival, fecundity) were assumed fixed at the final values estimated by the model-fitting i.e. those for 2012.

The effects of changes in survival and fecundity rates of different age classes in the population were also explored. Demographic trend predictions for the next 10 years, with different scenarios, were carried out. The scenarios were:

- Change in the fecundity rate.
- Change in the survival rate in different age classes.
- Direct mortality of individuals in different age classes.
- Impact of the local grey seal population size.

The results of fitting the baseline model suggest that of the demographic trends, the fecundity rate appears to be the most variable in time and it is suggested that this parameter is most sensitive to environmental change, so may be of particular interest. To simulate the impact of a decreasing fecundity rate on the population the survival rates were sampled from the posterior distribution produced by fitting the baseline model. For each simulation the fecundity rate was fixed, and it was decreased from 5% to 50% of the predicted value in 2012 (by steps of 5%). After each simulation the net population growth rate over 10 years with these specific parameters was calculated.

The same simulation was carried out with a fecundity sampled from the posterior distribution, but the survival rates of the pups, juveniles, adult males and adult females were fixed with values ranging from 50% to 95% of the estimated survival rates in 2012.

Finally to simulate the impact of removals of known numbers of animals e.g. due to boat collisions or predation, between 1 to 50 individuals of a specific age class were removed every year, and the impact on the growth rate of the population over 10 years was recorded.

Objective 2

The model was adapted with the objective of quantifying the possible contribution of specific environmental mechanisms (for example inter-species competition, prey availability, climate variation, biotoxin exposure) to shaping observed dynamics. The effects of four categories of direct or indirect covariates were explored:

- Prey covariates: herring (*Clupea harengus*), Atlantic cod (*Gadus morhua*), sprat (*Sprattus sprattus*) and sandeels (*Ammodytes marinus*).
- Environmental covariates: sea surface temperature and North Atlantic oscillation winter index
- Interaction covariates: counts of grey seals in northern Moray Firth
- Biotoxin data: concentration of saxitoxin (STX) and domoic acid (DA) in mussels

The extent of the time series coverage of the different covariates are summarised in Table 1.

The prey species tested covered the range of different types and nutritional qualities present in the diet of harbour seals (Wilson, 2014). Atlantic cod are considered to be of low nutritional value (Fritz & Hinckley, 2005; Wilson, 2014). Herring, sprat and sandeels are all classified as highly nutritious (Wilson, 2014). Fish data were collected from the International Council for the Exploitation of the Seas (ICES) website (http://www.ices.dk/ accessed July 2015) and the standing stock biomass (SSB) value of the year for the North Sea was used as an indicator of the fish abundance in the Moray Firth area (ICES, 2012). Sandeels are an important part of harbour seal diet, but it was not possible to find reliable data on the sandeel stock in the Moray Firth. However, as several studies have shown a correlation between black-legged kittiwake (*Rissa tridactyla*) breeding success and sandeel availability (Frederiksen et al. 2005, Harris & Wanless, 1997), this parameter was used as a proxy for sandeel abundance.

The sea surface temperature (SST) and the North Atlantic oscillation (NAO) winter index were collected respectively from the National Oceanic and Atmospheric Administration (NOAA) and University Corporation for Atmospheric Research (UCAR), Climate Data Guide websites (http://www.ospo.noaa.gov/Products/ocean/sst/contour/ and https://climatedataguide.ucar.edu/climate-data/ , accessed July 2015).

The count of grey seals hauling out on the study sites started in 2006. Counts varied between surveys, so where counts were available, the average number of grey seals observed in a given year was used in the model. Prior to this date no local data were available. So the ratio (estimated using a linear model estimated from the count data and abundance estimate for the North Sea, from 2006 to 2013) between the count of grey seals in the northern Moray Firth and the estimate of the total population size of grey seals in the North Sea was used and multiplied by the estimate of the North Sea population before 2006 (Thomas, 2012).

The last set of covariates tested focussed on the impact of biotoxins produced during harmful algal blooms (HABs). The STX and DA biotoxins were selected as they are regularly detected in the Scottish waters. The presence of STX in the water column and particularly if ingested in the prey can result in very rapid mortality. DA can have both acute and chronic effects and has caused mass mortalities among other pinnipeds worldwide (Hall & Frame, 2010). DA can also potentially have an impact on the reproductive success of pinnipeds as exposed females can abort their pups (Hall & Frame, 2010).

Each covariate was tested individually by including it in the linear predictor for the fecundity function or the survival function, with an associated multiplier which was estimated during model fitting. If this coefficient was significantly different to zero then this is evidence that the covariate has an effect on survival or fecundity. In addition, for some models, a prior sensitivity analysis was conducted.

Objective **3**

Two different scenarios of 'reduced' data were investigated (extracted from the original data set).

- Only the moult data.
- The number of breeding surveys was decreased to determine if it is possible to identify a minimum frequency of breeding surveys required for reliable results. The different scenarios of breeding survey pattern tested are presented in Table 4.

For the model run with only moult data either the fecundity rate using the value estimated by Cordes (2011) was fixed or an informative prior for the fecundity rate was set.

Results

More than 35 models were tested. Table 2 presents a summary of the models, with the median and credible interval for the covariate coefficients.

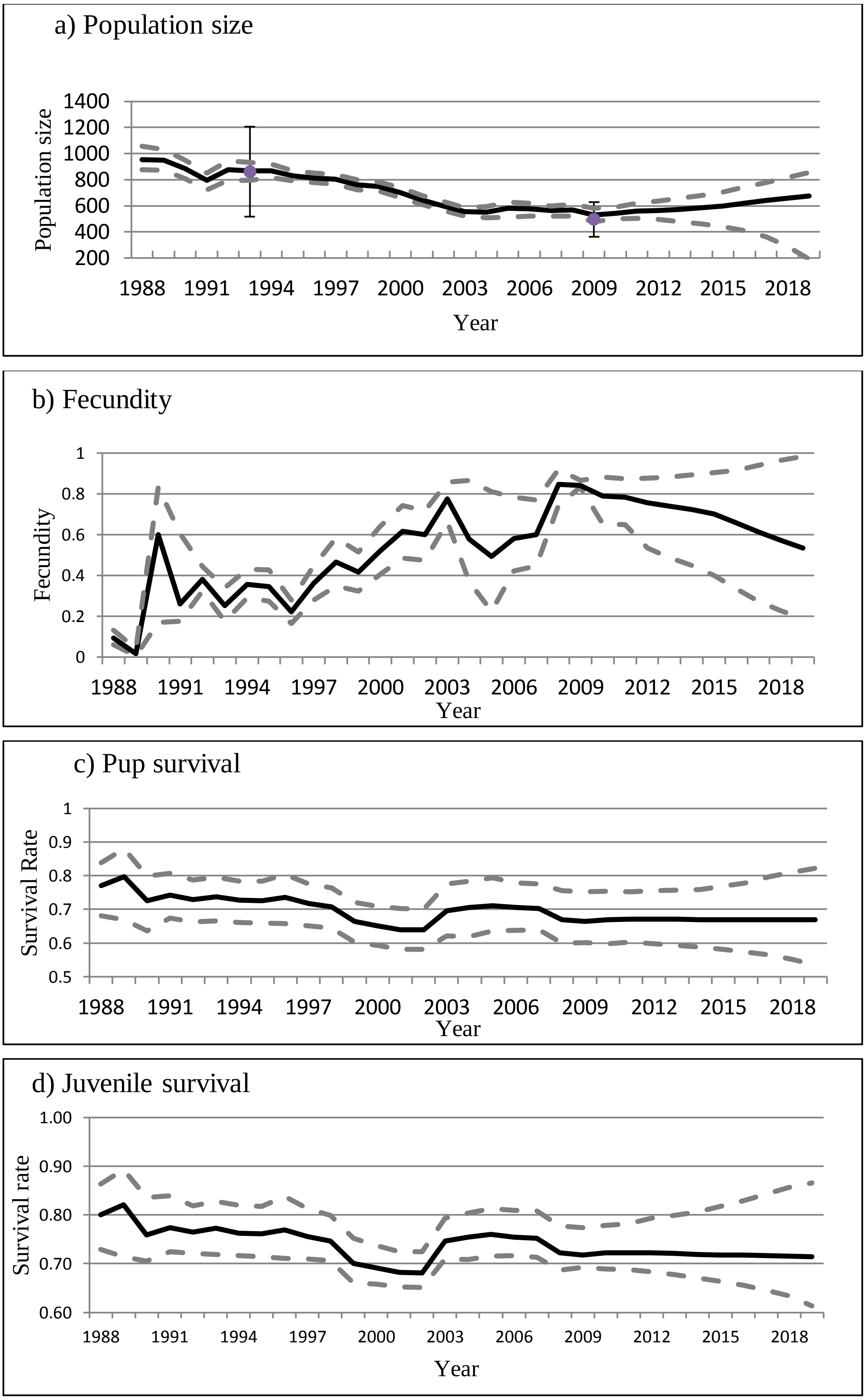
Fitting the baseline model highlighted the effect of shooting on the Moray Firth population. This can be seen in the survival rates of all age classes. There was also considerable variability in estimates of fecundity compared with other demographic rates which were more stable. The near-future trend in the Moray Firth population was projected to be slightly positive on average, though 95% Bayesian CIs spanned both positive and negative trends (Figure 2).

In determining population trends, the model predictions suggest that the most important class in the population are the adult females, a conclusion consistent with earlier studies (Harwood & Prime, 1978). If the adult female survival rate decreases by 5% then the population will decline (Figure 3). By contrast, the other demographic parameters would need to decrease between 20% (juvenile) and more than 50% (adult males) for the population to decline. Adult male mortality has the lowest impact on the population growth rate, even with a drop of 50% in the adult male survival rate, after 10 years the population will still be growing but at a slower rate (1% per annum). In contrast, any additional adult female mortality will have a substantial impact on the total population size. If an extra 12 females in the northern Moray Firth population die every year over a 10 year period, the population will decline. More generally if there is a factor generating a decrease of 3% in the survival rate for all age classes, the population will decline in 10 years.

Table 2. Models tested to identify correlations between covariates and demographic trends. The column headed 'Hypothesis' explains briefly how the covariates may influence fecundity or survival. The 'In model' column describes which demographic rate is influenced by the covariate (i.e. has the covariate as a term in the linear predictor). The "Covariate prior value" shows the prior values used when sensitivity analyses were conducted. The posterior values are the Bayesian credible intervals and medians for the covariate parameters. Highlighted in grey are the parameters whose posterior differs significantly from zero.

	Hypothesis	Covariates	In model	Covariate prior value	Posterior values		
Prey	Lack of food impacts fecundity and/or pregnancy success and pup survival	Herring	Fecundity	*cov=0.03*	-0.02669	-0.009804	0.01897
			Pup surv	*cov=0.03*	-0.08213	-0.002418	0.08156
		Cod	Fecundity	*cov=0.03*	-0.02073	0.007702	0.02644
			Pup surv	*cov=0.07*	-0.02386	0.001169	0.02495
			Pup surv	*cov=0.03*	-0.02386	0.001169	0.02495
		Sprat	Fecundity	*cov=0.03*	-0.02192	0.005099	0.0258
			Fecundity	*cov=0.07*	-0.07239	0.02342	0.08761
			Pup surv	*cov=0.03*			
		Sandeels	Fecundity	*cov=0.03*	-0.02171	0.006605	0.02619
			Fecundity	*cov=0.1*	0.000669	0.07019	0.09556
			Pup surv	*cov=0.1*	-0.07675	0.01131	0.08473
			Pup surv	*cov=0.03*	-0.02385	0.001368	0.02478
			Fecundity + Pup surv	*none*	-0.01916	0.009601	0.02684
Predator	Inter-specific competition for food – results in increased pup mortality	Grey Seals	Pup surv from 1988	*cov=0.05*	-0.04348	-0.007327	-0.0023
			Pup surv from 1988	*cov=0.03*	-0.00503	-0.002569	-0.00112
Environment	SST and NAO may have a direct impact on plankton production and on the food web	SST	Fecundity	*Mch-May*	-0.04432	-0.0152	0.03144
			Fecundity		-0.09153	-0.04076	0.05875
			Pup surv	*Mch-Sept*	-0.04008	0.000427	0.0404
		NAO (winter index)	Fecundity	*cov=0.03*	-0.03386	0.01198	0.04355
			Pup surv	*cov=0.05*	-0.04044	2.72E-05	0.04064
			Pup surv	cov=0.1	-0.08308	-0.004108	0.0794
			Breeding	cov=0.1	-0.07997	0.02144	0.08965
			Non pup surv	cov=0.1	-0.08452	-0.006132	0.08088
	Causes neurological effects - disorientated animals and impacts on fecundity	DA	Fecundity	cov=0.1	-0.08782	-0.00624	0.08357
			Non-pup surv	cov=0.1	-0.08781	-0.02022	0.07694
			Pup surv	cov=0.1	-0.08308	-0.004108	0.0794

Figure 2. Demographic trends (a) population size (excluding pups) (b) fecundity rate (c) pup survival (d) juvenile survival, (e) adult female survival estimated by the baseline model with 95% confidence intervals (grey dashed lines). From 2013 onwards, the model predicts forward in time. In (a) the two points are the independent estimates of population size excluding pups. The level of shooting in the area was high between 1999 and 2003.



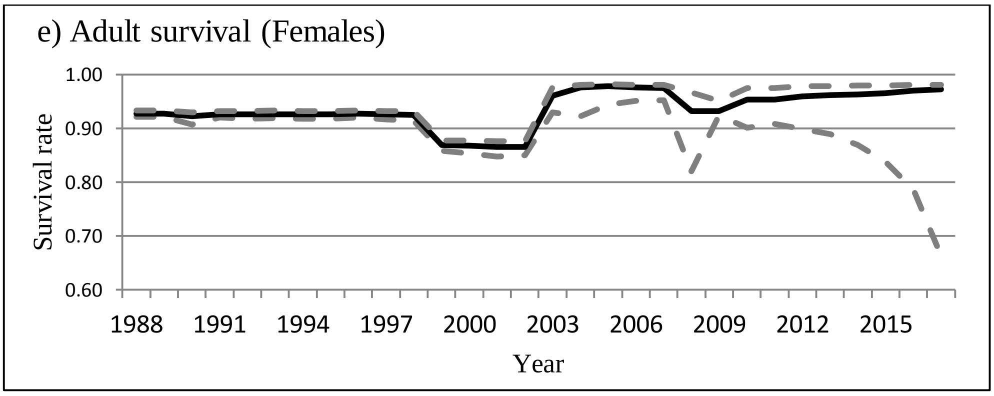
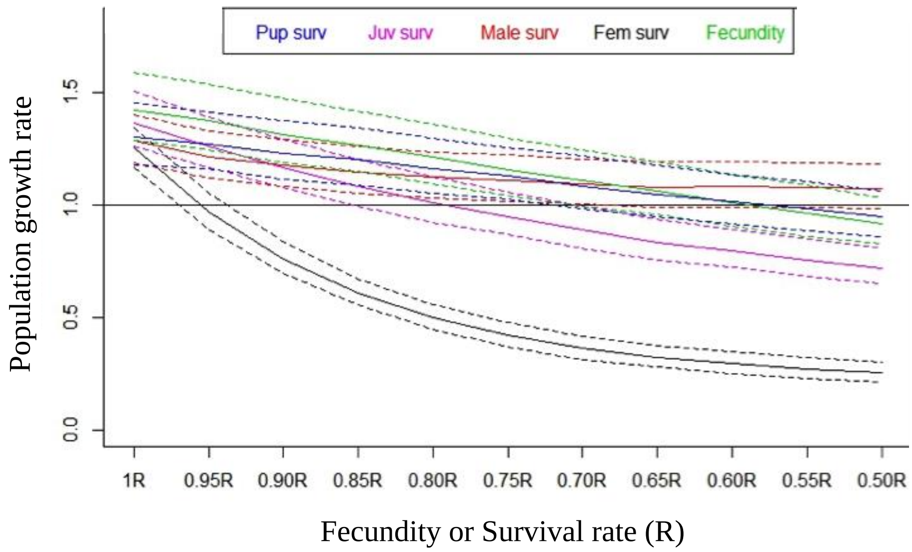


Figure 3. Population growth rate (over 10 years) under different scenarios of fecundity and survival. Starting with estimates from the Baseline model, vital rates are reduced in turn to determine the effect on forecast population growth over 10 years. (Pup surv: pup survival is reduced, blue curve; Juv surv: juvenile survival is reduced magenta curve, Male surv: adult male survival is reduced, brown curve; Fem surv: adult female survival is reduced, black curve; Fecundity (pups born per female) is reduced, green curve).



More than 35 models were tested. Table 3 presents a summary of the models, with the median and credible interval for the covariate coefficients. In almost all the models two covariates were significantly different from zero, indicating a correlation between (a) grey seal abundance and harbour seal pup survival and (b) sandeel abundance and harbour seal fecundity.

Table 3. Summary of results when the model was fitted with reduced datasets.

Data used	Breeding survey pattern	Summary of results
Moult data only	Fecundity fixed at a constant value	Over-estimation of population size; pup and juvenile survival very variable, falling outside credible intervals of the baseline model
	Fecundity variable	
Full dataset	Remove 1/5 breeding surveys	Trends within credible interval of baseline model
	Retain one breeding survey per year	Trends close to or within credible interval of baseline model, minor over-estimation

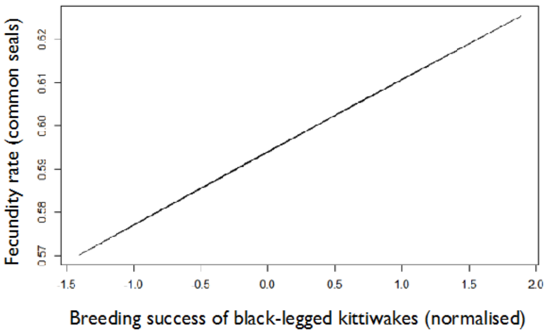
Although the sandeel covariate was positively correlated with the fecundity rate (Figure 4(a)), the realised impact on the breeding rate was small. An increase from -1.5 to 2 in the sandeel index generated an increase of 5% in the harbour seal fecundity rate. All the demographic trends showed only a very small difference compared to the baseline trends.

When grey seal count was included in the model, the trend in the pup survival rate was very different to the one in the baseline model with a decreasing pup survival rate linked to an increase in the grey seal population (Figure 4(b)). The abundance estimate of the non-pups is then closer to the 2009 independent estimate than with the baseline model. The most important difference between the two models can be observed at the end of the time series. Indeed from 2008 the abundance trends start to differ between the two models, with an increase in the population with the baseline model and a decrease in the population with the model including the grey seal data. The prediction after 2012 suggested a succession of consecutive years with fewer pups, generating lower juveniles and adults combined. If there are more than 800 grey seals hauling out for 10 years, the harbour seal population declines. Since 2006, grey seal counts were approximately 800 or more in 60% of the years. The demographic trends when the grey seal covariate was added to the baseline model for pup survival are shown in Figure 5.

“Reducing” the data set had substantial implications for the model-fitting. With moult data only and fecundity fixed or tightly constrained, the model over-estimated harbour seal abundance. In particular pup abundance was greatly overestimated (non-pup population size is just over the credible interval of the baseline model (Figure 6a)). However, the abundance trends were similar to those estimated from the full dataset (Figure 6, Figure 2). The model did not contain any pupping data to influence the fecundity prior distributions, and consequently the fecundity rate was based on the prior, which was relatively high. Including even one breeding survey per year greatly improved the results. This time, non-pups were slightly over-estimated, but the fecundity and the pup survival trends were within or very close to the credible interval of the baseline model.

Figure 4 (a) Correlation between the fecundity rate and the normalised breeding success rates for black-legged kittiwakes (b) Correlation between the harbour seal pup survival rate and the number of grey seals on the haul-out sites

(a)



(b)

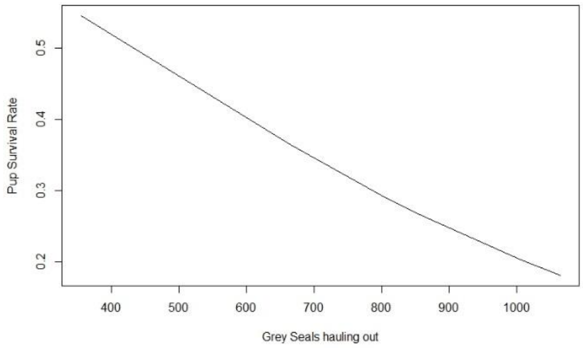
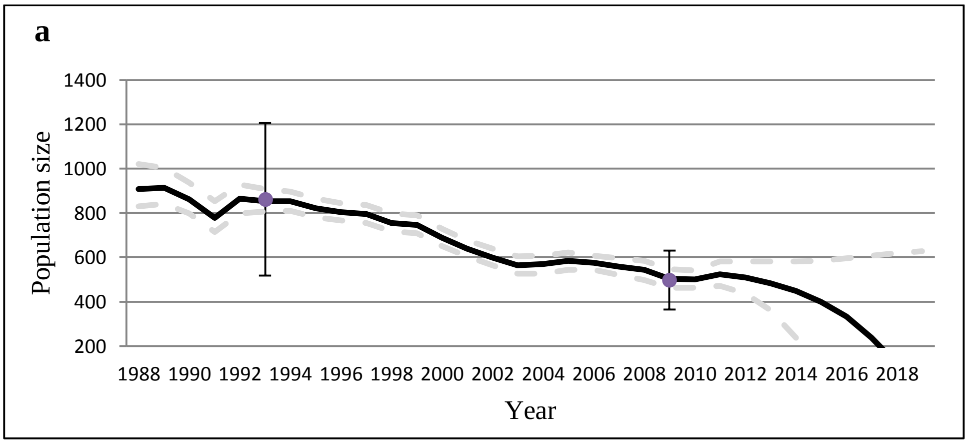


Figure 5. (a) Population size and (b) pup survival rate estimated by the model fitted to counts including the grey seal covariate. In the pup survival plot, the black dotted line is the average number of grey seals counted at local haul-out sites. Grey dashed lines represent 95% Bayesian credible intervals.



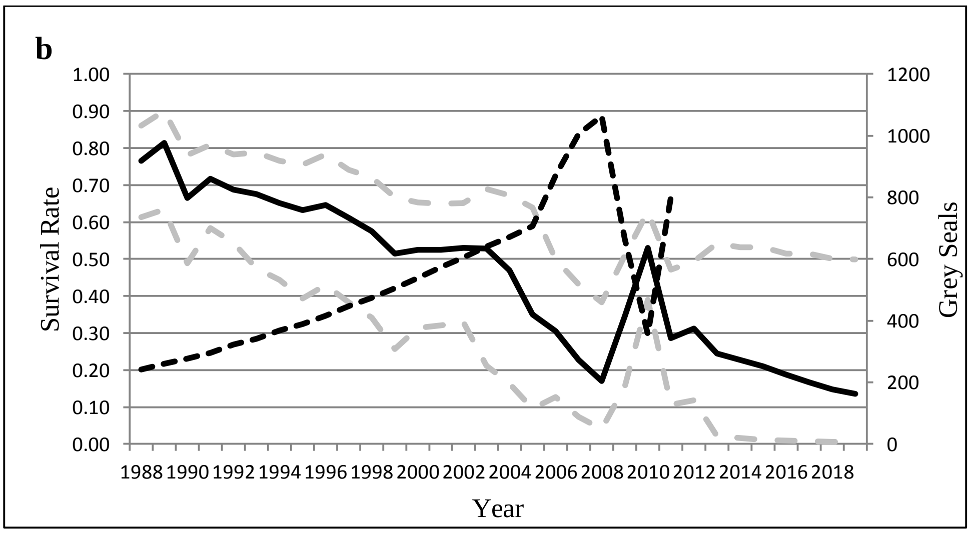
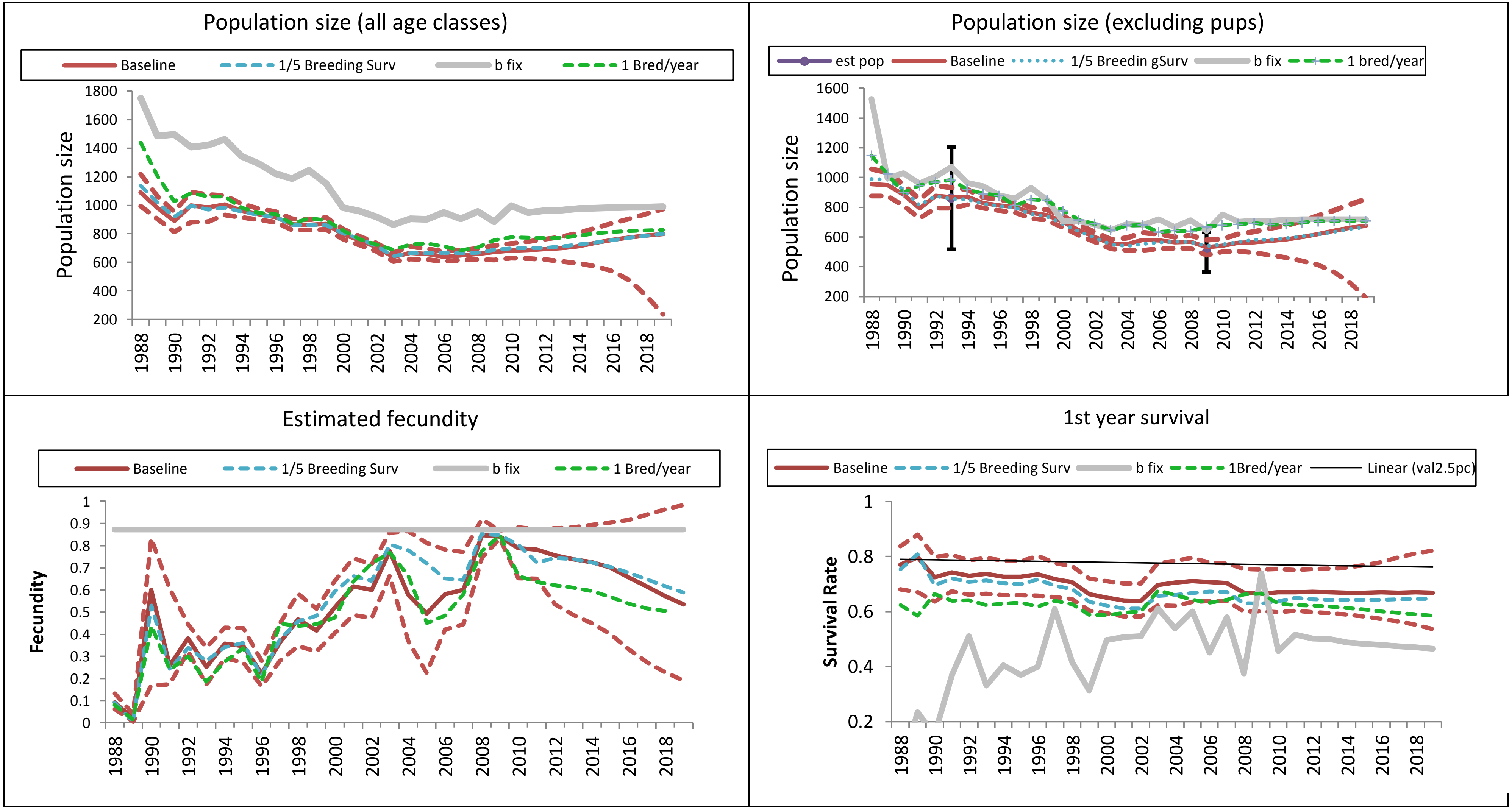


Figure 6. Effect of reducing the amount of data used to fit the model, on model predictions. The results of the baseline model (in dark red) are compared with results using 1/5 of the available breeding survey data (cyan), or with using 1 breeding survey only each year (green), or with results estimated in the absence of breeding survey data (grey).



Discussion

This work demonstrates the use of the state-space modelling approach to predict changes in the Moray Firth harbour seal population under the influence of external drivers such as additional mortality. A complex, integrated analysis of different types of count data, information in the form of prior distributions and 'independent estimates' of population size was possible for this well-studied system.

The predictions of the effects of different vital rates helped to identify which age classes are critical for the survival of the population and affirmed that adult females play a crucial role. For example if 5% of the current adult females in the northern Moray Firth (an additional 12) are killed every year, then this population is likely to decline.

The covariate study indicated that sandeels and local grey seal abundance each have an impact on the population. Wilson (2014) confirmed that the diet of harbour seals in the Moray Firth remains predominantly sandeels so they are a very important (and high-energy) food source for this population. Data on the abundance and distribution of sandeels in the Moray Firth would greatly help in understanding the relationship between this prey species and seals.

Some of the grey seal abundance data used were not from direct counts (from 1988 to 2006 an approximation was used). Consequently for most of the time series the increase in grey seals was a smooth time trend, which is unrealistic, given that the data after 2006 suggests the number of grey seals can vary substantially between years. The strong correlation observed should perhaps lead to further investigation especially in the light of recent evidence that adult grey seals can cause additional direct mortality in grey seal pups, suggesting the possibility of grey seal induced mortality also for harbour seals. An investigation of possible effects of grey seals on adult female harbour seal mortality may also be warranted.

The grey seal and sandeel indices are the only ones for which there were local values for a long period of time, and these are both 'indirect' quantities (for part of the time series, in the case of the grey seals). The identification of these covariates through our modelling work therefore should perhaps be seen mainly as motivation for future data collection and investigation relating to these covariates. Better geographical resolution to enable estimation of the local abundances of different fish prey would also be useful, because it is very possible that prey other than sandeels contribute substantially to the energetic budget and nutritional status of harbour seals.

No correlation was found between harbour seal demographic trends and biotoxin levels in mussels on the east coast of Scotland. Biotoxins in mussels are regularly monitored as part of the Shellfish Monitoring Regulations and therefore provided surrogate covariate data for the time series study. However, biotoxin concentrations in mussels may not be representative, due to the quantity and variation in contaminated prey consumed by harbour seals. It is possible that investigating the role of biotoxins using data from seals directly as seen from recent studies (Hall and Frame, 2010) may allow for the detection of effects at population level.

Nonetheless, this work illustrates how the modelling approach has the potential not only to estimate demographic rates, but also to understand what processes may be driving the demographic trends. Such understanding will assist in the prediction of future population trajectories under different environmental scenarios.

A comprehensive data set exists for the Moray Firth area. However, for the other harbour seal haul-out sites around Scotland, most surveys conducted have been during the moult only. The final part of the study highlights the importance of having a sufficiently large dataset at the appropriate spatial scale to detect and understand population demographic changes. With moult survey data only it is possible to estimate population trends and potentially infer changes in survival rates, but, because of the difficulty in estimating fecundity rates, it may not be possible to explain why the population is declining or increasing, so that it becomes more difficult to predict the consequences of environmental variability and regime shifts. With at least one breeding survey per year, however, the estimates of the demographic rate will be close to reality.

In conclusion: if the objective is to understand what parameters drive harbour seal vital rates (fecundity and survival) and to predict demographic trends, it is very important to collect:

- Annual harbour seal counts, including at least one annual pup count.
- Matching time series of local covariate data, especially continuing monitoring of the local abundance of grey seals.

Further mark-recapture data for harbour seals at the northern Moray Firth and elsewhere, to add information to the model in the form of priors on fecundity and survival rates, which may vary between sites.

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Provisional Regional PBR values for Scottish seals in 2016

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Abstract

This document estimates PBR values for the grey and harbour seal "populations" that haul out in each of the ten Seal Management Areas in Scotland. Sets of possible values are tabulated for each area using the equation in Wade (1998) with different values of that equation's recovery factor. A value is suggested for this parameter in each population, the resulting PBR is highlighted, and a rationale is provided for each suggestion. The PBR values are calculated using the latest confirmed counts in each management area.

Changes since last year: The large increase in harbour seal counts in the West Scotland region has resulted in increased PBRs for that region. The Moray Firth count decreased slightly reducing the PBR. A similar decrease in grey seal numbers in the Moray Firth has reduced the PBR by approximately 50% while the West Scotland numbers and PBR have increased.

Introduction

Potential Biological Removal is a widely used way of calculating whether current levels of anthropogenic mortality are consistent with reaching or exceeding a specific target population, chosen to be the Optimum Sustainable Population. It is explicitly given, in an amendment to the US Marine Mammal Protection Act, as the method to be used for assessing anthropogenic impacts in the waters around that country. The method has been supported by simulations demonstrating its performance under certain assumptions (Wade 1998). The formulation of the equation allows for small anthropogenic takes from any population, however much it is depleted or fast it is declining.

Scottish Government uses PBR to estimate permissible anthropogenic takes for each of the ten seal management regions and uses this information to assess licence applications for seal control and for other licensable marine activities.

Materials and Methods

The PBR calculation:

PBR = Nmin.(Rmax/2).FR

where:

> PBR is a number of animals considered safely removable from the population.
>
> Nmin is a minimum population estimate (usually the 20th percentile of a distribution)
>
> Rmax is the population growth rate at low densities (by default set 0.12 for pinnipeds), this is halved to give an estimate of the growth rate at higher populations. This estimate should be conservative for most populations at their OSP.

FR is a recovery factor, usually in the range 0.1 to 1. Low recovery factors give some protection from stochastic effects and overestimation of the other parameters. They also increase the expected equilibrium population size under the PBR.

The approach and calculation is discussed in detail in Wade (1998).

Data used in these calculations

Nmin values used in these calculations are from the most recent summer surveys of each area, for both species:

- Harbour seals: The surveys took place during the harbour seal moult, when the majority of this species will be hauled out, so the counts are used directly as values for Nmin. (An alternative approach, closer to that suggested by Wade (1998), would be to rescale these counts into abundance estimates and take the 20th centile of the resulting distributions. Results of a recent telemetry study in Orkney (Lonergan et al., 2012) suggest that would increase the PBRs by between 8%, if the populations are predominantly female, and 37%, if most of the animals are male.)

- Grey seals: Analysis of telemetry data from 107 grey seals tagged by SMRU between 1998 and 2007 shows that around 31% were hauled out during the survey windows (Lonergan et al. 2011a). The 20th centile of the distribution of multipliers from counts to abundances implied by that data is 2.56.

Rmax is set at 0.12, the default value for pinnipeds, since very little information relevant to this parameter is available for Scottish seals. A lower value could be argued for, on the basis that the fastest recorded growth rate for the East Anglian harbour seal population has been below 10% (Lonergan et al. 2007), though that in the Wadden Sea has been consistently growing at slightly over 12% p.a. (Reijnders et al. 2010). Regional pup production estimates for the UK grey seal population have also had maximum growth rates in the range 5-10% p.a. (Lonergan et al. 2011b). However the large grey seal population at Sable Island in Canada has grown at nearly 13% p.a. (Bowen et al. 2003).

$\mathbf{F_R}$ needs to be chosen from the range [0.1, 1]. Estimated PBR values for the entire range of F_R values are presented. A recommended F_R value is indicated for each species in each region, together with a justification for the recommended value.

Areas used in the calculations

Figure 1 and Table 1 shows the boundaries of the Seal Management Areas.

Table 1. Boundaries of the Seal Management Areas in Scotland.

Seal	Management Area	Area covered
1	South-West Scotland	English border to Mull of Kintyre
2	West Scotland	Mull of Kintyre to Cape Wrath
3	Western Isles	Western Isles incl. St Kilda, Flannan Isles, North Rona
4	North Coast & Orkney	North Mainland coast & Orkney
5	Shetland	Shetland incl. Foula & Fair Isle
6	Moray Firth	Duncansby Head to Fraserburgh
7	East Coast	Fraserburgh to English border

Particularly for grey seals, there will probably be substantial movement of animals between these areas. The division is a pragmatic compromise that attempts to balance: current biological knowledge; distances between major haul-outs; environmental conditions; the spatial structure of existing data; practical constraints on future data collection; and management requirements.

Results

PBR values for grey and harbour seals for each Seal Management Area. Recommended F_R values are highlighted in grey cells.

Table 2. Potential Biological Removal (PBR) values for harbour seals in Scotland by Seal Management Unit for the year 2016

				PBRs based on recovery factors (F_R) ranging from 0.1 to 1.0										Selected F_R	PBR
Seal Management Unit	**Count**	**Years Surveyed**	**N_{min}**	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1		
1 Southwest Scotland	834	2007	834	5	10	15	20	25	30	**35**	40	45	50	0.7	35
2 West Scotland	13878	2007-2014	13878	83	166	249	333	416	499	**582**	666	749	832	0.7	582
3 Western Isles	2739	2011	2739	16	32	49	65	**82**	98	115	131	147	164	0.5	82
4 North Coast & Orkney	1938	2013	1938	**11**	23	34	46	58	69	81	93	104	116	0.1	11
5 Shetland	3039	2009	3039	**18**	36	54	72	91	109	127	145	164	182	0.1	18
6 Moray Firth	733	2014	733	**4**	8	13	17	21	26	30	35	39	43	0.1	4
7 East Scotland	194	2007-2013	194	**1**	2	3	4	5	6	8	9	10	11	0.1	1
SCOTLAND TOTAL	**23355**		**23355**	138	277	417	557	698	837	978	1119	1258	1398		**733**

Table 3. Potential Biological Removal (PBR) values for grey seals in Scotland by Seal Management Unit for the year 2016

				PBRs based on recovery factors (F_R) ranging from 0.1 to 1.0										**Selected F_R**	**PBR**
Seal Management Unit	**Count**	**Years Surveyed**	**N_{min}**	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1		
1 Southwest Scotland	374	2007	957	5	11	17	22	28	34	40	45	51	**57**	1	57
2 West Scotland	4095	2007-2014	10483	62	125	188	251	314	377	440	503	566	**628**	1	628
3 Western Isles	3743	2011	9582	57	114	172	229	287	344	402	459	517	**574**	1	574
4 North Coast & Orkney	8035	2013	20569	123	246	370	493	617	740	863	987	1110	**1234**	1	1234
5 Shetland	1536	2009	3932	23	47	70	94	117	141	165	188	212	**235**	1	235
6 Moray Firth	532	2014	1361	8	16	24	32	40	48	57	65	73	**81**	1	81
7 East Scotland	2134	2007-2013	5463	32	65	98	131	163	196	229	262	295	**327**	1	327
SCOTLAND TOTAL	20449		52347	310	624	939	1252	1566	1880	2196	2509	2824	3136		**3136**

Rationale for the suggested recovery factors

The original PBR methodology leaves the setting of the recovery factor as a subjective choice for managers. Factors such as the amount of information available about the population (and in particular its maximum annual growth rate), recent trends in local abundance, and the connections to neighbouring populations are relevant to setting this. The main factors affecting the value suggested for each species in each area are given below:

Harbour seals

1) Shetland, Orkney + North Coast and Eastern Scotland (F_R= 0.1)

F_R set to minimum because populations are experiencing prolonged declines.

2) Outer Hebrides (F_R = 0.5)

Population was undergoing a protracted but gradual decline but the most recent count was close to the pre-decline numbers. The population is only partly closed being close to the relatively much larger population in the Western Scotland region, and the R_{max} parameter is derived from other seal populations.

4) Western Scotland (F_R = 0.7)

The population is largely closed, likely to have limited interchange with much smaller adjacent populations. The population is apparently stable and the intrinsic population growth rate is taken from other similar populations.

4) South West Scotland (F_R = 0.7)

The population is apparently stable, is closed to the south and the adjacent population to the north is apparently stable. The intrinsic population growth rate is taken from other similar populations.

5) Moray Firth (F_R= 0.1)

Counts for the Moray Firth showed large inter annual fluctuations after a period of gradual decline from 2000. However, the counts in the last 3 years have shown a substantial decline and the 2014 moult count was the lowest since recent surveys began in the 1980s. The neighbouring Orkney and Tay populations are continuing to undergo unexplained rapid and catastrophic declines in abundance. Data available from electronic telemetry tags suggest there is movement between these three areas. The PBR was set at 17 for 2014, permits for 10 harbour seals were granted and 6 were shot. We suggest that based on the recent decline in counts the F_R should be set to a value of 0.1.

Grey seals

All regions (F_R = 1.0)

There has been sustained growth in the numbers of pups born in all areas over the last 30 years, with some now appearing to be at or close to their carrying capacities (Lonergan et al. 2011b). Available telemetry data and the differences in the regional patterns of pup production and summer haulout counts (Lonergan et al. 2011a) also suggest substantial long-distance movements of individuals.

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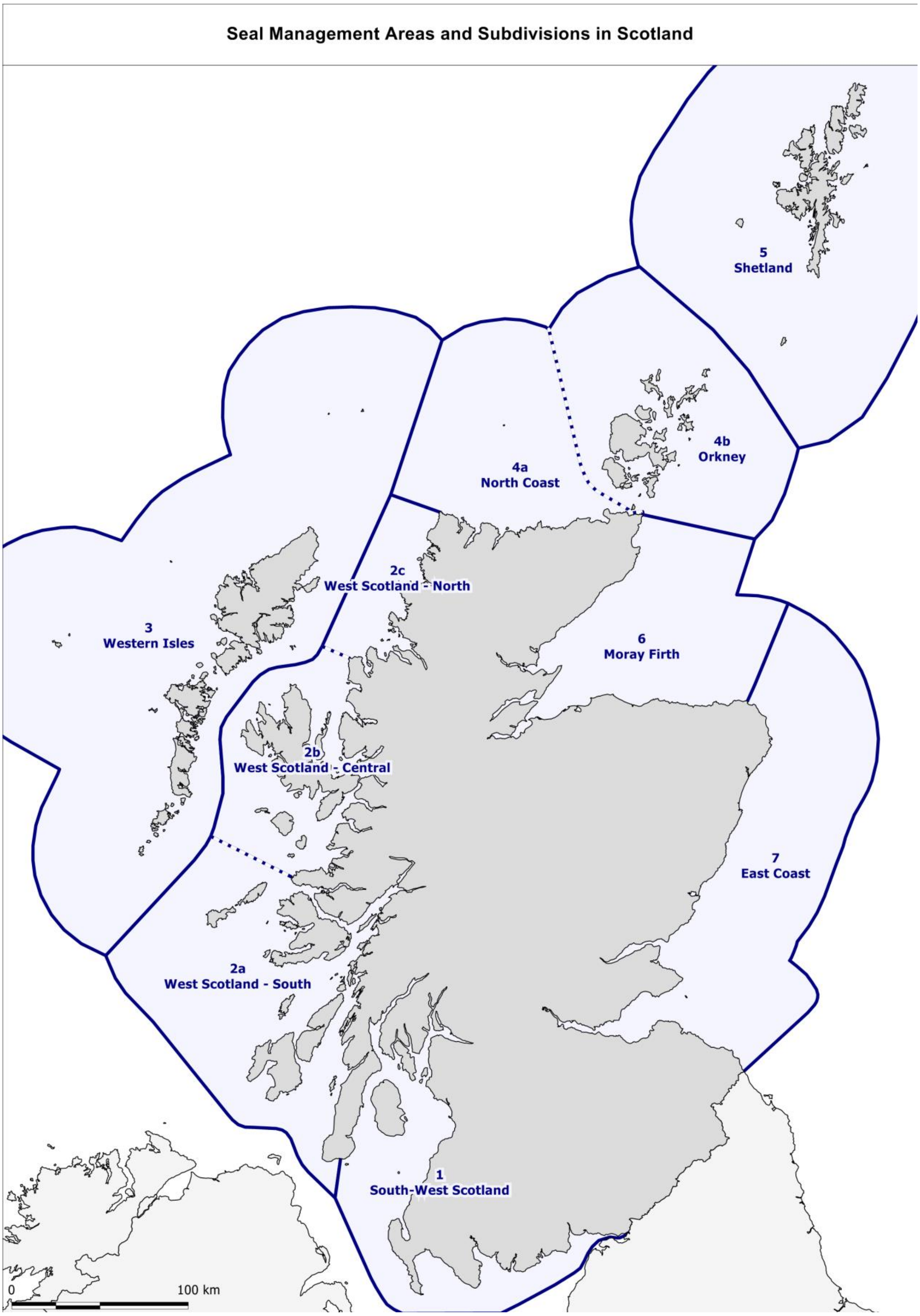
Figure 1.Seal management areas in Scotland.



Report on UK contribution to Marine Strategy Framework Directive seal indicators

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Abstract

The latest available data from the UK were used to perform a preliminary assessment of MSFD indicators M-3 and M-5 describing changes in grey seal and harbour seal population abundance and distribution. Simple models were fitted to count data and 95% confidence intervals of the specified metrics were calculated from bootstrap resamples of the data to provide estimates of the uncertainty surrounding each metric. In some cases, wide confidence intervals that include target values indicate that confidence in the assessment is low. Targets that use both rolling and stationary baselines are presented and give added information about (nonlinear) population trends. It was necessary to arbitrarily subdivide UK Assessment Units into smaller subareas to calculate distribution metrics for harbour seals. The distribution metrics showed no catastrophic contraction or shift in distribution has occurred for either grey or harbour seals in any Assessment Unit. These simple metrics – with added information about uncertainty and number of surveys – should prove applicable to other European datasets, as well as being understandable and useful to policy-makers.

Introduction

Under the Marine Strategy Framework Directive (MSFD), Member States are responsible for coordinating strategies to protect and restore the marine environment to 'Good Environmental Status'. To achieve this, a suite of indicators of marine environmental health has been adopted and will be monitored across European Member States.

Quantitative metrics of the state of grey and harbour seal populations are to be included in the MSFD assessment of environmental status in the North Sea and Celtic Sea under Descriptor 1: *Biological diversity is maintained*. The relevant indicators (and corresponding MSFD criteria and targets) are (Defra 2015):

- M-3: Abundance and distribution each of harbour and grey seals (1.1 Species distribution, 1.1.2 Distributional pattern within range; 1.2 Population size, 1.2.1 Population abundance);
    - "At the scale of the MSFD sub-regions the *distribution* of seals is not contracting as result of human activities: in all of the indicators monitored there is no statistically significant contraction in the distribution of marine mammals caused by human activities"
    - "At the scale of the MSFD sub-regions *abundance* of seals is not decreasing as a result of human activity: in all of the indicators monitored, there should be no statistically significant decrease in abundance of marine mammals caused by human activities"
- M-5: Grey seal pup production (1.3 Population condition, 1.3.1 Population demographic characteristics).

- "At the scale of the MSFD sub-regions seal populations are in good condition: there is no statistically significant decline in seal pup production caused by human activities"

At the North-East Atlantic regional level, progress towards defining and achieving GES for these indicators is coordinated by the Commission for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) across Contracting Parties (CPs), with technical advice from the International Council for the Exploration of the Sea (ICES). The UK acts as lead developer for the seal indicators and the Joint Nature Conservation Committee (JNCC) coordinates this work.

To facilitate coordination across CPs, specification of the metrics and associated targets and baselines for these indicators was addressed at a dedicated workshop held at the University of St Andrews in March, 2015. Representatives attended the workshop from the UK, France, Germany, Netherlands, and Denmark with additional remote input from Ireland and Norway. Attendees were individuals responsible for, or involved with, monitoring seal populations in their country and thus were expected to be familiar with the survey techniques and data format. The attendees discussed and agreed upon a preliminary format for the assessment of GES, and committed to providing the necessary data. Once compiled, the database will contain abundance and distribution data for grey seals and harbour seals across the Northeast Atlantic.

Targets and baselines

M-3 Abundance: For harbour seals, the number of seals counted hauled out during the moulting period will be assessed in each of 19 Assessment Units (Figure 1) where sufficient data exists. For grey seals, a total European population size will be estimated using the integrated Bayesian state-space model developed at SMRU and CREEM. For both species, a two-target approach was proposed, similar to those stipulated by reporting of 'Favourable Conservation Status' under Article 17 of the Habitats Directive. The targets are: **no decline of > 1% per year within the 6-year period** (rolling baseline), and **no decline of > 25% since the fixed baseline** at the start of the Habitats Directive in 1992 (or closest value).

M-3 Distribution: For harbour seals, presence of animal hauled out will be used as an indicator of occupancy of an area. Presence of breeding females at a site will be used as an indicator of colony occupancy for grey seals and the number of occupied colonies will be used to assess distributional changes. Two distribution metrics have been adopted: percentage change in occupancy and a measure of distributional pattern (shift index, described in detail in the Methods section). Because seal surveys in all CPs are designed primarily to estimate total population size rather than distribution, the distribution metrics will be provided as a 'surveillance indicator'. As a surveillance indicator, no formal targets are set but the metrics provide a quantitative description of distribution to be presented alongside population abundance. Assessment of distribution metric will be made for each of 19 Assessment Units where sufficient data exists.

M-5 Pup production: The targets for pup production will be same as for M-3 abundance, but for each of 19 Assessment Units where sufficient data exists so that local changes in grey seals populations are represented in the Assessment (Figure 1).

Reporting uncertainty

Quantitative targets can be set for trends in seal abundance, but formally assessing whether or not these have been met is more problematic. This is particularly true when data is sparse as can be the case in areas where seal occurrence, and hence survey effort, is low. Delegates to the seal indicator workshop, and past ICES WGMME reports, have expressed concern over the ability to detect '*statistically significant'* declines in abundance as stated in the MSFD legislation, and suggested that retrospective power analyses be conducted in order to determine if a particular target (effect size) is detectable with a minimum power of 80% (percentage chance of concluding that no '*significant*' trend in abundance has occurred when in fact it was). For more details of power analyses related to the detection of trends in seal population abundances, please see the response to SCOS question 3, 'seal population trends'. In this briefing paper, we do not engage in formal statistical hypothesis testing, but instead use 95% confidence intervals generated from available data to provide a measure of spread in the parameters estimated. We highlight Assessment Units where parameter confidence intervals encompass a particular target effect size as areas where the target was not met.

Aims

The present report focuses on seal abundance and distribution data collected by the Sea Mammal Research Unit (SMRU) at the University of St Andrews and funded by the Natural Environment Research Council (NERC), Scottish Natural Heritage (SNH) and Natural England. This programme estimates the abundance and distribution of the major grey and harbour seal populations in the UK and receives additional information about the other minor populations from various non-governmental groups across the country.

A trade-off exists between providing the best possible assessment of seal indicators using all available data and statistical tools, and the feasibility of conducting such an ambitious analysis. As a first step towards an integrated Intermediate Assessment in 2017, simple numerical indices of seal metrics restricted to data derived from regular (terrestrial) monitoring programmes will be compiled and assessed. This report describes a preliminary example of an Intermediate Assessment of UK data related to indicators M-3 and M-5.

Methods

Abundance

- *Target 1*: To estimate the annual population growth rate within a 6-year reporting round a linear trend was fitted to available data in each Assessment Unit for the round 2007-2012. Generalised linear models (GLMs) were fitted to count data with a quasi-Poisson error distribution and log link. Annual growth rate (%) and 95% confidence intervals were estimated for AUs with more than three surveys in the period.

- *Target 2*: To determine the change in abundance of seals since the baseline year, generalised linear models (GLMs; harbour seal data) or generalised additive models (GAMs; grey seal pup production data) were fitted to count data with a quasi-Poisson error distribution and log link using all available survey data in the range 1992-2012. The percentage change in abundance since reference year (Δ_{ref}) was calculated from fitted values. 95% confidence intervals were calculated from 10,000 bootstrap resamples of the data.

$$\Delta_{ref} = \left(\frac{\hat{y}_{recent} - \hat{y}_{reference}}{\hat{y}_{reference}}\right) * 100$$

where $\hat{y}_{recent}$ is the count fitted by the model in the most recent survey year and $\hat{y}_{reference}$ is the count fitted by the model in the reference year.

Distribution

To explore changes in seal distribution from available survey data, it was necessary to further subdivide the Assessment Unit area into subunits. 5km and 10km grid cells were considered for the UK; however, in some other countries (e.g. surrounding the Wadden Sea) the coordinates of individual haulout groups are not necessarily recorded and seal numbers are reported at the level of national management units or subareas.

In the UK, the proposed MSFD Assessment Unit boundaries correspond to the national management unit boundaries so subdivisions of management units were created to tabulate finer-scale resolution presence/absence data (noting that there is a third possibility of 'not surveyed'). The borders of subunits were arbitrarily assigned, but with the intention of aggregating haul-out sites by seaward proximity and the likelihood that seals would travel between sites rather than being based on any pre-existing municipal boundaries. To date, this subdivision has only been carried out for Scotland, where the majority of the seal population is found. Figure 1 shows the subareas within Assessment Units.

Describing the (terrestrial) distribution of seals from surveys that are designed primarily to assess abundance is problematic and any distribution metric based on these data will have inherent limitations arising from three main areas:

- <u>Spatial coverage</u>: Seal abundance surveys necessarily census animals seen hauled out on land and do not address the distribution at sea. To estimate at-sea usage, long-term telemetry data is necessary (e.g. Jones et al. 2011).

- <u>Sampling effort</u>: In Scotland, harbour seal moult surveys cover the entire coastline approximately every 5 years, but outside Scotland the surveys do not necessarily cover potential haul-out sites or breeding colonies in a systematic way. Haul-out and breeding sites are sampled preferentially based on past experience of seal occurrence. This means that the surveys will not necessarily detect changes in distribution; new haul-out or breeding sites are only added to the survey coverage as anecdotal data on seal occurrence accumulate. This could lead to a bias in seal distribution metrics due to *preferential sampling*.

- <u>Temporal coverage</u>: the surveys cover narrow windows during key life-stages such as moulting, breeding and pupping seasons.

These general limitations are applicable to most studies of animal abundance and distribution (Fortin et al. 2005, Thomas 2009). Despite these limitations, survey data may be useful to detect large-scale contractions in population distributions in terms of reduced use or abandonment of haul-out or breeding areas, depending on the resolution with which presence/absence data are reported. Shifts in distribution within the area covered by the surveys can also be described at the spatial resolution provided in the data.

To estimate the percentage change in occupancy of Assessment Unit subareas (harbour seals) or colonies (grey seals) a metric Δ_{occ} was calculated:

$$\Delta_{occ} = \left(\frac{B}{N} - \frac{A}{N}\right) * 100$$

where A is the number of spatial units (e.g. subareas of Assessment Units) occupied by seals during reference period A and B is the number of units occupied in a subsequent period. N is the total number of spatial units within the Assessment Unit. Δ_{occ} was calculated both with period A defined as the previous reporting period (in this case 2001-2006) . Comparison with earlier periods was not made due to differences in survey effort. and as the reference period (the earliest 'complete' reporting round since 1992, which is 1994-2000).

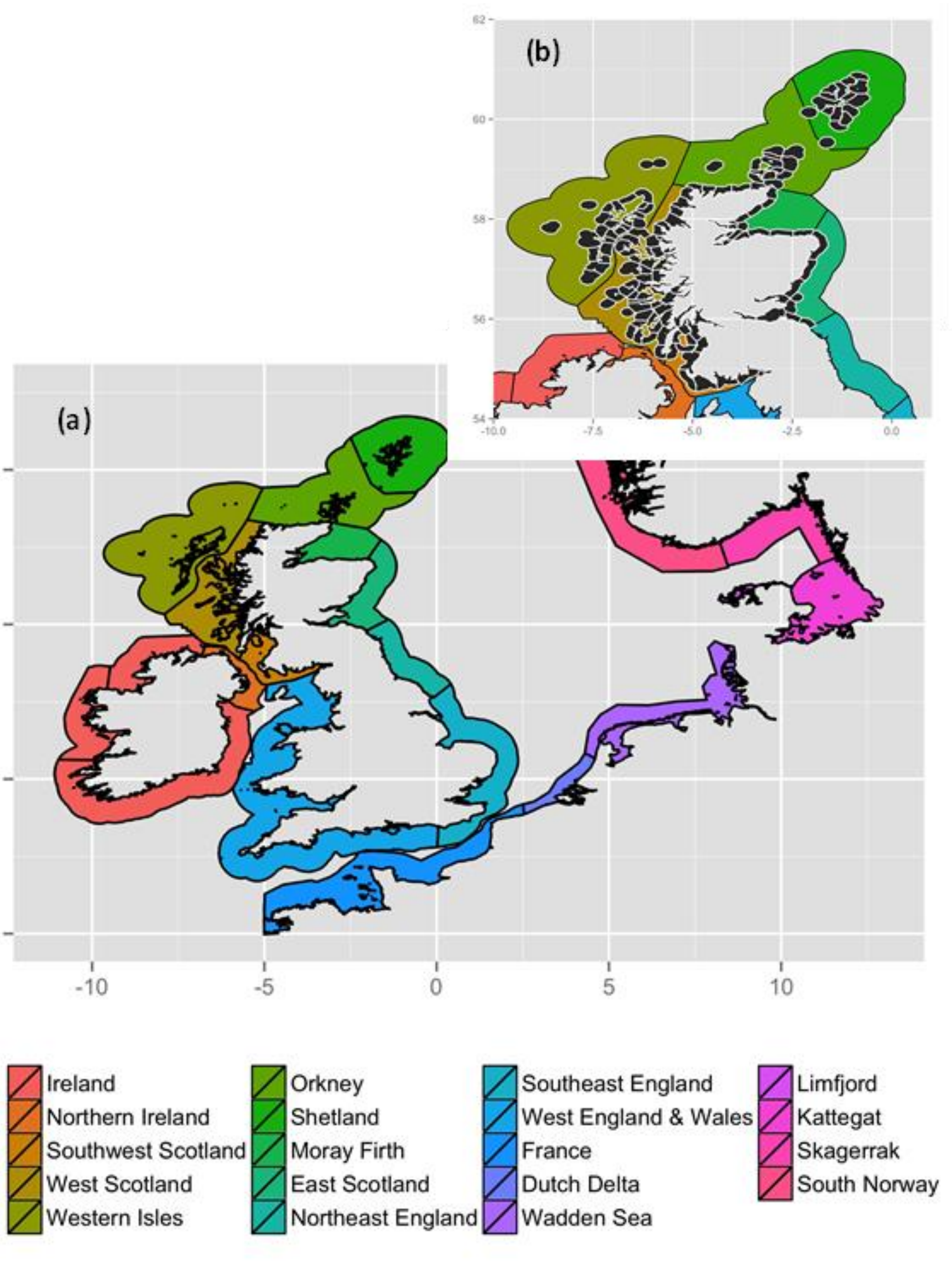
A metric of distributional shift, the shift index, was calculated:

$$Shift = 2 * \left(\frac{B}{N} - \frac{A}{N}\right)\frac{2(A\&B)}{A + B} * 100$$

where A is the number of spatial units (e.g. Assessment Unit subareas, haul-out sites, or colonies) occupied by seals during reference period A and B is the number of units occupied in a subsequent period. A&B is the number of units occupied in both periods.

A shift index of 1 indicates that there has been no change in occupancy and that the same spatial units were occupied in previous period. A shift index of 0 indicates that a complete shift in distribution has occurred (note that this does ***not*** mean the distribution has shrunk, just that it has changed). A 0 value can be caused by, for example, a colony or subarea being surveyed for the first time in that period.

Figure 1. Proposed Marine Strategy Framework Direct (MSFD) Assessment Units (a) and detail of Assessment Unit subdivisions in Scotland (b



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Results

M-5 Pup production

Generalised additive models fitted to grey seal pup production data from each Assessment Unit are presented in Figure 2. Table 1 gives the estimates of change in abundance and distribution detailed in the Methods section.

Figure 2. Grey seal pup production in 9 UK Assessment Units between 1992 and 2012. Generalised additive model fits (dashed line) and 95% confidence intervals (shaded region) are presented.

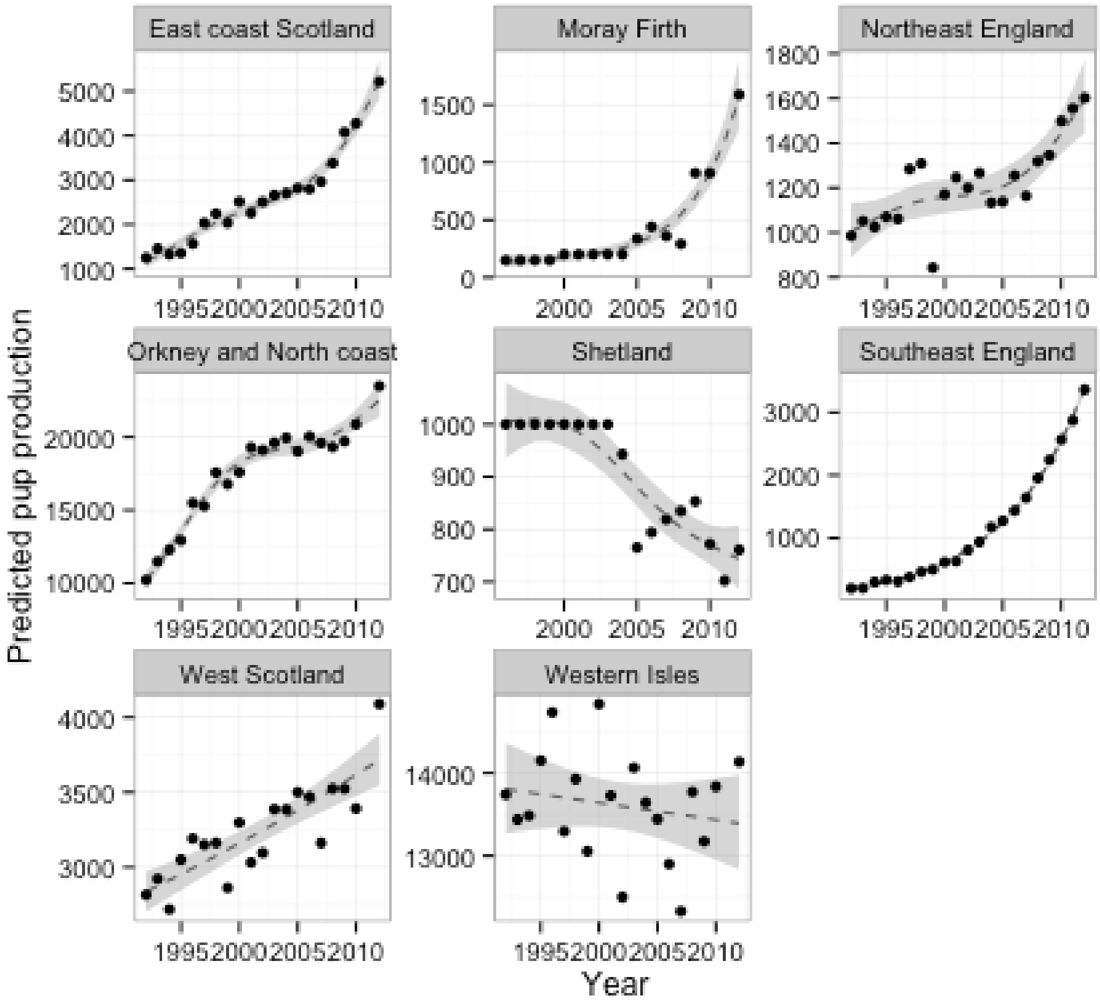

Table 1. Parameters of change in pup production and distribution of colonies estimated. For pup production metrics, the parameter estimate is presented with 95% confidence intervals in bracket. The number of surveys included in the time period specified is also given. Orange cells indicate those Assessment Units where 95% confidence intervals encompass the threshold value set in the target. Blue cells indicate AUs where estimated change confidence intervals were greater than the target value. Cells were symbol coded to indicate if occupancy rate has increased between the two time periods (🡅), decreased (🡇) or stayed the same (🡄🡆). The shift index (ranging from 0 to 1) is given in the final column.

	Abundance		Distribution	
Assessment Unit	Growth 2007:2012 (%)	Δ_{ref} (%)	Δ_{occ} (%) compared 2001:2006	Shift compared 2001:2006
East Scotland	+12 (9,14) n = 5	+347 (250, 465) n = 20	🡄🡆	1
Moray Firth	+36 (19, 55) n=5	+1075 (591, 1599) n = 16	🡄🡆	0.4
North Coast & Orkney	+4 (2, 6) n = 5	+121 (99, 160) n = 20	🡄🡆	1
Northeast England	+6 (5, 8) n=6	+59 (35, 83) n = 21	🡄🡆	0.92
Shetland	-3 (-5, -0.2) n=6	-26 (-36, -16) n = 17	🡄🡆	0.70
Southeast England	+15 (14, 16) n = 6	+1624 (1442, 1878) n = 21	🡄🡆	1
West Scotland	+4 (2, 7) n = 5	+31 (13, 46) n = 20	🡄🡆	0.89
Western Isles	2 (0.3, 4) n = 5	-3.0 (-14, 3.4) n = 20	🡄🡆	0.97

M-3 Grey seals

Indices of changes in the distribution of breeding colonies are presented in Table 1. For IA2017, the population abundance aspect of indicator M-3 will pool grey seal pup production and other count data to produce a total European population size estimate using the CREEM/SMRU state-space model (Thomas 2014). This work is ongoing.

M-3 Harbour seals

Generalised linear models fitted to harbour seal moult count data from each Assessment Unit with more than three surveys are presented in Figure 3. Table 2 gives the estimates of change in abundance and distribution detailed in the Methods section.

Figure 3. Harbour seals counted during UK aerial surveys in August 1996 to 2012. Generalised linear model fits (dashed line) and 95% confidence intervals (shaded region) are presented.

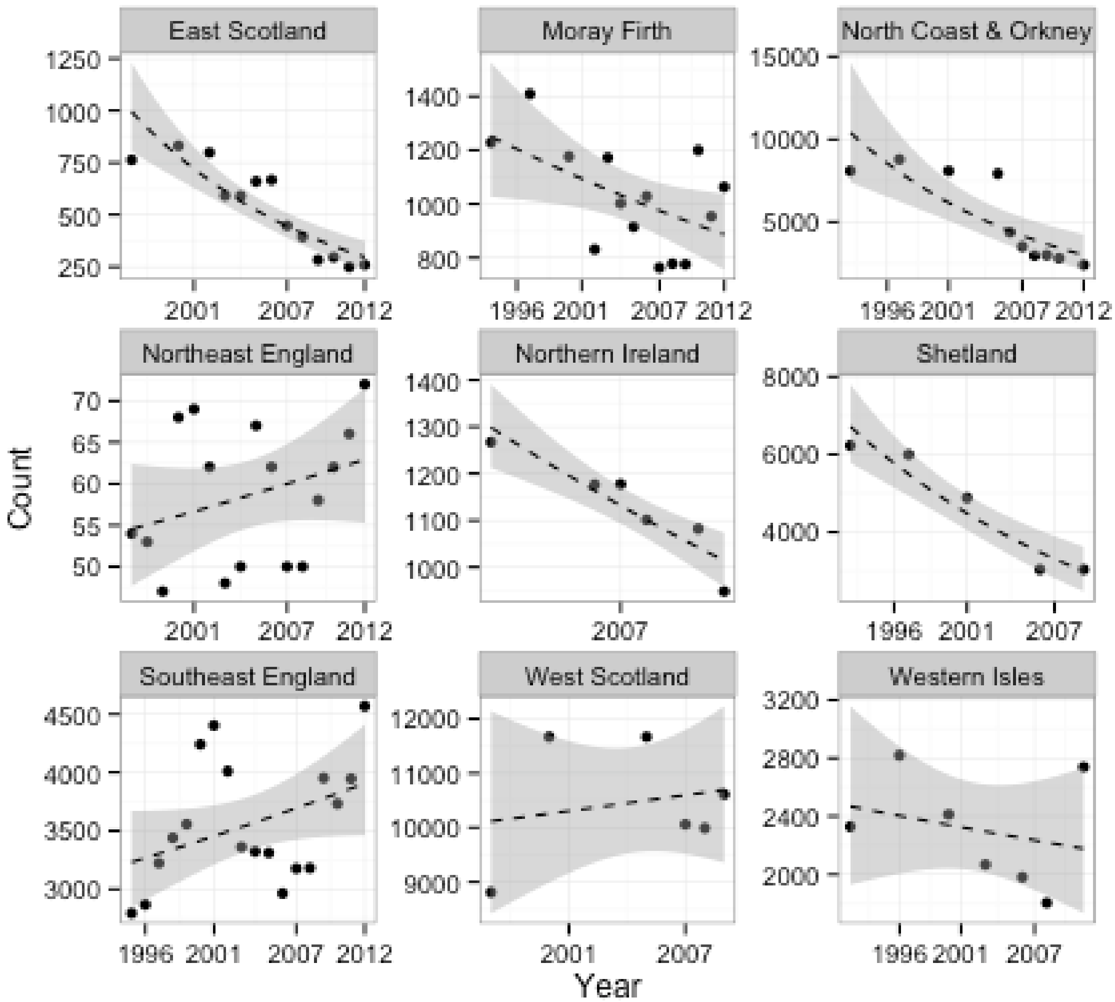

Table 2. Parameters of change in abundance and distribution estimated (abundance) or calculated (distribution) from available data. For abundance metrics, the parameter estimate is presented with 95% confidence intervals in bracket. The number of surveys included in the time period specified is also given. Orange cells indicate those Assessment Units where 95% confidence intervals encompass the threshold value set in the target. Blue cells indicate where estimated change confidence intervals were greater than the target value. Grey indicates sample sizes were too small to complete the assessment. Cells were symbol coded to indicate if occupancy rate has increased between the two time periods (🡅), decreased (🡇) or stayed the same (🡄🡆). The shift index (ranging from 0 to 1) is given in the final column.

	Abundance		Distribution	
Assessment Unit	Growth 2007:2012 (%)	Δ_{ref} (%)	Δ_{occ} (%) compared 2001:2006	Shift compared 2001:2006
East Scotland	-11 (-16, -7.0) n = 6	-70 (-83, -54) n = 13	🡇	0.83
Moray Firth	+7.9 (0.4, 16) n=6	-29 (-59, -3.6) n = 14	🡅	0.92
North Coast & Orkney	-6.6 (-8.8, -4.4) n = 5	-71 (-87, -50) n = 10	🡄🡆	0.93
Northeast England	+8.1 (6.6, 9.6) n=6	+15 (7.2, 39) n = 16		
Northern Ireland	-4.3 (-7.0, -1.6) n = 4	-22 (-33, -8.8) n = 6		
Shetland	n = 1	-55 (-76, -35) n = 5	🡄🡆	0.97
Southeast England	+7.1 (4.0, 10) n = 6	+21 (1.3, 46) n = 18		
Southwest Scotland	n = 1	n = 3		
West Scotland	2.7 (-1.0, 6.7) n = 3	5.7 (-31, 49) n = 6	🡄🡆	0.63
Western Isles	n = 2	-12 (-82, 57) n = 7	🡄🡆	1.0

In general, the results of the target assessments were unsurprising; for grey seals, nearly all populations are experiencing positive growth rates and thus meet the proposed targets for abundance. Harbour seal populations experiencing well-characterised long-term declines 'fail' to meet targets as expected (East Coast, Shetland, Orkney), but three other Assessment Units stand out. The Moray Firth 'passed' abundance target 1 with a rolling baseline but 'failed' to meet abundance target 2 which used a fixed baseline population reference level from 1992. This reflects the nonlinear pattern of growth in this population, which was negative until ~2003 and thereafter appeared to stabilise (Matthiopoulos et al. 2014) and highlights the potential for drawing erroneous conclusions about a population based on comparison with only one type of baseline.

The bootstrapped confidence intervals calculated for change in abundance for the West Coast and Western Isles harbour seal populations were wide, spanning negative and positive growth. This reflects the fact that counts in these areas are fairly stable, but variable.

Discussion

Seals are an important component of marine biodiversity. As top predators they integrate information about the state of the marine ecosystem. Their abundance and distribution can respond to various natural and anthropogenic drivers including disease, interspecific competition, shifts in resources, disturbance, and fisheries interactions. Thus, monitoring changes in their abundance and distribution is an important part of any assessment of 'Good Environmental Status'. However, in many cases detailed characterisation of the pressures affecting the state of the population is lacking due to the inherent difficulty in assessing wild population demographic parameters, and the fact that populations can be responding to multiple drivers. Thus changes in the abundance and distribution of apex predators as general indicators of ecosystem health should be viewed in the context of changes to other biodiversity indicators, as it is often difficult to pinpoint the specific causes of change.

Simple models were fitted to (non-linear) count data from each AU to assess both abundance targets. By bootstrapping 95% confidence intervals of abundance metrics from these models, rather than simply reporting the observed percentage change, we aim to provide a better estimate of the potential variability in MSFD abundance metrics. The width of these intervals should reflect the quality and quantity of the data. Because the number of surveys varies widely between Assessment Units both within the UK, and across the Northeast Atlantic region, the precision of estimated parameters will also vary and may greatly exceed the effect size stated within the target. Within the Marine Strategies Framework, it is possible to set unique targets for each AU individually and this approach was recommended by the ICES WGMME. However, setting appropriate and non-arbitrary conservation targets or reference levels is difficult, and the decision ultimately is a policy one. Here, targets and baselines were set after workshop discussion and on the basis of previously used or recommended metrics.

The distribution 'surveillance' indicators should be useful in some instances for identifying or eliminating potential drivers of population change; for example, in areas where population abundance is declining but distributional pattern is unchanged (e.g. harbour seals in Shetland), it is unlikely that severe habitat loss or disturbance is a significant driver of population change. Interpretation of the distribution metrics needs some care, however, and knowledge of the spatial relevance of the subdivisions used.

Looking ahead: Responses from other countries to the OSPAR formal data request sent at the end of May, 2015 have been received, but there remains a considerable amount of work to format the datasets. Aside from providing the data necessary to complete a preliminary Intermediate Assessment 2017 for OSPAR, however, the formation of a seal database at the European-level is expected to have wide-ranging uses outside of those immediately set by the MSFD.

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Monitoring harbour seal haul-out sites in the Sound of Islay

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Abstract

This paper describes a study of the effects of disturbance of harbour seals at haul-out sites in the Sound of Islay. A series of controlled disturbance trials were carried out to assess the effect of disturbance by increased boat activity on haul-out behaviour. A sample of 10 seals were fitted with GPS/GSM transmitters to monitor their haul-out and movement patterns during normal undisturbed haul-out events and in response to being chased into the water by direct approaches by a boat.

Concurrent monitoring of haul-out sites using remote camera systems recorded behavioural responses of non-tagged seals to disturbance trials as well as providing continuous monitoring of seal counts at particular sites.

Modelling of transition probability indicated that controlled disturbance trials did not affect the probability of harbour seals transiting from one haul-out site to another. Seals generally displayed a high degree of site fidelity. The relationship between site fidelity and transition probability varied with whether seals hauled out again on the same or on a subsequent low tide period. Overall seals were more likely to transit from one haul-out site to another if the trip in between included at least one high tide period.

The levels of disturbance in these trials were high relative to any expected boat based disturbance during construction. The results of this study suggest that increased boat activity during the construction phase of the proposed tidal turbine development is unlikely to cause individual seals to transit from one haul-out site to another. If seals are flushed from their haul-out they are likely to return during the same or subsequent low tide periods. The lack of redistribution after disturbance events suggests that monitoring effort to detect disturbance effects of boat activity need only be on a local scale relative to any proposed development.

Introduction

Several studies have described the normal haul-out pattern of harbour seals in relation to environmental conditions (Watts 1992, Grellier et al. 1996), tidal state (Pauli & Terhune 1987), diurnal activity (Watts 1996) and seasonal events such as the breeding and moult periods (Thompson et al. 1989). The expected haul-out pattern of harbour seals is therefore well understood. Understanding what happens when that normal haul-out pattern is affected by anthropogenic activity that causes disturbance is key to targeting mitigation to minimise the impact of disturbance on seals. Previous studies looking at the causes of disturbance of seals at haul-out sites have typically focussed on factors such as the distance at which seals are disturbed by boats (Jansen et al. 2010), the type of boat activity that causes disturbance (Johnson & Acevedo-Gutierrez 2007) and disturbance by pedestrians (Osinga et al. 2012). However, having identified the causes of disturbance it is important to then quantify the associated effects in terms of behavioural changes in the seals being studied.

This study assesses the impact of repeated disturbance of harbour seals from their haul-out sites and how this might be relevant to the proposed tidal turbine development in the Sound of Islay. The eventual aim of the study is to develop a protocol for monitoring haul-out sites that is both capable of detecting localised disturbance effects and of being delivered by developers and/or their consultants. The disturbance trial results will be combined with information on spatial and temporal patterns of haul-out behaviour to estimate the geographical range of influence of localised disturbances and to determine the scale of monitoring required to identify causal agents of disturbance.

Methods

Existing telemetry data

A total of 17 harbour seals were tagged in and around the Sound of Islay in 2011 and 2012. This telemetry data shows that there is some degree of interchange between haul-out sites surrounding the proposed development in the Sound of Islay and the South-East Islay Skerries SAC. Figure 1 shows the GPS tracks of individuals tagged in the Sound of Islay in 2012 demonstrating how seals used the Sound either for transit/foraging or to haul out and the geographical extent of their movement beyond the Sound during the tag deployment. The movement of seals tagged in 2011 and 2012 is briefly described in Sparling (2013). These data, along with aerial survey data, were used to determine the haul-out sites within the Sound of Islay that are most frequently used by harbour seals, especially those in the vicinity of the proposed tidal turbine development. This was with the aim of choosing the best locations for a new deployment of telemetry tags and for setting up remote camera systems.

Monitoring haulouts using remote cameras

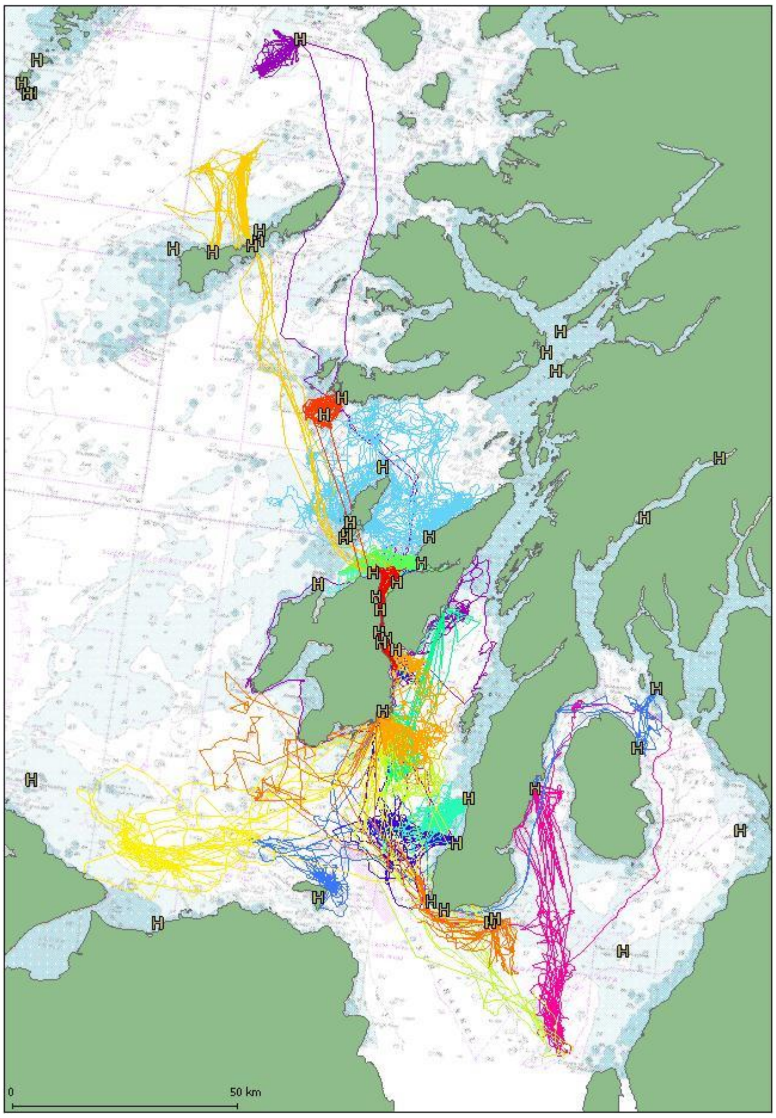
Monitoring all haul-out sites used in the Sound of Islay according to the 2011/2012 telemetry data would be prohibitively expensive and, at least at some lesser used sites, unnecessary. Remote time-lapse cameras were set up at vantage points overlooking the two most frequently used haul-out sites within the Sound of Islay in 2011/2012. These sites were Rubha Bhoraraic (RBR) and Bunnahabhain (BHN), either side of the proposed development. Time-lapse photography was collected continuously throughout the study period. Data collection commenced at BHN on the 23/04/2014 and at RBR on the 24/04/2014. Camera systems were recovered on the 22/07/2014. Each camera system consisted of two Canon EOS 1100 DSLR cameras in a single weatherproof housing, each with one camera equipped with an 18-55mm lens and the other with a 70-300mm lens. This provided both a wider scale view of activity around the haulout and a view more focussed on the haulout itself. Time-lapse photographs were taken at a rate of one per minute. Background counts could then be done for those sites during daylight hours while also recording the recovery of seals hauling out again within the same low tide period after being disturbed into the water. A high time-lapse frequency also allowed for observations of disturbance events other than those carried out purposefully during this study.

GPS/GSM phone tag deployment

In order to monitor changes in seal distributions at multiple haulouts that may result from anthropogenic sources, we first need to know the geographical extent to monitor. For example, if individual seals were faithful to only one haulout there is only a need to monitor counts at the haul-out sites nearest (within foraging trip distance of) the source. On the other hand, if seals regularly transited amongst distant haul-out sites then the monitoring extent would have to be appropriately increased. By using telemetry devices to monitor the movements and haul-out patterns of harbour seals this study aims to determine the geographical extent over which animals move in relation to the proposed Sound of Islay tidal turbine development. This study also aims to quantify the frequency with which animals' transit from one haulout to another within that geographical context.

In April 2014 a total of eight SMRU GPS/GSM phone tags were deployed on adult female harbour seals. Animals were captured at either BHN or RBR because the 2011/2012 telemetry data suggest that these sites are the most frequently used haul outs within the Sound of Islay. Catching and tagging seals at these two sites increased the likelihood of finding those animals again to carry out disturbance trials. Also, their haul-out behaviour in relation to other non-telemetered seals would be more likely to be recorded concurrently using the remote camera systems. This approach aimed to quantify the normal haul-out and movement behaviour of harbour seals at the two most frequently used haul-out sites within the Sound of Islay and to what extent those behaviours were affected by disturbance.

Figure 1. Tracks of 10 harbour seals (colour-coded by individual) tagged within the Sound of Islay in 2012. H represents a subset of the 'known haulout' list showing locations in and around the Sound of Islay where telemetered harbour seals have hauled out historically.



Controlled disturbance trials

The type of disturbance most relevant to the proposed tidal turbine array at the Sound of Islay is a higher than normal exposure to boat traffic during the construction phase. To simulate this type of disturbance, experimental trials were carried out by approaching hauled out seals in a 14ft RIB at a speed of five knots. Approaches were initiated at a distance of approximately 300m and continued in a straight line until the haul-out site was reached and all seals were flushed into the water. Seals were approached at an angle that provided the clearest line of sight between animals on the haulout and the approaching boat. This method of disturbing seals into the water is extreme in the sense that at the two sites overlooked by time-lapse cameras (RBR and BHN) animals were disturbed into the water whether telemetered animals were present or not. However, at all other sites a telemetered animal had to be present before a disturbance trial was carried out. Disturbance trials were restricted to approximately two hours before low tide to allow time for animals to haul out again within the same low tide period.

To maximise the efficiency with which telemetered animals could be found GPS/GSM tags were programmed to transmit regular and frequent updates on their location at approximately 1 hourly intervals. This allowed the disturbance events to be targeted on sites with tagged animals present. Telemetry data from sites at which seals were flushed into the water provided information on the effects of disturbance in terms of their subsequent haul-out behaviour and whether or not disturbance caused animals to transit between haul-out sites more frequently. Tags deployed in 2014 added to data collected in 2011/2012 to better quantify the amount of interchange between haul-out sites in the Sound of Islay and the nearby South-East Islay Skerries SAC.

Analysis of haul-out behaviour

Data

SMRU GPS/GSM tags record haul-out events. From the track data we can assign a location to each haul-out event. GPS fixes are obtained at irregular intervals and when no location fixes were obtained during a given event an approximate location was assigned using linear interpolation and then snapped to estimated location of the nearest known haul-out site. Where there was a large mismatch (snapDist > 2km) and the haulout was on land, we added this new location to the known haulout list. Apparent at sea haulouts (as defined by the tags' > 10 minutes continuous dry rule) were omitted from this analysis.

A haul-out event was defined as having ended when the tag was wet for > 10 minutes. An animal was then defined as being on a trip. The location of and time until a subsequent haul-out event then determined if an animal had returned to the same haul-out site or transited to a different haul-out site and in what timeframe either of those events occurred.

Analysis

The first week of data was excluded from the final dataset. All statistical analyses were carried out using the R package (R Development Core Team 2014). We examined how transition probability, i.e. seals moving from one haul-out site to another, was influenced by: Julian day, site fidelity, tidal cycle and disturbance. Julian day was included because there may be seasonal changes in the propensity of seals to switch haul-out location. Site fidelity may also influence transition probability and was defined as the percentage of haul-out events in the previous week that were at the current haul-out location. Both Julian day and site fidelity were input as smooth terms. Whether or not seals hauled out during the same or a subsequent low tide period was included as a factor to determine to what extent seals repeatedly haul out and how often they switch haul-out sites within a single low tide. In the context of disturbance this is relevant in that once disturbed into the water seals have four choices; (i) haul out within the same low tide period at the same haul-out site, (ii) haul out again within the same low tide period at a different haul-out site, (iii) haul out on a subsequent low tide period at the same haul-out site and (iv) haul out on a subsequent low tide period at a different haul-out site. Disturbance was included as a factor defined as whether or not seals were flushed into the water during a haulout event while carrying out controlled disturbance trials. The maximal model also included an interaction between site fidelity and tidal cycle because the effect of site fidelity on transition probability may depend on whether the animal is hauling out in the same or a subsequent low tide period. A Generalised Additive Mixed Model (GAMM) framework, within the R package mgcv (Wood 2004), was used with an AR1 correlation structure incorporated to account for autocorrelation among trips within each individual. Backward model selection was carried out using Akaike's Information Criterion (AIC) selection. The primary aim was to determine if repeated disturbance events affected seals' haul-out behaviour. However, this modelling approach allowed identification of variables that influence probability of transit to different haul-out sites regardless of the presence or absence of disturbance.

Results

Monitoring haulouts using remote cameras

General counts

When conditions permitted, the number of seals on haulouts was counted between the hours of 04:00 and 22:00 at both BHN and RBR. Tidal cycles at each site were defined as 6 hours either side of low tide at Port Askaig and counts were timed relative to the closest low tide. Counts of seals were then grouped by month and split into spring or a neap tide categories. Figures 2 and 3 show the mean and maximum counts during spring and neap tides in May, June and July at BHN and RBR respectively. During the greatest spring high tides both haul-out sites are completely submerged, truncating the time available to seals to haul out during a tidal cycle. However, during neap high tides at least a small amount of land protrudes from the water meaning seals can choose to remain hauled out at high tide. This pattern of haul-out time being truncated during spring tides and more widely spread during neap tides was quite evident at BHN. However, at RBR the pattern was less clear, perhaps due to the lower use of that haul-out site by seals as the study progressed.

Disturbance trials

On disturbance trial days counts were made every minute in the lead up to and beyond the time of disturbance. Counts were made until 6 hours after the low tide period in which disturbance trials took place, allowing for an assessment of the recovery of haul-out numbers from pre-disturbance to post-disturbance levels.

At BHN a total of 17 disturbance trials were recorded using time-lapse photography. The post-disturbance recovery of seal numbers on the haulout to pre-disturbance levels is shown for BHN in Figure 4. Mean counts returned to ~50-60% of pre-disturbance numbers within 30 minutes and ~90-100% of pre-disturbance numbers within 240 minutes. Beyond that time the influence of the rising tide caused mean counts to decline. Time-lapse photography indicated that BHN was regularly used as a haul-out site throughout this study with some seals present on most days. Seals were therefore available for disturbance trials on almost every occasion the site was visited. Zero counts occurred on only 2 days in May, 3 in June and 1 in July.

Figure 2. Mean counts of hauled out seals (solid red) with 95% confidence intervals (dashed red lines) over minutes relative to low tide at BHN. Mean peak counts are also given (solid black). Data are divided into spring and neap tide periods for May, June and July.

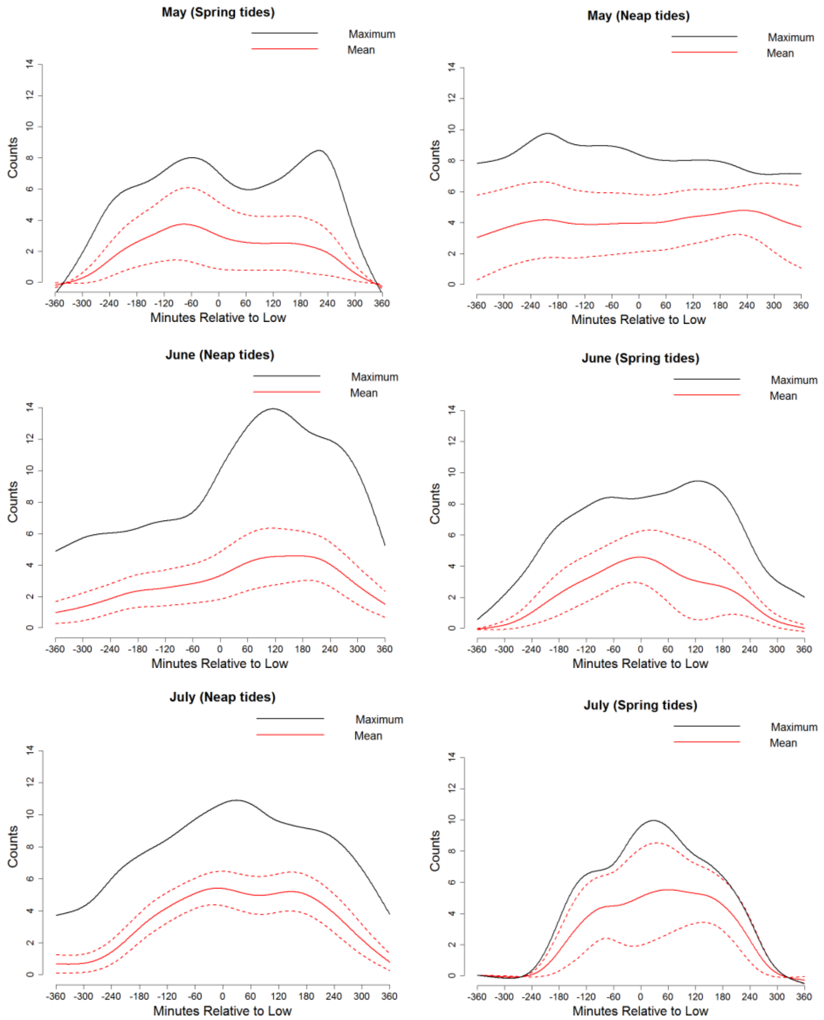
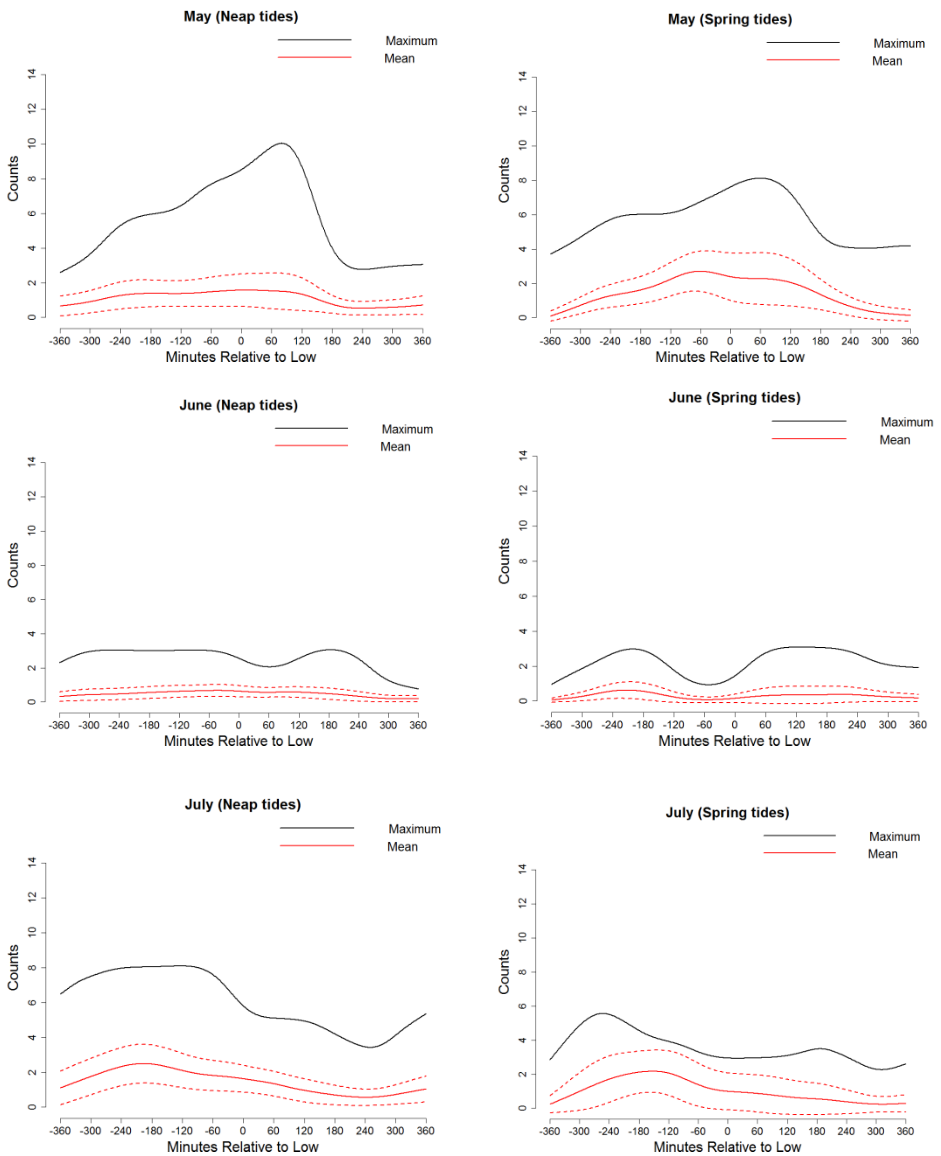


Figure 3. Mean counts of hauled out seals (solid red) with 95% confidence intervals (dashed red lines) over minutes relative to low tide at RBR. Mean peak counts are also given (solid black). Data are divided into spring and neap tide periods for May, June and July.



At RBR a total of 10 disturbance trials were recorded using time-lapse photography. The lower number of trials at RBR was due to this site being used less as the season progressed. Several disturbance trials were cancelled as there were no animals on the haulout. Mean counts at RBR were quite low during the study period, especially in June (Figure 3). Only one of the telemetered animals in this study visited RBR beyond April, making short visits (<2hr) and did not return to the site on the next haulout. There were no seals hauled out at low tide on 11 days in May, 17 days in June and 11 days in July.

The response of seals to disturbance differed between RBR and BHN. In all 10 of the disturbance trials at RBR no seals hauled out again within the following 30 minutes and on only one occasion did a seal haul out again within the next 60 minutes. The number of seals being disturbed at any one time at RBR was generally quite low. On four occasions only one seal was disturbed into the water. On two occasions there were relatively high numbers of seals on the haulout, 6 and 8 on the 11/06/14 and 15/07/14 respectively. However, even in these disturbance trials recovery of the counts of seals to pre-disturbance levels did not occur. As none of the telemetered animals were disturbed at RBR it is not possible to say what seals did after entering the water.

The mean peak counts of seals at BHN and RBR were compared for the day before, the day of and the day after disturbance trials (Figure 5). Data for BHN and RBR were combined due to the low number of disturbance trials at RBR. Mean peak counts were slightly lower on disturbance trial days as compared with the day before and the day after but the difference was not significant ($F(2,54)=0.621$, $p>0.05$). Fewer seals on the haulout during disturbance trial days might be expected if the disturbance trial itself reduces the amount of time available to seals to haul out in a given low tide period. Also, the telemetry data suggests that a proportion of animals will leave the site of disturbance which again would result in lower peak counts during disturbance trial days.

Figure 4. Counts of seals during 6 hours post disturbance expressed as percentage of the number of hauled out seals (solid black line) at the time of disturbance, with 95% confidence intervals (dashed red lines). Data are for BHN.

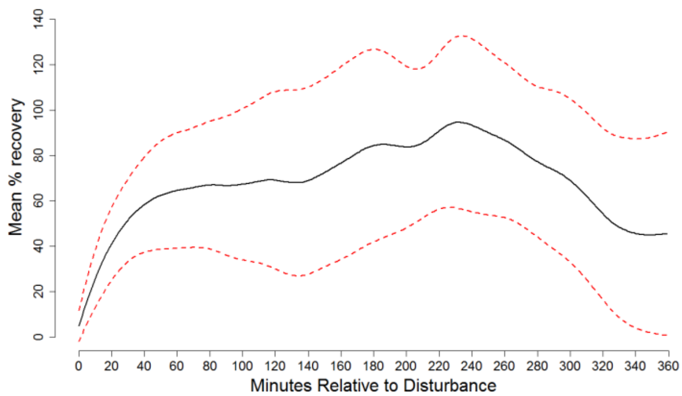
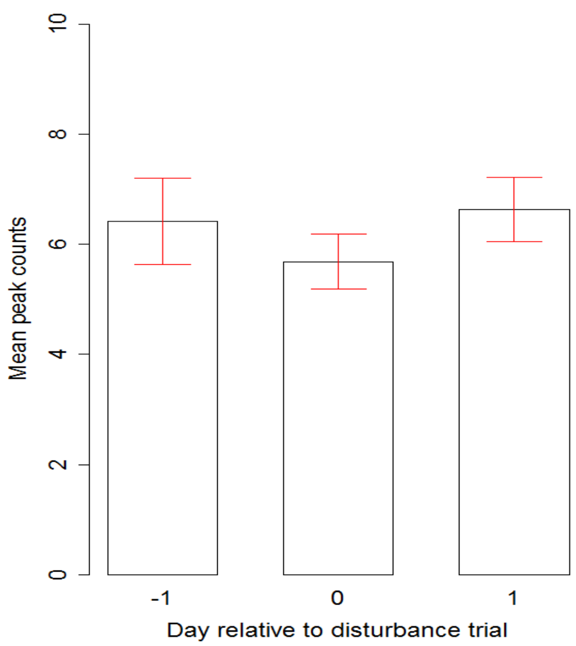


Figure 5. Mean peak counts of seals at BHN and RBR on the day before, the day of and the day after disturbance trials were carried out. Error bars are +- 1 standard error.



GPS/GSM phone tag deployment

In the 2014 telemetry data a total number of 626 days of data were collected from 8 adult female harbour seals. The mean duration of tag deployment was 78 days (range = 41 to 107 days, SE=6.98).

Use of haul-out sites

Overall, 16 haul-out sites were used throughout the study with individual seals using a mean of 5 haul-out sites (range=3 to 9, SE=0.77). There were a total of 634 haul-out events separated by more than 10 minutes. The mean duration of haul-out events not including those in which disturbance trials were conducted was 5.2 hours (SE=0.28). Some degree of site fidelity was shown in both the overall mean total of haulout time spent at each site and the mean number of times each site was visited. Table 1 shows these figures for each of the haul-out sites used as well as giving the number of individuals that visited each site throughout the duration of the 2014 study period.

Movement between haul-out sites

After filtering the data a total of 626 trips were identified in the 2014 dataset. A total of 162 (26%) trips resulted in seals transiting to different haul-out site with a mean trip duration of 34.1 hours (SE=4.58). On the remaining 464 (74%) trips the seals returned to the same site, with a mean trip duration of 14.25 hours (SE=0.95). The return trip duration was shorter because some seals did not leave the local area, but stayed close by before hauling out again. Also, shorter trips with higher site fidelity were increasingly observed as the pupping season progressed.

For trips that resulted in a transition to another haul-out site the mean number of times that seal had hauled out at that site in the previous week was 2.6 (SE=0.27) compared to 7.2 (SE=0.28) for return trips. This suggests that on a short temporal scale seals were more likely to haul out repeatedly in the same place. However, it was apparent that seals did make broad-scale seasonal changes in haul-out site usage with several animals moving to the north end of the Sound of Islay and into Loch Tarbert on the West side of Jura from the beginning of June onwards.

Of the 162 trips that resulted in a transition to a different site only 13 were transitions within the same low tide period. Two of these occurred after a controlled disturbance trial. The remaining 149 transitions occurred on a subsequent low tide. Of the 464 return trips, 51 occurred within the same low tide period. Some of these trips may be explained by seals entering the water for a short period and then hauling out again on a different part of the same haulout. Eleven of these trips occurred after controlled disturbance trials. The remaining 413 return trips occurred on a subsequent low tide period.

Table 1. Listed are site code abbreviations, their location and the latitude, longitude co-ordinates for all sites used by telemetered seals as haul-out sites in 2014. Also included are the number of times each site was visited, the mean duration of haulouts at those sites and how many individuals visited each site during this study.

Site Code	Site Name	Location	Lat	Lon	No. of visits	Mean haulout duration (hours)	No. of individuals
BDH	Bagh an Da Dhoruis	Islay	55.93559	-6.15097	87	3.2	3
BHN	Bunnahabhainn	Islay	55.891175	-6.131105	123	5	7
BRP	Brein Phort	Jura	55.922896	-6.064843	23	5.3	3
CAS	Carragh an t-Struith	Jura	55.87061	-6.096444	4	2.3	2
CON	Colonsay North	Colonsay	56.14747079	-6.167427959	2	4.7	1
EGH	Eileanan Gainmhich	Islay	55.864512	-6.110327	59	3.9	6
EGR	Eilean Gleann Righ	Jura	55.968332	-5.986099	230	6.2	6
EST	Eileanan Stafa	South Uist	57.39659	-7.288119	35	6.9	1
HAU	Haun	South Uist	57.090523	-7.296631	8	3.5	1
HOU	Hough Skerries	Tiree	56.52	-7.020000047	1	0.6	1
HRT	Hairteamul	outh Uist	57.084119	-7.229136	1	1.1	1
ISL	Nave Island	Islay	55.8991244	-6.34078397	1	0.5	1
RBL	Rubha Liath	Jura	55.962461	-5.950904	22	5.6	2
RBR	Rubha Bhoraraic	Islay	55.819718	-6.103997	4	1.6	3
SAN	Sanda Island	Kintyre	55.284856	-5.571027	4	2.9	1
SGB	Sgeiran a Bhudragain	Jura	55.958036	-5.946192	22	4.5	3

Controlled disturbance trials

Disturbance trials commenced on 26/05/2014 and were repeated every three days (weather permitting) for a total of 20 disturbance trial days.. There were 22 seal disturbance events when seals were flushed into the water and the telemetry data was successfully uploaded from the tag (Figure 6). Of those, 13 resulted in animals hauling out again within the same low tide period. On 12 of those occasions seals returned to the same haul-out location. The remaining nine seal disturbance events resulted in seals starting a trip that lasted at least one tidal cycle. When on these trips seals seemingly behaved normally, visiting the same areas they would use for what may be foraging trips. On eight of these occasions the seal returned to the same haul-out site.

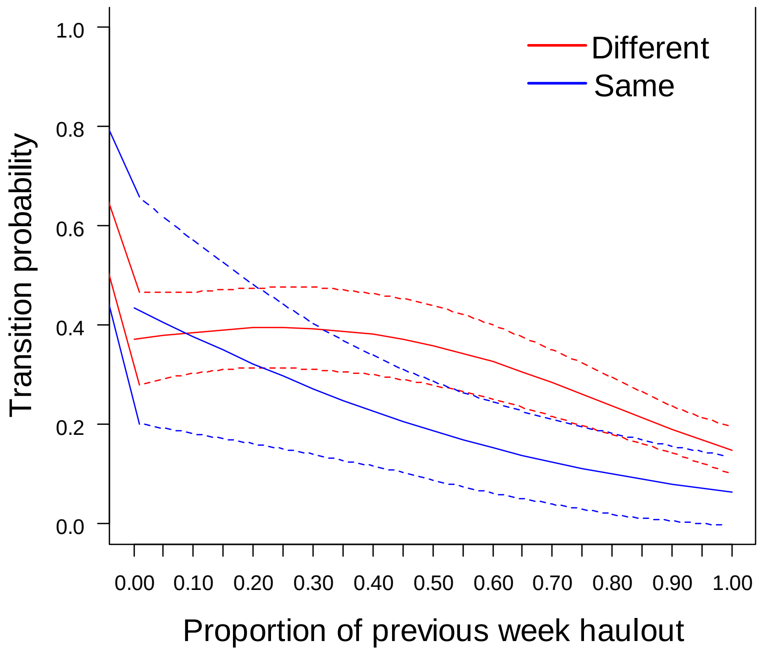
Figure 6. Haulout/trip transition matrix showing where seals departed from and where they hauled out again after simulated disturbance trials. The total number of disturbance trials resulting in each scenario are given. In the upper part of the matrix in grey are trials where seals hauled out again within the same low tide period after being disturbed into the water. In the lower part of the matrix in pink are trials where seals hauled out again during subsequent low tide periods.

		Arrive			
		BHN	BDH	EGR	BRP
Depart	BHN	5			1
	BDH		1		
	EGR			6	
	BRP				
	BHN’	5		1	
	BDH’		1		
	EGR’			2	
	BRP’				

Analysis of haul-out behaviour

By the process of backwards model selection using AIC, only the interaction between site fidelity and tidal cycle was retained in the final model. It is likely that Julian day was not retained because although seals did change the site at which they preferred to haul out during this study, this was mostly associated with the onset of the pupping season when broad scale changes were made. When seals did switch haul-out sites they switched preference for one haul-out site to another rather than regularly switching between different haul-out sites. This was reflected in the fact that that site fidelity was retained showing that the higher the proportion of haul out events in the previous week that occurred at a particular site, the more likely they were to haul out there again rather than elsewhere. However, the shape of the relationship between site fidelity and transition probability varied with whether seals hauled out again on the same or on a subsequent low tide period (Figure 7). Overall seals were more likely to transit from one haul-out site to another if the trip in between included at least one high tide period.

Figure 7. Shown are the transition probabilities for seals dependent on the proportion of times they had hauled out at the site of departure within the previous week. Transition probabilities are shown for the two scenarios of either hauling out again on the same (blue) or on a subsequent (red) low tide period. Solid lines are model predictions with 95% confidence intervals as dashed lines.



Discussion

These results suggest that repeated disturbance of harbour seals in the Sound of Islay did not cause broad-scale changes in the location at which they choose to haul out. Instead, seals usually either hauled out relatively quickly in the same location, within the same low tide period, or went on a trip before hauling out in a subsequent low tide period, usually at the location from which they were disturbed. The main effect of disturbance therefore seems to be to reduce the amount of time seals spend hauled out around the point of disturbance. In the case where seals haul out again within the same low tide period that effect will be minimal. However, at the higher usage haul-out sites where the mean duration of haulout was 5-6 hours a disturbance event that resulted in seals heading out to sea would reduce haul-out time within that low tide period. This is particularly true at haul-out sites that are submerged at high tide. A similar study by Andersen et al. (2012) suggested that extended inter-haul-out trips that occurred directly after a disturbance event were foraging trips, indicating a behavioural adaptation to offset the cost of being disturbed into the water. The 2014 telemetry data from the animals in the Sound of Islay suggest this may be true as the areas visited after disturbance events and the observed behaviours while in those areas were similar to those during normal trips outwith the disturbance trial days.

Previous studies that assessed the effects of disturbance provide evidence that anthropogenic activities can alter the haul-out behaviour of harbour seals. For example, Henry and Hammill (2001) suggested that increased human activity during good weather or during summer vacations increases the number of disturbance events of harbour seals in Métis Bay, Canada. Similarly, Lonergan et al.

(2013) suggest that harbour seals on the west coast of Scotland haul out less at the weekends as opposed to during weekdays, which may be attributable to differing levels of anthropogenic activity. It has also been suggested that harbour seals may switch to a nocturnal haul-out pattern to avoid hauling out during the day when anthropogenic activity is high (London et al. 2012). Repeated disturbance can therefore illicit broad-scale changes in the timing and frequency with which harbour seals haul out during their annual life cycle. This study found that repeatedly flushing seals into the water did not cause seals to transit from one haul-out site to another. However, conclusions drawn from this result that are relevant to potential marine renewable developers only apply to activity that involves boat work in close proximity to haul-out sites. Other types of anthropogenic activity that may be thought to cause disturbance would have to be assessed separately.

The seals in this study displayed a high degree of site fidelity, repeatedly returning to the same haul-out sites after trips. Site fidelity in harbour seals has also been observed in other studies (Thompson et al. 1989) and has been observed to change seasonally (Thompson et al. 1994, Lowry et al. 2001). The results of this study show that a relatively intense level of disturbance did not adversely affect site fidelity. Even when repeatedly flushed into the water seals generally returned to the haul-out site from which they were disturbed either immediately or during a subsequent low tide period.

The fact that repeated disturbance in this study didn’t cause seals to switch haul-out sites suggests frequent disturbance events at a particular haul-out site are likely to cause the same animals to be flushed into the water repeatedly. There was a seasonal change observed in the spatial distribution of seals with a shift north to Loch Tarbert for the pupping season being most notable. However, this did not result in an increase in the probability of seals transiting between haul-out sites due to the behavioural change being a sudden and sustained switch of haulout preference from one site to another. Only one site, BHN, was used as a pupping location within the Sound. All other pups were observed either in the South-East Islay Skerries SAC or in Loch Tarbert in Jura. Harbour seals tagged in the Sound of Islay shifting to a more northerly distribution was also noted in Sparling (2013) based on the 2012 telemetry data.

The mode of disturbance used in controlled trials for this study was to approach seals on haul-out sites by boat until they flushed into the water. This was to simulate the type of disturbance that might result from increased boat activity associated with a tidal turbine development in the Sound. It is possible that pedestrian disturbance at the same intensity as that implemented through controlled disturbance trials by boat may have had a stronger effect. However, this was not found to be the case in a comparison of boat versus pedestrian disturbance in a study by Andersen et al. (2012). At the two sites where most seals were observed in the vicinity of the proposed development, BHN and RBR, pedestrian access is very limited and so it should be expected that disturbance by pedestrians would be minimal. This was found to be the case according to the time-lapse photography data at both these sites which demonstrated that pedestrians rarely approach these haul-out sites. Generally, seals in the Sound of Islay are exposed to boat traffic on a daily basis at a distance that doesn’t cause them to flush from their haul-out sites. The time-lapse photography data shows that neither of the two haul-out sites BHN or RBR were ever approached by boats at a proximity similar to that during controlled disturbance trials. It is also unlikely that boat activity associated with the proposed tidal turbine development would be in such close proximity to haul-out sites within the Sound.

Conclusions

Planned construction at the Sound of Islay is likely to overlap with important periods in the life cycle of seals that regularly use the area to haul out. The results of this study suggest that despite repeated disturbance events seals will return to the same haulout location either immediately within the same low tide period or during a subsequent low tide period having gone on an inter-haul-out trip. It should therefore be expected that increased anthropogenic activity at or near a particular haul-out location will not cause individual seals that use that haul-out site to transit to another haul-out site as a result. In this study seals were exposed to controlled disturbance trials that resulted in animals being flushed into the water by an approaching boat. It is unlikely that during the construction phase of the proposed development at the Sound of Islay that seals would be exposed to such an extreme form of disturbance with any regularity. By quantifying behavioural changes associated with frequent disturbance events this study shows that the expected effect of increased boat traffic on the probability of seals transiting from one haul-out site to another will be negligible. Also, this study suggests that monitoring effort to mitigate any perceived risk of increase in disturbance levels need only be on a local scale due to the continued site fidelity of harbour seals when exposed to disturbance caused by approaching boats.

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ANNEX

Scottish Government questions and additional information relating to request for advice on PBR

<u>Request for advice</u>

We would like to understand SCOS's opinion on the issues detailed in the attached supporting information and whether they give rise to sufficient concern regarding the use of PBR, such that they would recommend an alternative framework that might make fuller use of best available evidence?

Marine Scotland currently uses PBR for both determining numbers of seals that can be licensed to be shot and for marine renewable assessments. Taking account of the points raised in the supporting information (particularly section A), does SCOS consider that PBR is suitable for both of these applications, and if not, do they consider that a single assessment framework accounting for both sets of licensable activities is required?

Given that the development and application of any framework would require close collaboration between policy makers and scientists, would SCOS consider it had a role in working with policy makers to deliver an alternative framework? If so, do SCOS have any recommendations at this stage on how best to make that happen?

In considering your response, please note that Marine Scotland do not request a definitive and final view from SCOS on all the issues raised. We anticipate that doing so would take considerable time and may require new work. A response that provides indicative consideration with respect to the overall purpose and aim of this request would be most useful at this stage.

(Responses given in italics)

<u>Supporting information</u>

<u>Background and purpose</u>

The Scottish Government uses advice based on PBR to determine the annual numbers of grey and harbour seals that can be removed from the populations through shooting. There are a number of other potential pressures upon seal populations that the Scottish Government wishes to assess with respect to regulation of the population consequences and the appropriate magnitude of effects. For example, the potential impacts of marine renewable energy, and of port developments also need to be appropriately regulated. The overall purpose of this question is to initiate fuller consideration of what constitutes the best available evidence and most suitable techniques for addressing potential impacts to seal populations. The aim is to consider the strengths and weaknesses of PBR and the potential for other approaches.

<u>Issues raised in association with PBR</u>

A list of concerns have been raised on several occasions about the utility of PBR for undertaking cumulative impact assessments. These issues are detailed in sections A to C below and SCOS are invited to comment on these, or any other issues associated with use of PBR. It would be particularly helpful if SCOS were able to provide an indication of whether or not these issues have reasonable foundation, whether anything could be done in the context of applying PBR to address them, and to also consider if they provide good reason to consider alternative approaches to assessment frameworks.

There are three main areas that these issues can be separated into:

A. the concept of using PBR as a framework for both determining annual numbers of shooting licences and for licensing renewable developments,

B. the underlying principles of PBR, and

C. the implementation of PBR as it is currently used by the Scottish Government under the advice of SCOS.

A. Issues relating to the concept of using PBR for licensing purposes:

1. Appropriate consideration of uncertainty associated with effects is a key aspect of any assessment. There are clearly very different levels of certainty associated with shooting seals and modelling collision risk. An issue is how best to account for these varying levels of uncertainty, and whether or not PBR provides the most useful threshold setting tool in light of the uncertainties associated with effects. In particular, PBR does not allow for nuances such as a probability of death.

This is correct. The PBR is a method for assessing thesimply estimates the number of individuals that can be removed level of removals from a population while still allowingthat will allow the population to grow (or decline) towards a pre-selected target known as the maximum net-productivity level (MNPL). If a probability of death can be assigned to an event, the annual mortality rate due to those events could be estimated in any situation where an effect can be translated into a probability of death or removal and could then be incorporated into the PBR process. If the probability of death is not known or cannot be estimated there is no facility to include it in a PBR process.

PBR deals only with removals and sub-lethal effects on fecundity cannot be incorporated. In any situation where there are sufficient data, and sufficient understanding of the population processes to allow a more realistic population dynamics model to be developed this could be used as the basis of some form of population viability analysis (PVA) to investigate the population consequences of those sub-lethal effects .

At present regulators take a precautionary approach where collision risk estimates are assumed to be equivalent to mortality estimates. However, currently the probability of mortality from collisions is unknown and realistically incorporating any uncertainty around such probabilities would not be possible within a PBR framework.

2. PBR is recalculated annually, based on latest population estimates. While this is appropriate for use in the iterative management of effects for issuing licences to shoot seals, it may be less suitable for a robust assessment of the effects to a population over the 20-25 year operational life span of a renewable energy development for the purposes of licensing decisions. In these situations, Marine Scotland must carry out an assessment that is competent of considering impacts throughout the lifetime of the project under scrutiny, and any consent granted would be difficult to revoke at a later stage. Is it reasonable to use PBR to assess effects occurring over forthcoming decades when it was originally designed for assessing effects annually?

We assume that this question relates only to actual removals (see above for discussion of non-lethal effects). PBR was not developed for long term prediction of population dynamics. This is likely to be a problem associated with the use of PBR for long term management PBR assumes a very simple density dependent population dynamics model based on a fixed intrinsic rate of increase and a fixed density dependent function. If these assumptions are robust then PBR could *be used iteratively, but if sufficient additional information exists to develop a more realistic population model that should be used to provide a more reliable means of predicting future population trends. , but only in situations where the underlying assumptions of density dependent population growth are violated. This becomes an issue where populations are depleted to significantly below historical levels and are continuing to decline, as for example with the harbour seal population of Orkney. Continued decline of such a population if the PBR is not exceeded could be due to a change (reduction) in the carrying capacity due to resource limitation, a new type of natural mortality or an unidentified anthropogenic take or to violation of the underlying assumption of density dependent growth.*

If a depleted population (below MNPL) is responding to density dependent effects, the current PBR should always be the minimum value and future projected PBRs should always be higher, unless carrying capacity changes.

Extrapolating the current population trend assuming that PBR is or is not taken each year, we can predict the counts and thus the likely PBR values in the coming years. Doing this for North Coast and Orkney area gives the following results. Please note that because this is based on a fitted line the estimated count for 2013 is slightly higher than the true count giving a higher PBR of 12 (rather than 11). Please also note the PBR values removed from the count were 72% of the PBR based on the estimate, from telemetry data, that 72% of seals are hauled out in the moult.

Year	PBR not taken		PBR taken	
	predicted count	PBR	predicted count	PBR
2013	2032	12	2032	12
2014	1841	11	1833	10
2015	1669	10	1654	9
2016	1513	9	1493	8
2017	1371	8	1347	8
2018	1243	7	1215	7
2019	1126	6	1096	6
2020	1021	6	989	5
2021	925	5	893	5
2022	839	5	806	4
2023	760	4	727	4
2024	689	4	656	3
2025	624	3	593	3
2026	566	3	535	3
2027	513	3	483	2
2028	465	2	436	2
2029	421	2	394	2
2030	382	2	356	2
2031	346	2	321	1
2032	314	1	290	1
2033	284	1	262	1
2034	258	1	237	1
2035	234	1	214	1
2036	212	1	193	1
2037	192	1	174	1
2038	174	1	157	0
2039	158	0	143	0

3. PBR assumes that the effect upon a population is via adult survival rates and that maximum productivity is achieved when the carrying capacity is reduced below a certain level. In practice, certain effects (e.g. noise) may impact productivity rates. A recent example is the application of PBR for seabird responses to wind farms, which included effects of displacement. Does PBR have any role to play in assessment of non-lethal effects, or in situations where an impact may have both lethal and non-lethal effects?

We assume that the first sentence was meant to say that maximum productivity is achieved when the population is reduced to some level below the carrying capacity. It does not currently take any account of non-lethal takes or changes in fecundity.

The underlying population process is not explicitly modelled beyond a simple assumption that growth is density dependent and that the population growth rate at MNPL will be approximately half the intrinsic rate of increase. The form of density dependence and the size of the carrying capacity will determine the size of the population at MNPL, but there is no requirement and therefore no facility built into the process to incorporate any explicit survival or fecundity functions. It would be plausible to replace the underlying density dependent growth assumption with a more complicated model that explicitly accounts for changes in survival probabilities and/or fecundity estimates. However, in that case a more realistic population model, as part of a PVA, would probably be more appropriate.

However, currently data are not available to determine the impact of marine renewable developments on fecundity for instance so any inclusion of such potential impacts would have to be based on expert knowledge (e.g. interim PCOD)

B. Issues relating to the underlying principles of PBR:

1. The density dependent response assumed in the PBR model assumes that maximum net productivity level (MNPL) is at half of carrying capacity. Whilst the density dependent response function assumed in the PBR model was clearly carefully selected based on sensitivity testing (against the objectives of PBR as stated in Wade 1998) the issue remains that we may reasonably expect species with varying life history traits to respond differently to perturbations and that the PBR model's simplification of density dependent response cannot be as readily tested with empirical data compared to other approaches.

The density dependent response used in the development of PBR produced a symmetrical sigmoid growth curve. In this case the NMPL is at or near half the carrying capacity. Changes to the way that density dependence affects population growth will alter this level. The few examples available for seal populations suggest a steeper approach to carrying capacity and a tighter inflection as density dependence kicks in. In this case the MNPL level will be higher than 50% of carrying capacity. The simulations conducted by Wade (1998) showed the output was robust to MNPL value at 0.45 and above.

It is not immediately clear that other approaches will be easier to test given the paucity of information on how fecundity and survival respond to population changes and the difficulties in estimating carrying capacities.

2. The PBR model assumes a fixed carrying capacity over a period of decades (up to 100 years). Populations of species, such as seals, respond to a number of spatio-temporal effects that give rise to legitimate questions about the suitability of regulating effects over local areas based on an assumption that the population can recover to a fixed level that will remain constant over decades. The effects of climate change on prey, or the potential for harbour seal populations to respond to inter-specific competition with grey seals could reasonably be

considered as examples of effects that are ultimately changing the capacity of the local environment to support a population at the current level. If it is only other anthropogenic effects that require regulation by society, then the issue arises of whether PBR provides the most suitable model, since we might assume that carrying capacity would change over time.

The PBR model does not need to assume a fixed carrying capacity through time. It simply assumes that density dependent effects will drive the population towards its carrying capacity. The PBR estimate can be updated at the same frequency as the available information on population size.

3. Populations' vital rates can reasonably be expected to respond to perturbations in a stochastic manner. Resource availability varies and species with differing life history traits are more or less likely to be able to respond over specified time periods. Point B1 above raised this issue with respect to the assumed population level at which MNPL occurs, but it may also be an issue with respect to the rate (or range of rates) at which populations may respond to change. As with point B2, is there a concern that embedding a simplification into assessments may give rise to the assessment framework acting as a barrier to progressive improvement in our understanding of how populations respond to perturbation?

These objections are all correct, but to date no practical alternative has been proposed. A population model with predictive power is required.

4. Productivity, and Rmax values, will similarly be expected to have spatial and temporal variation. This leads to questions about the applicability of published references to local circumstances.

The Rmax value is expected to be similar for different populations of seals within the UK. Observed productivity will of course vary between locations and times as it is essentially the product of the density dependent growth rate and the population at a particular time. The applicability of estimates of demographic parameters to populations other than the sampled population is a general problem for any demographic model or management procedure.

5. The PBR equation uses a denominator of 2. Is this because MNPL is half of carrying capacity? Or because MNPL is based on the proportion of the population that is female?

It is because the growth rate at MNPL is assumed to be approximately half Rmax if MNPL is at half the carrying capacity. If MNPL is at a higher proportion of the carrying capacity, then this value is conservative. Therefore removing a number of animals equal to half of the product of Rmax times and the conservative population estimate should allow will allow growth for populations below MNPL.

- If species have an MNPL that is a different percentage of carrying capacity, or an unbalanced sex ratio, should this ideally be reflected in the denominator?

No.

- The objectives that PBR are intended to achieve, relate to assumptions about how the level of the starting population compares to carrying capacity, a period of recovery after carrying capacity and a population level that would be achieved with respect to carrying capacity at the end of the recovery period.

Irrespective of the starting population size, PBR should cause a population to recover to or decline to NMPL.

Each of these factors may differ from the circumstances associated with a cumulative impact assessment (e.g. it may not be considered useful to assume any, or the same, period of recovery). The PBR objectives were specifically developed to meet the statutory requirements of the US Marine Mammals Protection Act. Do SCOS have any comments to make with respect to the usefulness/appropriateness of managing populations to the PBR's objectives under other statutory frameworks?

In light of the discussions at SCOS it was recommended that a workshop be held to bring together experts on seal population dynamics, population modelling and population management to provide a recommended approach or approaches to managing anthropogenic impacts on UK seals. This should be held as soon as possible in 2016.

C. Implementation of PBR as it is currently used by the Scottish Government under the advice of SCOS

1. The minPop value to be used in PBR is the 20th percentile of the most recent population estimate. It is our understanding that the purpose is to add a conservative measure given the uncertainty associated with the size of many marine mammal populations (especially cetaceans). Firstly, is this approach taken by SCOS or are average values used? Secondly, if average values are used, has the sensitivity been tested, and is there a risk that the assumptions of PBR are violated by using the most robust estimates of population size? Thirdly, if we consider that there is a relatively small spread of uncertainty associated with seal population estimates, is there a risk of the assumptions of PBR being violated even when the 20th percentile is used?

   a. *The method and parameters used to estimate PBRs for both seal species for each management region in Scotland are presented in briefing papers to SCOS each year (see SCOS-BP 14/05 and 15/08). We do not use the 20th percentile of the most recent population estimate. Generally this approach is chosen to ensure that there is approximately a 1/5 chance that the true population is lower.*

   For some areas we had so little data that any estimate of the lower 20th percentile would have been spurious.

   For harbour seals our approach is conservative. We use moult counts as an index of population size. These are direct counts and are not susceptible to double counting, so for any survey year the count represents the minimum possible population size. Several independent studies have produced similar estimates of the proportion of seals hauled out during the moult, consistently around 0.7.

   An alternative approach, closer to that suggested by Wade (1998), would be to rescale these counts into abundance estimates and take the 20th centile of the resulting distributions. Results of a recent telemetry study in Orkney (Lonergan et al., 2013) suggest that would increase the PBRs by between 8%, if the populations are predominantly female, and 37%, if most of the animals are male. If we had more data and could confidently estimate the lower 20th percentile we could use it. This would probably produce a slightly higher PBR value. For example, the estimated confidence intervals from the telemetry scaling factor suggest that the moult count is 72% of the population whereas the lower 20th %ile would be around 89%. For Orkney and North Coast this would increase the PBR from 11 to 13. However, SCOS has, in the past, endorsed the more conservative approach.

For grey seals we have counts of pups that are used to produce population estimates, but they do not represent the distribution of seals throughout most of the year when they are likely to be interacting with human activities. We therefore use the summer haulout counts obtained during the harbour seal moult surveys. To be consistent with the approach taken with the harbour seals, we used telemetry data to establish that 31% of the population were observed during survey windows (Lonergan et al. 2013). The 20th centile of the distribution of multipliers from counts to abundances implied by that data is 2.56 and doubled that estimate to provide an estimate equivalent to the 70% value for harbour seals. Again, this is more conservative than the 20th %ile value and could be increased if needed. However, again SCOS have endorsed this more conservative approach.

b. *No sensitivity analysis is required as the method is more conservative than the PBR standard methodology.*

c. *Not relevant as we do not use the 20th percentile.*

2. F values. What criteria do SCOS use to propose new values? How are the criteria classified to arrive at decisions? How consistent is application of these criteria with other users of PBR? Could the decision making associated with F values be made more transparent and objective?

Setting the F_R value is often regarded as arbitrary. For example it has a minimum value of 0.1 meaning that for a phocid seal population with an assumed R_{max} of 12% p.a. there will be an estimated PBR equal to at least 0.6% of the population estimate, irrespective of the status of that population.

The original PBR methodology leaves the setting of the recovery factor as a subjective choice for managers. Factors such as the amount of information available about the population (and in particular its maximum annual growth rate), recent trends in local abundance, and the connections to neighbouring populations are relevant to setting this. The main factors to be taken into account when deciding on the value were presented to SCOS in 2010 (SCOS-BP 10/08). The suggested values and the rationale for selection for the recovery factor and the other parameters used for each species in each area is presented to SCOS and to MS each year and are presented in SCOS-BP 15/09 for the current estimates..

For information, the recovery factors used in the 2015 calculations (presented in SCOS-BP 14/08) are shown below:

Harbour seals

2) *Shetland, Orkney + North Coast and Eastern Scotland (F_R= 0.1)*
F_R set to minimum because populations are experiencing prolonged declines.

3) *Outer Hebrides (F_R = 0.5)*
Population was undergoing a protracted but gradual decline but the most recent count was close to the pre-decline numbers. The population is only partly closed being close to the relatively much larger population in the Western Scotland region, and the R_{max} parameter is derived from other seal populations.

4) *Western Scotland (F_R = 0.7)*

The population is largely closed, likely to have limited interchange with much smaller adjacent populations. The population is apparently stable and the intrinsic population growth rate is taken from other similar populations.

5) *South West Scotland (F_R = 0.7)*

The population is apparently stable, is closed to the south and the adjacent population to the north is apparently stable. The intrinsic population growth rate is taken from other similar populations.

6) *Moray Firth (F_R= 0.3)*
The recent counts for the Moray Firth show large inter annual fluctuations after a period of gradual decline. The higher counts in some years suggest that this population may be slowly recovery from the declines that occurred in the years around 2000. The neighbouring Orkney and Tay populations are continuing to undergo unexplained rapid and catastrophic declines in abundance. Data available from electronic telemetry tags suggest there is limited movement between these three areas. The PBR was set at 17 for 2013, permits for 16 harbour seals were granted and 3 were shot. We therefore, suggest that the F_R should be again set to a value of 0.3.

Grey seals

All regions (F_R = 1.0)

There has been sustained growth in the numbers of pups born in all areas over the last 30 years, with some now appearing to be at or close to their carrying capacities (Lonergan et al. 2011b). Available telemetry data and the differences in the regional patterns of pup production and summer haulout counts (Lonergan et al. 2011a) also suggest substantial long-distance movements of individuals.

Our setting of the recovery factor is consistent with other users of PBR. For example, for the US set a recovery factor of 0.1 for whales which are listed as "Endangered" due to their low population size caused by whaling. This value was set to ensure there would not be more than a 10% increase in recovery time (Wade 1998).

However, it should be noted that for populations which are currently declining for unknown reasons, the assumptions of PBR are violated.

Other considerations

PBR does not take into account any previous data on population size. Its assumptions are implicit within the model. Another potential yield based framework, used by the IWC, which could be considered is the Revised Management Plan.

References

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

Lonergan, M., Duck, C., Moss, S., Morris, C., & Thompson, D. (2013). Rescaling of aerial survey data with information from small numbers of telemetry tags to estimate the size of a declining harbour seal population. Aquatic Conservation-Marine and Freshwater Ecosystems, 23, 135-44.