

**Scientific Advice on Matters Related to the
Management of Seal Populations: 2021**

**Natural Environment Research Council
Special Committee on Seals**

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 UK 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 2021, thirty-six questions were received from Marine Scotland, Defra and Natural Resources Wales. SCOS's answers to these questions are provided in detail in the main Advice below and summarised here.

Current status of British grey seals (*Halichoerus grypus*)

Grey seal population trends are assessed from counts of pups born during the autumn breeding season, when females congregate on land to give birth. Outside of the breeding season animals may re-distribute themselves, thus, 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 synoptic census of the principal grey seal breeding sites in Orkney, the Inner and Outer Hebrides, the Firth of Forth and sites in eastern England was carried out in 2019. The results, together with a correction for less frequently monitored sites, produce an estimate of 67,850 (approximate 95% CI: 60,500-75,100) pups born throughout the UK in 2019 (Table S1).

The pup production estimates are converted to estimates of total population size (1+ aged population at the start of the breeding season) using a mathematical model. The population model provided an estimate of **157,300** individuals (approximate 95% CI 144,600-169,400). The UK currently holds approximately 35% of the world population and 82% of the European population of grey seals.

Table S1. Grey seal pup production by country (based on 2019 pup production estimates), and total population estimates at the start of the 2020 breeding season. Pup production numbers rounded to nearest 50 pups and total population rounded to nearest 100.

Location	Pup production in 2019	2020 Population estimate
England	11,300	30,700
Wales	2,250	5,200
Scotland	54,050	120,800
Northern Ireland	250	600
Total UK	67,850	157,300

The overall UK pup production increased by <1.5% p.a. between 2016 and 2019. Growth was mainly limited to the North Sea colonies along the east coast of Scotland and England. The combined 2019 pup production estimate in the Inner and Outer Hebrides and Orkney was 3.3% lower than the 2016 estimate, whereas the production for the North Sea colonies increased by 23% over the same period.

Current status of British harbour seals (*Phoca vitulina*)

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. Due to Covid restrictions through summer 2020, no large-scale surveys of

Scottish harbour seal populations were undertaken. However, a complete survey of the East Anglian coast from Donna Nook to Scroby Sands was completed in 2020 and three further surveys of that area were carried out in August 2021.

The best estimate of the UK harbour seal population in 2020 is 43,750 (approximate 95% CI: 35,800-58,300). This is derived by scaling the most recent composite count of 31,500, (based on surveys between 2016 and 2021) by the estimated proportion hauled out during the surveys (0.72 (95% CI: 0.54-0.88)). Overall, the UK population has increased since the late 2000s and is close to the late 1990s level prior to the 2002 Phocine Distemper Virus (PDV) epizootic. However, there are significant differences in the population dynamics between regions.

Until recently, harbour seal populations along the English East coast had generally increased year on year, with those increases punctuated by major declines associated with two major PDV epizootics in 1988 and 2002. However, the 2019 count in the large Southeast England Seal Management Unit (SMU) was approximately 25% lower than the mean of the previous five years. Counts for 2020 and 2021 confirm that the population has declined. The total count for the sites between Donna Nook in Lincolnshire and Scroby Sands in Norfolk, has declined by approximately 38% compared to the mean of the previous five years (2019–2021 mean = 3080; 2014-2018 mean = 4296). This decline is a clear cause for concern and emergency funding for additional surveys has been provided by Defra. A proposed programme of research to investigate the causes of this decline is being developed.

Populations along the east coast of Scotland and in the Northern Isles have generally declined since the early 2000s. The recorded declines have differed in intensity but in all areas the current population size is at least 40% below the pre-2002 level. Populations in North Coast & Orkney SMU and in the Tay and Eden SAC are continuing to decline. Although continued declines are not evident in Shetland or the Moray Firth, there is no indication of recovery.

Populations in western Scotland are either stable or increasing. Counts in the central and northern sections of the large West Scotland SMU and the Southwest Scotland SMU have been increasing since the 1990s and in all other areas they have remained stable. In Northern Ireland, the population appeared to have declined slowly after 2002 but has been apparently stable since 2011.

Table S2. UK harbour seal population estimates based on counts during the moult; rounded to the nearest 100.

Location	Most recent count (2016-2021)	Total Population estimates with 95% CIs
England	3,600 ¹	5,000 (95% CI 4,100-6,700)
Wales	<10 ²	<15
Scotland	26,800 ³	37,200 (95% CI 30,400-49,600)
Northern Ireland	1,000	1,400 (95% CI 1,100-1,900)
Total UK	31,500	43,750 (95% CI 36,000-58,700)

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 the moulting surveys.

Information on the causes of the declines in harbour seals in some Scottish SMUs is required for SCOS to advise on appropriate conservation actions. A wide range of potential causes have been discussed at previous SCOS meetings. Details of the current state of knowledge for each of the potential drivers of decline were discussed and a summary is presented in Table 9. This identifies three ultimate causes as likely drivers of the declines; prey quality and availability,

competition with other marine predators, and predation by killer whales and grey seals. Other potential contributing factors include disease and exposure to toxins from harmful algae. Importantly, several factors have been ruled out or are considered unlikely to be driving the declines, these include fisheries bycatch, deliberate killing, disturbance at haulout sites, entanglement, ingestion of micro-plastics and Persistent Organic Pollutants (POPs).

Seal management

Conservation orders for harbour seals are currently in place for the Western Isles, Northern Isles and down the Scottish East Coast as far as the border. SCOS discussed the requirement for continuation of the Seal Conservation Area designations in Scotland and recommended that orders for the Northern Isles and East Scotland SMUs should remain in place. However, the continued increases in the Outer Hebrides harbour seal population means that the designation could be removed. SCOS also provided advice on scientific criteria for designating and revoking Seal Conservation Area designations.

The Potential Biological Removals (PBR) is a relatively simple metric developed to provide advice on the levels of removals from a marine mammal population that would still allow the population to approach a defined target. PBR estimates for both harbour and grey seals for each seal management unit in Scotland are presented. As there were no changes to the harbour seal or grey seal summer population estimates from Scotland the values are unchanged from last year's recommendations.

The SCOS discussed the merits of altering the existing Seal Management Unit areas and concluded that there was no scientific merit in coalescing units. SCOS recognised the difficulty of managing geographically widespread threats such as bycatch but concluded that these issues can best be addressed by combining the individual SMU populations where and when appropriate.

SCOS also discussed the need to designate additional Sites of Special Scientific Interest (SSSIs) for seals and provided advice to Defra and Natural England on the most important seal sites in each SMU.

Seal Bycatch

The most recent estimated bycatch of seals in UK fisheries was in 2019. The total estimate was 488 animals (95% CI: 375-872). This is almost exclusively in gill net fisheries and 81% of the bycatch occurs in the southwest, in ICES area VII.

Statistical analyses have not found any strong seasonal signal to seal bycatch rate. All recorded species IDs in the southwest are of grey seals, as there are few harbour seals west of the Solent area. Most bycaught animals are small. SCOS recommend that effort should be directed towards identifying the species and if possible, the sex and age structure, and genetic information from the bycaught seals. This could be achieved by obtaining photographs of the animals and taking a skin sample.

Estimated bycatch levels in the Western Channel and Celtic Sea exceed the PBR for the combined grey seal populations of SW England, Wales, and Ireland. Despite the bycatch, grey seal populations in Wales and Ireland are probably stable, suggesting that bycaught seals include animals that may have originated from the large, adjacent breeding populations in western Scotland.

Interactions with Fisheries

SCOS discussed a range of topics related to seal interactions with fisheries, aquaculture, and the wider marine environment.

Interactions with Marine Renewable Energy developments

SCOS discussed the current state of knowledge on seal interactions with marine renewable energy devices, including recent issues of seal entrapment in underwater structures. An update on interactions between seals and marine renewables is presented along with a review of emerging technologies and methodologies that may be useful for investigating the behavioural and physiological consequences of interactions.

Threats to UK seals

SCOS discussed the available information on the likely impacts of climate change on UK seal populations and an updated review of likely impacts is presented together with a review of the current and potential future threats to UK seal populations. This includes available information on effects of macro- and micro-plastic pollution, entanglement, pollutants including POPs, plasticizers and pharmaceuticals, harmful algae, fisheries interactions, disturbance, infectious diseases, and predation risk.

There was considerable discussion on the likely effects of disturbance. SCOS recognise the increasing public concern over disturbance, but conclude that, while disturbance can clearly affect individual animal welfare, there is no evidence that disturbance at haulout sites is currently a concern at the population level. An extensive review of the available information on disturbance of seals is presented.

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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 2021. It begins with some general information on British seals, gives information on their current status, and addresses specific questions raised by Marine Scotland (MS) and the Department of the Environment, Food and Rural Affairs (Defra) and Natural Resources Wales (NRW).

Briefing papers which provide additional scientific background for the advice are appended to the main report (Annex III).

SMRU's long-term funding has recently seen a substantial reduction. This will have an impact on the frequency and types of advice that SMRU will be able to deliver and research activities are being reprioritised as necessary.

General information on British seals

Two species of seal live and breed in UK waters: grey seals (*Halichoerus grypus*) and harbour (also called as 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 that occasionally occur in UK coastal waters, include ringed seals (*Pusa hispida*), harp seals (*Pagophilus groenlandica*), bearded seals (*Erignathus barbatus*), hooded seals (*Cystophora cristata*) and walrus (*Odobenus rosmarus*), 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 300 kg while the females weigh around 150-200 kg. 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 seabed at depths of up to 100 m, although they are 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 for an adult 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 100 km 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 100 km 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 haulout sites in the Outer Hebrides have been recorded as well as movements from sites in Wales and NW France, to the Inner Hebrides.

Globally there are three centres of grey seal abundance: one in eastern Canada and the north-east USA, a second around the coast of the UK, especially in Scottish coastal waters, and a third, smaller group in the Baltic Sea. All populations 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. In the UK and Canadian populations, there are clear indications of a slowing down in population growth in recent years.

Approximately 36% of the world's grey seals breed in the UK and 80% of these breed at colonies in Scotland with the main concentrations in the Outer Hebrides and in Orkney. There are large and rapidly growing breeding colonies on the east coast of Scotland and England with fastest growth in the central and southern North Sea. There are also breeding colonies in Shetland, on the north and northeast coasts of mainland Britain and smaller populations in Wales and southwest England.

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 the past, grey seals have been 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 became habituated to human disturbance in the 1990s and that tolerance of human activity has spread as the population has grown in the southern North Sea colonies. Several mainland colonies now receive tens of thousands of visitors each breeding season with no apparent impact on the number of 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 October; in north and west Scotland pupping occurs mainly between September and late November; and in 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. They 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 adult seal per day depending on the prey species.

Harbour seals come ashore in sheltered waters, often 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 32% of European harbour seals are found in the UK. The proportion has declined from approximately 40% in 2002 due to the more rapid recovery and higher sustained rates of increase in the Wadden Sea population. 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, the Firths of Forth and Tay, and the Moray Firth. Scotland holds approximately 85% of the UK harbour seal population, with 12% in England and 3% 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) epizootic. A second epizootic 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 immediate recovery from the 2002 epizootic and continued to decline until 2006. The counts increased rapidly from 2006 to 2012 but appeared to have remained relatively constant since until a decline began in 2019. In contrast, the adjacent European colonies in the Wadden Sea experienced continuous rapid growth after the epizootic, but again, the counts over the last 5 years suggest that the rate of increase has slowed dramatically.

Major declines have now been documented in several harbour seal populations around Scotland, with declines since the late 1990s of 85% in Orkney, 47% in Shetland and 95% in the Firth of Tay. However, the pattern of declines is not universal. The Moray Firth count apparently declined by 50% before 2005 and has fluctuated since, showing no significant trend since 2003. The Outer Hebrides apparently declined by 35% between 1996 and 2008 but has shown no significant trend over the entire time series. The West Scotland population is now the largest population in the UK and in 2018 was approximately twice the size it was in the mid-1990s. The recorded declines are not thought to have been linked to the 2002 PDV epizootic as there was very little recorded mortality of harbour seals in Scotland in 2002.

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. 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 epizootics 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.

In Scotland, the Conservation of Seals Act was superseded by the Marine (Scotland) Act 2010. As a result, the conservation orders in Scotland have been superseded by the designation of seal conservation areas under the provisions of the Marine (Scotland) Act 2010. Conservation areas have been established for the Northern Isles, the Outer Hebrides and the East coast of Scotland. In general, seals in Scotland are afforded protection under Section 6 of the Act which prohibits the killing or taking of seals except under licence. In the original version of the Act, licences could be granted for ten specific reasons, including to conserve natural habitats, for scientific, research or educational purposes, to protect the health and welfare of farmed fish and to prevent serious damage to fisheries or fish farms' aquaculture activities. Recent legislative changes in Scotland, via the Animals and Wildlife (Penalties, Protections and Powers) (Scotland) Act 2020, have amended the Marine (Scotland) Act 2010 to remove the provision to grant licences authorising the killing or taking of seals to protect the health and welfare of farmed fish, and to prevent serious damage to fisheries or fish farms.

Similar legislative changes in England and Wales, and Northern Ireland via Schedule 9 of the Fisheries Act 2020, amends the Conservation of Seals Act 1970 and the Wildlife (Northern

Ireland) Order 1985, prohibiting the intentional or reckless killing, injuring or taking of seals and removing the provision to grant licences for the purposes of protection, promotion or development of commercial fisheries or aquaculture activities. These changes were enacted to ensure compliance with the US Marine Mammal Protection Act Import Provision Rule.

In Scotland it also is now an offence to 'intentionally or recklessly harass' seals at designated haulout sites. NERC (through SMRU) provides advice on all licence applications and haulout designations.

In Northern Ireland It is an offence to intentionally, or recklessly disturb seals at any haulout site under Article 10 of Wildlife and Natural Environment Act (Northern Ireland) 2011.

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 six-yearly SAC reporting cycle requires formal status assessments for these sites. These were last completed in 2019.

Questions

Seal Populations

1. What are the latest estimates of the number of seals in UK waters?	MS Q1 Defra Q1 NRW Q1
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Current status of British grey seals

The total UK grey seal population of at the start of the 2020 breeding season (before pups are born) is estimated at 157,300 (approximate 95% CI 144,600-169,400). The estimate is based on the most recent pup production estimates in 2019 for aerial surveyed colonies in Orkney, the Inner and Outer Hebrides and the Firth of Forth, and from ground surveyed colonies and the colonies on the east coast of England.

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. Outside of the breeding season animals may re-distribute themselves, thus, 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 synoptic census of the principal grey seal breeding sites in Orkney, the Inner and Outer Hebrides, the Firth of Forth and sites in eastern England was carried out in 2019. The results, together with a correction for less frequently monitored sites, produce an estimate of 67,850 (approximate 95% CI 60,500-75,100) pups born throughout the UK (Tables 1 & 2) in 2019.

The regional pup production estimates for 1984 to 2019 for the Inner Hebrides, Outer Hebrides and Orkney and the North Sea colonies were converted to estimates of total population size (1+ aged population, referred to as 'adult population') at the start of the 2020 breeding season, using a mathematical model of British grey seal population dynamics. The population estimate is then corrected to account for pup production at less frequently monitored colonies. The stages in the process, the fitting of the pup production model and the observed trends are described below and presented in SCOS BPs 21/05, Russell *et al.* (2019) and Thomas *et al.* (2019).

The overall UK pup production increased by <1.5% p.a. between 2016 and 2019. Growth was mainly limited to the North Sea colonies along the east coast of Scotland and England. The combined 2019 pup production estimate in the Inner and Outer Hebrides and Orkney was 3.3% lower than the 2016 estimate, whereas the production for the North Sea colonies increased by 23% over the same period.

Pup Production

The pup production estimates from 2019 aerial surveys ground counts combined with estimates from less frequently aerially surveyed colonies, indicated that approximately 67,850 (approximate 95% CI 50,250-85,400) grey seal pups were born in 2019 across all UK colonies, including the Isle of Man.

Major colonies in Scotland are now surveyed biennially (see SCOS-BP 14/01). Aerial surveys to estimate grey seal pup production were carried out in Scotland in 2019, using a digital camera system (SCOS-BP 21/01). Counts then go into a model to estimate pup production on the biennially monitored colonies around Scotland. Pup- production estimates for colonies on the East coast of England were obtained from ground counts in 2019.

Table 1. Grey seal pup production by country (based on 2019 pup production estimates), and total population estimates at the start of the 2020 breeding season. Numbers rounded to nearest 50 pups.

Location	Pup production in 2019	2020 Population estimate***
England**	11,300*	30,700
Wales	2,250*	5,200
Scotland	54,050*	120,800
Northern Ireland	250*	600
Total UK	67,850	157,300

*Includes estimated production for less frequently monitored colonies, see SCOS-BP 21/01 and 20/04 for details. Populations associated with these estimates were based on the region-specific ratios of pups to total population for the regularly monitored sites, while the UK-wide average ratio was used for the less frequently monitored sites.

** Isle of Man count included with England

*** Populations derived from the 2019 pup production estimates and represents the total population alive on first day of 2020 breeding season. Confidence intervals are not provided as the national populations have been derived from regional population estimates scaled by proportions of that region's pup production in each country. Estimates were rounded to nearest 100 seals.

Regional pup production estimates in 2019 at biennially air surveyed and annually ground counted colonies (rounded to nearest 50 pups) were: 4,450 (approximate¹ 95% CI 3,300-5,600) in the Inner Hebrides, 16,100 (95% CI 12,000-20,300) in the Outer Hebrides, 22,150 (95% CI 16,400-27,900) in Orkney and 18,000 (95% CI 13,300-22,600) at the North Sea colonies (including Isle of May, Fast Castle, Farne Islands, Donna Nook, Blakeney Point and Horsey/Winterton) (SCOS-BP 21/01).

An additional 7,200 pups were estimated to have been born in Wales and at less frequently surveyed colonies in Southwest England, Northern Ireland, Shetland, and at scattered locations throughout Scotland (SCOS-BP 20/04; 21/01).

Trends in pup production

There has been a continual increase in the total UK pup production since regular surveys began in the 1960s (Figure 1) (see SCOS-BP 18/01 & Russell *et al.* (2019) for details). This increase has continued over the last survey interval, but the overall increase is small, <1.4% p.a. and is mainly limited to the North Sea colonies along the east coast of Scotland and England. The combined 2019 pup production estimate in the Inner and Outer Hebrides and Orkney was 3.3% lower than the 2016 estimate (equivalent to a 1% p.a. decrease), whereas the production for the North Sea colonies increased by 23% over the same period (equivalent to a 7% p.a. increase) (Table 2).

¹ Approximate CIs based on the overall CI of the total pup production estimated by the population dynamics model: see SCOS-BP 18/03. This will likely overestimate the CI for individual regions

Interpretation of the trends in pup production are complicated by a transition to a digital camera system and reduced survey altitude between 2010 and 2012. This affected both the efficiency of counting and the stage classification of pup images. In all three regions where the pup production is estimated entirely from aerial survey counts there was an apparent step change coincident with this transition. For logistical and technical reasons, it has not been possible to directly cross-calibrate the two methods. However, as the new time series extends it becomes easier to estimate the magnitude and nature of these changes, and therefore to determine appropriate correction factors to be applied to obtain consistent time series.

To make it easier to compare population estimates during the August surveys and the pup production data it is suggested that the previous naming convention for grey seal population model regions should be altered to match the Seal Management Units (SMUs) in which they are found: the Inner Hebrides is equivalent to West Scotland SMU, Outer Hebrides is equivalent to Western Isles SMU, Orkney is equivalent to the North Coast and Orkney SMU and Firth of Forth colonies are equivalent to Southeast Scotland SMU. For the rest of this section the SMU names will be used.

Russell *et al.* (SCOS-BP 21/03) fitted a series of models to the pup production estimates for each SMU. For Scottish SMUs where the pup productions were estimated from SMRU aerial surveys (all except Shetland and Moray Firth), the model fitted a step increase in pup abundance between 2010 (the last film survey) and 2012 (the first digital survey) to account for any artificial increase in pup counts that resulted from the change in aerial survey method. To maximise the data available to fit this jump, all applicable SMUs were modelled within a single GAM (number of knots limited to $k=5$), allowing a different temporal trend for each SMU but a single adjustment for the change in survey methods. Once fitted, the single adjustment allows the trends in each SMU to be examined excluding this jump.

The final model estimating trends in grey seal pup production for aerially surveyed SMUs included an estimated 27 % jump (95% CI: 16.7 – 37.5) in pup production associated with the change from film to digital (delta AIC of -30 compared to a model without the jump).

A detailed description of the trends in pup production up to 2010, at regional and colony levels was presented in Russell *et al.* (2019) and summarised in SCOS 2020. The recent analysis extends the fitted trends through the change in methodology in 2012, allowing examination of trends through the entire time series including the past decade.

Figure numbers here refer to figures in SCOS-BP 21/03, where a full description of the model selection process and the resulting trends can be found. Briefly, pup production had levelled off in West Scotland (early to mid - 1990s; Fig 2i c SCOS-BP 21/03) and Western Isles (mid 1990s; Fig 3c SCOS-BP 21/03) (Russell *et al.*, 2019), but the 2016 and 2019 estimates were higher than the first two digital survey estimates (2012 and 2014). For the Western Isles this resulted in a slight recent increase in the mean predicted trend. This apparent increase is reflected in the Monach Islands SAC which accounts for >75% of the SMU pup production. In contrast, pup production in North Rona is continuing to decline.

In the North Coast & Orkney SMU (Fig 4c SCOS-BP 21/03), pup production has remained stable since around 2000. The Faray & Holm of Faray SAC estimates indicate that the colony may be in decline. A declining trend was fitted for Shetland (Fig 5c SCOS-BP 21/03). However, the time-series comprised a subset of colonies and was based on peak counts (which are sensitive to effort, i.e., number and timing of counts) and thus there are doubts as to how robustly these trends represent Shetland as a whole.

The Moray Firth SMU (Fig 6c SCOS-BP 21/03) estimates show pup production is increasing though it should be noted that there is a limited temporal extent to the data and pup production within this SMU is difficult to accurately estimate.

The East Scotland SMU (Fig 7c SCOS-BP 21/03) is continuing to increase rapidly (mean estimate of c. 28% between 2014 and 2019), but the two SACs that represent the vast majority of production in the SMU show differing patterns in abundance. The Isle of May SAC, which essentially held all of the SMUs pup production until the mid-1990s appears to be stable or potentially declining. In contrast, the Fast Castle colony within the Berwickshire & North Northumberland Coast SAC is showing rapidly increasing pup production.

Pup production in Northeast England, which is entirely encompassed by the Farne Islands component of the Berwickshire & North Northumberland Coast SAC, is also increasing rapidly (mean estimated increase of 53% between 2014 and 2019).

Pup production within the Southeast England SMU is continuing to increase exponentially (mean estimate c. 75% between 2014 and 2019,) but this is in large part due to increases in Blakeney Point and Horsey. The increase at Donna Nook (Humber Estuary SAC) which, up until c. 2000 accounted for the SMUs entire pup production, is now slowing.

Monitoring of grey seals in Wales is split into two areas: North Wales (Dee Estuary- Aberystwyth) and West Wales (Aberystwyth - Caldey Island). Details of the available data, data sources and derivations of pup production estimates are given in SCOS-BP 20/04.

There are no or very few grey seals in south Wales (Caldey Island – Bristol Channel). Intensive monitoring of pup production is primarily focussed at three sites: Bardsey Island, parts of Ramsey Island, and Skomer Marine Conservation Area. Other areas have been monitored more sporadically, and within a season, less intensively. North Wales wide surveys were conducted in 2001, 2002 and 2017. The latest pup production estimate for 2017 was 216. West Wales wide surveys were conducted in 1992, 1993, and 1994.

It is not possible to estimate trends in pup production on a SMU scale in Wales. Pup production at Ramsey Island indicator sites has been variable but shown little trend. There is an upward trend in pup production at Skomer MCZ, though the trend is variable. The pup production estimate for Skomer and the adjacent Marloes peninsula increased slightly from 408 in 2019 to 422 in 2020 (Wilkie & Zbijewska, 2020).

Scalars between pup production in West Wales and indicator sites (in mainland north Pembrokeshire sites, Ramsey Island, and Skomer MCZ), in 1993 and 1994, were used to generate a total pup production estimate for West Wales. It should be noted, this was generated using the most recent available estimates for indicator sites, rather than predictions from fitted trends at these sites. Combined with the most recent estimate of North Wales, and rounding up to the nearest 50, this results in a pup production estimate of c. 2,250. Almost half of the SMU estimate of pup production is from sites not surveyed since the early 1990s.

To produce a robust estimate of pup production, scalars between indicator sites and irregularly monitored colonies need to be updated. This is particularly important when there are multiple habitat types (e.g. caves, open beaches) in an area. Cryptic sites (such as caves, small coves) can often support much smaller colonies and thus their trends, especially in the longer term, may differ from more open sites that are also easier to monitor. Indeed, for North Wales, Robinson *et al.* (NRW unpublished) found that a much lower proportion of pup production was at cryptic sites than found previously (Stringell *et al.*, 2014).

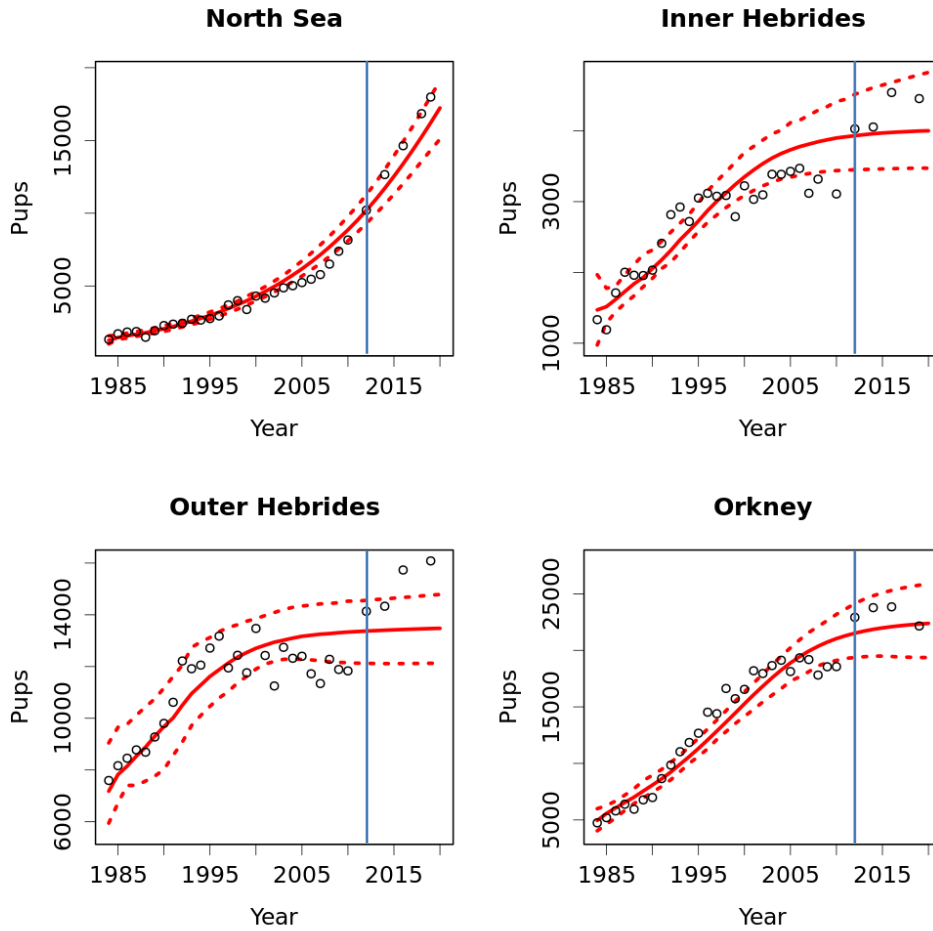


Figure 1. Posterior mean estimates of pup production (solid lines) and 95% Confidence Intervals (dashed lines) from the model of grey seal population dynamics, fit to pup production estimates for regularly monitored colonies (SCOS-BP 18/01 and Table 2 below), from 1984-2016 (circles) for colonies in Orkney and the Inner and Outer Hebrides, and for 1984-2018 for the colonies in the North Sea, and two independent total population estimates from 2008 and 2014 (see text for details). The vertical blue line at 2012 indicates the change to a new digital camera system.

Table 2. Grey seal pup production estimates from 2019 aerial surveys for the regularly monitored colonies in Orkney and the Inner and Outer Hebrides and Firth of Forth colonies and ground counts for English North Sea colonies, combined with most recent data from less regularly monitored colonies (see main text and SCOS-BP 21/01 and 20/04 for details). These estimates are compared with similar production estimates from 2016.

Location	Pup production in 2019	Pup production in 2016	Average annual change 2016 to 2019
Inner Hebrides	4,455	4,541	- 0.6%
Outer Hebrides	16,083	15,732	+ 0.7%
Orkney	22,153	23,849	- 2.4%
Firth of Forth	7,261	6,426	+ 4.2%
Regularly monitored Scottish colonies	49,952	50,548	- 0.4%
Other Scottish colonies ¹ (incl. N & NE mainland & Shetland)	4,112	4,193	- 0.6%
Total Scotland	54,064	54,741	- 0.4%
Farne Islands	2,823	2,295	+ 7.1%
Donna Nook, Blakeney, Horsey	7,902	5,918	+10.1%
Annually monitored colonies in eastern England	10,725	8,213	+ 9.3%
SW England ^{1,2}	450	250	
Small sites in E and NW England ^{1,3}	50	50	
Total England	11,225	8,513	+ 9.7%
Wales ^{1,4}	2,250	1,650	
Northern Ireland ¹	250	150	
Total UK	67,789	65,054	+ 1.4%
Isle of Man	69	84	

¹ Includes estimated production for colonies that are rarely monitored from different years

² Includes estimates for Scilly Isles, Lundy, various sites in Devon & Cornwall

³ Includes Coquet Island, Ravenscar, Scroby Sands, South Walney

⁴ Multiplier derived from indicator colonies surveyed in 2004 and 2005 and applied to other colonies last monitored in 1994 (SCOS-BP 20/04)

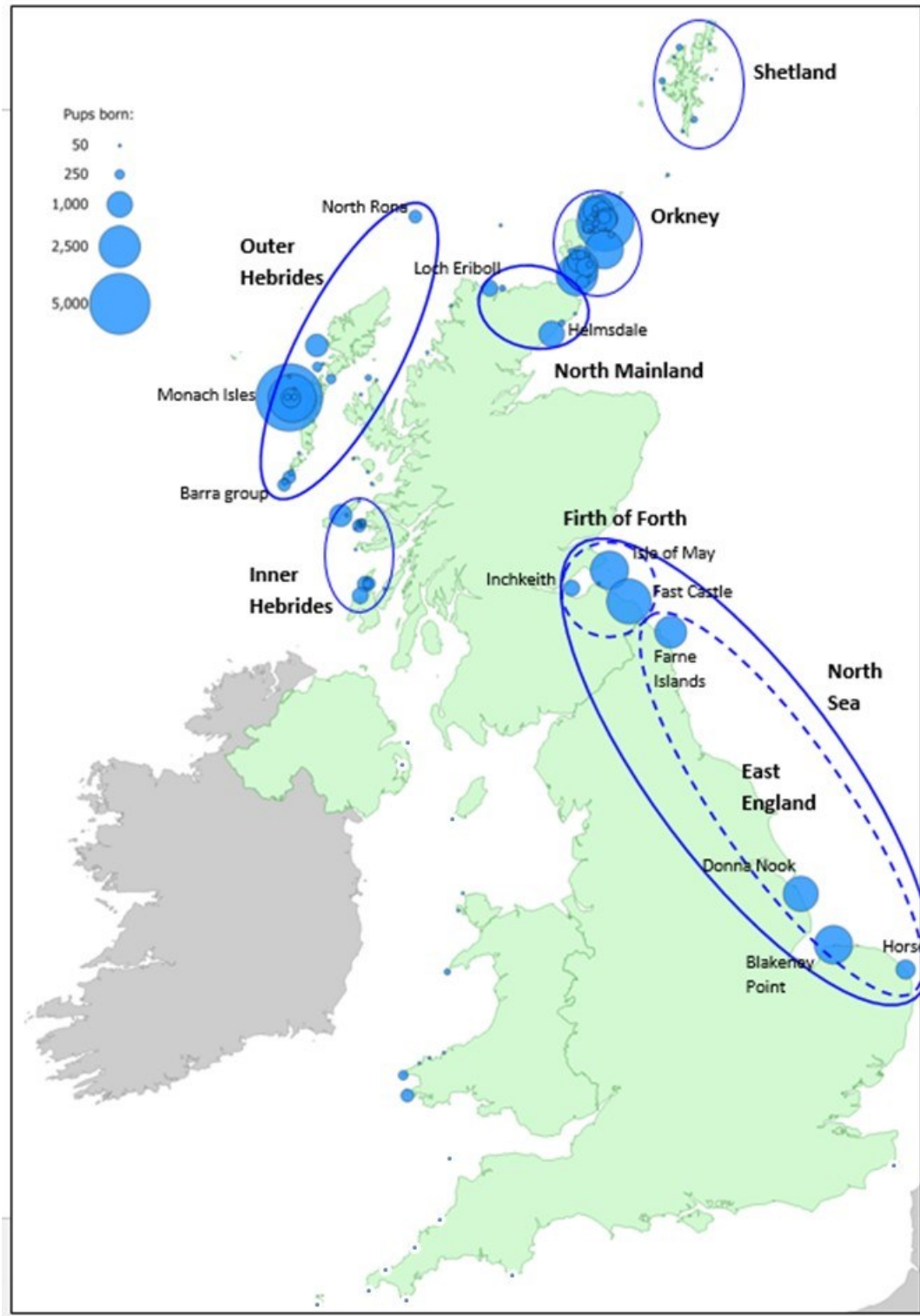


Figure 2. Distribution and estimated pup production of the main grey seal breeding colonies. Solid blue ovals indicate groups of regularly monitored colonies within each region, dashed ovals show sites in the north that are routinely monitored by aerial survey and those in the south that are routinely monitored by ground counts.

Population size

The raw data for estimating the total grey seal population are currently the region specific (Inner Hebrides, Outer Hebrides, Orkney and North Sea) pup production estimates derived from aerial surveys and ground counts at all major colonies around Scotland and eastern England.

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.

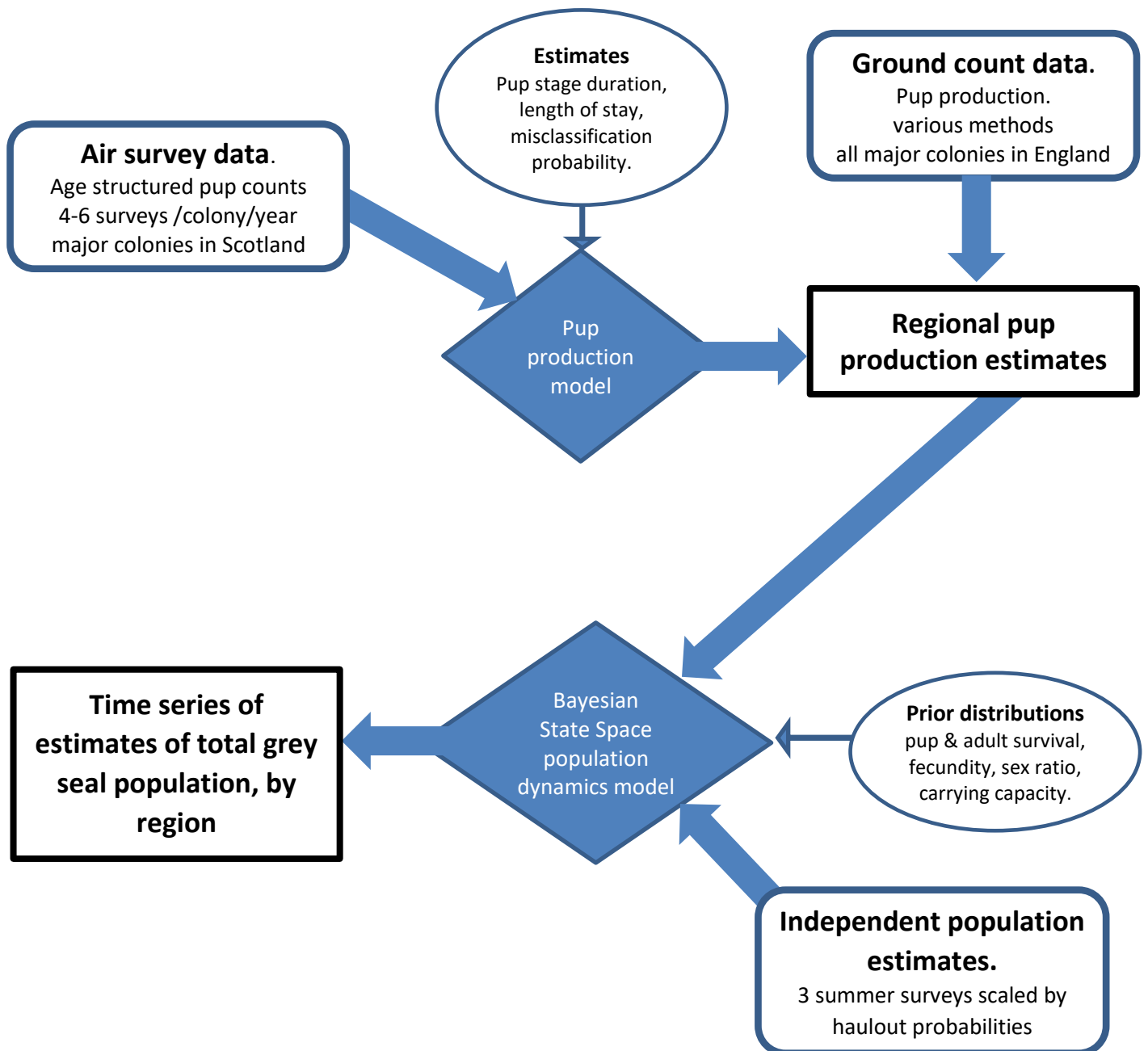


Figure 3. Schematic diagram of steps involved in estimating total grey seal population size from pup counts.

Using appropriate estimates of fecundity rates, both pup and non-pup survival rates and sex ratio 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 demographic parameters and population size.

The time series of pup production estimates from the regularly monitored colonies indicate that from at least 1984 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-BPs 09/02, 10/02 and 11/02) has resulted in reduced population growth rates in the Northern and Western Isles.

To estimate the population size, a Bayesian state-space model of British grey seal population dynamics was fitted to the pup production data. Initially, alternative models with density dependence acting through either fecundity or pup survival were tested, but results indicated that the time series of pup production estimates did not contain sufficient information to quantify the relative contributions of these factors (SCOS-BPs 06/07, 09/02). In 2010 and 2011 we incorporated additional information in the form of an independent estimate of population size. This was based on counts of the numbers of grey seals hauled out during the summer and information on their haulout behaviour, which provides an estimate of the proportion of the population available to be counted during the aerial surveys (SCOS-BP 10/04 and 11/06). Between 2007 and 2009, 26,699 grey seals were counted during harbour seal moult surveys across the UK (excluding southwest UK). Using telemetry data, it was estimated that 31% (95% CIs: 15 - 50%) of the population was hauled out during the survey window and thus available to count (Lonergan *et al.*, 2011a; b). Assuming 4% of the population were in southwest UK, this led to a UK independent population estimate in 2008 of 91,800 (95% CI: 78,400 - 109,900).

Inclusion of the first independent estimate in 2008 allowed us to reject the models that assumed density dependent effects operated through fecundity and all estimates were therefore based on a model incorporating density dependent pup survival. However, SCOS felt that the independent estimate appeared low relative to the pup production and its inclusion forced the model to select extremely low values of pup survival, high values of adult female survival and a heavily skewed sex ratio, with few surviving male seals.

Additional independent estimates were obtained in 2014 (SCOS-BP 16/04) and 2017 (SCOS-BP 21/02). A new analysis of haulout patterns including data from an additional 60 new deployments of improved GPS/GSM tags on grey seals is presented in SCOS-BP 21/02 and SCOS-BP 21/03.

The revised analyses resulted in an estimate of the proportion of the population hauled out during the survey window of 25.15% (95% CI: 21.45-29.07%) compared to 23.9% (95% CI: 19.2-28.6%) used previously. As per the previous analyses there was no effect of region, length of individual (regarded as a proxy for age), sex or time of day on the conversion factor/ scalar. However, observed count variability appears higher than suggested by the estimated variance of haulout probabilities. This may indicate a lack of independence in the haulout patterns between individuals. If true, this would increase the confidence intervals on the scalar.

The updated scalar resulted in slightly reduced mean population estimates for 2008 (96,028 compared to 101,196) and 2014 (138,437 compared to 145,889; Russell *et al.*, 2016; Table 2). The total count and population estimate for 2017 was 40,347 and 160,425, respectively, representing a 16% increase compared to 2014.

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 the independent estimates of total population size from the summer surveys. Work on updating these priors is continuing and an annual update is presented in SCOS-BP 21/04.

In 2014, SCOS adopted a set of revised priors, including a different prior on adult sex ratio, to generate the grey seal population estimates (SCOS-BP 14/02). 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 what was considered to be 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 upper bound of the adult survival prior was increased slightly to 0.98 in line with revised survival estimates.

The model and fitting methods used here are the same as those employed in recent years and are described in detail in Thomas *et al.* (2019 and SCOS-BP 21/05); the prior distributions on model parameters are the same as those used for the last two years (see SCOS-BP 21/04 & 21/05 for details). The data are a time series of regional pup production estimates for the regularly monitored colonies in the Inner and Outer Hebrides, Orkney, and the North Sea, for the years 1984-2016, 2018 (North Sea region only) and 2019, and three independent estimates of total population size (2008, 2014 and 2017).

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. The same model and prior distributions for demographic rates were used, including a prior on sex ratio and a constraint on adult survival to the range 0.80-0.98. The revised prior on North Sea carrying capacity of 20,000 was used as the population produced over 14,000 pups but continues to increase rapidly, indicating that it was not close to carrying capacity. Carrying capacity is taken to mean the average population size below which numbers tend to increase and above which numbers tend to decrease due to resource limitations.

Grey seal population estimate

From the standard model run, the estimated adult class population size (here taken to mean the total 1+ age population) in the regularly monitored colonies at the start of the 2020 breeding season was 140,700 (95% CI: 129,300-153,500). This estimate is produced by a model incorporating density dependent pup survival, using the revised priors, and including the independent estimates for 2008, 2014 and 2017 (details of this analysis and posterior estimates of the demographic parameters are given in SCOS-BP 21/05).

A comprehensive survey of data available from the less frequently monitored colonies was presented in SCOS-BP 18/01 and revised estimates for Southwest England, Wales, Northwest England, and Northern Ireland are presented in SCOS-BP 20/04 and presented in Table 1. Total pup production at these sites was estimated to be approximately 7,150. The total population associated with these sites was then estimated using the average ratio of pup production to population size estimate for all annually monitored sites in 2019. Approximate confidence intervals were estimated by assuming that they were proportionally similar to the population dynamics model confidence intervals for the standard model run. This produced a population estimate for these sites of 16,600 (approximate 95% CI: 15,300 to 17,900). This will undoubtedly under-estimate the uncertainty in the estimate, but it represents a relatively small proportion (12%) of the total.

Combining the annually monitored sites with the estimate for the less regularly monitored sites gives an estimated 2020 UK grey seal population of 157,300 (approximate 95% CI: 146,000-169,400).

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 senescence could be causing a mismatch between the estimates from the model and the pup production data.

In 2018, the mode of the posterior distribution on adult survival from the population dynamics model was close to the upper bound 0.97 of the prior. In addition, mark-recapture-based estimates of adult female survival at Sable Island in Canada were higher than this upper bound (0.976, SE 0.001) (den Heyer & Bowen, 2017). Hence, the prior for adult female survival was increased to 0.98 for last and this year's model runs.

Thomas *et al.* (2019) discussed how sensitive the estimate of total population size may be to the parameter priors, and concluded that fecundity and adult male:female ratio are two parameters that strongly affect total population size but for which the prior specification is particularly influential. Hence a renewed focus on priors for these parameters may be appropriate.

In addition, the model assumes a fixed CV for the pup production estimates and obtains this value from an initial model run. Ideally, region-level estimates of pup production variance would be produced as part of fitting the pup production model to the aerial pup count data. These developments are ongoing. One factor that will require consideration is how to incorporate uncertainty in the ground counts made at some North Sea colonies. A set of four aerial surveys were carried out for each of these ground-counted North Sea colonies. Counts and comparison with the 2018 ground counts are ongoing and will be presented to SCOS 2021. A revised pup production model is being developed with the aim of re-estimating pup production for the entire count data set.

Population trends

Model selection criteria suggest that density dependence is acting mainly on pup survival (see SCOS-BP 09/02). Fitting to the three independent population estimates confirms that the density dependent pup survival model is a better fit than a model incorporating density dependent fecundity. A corollary of this density dependent pup survival is that the overall population should closely track the pup production estimates when experiencing density dependent control, as well as during exponential growth. This is borne out by the similarities in the fitted population model trends (Figure 1) and the pup production trends (SCOS-BP 21/03). The population trend in each region/SMU will therefore follow the trends in pup production estimates described in detail above and in SCOS-BP 21/03.

The factors influencing the dynamics of the different populations are not well known. The population dynamics model currently assumes that demographic rates are either fixed or respond to density dependent factors related simply to population size. However, it is likely that demographic parameters will be subject to environmental factors. For example, female fecundity is likely to be influenced by environmental factors regulating prey availability and seals' ability to gain fat reserves before breeding. A preliminary investigation was carried out of the relationship between fluctuations in pup production around the modelled trend and the NAO index from the previous winter, and also lagged by a further year (SCOS-BP 20/01). No association was found between NAO

and variation in pup production. However, NAO changes may not be a sensitive indicator of changes in seal prey and hence seal fecundity. Further investigations of this and other potential indices of environmental conditions should be pursued once revised estimates of pup production are available.

UK grey seal population in a world context

The UK grey seal population represents approximately 34% of the world population on the basis of pup production estimates. The other major populations in the Baltic and the western Atlantic are also increasing (Table 3).

Table 3. Relative sizes and status of grey seal populations using pup production as an index of population size.

Region	Pup Production	Year	Possible population trend
UK	67,800	2019	Increasing
Ireland	2,100	2012 ¹	Increasing
Wadden Sea	1,750	2020 ²	Increasing
France	70	2019 ⁴	increasing
Norway	700	2015-20 ³	Possible decline
Russia	800	1994	Unknown
Iceland	1,450	2017 ⁸	Declining
Baltic	8,000	2019 ^{4,5}	Increasing
Europe excluding UK	14,870		unknown
Canada - Scotian shelf & Gulf of Maine	92,300	2016 ⁶	Increasing
Canada - Gulf St Lawrence	9,800	2016 ⁶	Increasing
USA	6,500	2019 ⁷	Increasing
WORLD TOTAL	191,270		Increasing

¹Ó 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 Galatius A., Brasseur S., Carius F., Diederichs B., Jeß A., Körber P., Schop J., Siebert U., Teilmann J., Bie Thøstesen C. & Kløpper S. (2021,) EG-Seals - grey seal surveys in the Wadden Sea and Helgoland in 2019-2020. Common Wadden Sea Secretariat, Wilhelmshaven, Germany. ³Nilssen, K.T. and Bjørge, A. 2017b. Status for kystsel. Anbefaling av jaktkvoter for 2018 [Status for coastal seals. Recommendation for harvest quotas for 2018]. Document to the Norwegian Marine Mammal Scientific Advisory Board, October 2017. 9 pp. ⁴ ICES. 2021. Working Group on Marine Mammal Ecology (WGME). ICES Scientific Reports. 3:19. 155 pp. <http://doi.org/10.17895/ices.pub.8141>.

⁵Baltic pup production estimate based on mark recapture estimate of total population size (38,000) and an assumed multiplier of 4.7 HELCOM fact sheets (www.HELCOM.fi) & http://www.rktl.fi/english/news/baltic_grey_seal.html

⁶ den Heyer, C.E., Bowen, W.D., Dale, J., Gosselin, J-F., Hammill, M.O., Johnston, D.W., Lang, S.L., Murray, K.T., Stenson, G.B. & Wood, S.A. (2020) Contrasting trends in gray seal (*Halichoerus grypus*) pup production throughout the increasing northwest Atlantic metapopulation. Marine Mammal Science, DOI: 10.1111/mms.12773. ⁷ Wood *et al.* 2020 Journal of Mammalogy, 101(1):121–128, 2020, DOI:10.1093/jmammal/gyz184. ⁸ Granquist, S.M. and Hauksson, E. 2019. Aerial census of the Icelandic grey seal (*Halichoerus grypus*) population in 2017: Pup production, population estimate, trends and current status. Marine and Freshwater Research Institution, HV 2019-02. Reykjavik 2019. 19 pp. <https://www.hafogvatn.is/static/research/files/1549015805-hv2019-02pdf>.

Table 3 shows the relative sizes and status of grey seal populations throughout their range. Pup production estimates are used as indices of population size because they represent a directly observable/countable section of the population, the largest populations are monitored by means of pup production surveys and because of the uncertainty in overall population estimates in

some cases. Total population estimates are derived from population dynamics models fitted to time series of pup productions in the two largest populations, i.e., Canada and the UK (Hammill *et al.*, 2017; Thomas *et al.*, 2011; 2019). However, although the models are similar, the published total population estimates are derived differently: in the Canadian population, total population refers to the number of 1+ age class animals alive at the end of the breeding season plus the total pup production for that year; in the UK, the total population is given as the total number of seals alive at the start of the breeding season, i.e., does not include any of that year's pup production. The published estimates therefore differ by around 20 to 30% for the same pup production estimate. It is not clear how the total population is derived in several populations. To avoid confusion, only the pup production values are presented here.

Current status of British harbour seals

Due to Covid restrictions through summer 2020 no large-scale surveys of Scottish harbour seal populations were undertaken. One survey of the Firth of Tay and Eden SAC was carried out in August 2020. In England a survey of the East Anglian coast from Donna Nook to Scroby Sands was completed in 2020. In 2020, the Firth of Tay and Eden estuary count was the same as the 2019 count and the East Anglian count was approximately 8% higher than the 2019 count. A series of three surveys of the coast from Donna Nook to Scroby Sands (by SMRU) and a single survey of the Greater Thames estuary (by the Zoological Society of London (ZSL)) were carried out in 2021 in response to observed declines in 2019 and 2020.

The best estimate of the UK harbour seal population in 2020 is 43,750 (approximate 95% CI: 35,800-58,300). This is derived by scaling the most recent composite count of 31,500, (based on surveys between 2016 and 2021) (Table 4) by the estimated proportion hauled out during the surveys (0.72 (95% CI: 0.54-0.88)). Overall, the UK population has increased since the late 2000s and is close to the late 1990s level prior to the 2002 PDV epizootic. However, there are significant differences in the population dynamics between regions. As reported in SCOS 2008 to 2020, there have been general declines in counts of harbour seals in several regions around Scotland, but the declines are not universal with some populations either stable or increasing.

Recent trends, i.e., those that incorporate the last 10 years show significant growth in both SMUs on the east coast of England up to 2018. However, the 2019 count in the large SE England SMU was approximately 25% lower than the mean of the previous 5 years. Counts for 2020 and 2021 confirm that the population has declined.

Populations in Orkney & North Coast SMU and in the Tay and Eden SAC are continuing to decline and in Shetland and the Moray Firth, the current population size is at least 40 % below the pre-2002 level with no indication of recovery. Populations in western Scotland are either stable or increasing. In Northern Ireland counts have declined slowly.

Until interrupted by the Covid pandemic, SMRU have carried out surveys of harbour seals during the moult in August each year. Recent survey counts and overall estimates were summarised in SCOS-BP 20/03. Given the length of the mainly rocky coastline around north and west Scotland it is impractical to survey the whole coastline every year, but SMRU aims to survey the entire coast every five years. Where there are indications of significant changes the survey effort has been increased and some regions, e.g., Orkney and the Moray Firth, have been surveyed more frequently. The English population, and Scottish east coast populations in the Moray Firth, and the Tay and Eden estuaries are surveyed annually, except for 2020 in the Moray Firth.

Seals spend a higher proportion of their time on land during the moult than at other times, thus counts during the moult are thought to represent the highest proportion of the population with the lowest variance. Initial monitoring of the population in East Anglia in the 1960s used these maximum counts as minimum population estimates. In order to maintain the consistency of the long-term monitoring of the UK harbour seal population, the same time constraints are applied throughout, and surveys are timed to provide counts during the moult. Most regions are surveyed using combined thermographic, video and HR still aerial imagery 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. Efforts are made to reduce the effect of environmental factors by always conducting surveys within 2 hours of low tides that occur between 10:00 and 20:00 during the first three weeks of August and only in good weather². A conversion factor of 0.72 (95% CI: 0.54-0.88) to scale moult counts to total population was derived from haulout patterns of harbour seals fitted with flipper mounted ARGOS tags (n=22) in Scotland (Lonergan *et al.*, 2013)

The conversion factor used here is close to the middle of the range (0.6-0.8) of values estimated for other populations in Europe and North America (e.g., Harvey & Goley, 2011; Huber, Jeffries, Brown, DeLong & VanBlaricom, 2001; Ries, Hiby, & Reijnders, 1998; Simpkins, Withrow, Cesarone & Boveng, 2003). The conversion factor is based on a sample of only 22 seals from a single year that only represents adult seal behaviour. SCOS recommend this conversion factor should be re-investigated when resources allow to examine sex and age differences as well as potential extension to surveys outside the moult.

Table 4. UK harbour seal population estimates based on counts during the moult; rounded to the nearest 100.

Location	Most recent count (2016-2021)	Total Population estimates with 95% CIs
England	3,600 ¹	5,000 (95% CI 4,100-6,700)
Wales	<10 ²	<15
Scotland	26,800 ³	37,200 (95% CI 30,400-49,600)
Northern Ireland	1,000	1,400 (95% CI 1,100-1,900)
Total UK	31,500	43,750 (95% CI 35,800-58,300)

¹ A complete survey of SEE_SMU completed in 2021

² There are no systematic surveys for harbour seals in Wales

³ Compiled from most recent surveys (2016-2019), see Table 5 for dates and details

The most recent counts of harbour seals by region are given in Table 5 and Figures 4, 5 & 6. These are raw counts and therefore represent minimum estimates of the British harbour seal population. Results of surveys conducted in 2019 were described in more detail in SCOS-BP 20/03. It has not been possible to conduct a synoptic survey of the entire UK coast in any one year. Data from different years are grouped into recent, previous and earlier counts to illustrate, and allow comparison of, the general trends across regions.

Combining the most recent counts (2016-2019) at all sites in Scotland and 2021 counts in Southeast England, approximately 31,500 harbour seals were counted in the UK: 85.4% in Scotland; 11.4% in England; 3.2% in Northern Ireland (Tables 4 & 5). Including the 4,000 seals counted in the Republic of

² The diurnal timing restriction is occasionally relaxed for sites in military live firing ranges where access is only at weekends or in the evening.

Ireland produces a total count of ~35,500 harbour seals for the British Isles (i.e., the UK and Ireland). Trends in individual SMUs are described in detail in SCOS-BP 21/03 and briefly in the following section.

Breeding season aerial surveys of the harbour seal population along the east Anglian coast are attempted annually, in addition to the surveys flown during the moult in August. In 2015 and 2016 the east Anglian coast was surveyed five times during the breeding season in June and July (Thompson *et al.*, 2016). These flights confirmed that the peak number of pups ashore occurred around the beginning of July. Due to a combination of aircraft availability and poor weather conditions no breeding season surveys were flown in the UK in 2019 and covid related travel and working restrictions also prevented survey flying in 2020 and 2021. Therefore, the most recent survey was that carried out over two days, 29th June and 2nd July 2018.

Table 5. The most recent August counts of harbour seals at haul-out sites in the British Isles by Seal Management Unit compared with four previous periods. The grey values for SMUs 10-13 are rough estimates. Details of sources and dates of surveys used in each compiled regional total are given in SCOS-BP 20/03.

Seal Management Unit / Country	Harbour seal counts				
	1996- 1997	2000- 2006	2007- 2009	2011- 2015	2016- 2021
1 Southwest Scotland	929	623	923	1,200	1,709
2 West Scotland ^a	8,811	11,666	10,626	15,184	15,600
3 Western Isles	2,820	1,920	1,804	2,739	3,532
4 North Coast & Orkney	8,787	4,388	2,979	1,938	1,405
5 Shetland	5,994	3,038	3,039	3,369	3,180
6 Moray Firth	1,409	1,028	776	745	1,077
7 East Scotland	764	667	283	224	343
SCOTLAND total	29,514	23,330	20,430	25,399	26,846
8 Northeast England ^b	54	62	58	91	79
9 Southeast England ^c	3,222	2,964	3,952	4,740	3,494
10 South England ^d	10	15	15	25	40
11 Southwest England ^d	0	0	0	0	0
12 Wales ^d	2	5	5	10	10
13 Northwest England ^d	2	5	5	5	5
ENGLAND & WALES total	3,290	3,051	4,035	4,871	3,628
NORTHERN IRELAND total ^e		1,176	1,101	948	1,012
UK total		27,557	25,566	31,218	31,486
REPUBLIC OF IRELAND total ^f		2,955		3,489	4,007
BRITAIN & IRELAND total		30,512		34,707	35,493

For data sources see SCOS-BP 20/03.

The 2018 count was 17% higher than the 2017 count and similar to the average for the preceding 5 years. This continues the pattern of high inter annual variability (SCOS-BP 19/04). These wide fluctuations are not unusual in the long-term time series and despite the apparently wide inter-annual variation, the pup production has increased at around 5.6% p.a. since surveys began in 2001 although the rate of increase may have slowed and may be reaching an asymptote (SCOS-BP 19/04). The absence of pup survey data for the past three years in the Wash & N Norfolk SAC population is unfortunate given the scale of the declines observed in the moult survey counts. A pup survey is planned for 2022 together with three moult surveys.

The ratio of pups to the moult counts remained high in 2018 (0.41:1), close to the previous five -year average (0.45:1), and more than double the same ratio in 2001 (0.17:1). This ratio can be seen as an index of the productivity of the population. Until recently, the index for the Wash was higher than for the larger Wadden Sea population. However, the ratio has increased rapidly in the Wadden Sea population since 2008 as moult counts stopped increasing while pup counts continue to grow and the ratio is now at a similar level to the Wash population (Galatius *et al.*, 2021). Previous attempts to explain the apparently high fecundity/productivity in the Wash as being due to seasonal movements between these populations can no longer explain the increase. A population-wide increase in the fecundity index could be due to a real increase in fecundity in both the Wash and Wadden Sea populations, or to a change in the ratio between the moult counts and the total population, or in a change in the ratio of maximum pup count and the total pup production. We do not have any current or historical information to determine the extent to which these metrics may have changed. Reliable estimates of fecundity would provide the basis for identifying and quantifying future changes. Accurate estimates of pup production and of the proportion of animals hauled out during the moult surveys could provide fecundity estimates. SCOS recommends further investigation to identify the underlying changes.

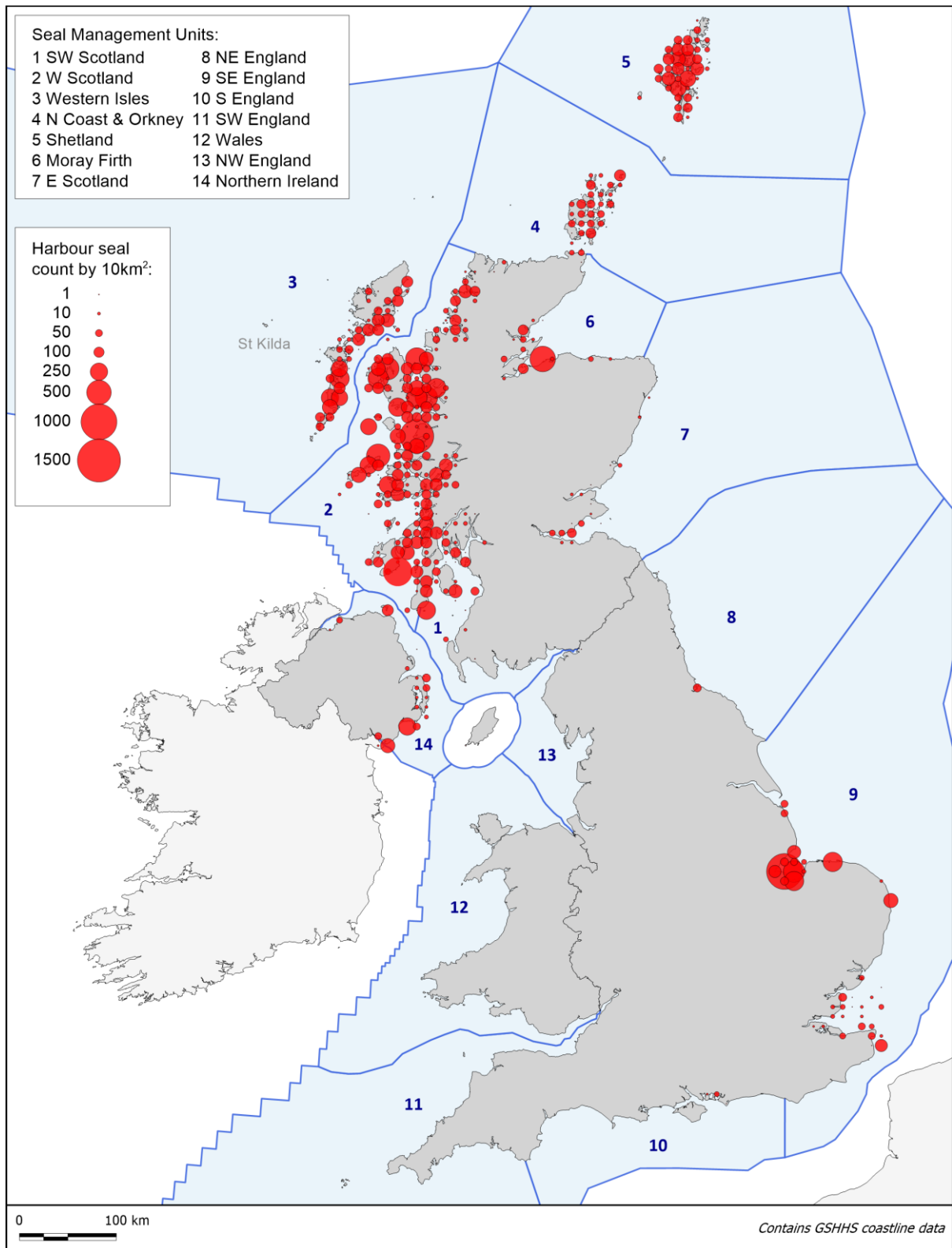


Figure 4. August distribution of harbour seals around the British Isles by 10km squares based on the most recent available haul-out count data collected up until 2019. Limited data available for SMUs 10-13; no data available for St Kilda.

Table 6. Estimates of harbour seal populations in the British Isles by Seal Management Unit. Estimates are based on the most recent August counts of harbour seals at haul-out sites scaled by the proportion of the population estimated to be hauled out during the survey window (0.72; 95% CI=0.54 – 0.88). The grey values given for SMUs 10-13 are rough estimates. Details of sources and dates of surveys used in each compiled regional total are given in SCOS-BP 20/03.

Seal Management Unit / Country		Harbour seal population					
		2007-2009		2011-2015		2016-2021	
1	Southwest Scotland	1281	95% C.I. (1048 - 1709)	1666	95% C.I. (1363 - 2222)	2373	95% C.I. (1942 - 3164)
2	West Scotland	14758	95% C.I. (12075 - 19677)	21088	95% C.I. (17254 - 28118)	21666	95% C.I. (17727 - 28888)
3	Western Isles	2505	95% C.I. (2050 - 3340)	3804	95% C.I. (3112 - 5072)	4905	95% C.I. (4013 - 6540)
4	North Coast & Orkney	4137	95% C.I. (3385 - 5516)	2691	95% C.I. (2202 - 3588)	1951	95% C.I. (1596 - 2601)
5	Shetland	4220	95% C.I. (3453 - 5627)	4679	95% C.I. (3828 - 6238)	4416	95% C.I. (3613 - 5888)
6	Moray Firth	1077	95% C.I. (881 - 1437)	1034	95% C.I. (846 - 1379)	1495	95% C.I. (1223 - 1994)
7	East Scotland	393	95% C.I. (321 - 524)	311	95% C.I. (254 - 414)	476	95% C.I. (389 - 635)
SCOTLAND total		28375	95% C.I. (23215 - 37833)	35276	95% C.I. (28862 - 47035)	37286	95% C.I. (30506 - 49714)
8	Northeast England	80	95% C.I. (65 - 107)	126	95% C.I. (103 - 168)	109	95% C.I. (89 - 146)
9	Southeast England	5488	95% C.I. (4490 - 7318)	6583	95% C.I. (5386 - 8777)	4852	95% C.I. (3970 - 6470)
10	South England	20	95% C.I. (17 - 27)	34	95% C.I. (28 - 46)	55	95% C.I. (45 - 74)
11	Southwest England		95% C.I. (0 - 0)		95% C.I. (0 - 0)		95% C.I. (0 - 0)
12	Wales	6	95% C.I. (5 - 9)	13	95% C.I. (11 - 18)	13	95% C.I. (11 - 18)
13	Northwest England	6	95% C.I. (5 - 9)	6	95% C.I. (5 - 9)	6	95% C.I. (5 - 9)
ENGLAND & WALES total		5604	95% C.I. (4585 - 7472)	6765	95% C.I. (5535 - 9020)	5038	95% C.I. (4122 - 6718)
NORTHERN IRELAND total		1529	95% C.I. (1251 - 2038)	1316	95% C.I. (1077 - 1755)	1405	95% C.I. (1150 - 1874)
UK total		25566	95% C.I. (29052 - 47344)	43358	95% C.I. (35475 - 57811)	43730	95% C.I. (35779 - 58307)
REPUBLIC OF IRELAND total				4845	95% C.I. (3964 - 6461)	5565	95% C.I. (4553 - 7420)
BRITAIN & IRELAND total				48204	95% C.I. (39439 - 64272)	49295	95% C.I. (40332 - 65727)

Population trends

The overall UK harbour seal population has increased over the last decade. Counts increased from 25,600 (rounded to the nearest 100) in the 2007-2009 period to 31,500 during the 2016-2021 period. As no count was available in Northern Ireland in the 1990s, a UK wide comparison is not possible, but the 2016-2021 count of 31,500 harbour seals in Great Britain (i.e., UK minus Northern Ireland) was similar to the 1996-97 count of 32,800 (Table 5). However, as reported in SCOS 2008 to 2019, patterns of changes in abundance have not been universal; although declines have been observed in several regions around Scotland some populations appear to be either stable or increasing. Details of fitted trends by MU and for SACs are given below and in SCOS-BP 21/03. To allow a simple visual comparison the raw count data for each SMU are shown in Figure 5.

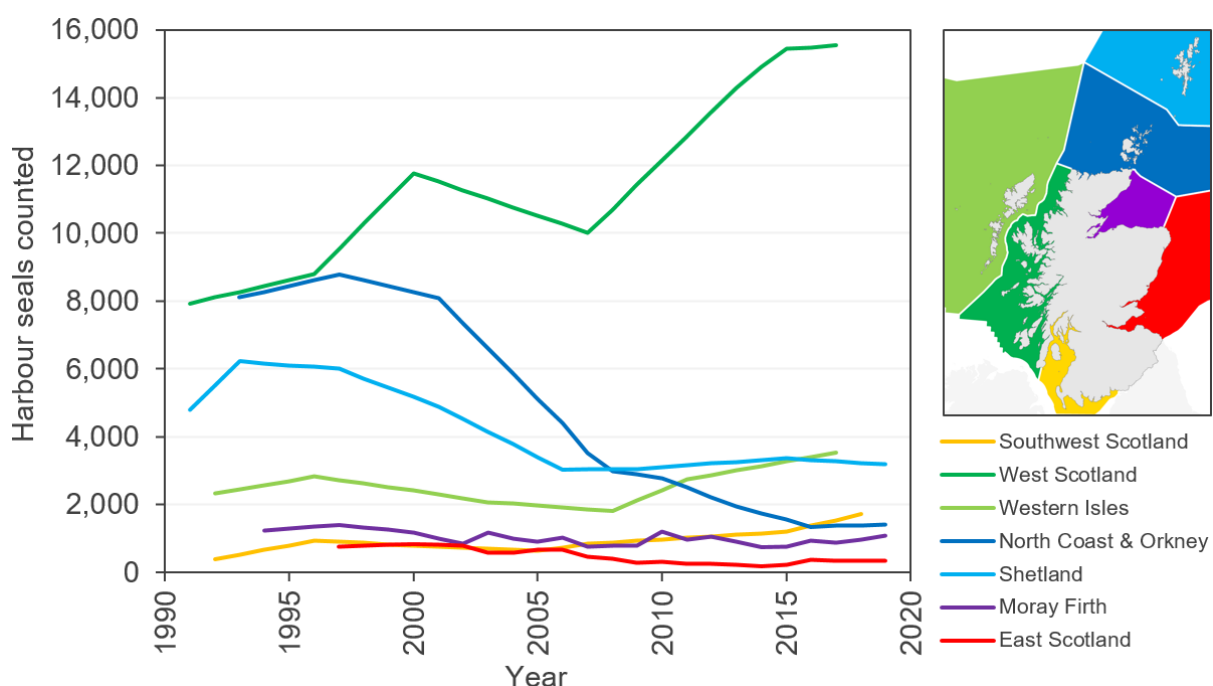


Figure 5. Comparison of August harbour seal counts in Scottish Seal Management Units (SMUs) from 1991 to 2019. Because SMA totals represent counts of seals distributed over large areas, individual data points may contain counts made in more than one year. Interpolated values are used for years with incomplete coverage.

Trends by Seal Management Unit (SMU).

Details of regional and local trend analyses, and model selection for each were given in Thompson *et al.* (2019) and the results presented here are from an extension of that analysis incorporating extra data and with a change in model selection criteria from AICc to AIC. At least three models were fitted for each SMU: a stable trend i.e., an intercept-only Generalised Linear Model (GLM), an exponential year effect within a GLM, and a nonlinear smooth year effect within a GAM. Details of the analysis and figures showing fitted trends for each SMU and SAC are presented in SCOS-BP 21/03.

In the Northeast and Southeast England SMUs Phocine Distemper Virus (PDV) caused sudden declines in 1988 and 2002. Additional models with a step change in abundance and/or trends associated with 2002 were fitted in these SMUs. Although the declines in north and east Scotland SMUs were not thought to be due to PDV, there were sudden drops or declines in Shetland and

North Coast & Orkney SMUs during multi-year gaps in surveys that spanned 2002, and a sudden change in trend around 2002 in East Scotland SMU. Because of the unknown nature of these declines, additional models were also fitted for SMUs 4 – 9 that allowed any combination of stable/exponential trends prior to and following 2002 (including the same trend across the time-series) and with/out a step change associated with 2002. For details of model fitting and model selection see SCOS-BP 21/03.

Western Isles: A complete survey of the Western Isles SMU carried out in 2017 produced the highest recorded count for the Western Isles (3,533) which was 29.0% higher than the previous (2011) count of 2,739 and approximately 40% higher than the average between 1993 and 2011. Relaxing the model selection criteria resulted in the best model being a GAM that shows a decline from the mid-1990s to around 2005 followed by a steep increase to 2017. The revised trends analysis is the basis for a suggested relaxation of the Seal Conservation Area designated for the Western Isles SMU (Answer to Q19 below).

West Scotland: Parts of the West Scotland SMU (North and part of Centre) were surveyed in 2017 and the remainder was surveyed in 2018. The harbour seal count for West Scotland - North was 1,084, for West Scotland - Centre was 7,447 and for West Scotland – South was 7,053, and the overall total for the West Scotland SMU was 15,600 (Table 5).

The 2015 West Scotland harbour seal count was 43% higher than the 2009 count. The best model, selected in the trend analysis shows a continuous increase from 1990 to 2017 at approximately 4.7% p.a. Over the last five years the rate of increase is estimated to be 3.9 % p.a.

Although the West Scotland region is defined as a single management unit, it is very large geographically in terms of total coastline and contains a large proportion of the UK harbour seal population; 49% of the most recent UK total count. The trajectories of counts within north, central and south sub-divisions of this large region differ:

- In the north of the region (Figure 4), the selected model for data up to 2017 indicates that counts have increased since the early 1990s, by approximately 4.9% p.a.
- In the central sub-region (Loch Ewe to Ardnamurchan) (Figure 4) the selected model indicates that counts have increased since the early 1990s. The average rate of increase has been approximately 4.0% p.a.
- In the south sub-region (Ardnamurchan to Scarba) (Figure 4) there was no detectable trend in the overall population since the early 1990s, with counts varying between approximately 5,000 and 7,000 over the period 1990 to 2018.

Southwest Scotland: All of the Southwest Scotland SMU was surveyed in August 2018. A total of 1,700 harbour seals were counted compared with 1,200 in 2015 and 923 in 2009 (Table 5). This was the highest count of harbour seals for the Southwest Scotland SMU, approximately three times higher than the 1990's count. The trend analysis selected a continuous increase since 1990. The rate of increase over the past five years was approximately 3.9% p.a.

North Coast and Orkney: Orkney was surveyed twice during the last round-Scotland census period. In 2016, 1,240 harbour seals were counted, and 1,296 in 2019 (Table 5). These are the two lowest counts to date and represent an 85% reduction from the highest count in 1997 (8,522). The 2016 and 2019 counts were similar. Although this could indicate that the decline has slowed this cannot be confirmed without additional counts. Trend analysis (Thompson *et al.*, 2019) indicates that counts were stable until 2001, then dropped by 46% between 2001 and 2006, and have declined continuously since 2006. The average rate of decrease over the past 5 years was approximately 8.5%

p.a. The North Coast section of the SMU was not surveyed in 2019 but few harbour seals are counted on the north coast section of the SMU.

Shetland: A complete survey was carried out in 2019 when 3,180 harbour seals were counted compared with 3,369 in 2015. The 2019 count was close to the mean of the 2009 and 2013 counts but was 47% lower than the 1997 count of c.6,000. The selected model for counts for the whole of Shetland incorporated a step change involving a drop of approximately 40% occurring between 2001 and 2005. Counts either side of the step change (1991-2001 and 2006-2019) do not show any obvious trend, though in both cases the sample size was limited (n=4 and 4, respectively).

Moray Firth: The total harbour seal count for the entire Moray Firth SMU in 2019 was 1025. This was 12% higher than the 2018 count. The majority of these harbour seals (60%) were observed between Culbin and Findhorn, confirming the continued importance of these sites and the dramatic and continuing redistribution within the inner Moray Firth.

The majority of the counts in the Moray Firth are from haul outs between Loch Fleet and Findhorn, an area that held approximately 98% of the SMU total in 2016. The selected model for this area suggests that counts were decreasing between 1994 and 2000, the rate of decline slowed to around 2010 and the population may now be increasing slowly.

East Scotland: The harbour seal count for the Firth of Tay and Eden Estuary SAC in 2019 was 41, equal to the mean of the previous 5 years' counts for this SAC. This represents a 94% decrease from the mean counts recorded between 1990 and 2002 (641).

In the East Scotland SMU (Figure 4) the population was mainly concentrated in the Firth of Tay and Eden Estuary SAC prior to 2000. Additional groups were also present in the Firth of Forth, Montrose Basin and at coastal sites in Aberdeenshire. Counts in the Firth of Forth have been sporadic but the fitted trend suggests a decline from the late 1990s to 2016.

A more extensive data set is available for the Firth of Tay and Eden Estuary SAC. The selected model indicates that counts in the SAC remained stable between 1990 and 2002, at which time they represented approximately 85% of the total SMU count. From 2002 to 2020 the counts in the SAC declined rapidly and monotonically: over the 18-year period counts fell from approximately 680 to less than 40, representing a 95% decline. By 2016 the SAC counts represented only approximately 15% of the SMU total.

Northern Ireland: Only three synoptic surveys of the entire harbour seal population in Northern Ireland have been carried out in 2002, 2011 and 2018, although data from a fourth survey in 2021 will be available for SCOS 2022. However, a subset of the population from Carlingford Lough to Copeland Islands has been monitored more frequently from 2002 to 2018. This area contained 80-85% of the total in the two years with complete coverage. This subset of the population declined slowly over the period 2002 to 2011 at an average rate of 2.7% p.a. However, the 2018 survey suggests that there had been no significant change since 2011.

Southeast England: A detailed description of recent survey results from 2020 and 2021 are given in SCOS-BP 21/06. Briefly, the combined counts for the Southeast England SMU (Figure 6) in 2019 (3,081) was 27.6% lower than the 2012 to 2018 mean count. Additional surveys in 2020 and 2021 confirmed the decrease. The total count for the sites between Donna Nook and Scroby Sands has declined by approximately 38% compared to the mean of the previous five years (2019–2021 mean = 3080; 2014-2018 mean = 4296). The count for the Wash and North Norfolk SAC has decreased by approximately 21% (2019 – 2021 mean = 2883; 2014-2018 mean= 3658) over the same time periods while Donna Nook showed a 57% decrease and Scroby Sands showed a 73% decrease.

The fitted trend for the Wash and North Norfolk SAC (figure 6) shows that the population recovered from the 2002 PDV epizootic, reached a maximum around 2014 to 2015 and has since declined rapidly.

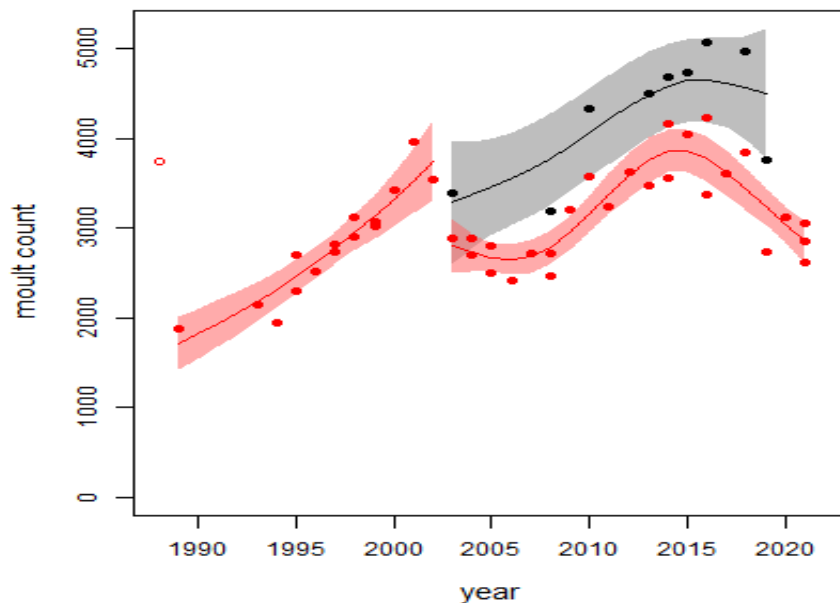


Figure 6. Trends in harbour seals counts in the Southeast England SMU (grey) and in The Wash and North Norfolk SAC (red), between 1988 and 2021 (shaded areas indicate the 95% confidence intervals for the fitted curves). For further explanation see text and SCOS-BPs 21/06. 2018 counts were similar to the previous 5 year's counts, but the 2019, 2020, and 2021 counts show a clear decline.

The 2018 count was the second highest ever recorded in the Wash and was consistent with the pattern of relatively stable population after 2010. However, the fitted trend suggests that the population may have been declining since 2015, but at present it is unclear whether the decrease represents a continuing decline or a step change decrease between 2018 and 2019. In the absence of any clear anthropogenic effects, this decline is dramatic. Recent counts from the rest of Southeast England Seal Management Unit (SEE_SMU) by ZSL (SCOS-BP 21/07) suggest that population may also be showing the start of a decline. Given that the survey area represents the majority of harbour seals in the SEE-SMU, including the population in the Wash & N Norfolk SAC, this likely drop in abundance is of immediate and serious concern. The SEE-SMU was the only one in the UK that was showing a sustained increase in abundance at a time when the majority of SMUs on the eastern and northern coasts had depleted or declining populations (Thompson *et al.*, 2019; SCOS-BP 21/06). SCOS recommend that research is required to determine the time course and potential causes of this reduction and recommend that SMRU should seek funding to establish an appropriate programme of research.

The Thames population, here taken to include all haulout sites between Hamford Water in Essex and Goodwin Sands off the Kent coast, have been surveyed sporadically since 2002 and annually since 2008. In August 2019, a total of 671 harbour seals were counted compared with an average of 742 for three surveys in 2016-2018, and an average of 474 for three surveys in 2013-2015. A GLM for the series of counts from 2002 to 2019 demonstrated an increase at an average of 9.0% p.a. (bootstrap 95% CI 6.8-11.2) (Cox *et al.*, 2020). No survey was carried out in 2020, but a survey in 2021 showed that the population has not grown over the past 4-5 years and may be starting to decline (SCOS-BP 21/07).

Table 7. 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 were obtained
Scotland	26,850	2016-2019
England	3,900	2019 ²
Northern Ireland	1,000	2018
UK	31,750	
Ireland	4,000	2017-18
France	1,150	2018
Wadden Sea-Germany	17,250	2021
Wadden Sea-Denmark	1,350	2021
Wadden Sea-NL	8,250	2021
Delta-NL	1,200	2017
Limfjorden	1,050	2019
Kattegat	9,900	2019
Skagerrak	7,300	2019
Baltic (Kalmarsund)	1,800	2019
Baltic Southwestern	1,100	2019
Norway	6,450	2012-18
Svalbard	1,900	2010
Iceland	9,450	2018
Europe excluding UK	68,150	
Total	99,900	

¹ Counts rounded to the nearest 50. 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.

² Includes an estimate of 55 seals for south England, Wales and north-west England compiled from sporadic reports

Data sources

ICES. 2021. Report of the Working Group on Marine Mammal Ecology (WGMME), ICES Scientific Reports. 3:19. 155 pp. <http://doi.org/10.17895/ices.pub.8141>. 120 pp; Desportes, G., Bjørge, A., Aqbal, 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 harbour seal (*Phoca vitulina*) in the Baltic Proper. NAMMCO Sci Pub 8:71-76.; Olsen MT, Andersen SM, Teilmann J, Dietz R, Edren SMC, Linnel A, and Härkönen T. 2010. Status of the harbour seal (*Phoca vitulina*) in Southern Scandinavia. NAMMCO Sci Publ 8: 77-94.; Galatius A., Brackmann J., Brasseur S., Diederichs B., Jeß A., Klöpffer S., Körber P., Schop J., Siebert U., Teilmann J., Thøstesen B. & Schmidt B. (2020) Trilateral surveys of Harbour Seals in the Wadden Sea and Helgoland in 2020. Common Wadden Sea Secretariat, Wilhelmshaven, Germany. ; 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; www.fisheries.is/main-species/marine-mammals/stock-status/; www.nefsc.noaa.gov/publications/tm/tm213/pdfs/F2009HASE.pdf; www.hafogvatn.is/en/research/harbour-seal/harbour-seal-census. www.nammco.no/webcronize/images/Nammco976.pdf, Nilssen K and Bjørge A 2017. Seals – grey and harbor seals. In: Bakkevig IE, Gjøsæter H, Hauge M, Sunnset BH and Toft KØ (eds). Havforskningsrapporten 2014. Fisken og havet, 2014(1). Merkel, B., Lydersen, C., Yoccoz, N. & Kovacs, K. (2013) The World's Northernmost Harbour Seal Population—How Many Are There? PLOS-ONE. <https://doi.org/10.1371/journal.pone.0067576>

Although the Southeast England population increased after the 2002 PDV epizootic and apparently levelled off at a similar size to its pre-2002 epizootic population, it grew at a much lower rate than the Wadden Sea harbour seal population, the only other major population in the southern North Sea. Counts in the Wadden Sea increased from 10,800 in 2003 to 26,788 in 2013, equivalent to an

average annual growth rate of 9.5% over ten years. Counts since 2014 indicate that the rapid growth since the 2002 PDV epizootic has stopped (Galatius *et al.*, 2021). Although there was an influenza-A epizootic that killed at least 1600 seals in 2014, it now seems highly likely that cessation of the previously rapid increase in the Wadden Sea population indicates that it has reached its carrying capacity. The coincidence of the timing of the slowdown in the Wadden Sea and SE England is notable, but the Wadden Sea counts have not shown a decrease since 2018.

UK harbour seal populations in a European context

The UK harbour seal population represents approximately 32% of the eastern Atlantic sub-species of harbour seal (Table 7). Since 2000, the declines in Scotland and coincident dramatic increases in the Wadden Sea mean that the relative importance of the UK harbour seal population has declined, although with the reduction in growth rates in the Wadden Sea this pattern may have stabilised.

<p>2. Please could SCOS provide an update on the Scottish regional harbour seal declines, including current and projected trends.</p>	<p>MS Q9</p>
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The most recent composite count for Scotland, for surveys in 2016 to 2019, was 6% higher than for the previous round of surveys (2011-2015) and 31% higher than the 2007-2009 composite count.

Trends in each SMU around Scotland and on the east coast of England are presented in answer 1 above and in detail in SCOS-BP 21/03.

The current UK harbour seal population is at a similar size to the estimates from the late 1990s, but there have been significant population declines in some regions and similar increases in others.

The composite count for all of Scotland, 26850 based on recent (2016-2019) surveys was 6% higher than for the previous round of surveys (2011-2015) and 31% higher than the 2007-2009 composite counts, representing approximately 3% p.a. increase (Figure 5; Table 5) and is similar to counts in the mid-1990s.

Trends by SMU are reported in SCOS-BP/03 in detail and briefly described in answer 1 above and shown in Figure 5 for Scottish SMUs and Figure 6 for the Southeast England SMU. Briefly, the populations in the West Scotland and Southwest Scotland SMUs have increased continuously since the 1990s. The Western Isles population declined in the late 1990s but has been increasing since approximately 2005. Shetland and the Moray Firth SMUs are apparently stable after a large, rapid decline in the early 2000s, but Moray Firth counts may now be increasing. North Coast and Orkney SMU is still declining. In the East Scotland SMU the population in the Tay and Eden SAC has declined rapidly since 2002 and the decline is apparently continuing. Less frequent counts in the Firth of Forth indicate that the whole SES_SMU may also be declining.

Large changes in relative density have resulted from differences in regional population trends. E.g., in 1996-1997 the West Scotland SMU and Orkney & North Coast SMU each held 27% of the UK population but now hold 50% and 4% respectively. Recent surveys in the Northeast England SMU and particularly in the large population in the Southeast England SMU have shown a sudden rapid decline since 2018, in what was, until recently, a rapidly increasing population. The Southeast England SMU population was approximately half that of the Wadden Sea in 1980 but by 2019 the Wadden Sea count was approximately eight times larger.

Given the variable patterns in harbour seal trends and very significant declines in some management units SCOS consider it prudent and timely to undertake risk assessments regarding the viability of local populations in relevant SMUs. These should be based on available scientific knowledge (e.g., breeding data, movements, immigration, emigration) and knowledge of pressures and threats. A further consideration would be to review resourcing, to ensure that adequate monitoring resources are deployed in SMUs considered “high risk” as a result of such an assessment exercise.

Due to Covid restrictions, no Scotland based surveys were carried out in 2020, so there are no updates on the trend information in any Scottish SMUs. One survey flight of the Tay and Eden SAC population was carried out during an aircraft re-positioning flight from Dundee to Kent. The survey produced a count of 39 harbour seals. This was similar to the mean of the three previous counts and there is therefore no change to the East Scotland SMU estimate.

At present there is no predictive model capable of projecting trends for any Scottish SMU population. In the absence of revised counts and a predictive model, SCOS defers the answer to the next SCOS meeting.

The current UK harbour seal population is at a similar size to the estimates from the late 1990s, but there have been significant population declines in some regions and similar increases in others. As reported in previous SCOS reports since 2008, there have been general declines in the counts of harbour seals in several regions around Scotland, but the declines are not universal with some populations either stable or increasing. Details of trends are presented in SCOS-BP 20/03 and Thompson *et al.* (2019).

<p>3. Are trends in common/harbour seal abundance considered to be declining in English waters and if so, what are the potential influencing factors and where is further research needed?</p>	<p>Defra Q1b</p>
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Harbour seal populations in the Wash and adjacent sites have declined rapidly since 2018. Counts in the rest of the SEE_SMU are also showing signs of the start of a decline. The decline is widespread throughout The Wash and adjacent sites and coincides with a similar change in grey seal numbers at the UK’s largest haulout site at Donna Nook.

Neither the mechanism of change (e.g., emigration, mortality, change in behaviour) nor the drivers of change are known. Grey seal abundance and the simultaneous slow down and possible decline suggest that the two population trajectories may be coupled.

Assigning cause to these changes will require a multi strand research programme.

The counts of harbour seals at sites in SSE_SMU from Donna Nook to Scroby Sands during the August survey in 2019 were approximately 30% lower than the five year mean for 2014 to 2018.

The same sites were surveyed in 2020. That count was 8% higher than the 2019 count but was still 21.5% lower than the 2014-2018 mean. In response to this decrease Defra funded additional surveys in August 2021. Three surveys were carried out in 2021 and the mean harbour seal count was close to the mean of 2019 and 2020 counts and confirms that there has been a decrease.

A detailed description of the surveys, the resulting count data, and trend analyses are presented in SCOS-BP 21/6 and briefly in answer 1 above. The total count for the sites between Donna Nook and Scroby Sands has declined by approximately 30% compared to the mean of the previous five years (2019–2021 mean = 3045; 2014-2018 mean = 4296). The count for the Wash and North Norfolk SAC

has decreased by approximately 23% (2019 – 2021 mean = 2862: 2015-2018 mean= 3712) over the same time periods while Donna Nook showed a 57% decrease and Scroby Sands showed a 73% decrease. The harbour seal decline is evident at all sites and appears to have affected all sub-sections of the Wash & N Norfolk SAC.

Recent surveys of the Greater Thames estuary by ZSL have also detected the first indications of a possible decline in the remainder of the Southeast England SMU population (SCOS-BP 21/7).

Grey seal numbers have increased dramatically over the past 20 years, but the large grey seal haulout group at Donna Nook, accounting for around 65% of the SEE_SMU total shows a similar levelling off and possible decline, coincident with the harbour seal decline. Over the past five years grey seals have been expanding their haulout range within the Wash and small groups are now appearing in the sheltered tidal creeks at the southern edge of the estuary, which are important pupping sites for harbour seals.

Neither the mechanism of change (e.g., emigration, mortality, change in behaviour) nor the drivers of change are known. Grey seal abundance and the simultaneous slow down and possible decline suggest that the two population trajectories may be coupled. Assigning cause to these changes will require a multi strand research programme. Natural England and Defra have funded a preliminary assessment of available information (Russell *et al.* 2021) and a preliminary series of additional surveys. On the basis of these preliminary actions SMRU have developed proposals for such a project and are seeking extra resources.

<p>4. What is the latest information about the population structure, including mortality, age and sex structure, and carrying capacity of grey and common/harbour seals in English waters? Is there any new evidence of grey or common/harbour seal populations or sub-populations specific to localised/regional areas? What is the latest understanding about the population structure, including survival, reproduction and age structure, of grey and harbour seals in European and Scottish waters?</p>	<p>Defra Q2</p> <p>MS Q2</p>
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SCOS are not aware of any new information on population structure, mortality, age or sex structure, or carrying capacity for harbour seals in European populations of harbour seals since the 2020 SCOS report. Other than a modelling study of survival and two published studies of breeding phenology there do not appear to be any new studies of population structure, mortality, age or sex structure, or carrying capacity for grey seals. For information the 2020 answer to these questions is included with minor additions.

Grey seals

There is evidence for regional differences in grey seal demographics (Smout *et al.*, 2019) but detailed information on vital rates are lacking. New resources should be identified to address questions around fecundity and first-year survival as they are likely drivers of UK grey seal population dynamics.

There is no new genetic information with which to assess the substructure of the breeding grey seal populations and therefore no new evidence of sub-populations specific to local areas.

Earlier studies indicated a degree of reproductive isolation between grey seals that breed in the south-west (Devon, Cornwall and Wales) and those breeding around Scotland, and within

Scotland, there were significant differences between the Isle of May and North Rona. There is therefore some indication of sub-structure within the UK grey seal population, but it is not strong.

Age and sex structure

While the population was growing at a constant (i.e., exponential) rate, it was assumed that the female population size was directly proportional to the pup production. Changes in the rate of increase in pup production imply changes in age structure and/or changes in fecundity. 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. An indirect estimate of the age structure, at least in terms of pups, immature and mature females is generated by the fitted population estimation model (SCOSBP 20/01). As currently structured the model fits single global estimates for fecundity, maximum pup survival (i.e., at low population size), and adult female survival, and fits individual carrying capacity estimates separately for each region to account for differing dynamics through density dependent pup survival.

Recently Bull *et al.*, (2021) suggested that changes in timing of births at the small grey seal colony on Skomer Island were being driven by changes in population age structure that was itself responding to changes in an index of sea surface temperature. It is not clear if this represented permanent changes in age structure, temporary immigration/emigration of breeding females of different ages or even interannual variation in fecundity. Nor is it clear whether this was a purely local effect due to movement or changes in recruitment patterns between Skomer Island and the nearby colony on the Welsh mainland. Bowen *et al.* (2020) studied phenology over a 30-year period at the much larger grey seal colony on Sable Island and showed much smaller magnitude changes. They ascribed the changes in timing of births to gradual demographic changes and showed that females of all ages responded to environmental forcing. They also concluded from their sample of 2768 pups that birth date had no impact on pup weaning mass. As weaning mass is related to pup survival, there is therefore unlikely to be a detectable link between birth date and pup survival.

Survival and fecundity rates

The only contemporary data that we have on fecundity and adult survival in UK grey seals has been estimated from long term studies of marked or identifiable adult females at two breeding colonies, North Rona and the Isle of May. Results of these studies together with branding studies in Canadian grey seal populations and historical shot samples from the UK and Baltic have been used to define priors for a range of demographic parameters (SCOS-BP 20/02).

Adult female survival: Estimates of annual adult survival in the UK, obtained by aging teeth from shot animals were between 0.93 and 0.96 (Harwood & Prime, 1978; Hewer, 1964; SCOS-BP 12/02). Capture-mark-recapture (CMR) of adult females on breeding colonies (Smout *et al.*, 2019) has been used to estimate female survival on North Rona and the Isle of May of 0.87 and 0.95 (SCOS-BP20/02 - Table 2). The population dynamics models fitted to the pup production time series, produced estimates of adult female survival close to the upper limit of that range (SCOS-BP 20/01). Interestingly, recent estimates from Sable Island suggest that adult female survival during the main reproductive age classes (4 to 24 years old) may be even higher. A Cormack-Jolly-Seber model was used to estimate age- and sex-specific adult survival from a long-term brand re-sighting programme on Sable Island (den Heyer & Bowen, 2017). Average adult female survival was estimated to be 0.976 (SE 0.001), averaged over all animals, but was higher for younger adults (0.989 with SE 0.001 for age classes 4-24) than older adults (0.904 SE 0.004 for age 25+).

Rossi *et al.*, (2021) used the branded animal data set for Sable Island to show that survival rates were higher for females compared to males for all age classes, though differences were small for ages 1–19. Females' annual survival rates were very high (>97%) until age 25, after which survival declines by 8% between ages 25–29 and by another 9% for ages 30 and above. Males similarly maintained

high survival rates (>95%) until age 25, though declines in male survival rates in older age classes were much steeper than in female rates. The estimated survival rates imply maximum ages of about 35 years for males and 45 years for females.

In the current population estimation model density dependence acts through pup survival only, so adult survival does not vary with time or between regions. The fitted posterior value for adult survival was a constant rate of 0.96 (SE 0.01), which is consistent with the findings of Rossi *et al.* (2021).

Fecundity: For the purposes of the population estimation model, fecundity is taken to be the proportion of breeding-age females (aged 6 and over) that give birth to a pup in a year (natality or birth rate). Pregnancy rates estimated from samples of seals shot in the UK (Hewer, 1964; Boyd, 1985) and Canada (Hammill & Gosselin, 1995) were similar, 0.83 to 0.94 and 0.88 to 1 respectively. However, these are pregnancy rates and may overestimate natality if there are significant numbers of abortions.

Natality rates estimated from direct observation of marked animals produce lower estimates, which may be due to abortions, but may also be due to unobserved pupping events (due to mark misidentification, tag loss, or breeding elsewhere) and may therefore under-estimate fecundity. Such studies, from Sable Island estimate fecundity to be between 0.57 and 0.83 (den Heyer & Bowen, 2017; Bowen *et al.*, 2006). UK estimates of fecundity rates adjusted for estimates of unobserved pupping events were higher; 0.790 (95% CI 0.766-0.812) and 0.816 (95% CI 0.787-0.841) for a declining (North Rona) and increasing (Isle of May) population respectively (Smout *et al.*, 2019).

In the current population estimation model, density dependence acts through pup survival only, so fecundity does not vary with time or between regions. The fitted posterior value for fecundity was 0.90 (SE 0.06) (SCOS-BP 20/01).

Four separate, recent studies have investigated the potential effects of environmental conditions on fecundity of grey seals:

- Kauhala *et al.* (2019) used samples from seals shot in Finland to demonstrate that pregnancy rates show significant interannual variation (between c0.6 and c0.95) and are significantly related to herring (*Clupea harengus*) and sprat (*Sprattus sprattus*) quality (weight), which in turn were influenced by sprat and cod (*Gadus morhua*) abundance and zooplankton biomass. Their results suggest strong coupling over three trophic levels in the Baltic and suggest that this is likely to influence fecundity rates.
- Smout *et al.* (2019) reported a similar link between likelihood of breeding and environmental conditions during the preceding year.
- In a parallel study, Hanson *et al.* (2019) showed high levels of variation in individual postpartum maternal body composition at two grey seal breeding colonies (North Rona and Isle of May) with contrasting population dynamics. Although average composition was similar between the colonies, it increased at the Isle of May where pup production increased and declined at North Rona where pup production decreased.
- Badger *et al.* (2020) investigated the effects of increasing population density on the reproductive performance of female grey seals over a period when the population was apparently approaching its carrying capacity. Counter to expectations, reproductive performance (measured by reproductive frequency and likelihood of successfully weaning a pup) increased with population size over a period when the population was approaching carrying capacity. However, individual heterogeneity was high and the difference in performance between females identified as either robust or frail on the basis of reproductive histories, increased with population size.

All four studies suggest that fecundity or reproductive performance is influenced by prevailing environmental conditions. The consequences in terms of population level fecundity estimates are not

clear, but SCOS recommends continued investigations into the effects of environmental variation on fecundity and the potential effects of such links on population projections for UK grey seal populations.

First year survival: In the context of the population estimation model, first year survival is defined as the probability that a female pup, will be alive at the start of the following breeding season. However, the model makes the simplifying assumption that annual survival from age 1 to age of recruitment into the breeding population is the same as adult survival. In practice the time series of pup production data contains no information on the pattern of mortality between birth and recruitment. This simplifying assumption means that all additional, pre-recruitment mortality is pooled into the pup survival estimate.

At present, density dependent effects in the UK grey seal population are thought to operate primarily through changes in pup survival. The currently used density-dependent pup survival population model therefore requires a prior distribution for the maximum pup survival, i.e., pup survival in the absence of any density dependent effects. The model then produces a single global posterior estimate of that parameter and region-specific estimates of the current pup survival under the effects of density dependence.

Estimates of maximum pup survival, from populations experiencing exponential growth and therefore presumed not to be subject to strong density dependent effects are given in SCOS-BP 21/04 (Table 2). Mean estimates of pup survival were between 0.54 – 0.76.

The fitted value for maximum unconstrained pup survival was 0.46 (SE 0.07) from the standard model run on the 1984-2016 dataset and data from the North Sea population in 2018 (SCOS-BP 20/01). This value increases slightly to 0.49 when the later pup production estimates were altered by changing the probability of misclassification (SCOS-BP 20/01). These values are substantially lower than estimates in the literature (SCOS-BP 21/04).

It is also possible to derive region-specific pup survival estimates, given the density dependent response to the region-specific population sizes. In the North Sea where density dependence is having little effect, the current pup survival estimate is 0.43, close to the maximum, unconstrained rate estimated by the model, but substantially lower than the published estimates (SCOSBP 21/04). In the other three regions where population growth has slowed or stopped the current estimate is much lower, being 0.11 in the Inner and Outer Hebrides and Orkney. Thomas *et al.*, (2019) estimated that pup survival for a population at carrying capacity will be around 0.1-0.14.

Investigations using the grey seal population dynamics model suggested that changes in first year survival rather than changes in fecundity are the main mechanisms through which density dependence acts on UK grey seal populations (Thomas, 2010; Thomas *et al.*, 2019). Fecundity at an increasing population at the Isle of May was only marginally higher than in a declining population at North Rona colony in Scotland, and fecundity has not changed as the Sable Island grey seal population reaches density dependent limits (den Heyer *et al.*, 2017; Smout *et al.*, 2019). Variation in fecundity may become increasingly important in areas where populations have reached carrying capacity, e.g., age of first recruitment appears to increase as populations reach carrying capacity (Bowen *et al.*, 2006; Pomeroy *et al.*, 2010) and the reproductive success of individuals becomes more variable (Badger *et al.*, 2020).

Regional data on fecundity and survival rates would allow us to further examine the drivers of population trends. Such data would feed into the population dynamics model, improving confidence in model predictions and enhancing our ability to provide advice on population status. Furthermore, such data could inform effective management by identifying the relative sensitivities associated with different life stages, in terms of population dynamics. SCOS 2019 recommended that new resources should be identified to investigate regional patterns and the effects of environmental covariates on both first-year survival and fecundity in UK grey seal populations.

Sex Ratio: The sex ratio effectively scales up the female population estimate derived from the model fit to the pup production trajectories, to the total population size. With the inclusion of two independent estimates of total grey seal population size, the fitted values of the demographic parameters and the overall population size estimates are sensitive to the population sex ratio for which we do not have good information. The reported values are produced by a model run with a prior on the sex ratio multiplier of 1.7 (SE 0.02), i.e., a female to male sex ratio of 1:0.7 or ten females to every seven males.

den Heyer and Bowen (2017) estimated survival rates of male and female branded seals at Sable Island, Canada. The differential survival of males and females would produce an effective sex ratio of 1:0.7 if maximum age is set to 40, reducing to 1:0.69 if maximum age is set to 45. The sex ratio estimate from the Canadian population is remarkably similar to the prior used in the 2016 model runs. Rossi *et al.* (2021) produced similar sex specific survival rates from the Sable Island brand re-sightings data, but an age structure derived from the survival estimates in Rossi *et al.* (2021) would result in a sex ratio of approximately 1:0.8 assuming equal first year survival for male and female pups.

Regional differences in grey seal demographics and genetics

The difference in population trends between regions for UK grey seals suggests underlying regional differences in the current values of demographic parameters. 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 (Walton & Stanley, 1997) and within Scotland, there are significant differences between grey seals breeding on the Isle of May and on North Rona (Allen *et al.*, 1995). There is therefore some indication of sub-structure within the UK grey seal population, but it is not strong.

Recent genetic data from the Baltic grey seals (Fietz *et al.*, 2016) suggest that a combination of previous management practices and local climate change effects may be moving the boundaries between the North Sea and Baltic subspecies of grey seal, with increasing encroachment of North Sea seals on areas previously occupied by the Baltic Sea subspecies.

The very rapid increases in pup production at colonies in the Southern North Sea in England, the Netherlands and Germany all point to large scale recruitment to those colonies from colonies in the Northern North Sea (Brasseur *et al.*, 2015). Similar immigration appears to be driving growth in southern colonies on the west side of the Atlantic. On the basis of mtDNA haplotype information Wood *et al.* (2011) could not differentiate between US and Canadian grey seal populations and concluded although grey seals are regarded as philopatric, their results indicate that the genetic structure of the northwest Atlantic grey seal population is not different from the null hypothesis of panmixia.

A study led by the Galway-Mayo Institute of Technology (GMIT) is currently investigating the genetic structure of both grey and harbour seals occupying Irish haul-out sites and coastal/marine waters, to determine their relationship to wider regional populations across Western Europe (Steinmetz *et al.*, in prep). New mitochondrial data from grey seals in Ireland, southwest England and the German/Danish North Sea coasts were combined with previously published data to generate a dataset including more than 2,000 individuals. Mitochondrial and nuclear diversity were high in all sub-regions. Genetic structuring results suggested that grey seals from the island of Ireland are part of a single interbreeding population. Southwest England was identified as a source of migrants to the island of Ireland. Southern North Sea populations from continental Europe were identified either as a source of migrants to the island of Ireland or as sharing a common source population. Considering these genetic findings, the authors suggest two distinct MUs are proposed for the Northeast Atlantic, comprising: (i) the Faroe Islands, Scotland and the North Sea; and (ii) the island of Ireland, southwestern UK and France. Two transition zones between these MUs are also proposed: (i)

Northwest Scotland and (ii) the English Channel/ Dutch North Sea. A similar analysis of genetic structure in grey and harbour seals in Norway is underway but at an early stage.

Harbour seals

Knowledge of UK harbour seal vital rates is limited and inferences about population dynamics rely on count data from moulting surveys. Information on vital rates would improve our ability to provide advice on population status but estimates for UK harbour seals are only available from one long term study at Loch Fleet in northeast Scotland. Additional studies are underway to obtain similar data from new sites in Orkney and western Scotland.

Indices of fecundity in both the Wash and Wadden Sea have increased suggesting that either demographic rates, or our indices of those rates, are changing and require further investigation. Recent genetic studies show that harbour seals in southeast England, north and east Scotland, and northwest Scotland form three distinct genetic clusters and population trend analyses suggest that these three groups show different population trends.

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 epizootics in 1988 and 2002 were aged, these were clearly biased samples that cannot be used to generate population age structures (Hall *et al.*, 2019).

Survival and fecundity rates

A long-term photo-ID study of harbour seals at Loch Fleet, NE Scotland produced survival rate estimates of 0.95 (95% CI 0.91-0.97) for adult females and 0.92 (0.83-0.96) for adult males (Cordes & Thompson, 2014; Mackey *et al.*, 2008).

A study investigating first year survival in harbour seal pups, using telemetry tags was carried out in Orkney and on Lismore in 2007. Battery life of the transmitters limited the study duration, but survival was not significantly different between the two regions and expected survival to 200 days was 0.3 (Hanson *et al.*, 2013). Harding *et al.* (2005) showed that over winter survival in harbour seal young of the year was related to body mass and to water temperature. Preliminary estimates of survival of harbour seals in Orkney and Skye should be available for SCOS 2022 from the ongoing harbour seal decline project under the Marine Scotland MMSS programme.

In South-east England there is evidence for changing demographic parameters in harbour seals. The apparent fecundity, i.e., the peak count of pups (as an index of pup production) divided by the moult survey count (as an index of total population size) of the large harbour seal population in The Wash has shown large changes since the early 2000s. The rate has been approximately twice that of earlier estimates and until recently was much higher than in the larger population in the Wadden Sea (SCOSBP 20/03). The fact that apparent fecundity of the much larger population in the Wadden Sea has now also increased, suggests that this is a real effect and not due simply to movement between breeding and moulting populations in the two areas. This is a crude metric for the productivity of a population of seals and may be influenced by changes in the timing or the pattern of haulout during the moult. It does however indicate that demographic rates, or our indices of those rates, are changing and require further investigation.

Growth

If harbour seal dynamics are the consequence of resource limits, e.g., because of reduced prey density or increased competition, it is likely that the growth rates of individuals would carry some signal of those effects. Resource limitations are likely to result in slower growth and later age at sexual maturity.

A comprehensive length-at-age dataset for UK harbour seals spanning 30 years, was investigated but showed no evidence for major differences, or changes over time in asymptotic length or growth parameters from fitted von-Bertalanffy growth curves, across all regions (Hall *et al.*, 2019). However, the power to detect small changes was limited by measurement uncertainty and differences in spatial and temporal sampling effort. Asymptotic lengths at maturity were slightly lower than published lengths for harbour seal populations in Europe, the Arctic and Canada, with females being on average 140.5cm (95% CI, 139.4, 141.6) and males 149.4cm (147.8, 151.1) at adulthood.

This lack of signal is in contrast to data from Danish and Swedish harbour seal populations. Comparison of somatic growth curves of 2,041 specimens with known age, length and population size at birth showed that while all populations were similar in 1988, by 2002 there were clear differences between populations (Harding *et al.*, 2018). While seals in the Kattegat showed similar asymptotic lengths as in 1988, seals in the Skagerrak were significantly shorter. Asymptotic lengths of both male and female harbour seals declined by 7 cm. The restricted growth may have been related to relative foraging densities of seals, which were three times greater in the Skagerrak compared to the Kattegat. The authors suggest that reduced growth in the Skagerrak may be an early signal of density dependence.

Genetics

Genetic data from a study directed toward resolving patterns of population structure of harbour seals from around the UK and adjacent European sites (Olsen *et al.*, 2017) has recently been added to (with funding from Scottish Natural Heritage) and combined with the population trend and telemetry data to investigate source-sink dynamics of harbour seal populations.

DNA samples were collected from approximately 300 harbour seals at 18 sites throughout the UK and the Wadden Sea (Olsen *et al.*, 2017) and were genotyped at 12 micro-satellite loci. Results suggested three distinct groups, one in the south equivalent to Southeast England SMU and the Wadden Sea, and a northern cluster that was further divided into a north-western cluster equivalent to the West Scotland, Southwest Scotland and Western Isles SMUs, and a north-eastern cluster equivalent to Shetland, Orkney, Moray Firth and the East Scotland SMUs.

The UK harbour seal population can be divided into similar regional sub-divisions to those seen in the genetics data on the basis of the observed population trends. The southern UK population equivalent to the English east coast shows continual rapid increase punctuated by major declines associated with PDV epizootics in 1988 and 2002. Populations along the East coast of Scotland and in the Northern Isles have generally declined while populations in western Scotland are either stable or increasing.

Nikolic *et al.* (2020) reported an analysis of the genetic structure of the Moray Firth harbour seal population. Their analysis revealed that the Moray Firth cluster is a single genetic group, with similar levels of genetic diversity across each of the localities sampled. Their estimates of current genetic diversity and effective population size were low, but they conclude that the Moray Firth population has remained at broadly similar levels following the population bottleneck that occurred after post-glacial recolonization of the area.

Carroll *et al.* (2020) used a combination of population trends, telemetry tracking data and UK-wide, multi-generational population genetic data to investigate the dynamics of the UK harbour seal metapopulation. Their results indicate that the northern and southern groups previously identified by Olsen *et al.* (2017) represent two distinct metapopulations. Carroll *et al.* (2020) also examined the dynamics of the northern metapopulation before and after the declines in the early 2000s. They identified two putative source populations (Moray Firth North Coast and Orkney, and Northwest Scotland) which provided recruits to three sink populations (East Coast, Shetland and Northern Ireland). Their results indicated a recent metapopulation-wide disruption of migration coincident with the start of the declines.

Steinmetz *et al.*, (2021) used mitochondrial DNA from 123 harbour seals in Ireland and Northern Ireland and 289 seals from the UK and Europe to investigate population structure. They identified three genetically distinct Irish populations characterised by high genetic diversity, in North-western and Northern Ireland (NWNl), South-western Ireland (SWI) and Eastern Ireland (EI). SWI and EI populations were genetically distinct from UK/European populations, but the NWNl population was indistinguishable from the northern UK metapopulation, with evidence of significant migration from Northwest Scotland to NWNl.

<p>5. What are the latest SAC relevant count/pup production estimates for the harbour and grey seal SACs, together with an assessment of trends within the SAC relative to trends in the wider seal management unit/pup production area?</p>	<p>MS Q3</p>
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The most recent survey data and descriptions of trends in harbour seal counts for all SACs in Scotland and England are presented in SCOS-BP 21/03. Grey seal pup production estimates and descriptions of trends at all SACs in Scotland and eastern England are presented in SCOS-BP 21/03. The relevant count/pup production estimates for SACs together with an assessment of potential trends (increasing, stable (i.e., flat), decreasing, and depleted (stable at a reduced level)) relative to SMU-wide trends in Scotland are shown in Table 8. SMU-wide trends in harbour seal August counts, and grey seal August counts and pup production have been estimated for Scotland (and for eastern England; see Russell *et al.* (2021)).

For grey seal SACs, the August and pup production trends were based on examination of the August aerial survey counts and pup production estimates, respectively.

Because the August counts of grey seals are inherently variable, it was not possible to assess potential trends for SACs with relatively small counts. Many grey seal SACs were designated on the basis of their breeding colonies, and do not host large haulout numbers.

For harbour seal SACs, potential trends were assessed on the basis of estimated trends up to 2017 (Thompson *et al.*, 2019) supplemented by more recent counts where available. The counts/pup production estimates for the SACs are displayed in Russell *et al.* (2021; Figure numbers as per the relevant SMU). A more detailed examination of harbour seal counts within both Scottish SACs and SMUs is given in Morris *et al.* (2021).

Harbour seals

Information on the available data, trend analyses and comparisons with survey data for adjacent areas up to 50km from the SAC together with similar data and analyses for all SMUs in Scotland form part of a report to NatureScot that will be published in 2021. For information the SAC relevant sections of that report were summarised in SCOS-BP 20/05.

Dynamics of SAC populations of harbour seals vary (see SCOS-BP 21/03, and Table 8 below and answer 1 above). Comparisons of the time series of harbour seals counted within SACs compared with numbers found within a 50km range show that SACs are not reliable indicators of trends in the wider population. This is especially evident for the Sound of Barra SAC, where harbour seal numbers have declined dramatically since the 1990s. In contrast, surrounding areas have seen a significant increase in numbers. To varying degrees, all SACs now represent a smaller proportion of the wider population than in the past.

Recent counts in the Wash and North Norfolk SAC show a dramatic reduction. The 2019 count was 27% lower than the preceding 5-year average. Preliminary results from 2020 suggest that this was a real decrease. SCOS have highlighted this population as a priority for additional research and increased monitoring.

Grey seals

A small number of grey seal breeding sites are designated as SACs and use pup production as a condition indicator. Trends in pup production in those SACs were described by Russell *et al.* (2019) and are briefly described here.

Treshnish Isles SAC (Inner Hebrides) produced over a third of the pups born in the Inner Hebrides in the late 1980s. Until the mid-1990s, the trend in pup production within the Treshnish Isles SAC mirrored the regional trend, after which pup production in the SAC showed indications of a gradual decline. From 2010 to 2016, the SAC produced approximately 25% of pups born in the Inner Hebrides.

Monach Isles SAC (Outer Hebrides) produced 79% of the pups born in the Outer Hebrides in 2016. As a consequence, the Outer Hebrides pup production trend closely mirrors the trend seen at Monach Isles which showed an increase of 7.4% p.a. (CIs: 6.3, 8.4) between the mid-1980s and mid-1990s before levelling off as the pup production approached an asymptote.

North Rona SAC (Outer Hebrides) used to be the biggest colony in the Western Isles (c. 2,000 pups in 1960s and 1970s), but has declined since 1995 at a rate of 5.1% p.a. (1995- 2010: CIs: 4.2, 6.0), with fewer than 400 pups born in 2016. Many of the other historical colonies in the Outer Hebrides underwent similar decreases in pup production (e.g., Causamul: -8% p.a. (CIs: 6.8, 9.3); Haskeir: 3.3% p.a. (CIs: 2.4, 4.1)). More recently, Gasker also declined (-4% p.a. (2000-2010; CIs: 3.87, 2.7, 5.3)). Conversely, newly established colonies (e.g., Berneray, Mingulay and Pabbay) in the south of the region increased.

Faray & Holm of Faray SAC (Orkney) produced approximately 15% of the pups born in Orkney in 2016. Pup production within the Faray & Holm of Faray SAC increased at a rate of 9.4% p.a. (1987-1995; CIs: 7.5, 1.4) reaching a maximum of 3,840 pups in the late 1990s before decreasing at a rate of 2% p.a. since 2000 (CIs: 0.8, 3.2). Production in Orkney reached an asymptote of 18,000 to 19,000 pups in c.2000 and has been stable ever since.

Isle of May SAC (East Scotland) The pup production in the central North Sea has increased since 1987 at an average rate of 5% p.a. between 1987 and 2010 (CIs: 4.4, 5.5). However, rates of increase at the three main colonies vary. Production at the Isle of May increased exponentially at 9.9% p.a. (CIs: 7.5, 12.3), since surveys began (1979), before reaching an asymptote of c.2,000 pups in the late 1990s.

Berwickshire and North Northumberland Coast SAC (East Scotland & Northeast England). Pup production in the Berwickshire & North Northumberland Coast SAC is continuing to increase and does not show any indication of reaching an asymptote. However, this SAC contains two large, discrete grey seal breeding populations with different histories and different recent dynamics. The Farne Islands have been an important breeding site since the Middle Ages, while Fast Castle is a recently established breeding site first colonised in the 1990s. Pup production at the Farne Islands increased from the beginning of the surveys in the 1950s until the mid-1970s, when production fell rapidly likely due to a series of culls (Summers, 1978) between 1967 and 1985 (pre-cull pup production between 1956-1965: 7.5% p.a.; CIs: 6.5, 8.5). Production increased at a slower rate of 4.2% p.a. in recent years (2005 – 2014; 95% CIs: 3.2, 5.2).

The Fast Castle colony has continued to increase at a rate of 16.9% p.a. (CIs: 15.2, 18.7).

Pembrokeshire Marine/ Sir Benfro Forol SAC. Pup production at Skomer, on the Marloes Peninsula and at the monitored sites on Ramsey Island have all increased (see SCOS-BP 20/04 for details and data sources). This increase persists despite significant bycatch that exceeds current PBR estimates for the wider SW British Isles population of grey seals (see answer 11 & 14 for detailed discussion).

Table 8. Latest harbour (8.a.) and grey (8.b.) seal data for Special Areas of Conservation (SACs) in Scotland by Seal Management Unit (SMU). SMU numbers also refer to the relevant Figure number in Russell *et al.* (2021). The trends are potential for each SAC and estimated for each SMU.

8.a. Harbour seal

SMU	SAC	Latest August count (year)	Potential SAC trends	SMU trend	
2	West Scotland	Ascrib, Isay and Dunvegan SAC	712 (2017)	stable	increasing
		Eileanan agus Sgeiran Lios mor SAC	238 (2018)	stable	
		South-East Islay Skerries SAC	706 (2018)	stable	
3	Western Isles	Sound of Barra SAC	132 (2017)	depleted/declining	increasing
4	North Coast & Orkney	Sanday SAC	77 (2019)	declining	declining
5	Shetland	Mousa SAC	7 (2019)	declining	depleted
		Yell Sound Coast SAC	209 (2019)	stable	
6	Moray Firth	Dornoch Firth and Morrich More SAC	62 (2019)	declining	stable/increasing
7	East Scotland	Firth of Tay and Eden Estuary SAC	37 (2020)	declining	limited data, likely declining

8.b. Grey seal

SMU	SAC	August counts			Pup production (latest data 2019)			
		Latest count (year)	Potential SAC trends	SMU trend	Latest estimate	Potential SAC trends	SMU trend	
2	West Scotland	Treshnish Isles SAC	160 (2018)	Not examined	increasing	1131	stable	stable
3	Western Isles	Monach Islands SAC	2701 (2017)	stable	stable	12511	stable /increasing	stable /increasing
		North Rona SAC	175 (2014)	Not examined		286	declining	
4	Orkney & North coast	Faray and Holm of Faray SAC	228 (2019)	Not examined	stable/ increasing	2186	stable/ declining	stable
7	East Scotland	Isle of May SAC	40 (2016)	Not examined	stable	1885	stable/ declining	increasing
		East Scotland component of Berwickshire and North Northumberland Coast SAC*	71 (2018)	Not examined		4499	increasing	

* The boundary of this SAC transects the Fast Castle colony. Here we have included all pup production within the total for the SAC.

<p>6. The frequency of grey seal surveys in some areas of Scotland are likely to be reduced in future years. Can SCOS advise on what a reduction in survey effort would mean in terms of the confidence of population estimates?</p>	<p>MS Q4</p>
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Reducing survey frequency will likely lead to an increase in the confidence intervals but is unlikely to substantially change the mean estimates. It is considered that the effects of further reducing survey frequency should be compensated to some extent by inclusion of additional independent estimates. Although estimating the population size is important, estimating trends and detecting changes in those trends is arguably more important. Rapid detection of changes in dynamics at appropriate spatial scales is essential for effective management of anthropogenic effects. The effect of reducing survey frequency in a stable population will be less than for a rapidly changing population. Reduced survey frequency may increase the time taken to detect changes.

Any decision to reduce survey frequency will take into account the need to maintain a good understanding of current trends and should, where possible, include an appropriate power analysis. A revised analysis of the likely effects will be carried out as part of the planning and decision-making process before any change in survey frequency is implemented.

Thomas & Harwood (SCOS-BP 05/3) investigated the effect of reducing the frequency of pup production estimates by re-fitting a suite of population dynamics model to a reduced data set comprising pup production estimates from 1984, 1985 and alternate years from 1987 to 2003. The predicted total population sizes for 2004 were similar to the estimates obtained using the entire dataset. However, the posterior credibility intervals were noticeably wider. In 2010 the monitoring programme was reduced to biennial surveys. Reducing the frequency further will likely lead to a further increase in the confidence intervals, but is, again, unlikely to substantially change the mean estimates.

It should be noted that the previous analysis showed only a limited impact of effectively halving the data. This was a worst-case scenario as the reduction in survey frequency only affects the later part of the time series. The models are fitting to an unbroken time series from 1984 to 2010 and biennial surveys since 2010 as well as to the future data. The model now also fits to three independent estimates of the grey seal population size, and this time series will be updated every five years. The effect of reducing survey frequency to biennial has apparently been compensated for by the inclusion of the independent estimates and by the extra data points since 2010. The approximate CV of the 2010 estimate of the overall UK population, based on pup production to 2009 and including one independent estimate was 0.12. The approximate CV of the 2018 estimate, based on pup production estimates up to 2016 (including three biennial surveys) and including two independent estimates was 0.065. This suggests that the effects of further reducing survey frequency should be compensated to some extent by inclusion of more independent estimates in future.

Although estimating the population size is important, e.g., for quantifying interactions with fisheries or industrial activities, estimating trends and detecting changes in those trends is arguably more important. Rapid detection of changes in dynamics at appropriate spatial scales is essential for effective management of anthropogenic effects. In such cases, comparisons are complicated by the fact that different populations are showing different dynamics and the effect of reducing survey frequency in a stable population will be less problematic than for a rapidly changing population. Any decision to reduce survey frequency will be an attempt to target the available survey resources more effectively, e.g., by reducing frequency of surveys in regions that are showing little change and

concentrating effort where rapid change has been observed or is expected. A revised analysis of the likely effects will be carried out as part of the planning and decision-making process before any change in survey frequency is implemented. Wherever possible such a reduction in survey frequency should be compensated with increased use of alternative information such as independent (i.e., not derived from pup production time series) estimates of population size and demographic parameters.

<p>7. Could SCOS provide advice on the most appropriate multiplier to use when estimating an all age population size from pup production in the Southwestern British Isles (including Ireland) region.</p>	<p>NRW Q3</p>
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The main source of uncertainty in the Southwestern population estimate is the absence of reliable pup production data for a large proportion of the Welsh grey seal population. Any population estimates and resulting PBR values derived from the existing pup production estimates should be treated with caution.

In the absence of either an independent estimate of total population size, or a time series of pup production estimates for Welsh grey seals, a method is required to convert single pup production estimates to total population size. Several conversion factors could be used, but SCOS identified the ratio of pup production at regularly monitored colonies in Scotland and eastern England to a population estimate derived from a population dynamics model as the most appropriate method. For future PBR estimates, SCOS recommend a count of seals in August, to align with the rest of the UK would be the best option, if possible.

The scalar for estimating 1+ age population from pup production based on the population dynamics model was 2.32 (CI 2.15 – 2.50). However, this includes additional uncertainty in the recent pup production estimates. A more conservative scalar of 2.08 (CI 1.93 – 2.24), based on the 2010 ratio between pup production and population estimates, avoids this additional uncertainty.

In the absence of data on the distribution and abundance of seals in Wales and Southwest England Seal Management Units (SMUs) outside the breeding season, a scalar has been used to generate total population and N_{min} estimates from estimated pup production in those SMUs. However, there are no time series of comprehensive/reliable pup production estimates for Wales or Southwest England with which to fit a population model to predict population size. An approximate population estimate has been proposed based on a multiplier, derived from the pup production and total population estimates from the regularly monitored populations in Scotland and the North Sea. In addition, the rationale for combining the Irish population with the Welsh and Southwest English populations is unclear; these are unlikely to form either a closed or fully mixed population.

The most recent nationwide estimate for pup production in Wales and SW England is 2,700 pups. derived from counts/estimates at indicator sites and a scaling factor (approximately 2) to convert the sum of these indices to total pup production (SCOS-BP 20/04). Thus, approximately half of this estimate is based on counts from the 1990s and an assumption that those sites have increased in line with the other half for which a time series of counts are available (SCOS-BP 20/04). There does not appear to be any information to support that assumption. The most recent published estimate for Ireland is 2100 pups based on pup counts carried out between 2009 and 2012.

SCOS are concerned that pup production estimates for sites that are currently thought to hold approximately half of the total Welsh grey seal pup production are based on 30 year old counts and that pup production estimates for Ireland are based on 10 year old data. The estimated pup production should therefore be treated with extreme caution. An analysis of newer pup production and population data from Ireland covering the period 2013-2017, and for which summer haul-out

count data have also been gathered in 2017-18, is ongoing and may help to inform this subject for future SCOS deliberations.

In the absence of comprehensive summer haulout survey data SCOS recommend a scaling factor for estimating total population size from pup production using the ratio of pup production to the population estimate derived from the population dynamics model (SCOS-BP 21/05) for the rest of the UK grey seal population. Pup production for the regularly monitored colonies was 60,700 in 2019. The model generated population associated with those colonies was 140,900 (95% CI 130,600-151,600). This produces a scalar of 2.32 (CI 2.15 – 2.50).

However, this estimate includes a large uncertainty due to the step change in pup production estimates associated with the change in methodology after 2010. To avoid that additional uncertainty, using the ratio of pup production to total population estimate from 2010 would be a safer, i.e., more precautionary approach. This would produce a pup production to total population scalar of 2.08 (CI 1.93 – 2.24).

Notwithstanding the concerns over the uncertainty in pup production estimates, these scalars could be used for calculating PBRs. The same process can be used to estimate approximate scalars from pup production to N_{min} equal to the lower 20th percentile of the distribution of the population estimate. The scalar/multiplier for pup production to N_{min} derived from the current population dynamics model is approximately 2.24. Using the 2010 ratio would produce a scalar of 2.00.

However, SCOS again stress that these numbers are speculative given the absence of a comprehensive pup production estimate for over 30 years. Using the ratio between overall pup production and population size for the rest of the UK is also problematic. We do not have an estimate of the growth rate for the Welsh population and the growth rate strongly influences that ratio. As a result, SCOS again urge extreme caution when applying these all-age population estimates for seal management.

As there are no new comprehensive pup production data and no comprehensive summer survey data, SCOS recommend leaving the $F_R = 0.5$. Although there is a perception that the Welsh population may be increasing slowly, CCW previously recommended setting the F_R to 0.5 based on uncertainty in population status and the use of parameter estimates from other populations (SCOS 2016 Q9). There are detailed time series for some of the larger sites, but there is still a great degree of uncertainty because a potentially sizeable proportion of the population is effectively uncounted, so the uncertainty has not decreased.

<p>8. Are there any technologies (existing or new/emerging) that could be considered as an alternative to aerial surveys that could help meet Net Zero aspirations, or does the method currently used remain the most appropriate vehicle?</p>	<p>MS Q5</p>
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New survey techniques are continually assessed for the potential to reduce the environmental costs and health and safety risks associated with SMRU’s aerial survey programme. Despite improvements in resolution, satellite imagery does not have the required resolution for species differentiation and for differentiation of different classes of seal pups.

Unmanned Aerial Vehicles or drones are becoming more affordable and reliable and offer the potential to carry out surveys in poorer weather conditions at lower level than fixed wing aircraft or helicopters. However current limitations of battery life, payload weight and legislation limiting use to line of sight limits the extent to which drone technology could replace the current aerial survey approach. The very large extent of individual colonies, often several kilometres, the

number of colonies that require synoptic surveys and the large distances between them render current drone technology unsuitable. SMRU will continue to monitor the capabilities and legislation surrounding drone use. Despite this, drones have significant potential to provide data to supplement SMRU's regular monitoring and collect specific information at individual colony level.

Other options to reduce the environmental impact of the aerial survey programme would be to reduce the frequency of surveys and/or to have the plane used for grey seal breeding survey stationed at Dundee airport throughout the season.

Efforts to reduce the carbon footprint of the existing surveys continue. From 2021 all east coast harbour seal surveys will be conducted using a single engine Cessna 172 aircraft. Improved manoeuvrability at slower speeds and ability to use local grass landing strips has improved survey efficiency and reduced fuel consumption by approximately 70%.

The Sea Mammal Research Unit continually review the capabilities of new techniques to conduct accurate, safe, efficient, and cost-effective population surveys. The need to reduce the environmental effect of research is also a driver for the investigation of new techniques.

The increasing resolution of satellite imagery has provided opportunities to assess wildlife populations from space (McMahon *et al.*, 2014, Bamford *et al.*, 2020). However, satellite-derived methods have difficulty resolving smaller or camouflaged animals. The best available resolution of 30 cm per pixel makes it feasible to count individual seals on sand (Moxley *et al.*, 2017), but does not allow the differentiation of seal species or different classes of grey seal pups on sand or even the detection of seals on rocky shorelines. Even though it is possible to count individual seals on some satellite images, the frequency at which usable imagery (highest resolution image of a specific location during low tide) would become available is unknown. Figure 7 shows the recent imagery available on Google Earth for a popular grey seal haul-out site at the mouth of the Ythan estuary, north of Aberdeen. Although large numbers of grey seals are visible on all six images taken since May 2016, only the image taken on 28th June 2018 has a resolution that allows most individuals to be counted confidently.

Another technique that is under continual review is the development of unmanned aerial vehicles (UAV) such as quadcopters and fixed wing aircraft, also known as drones.

It has only been in the past few years that commercially available drones have become affordable and reliable for professional use, allowing researchers to conduct highly detailed aerial surveys on a routine basis (Dickens *et al.*, 2021). However, these remain limited in terms of battery life, and associated flight time, and payload weight for camera equipment. Currently, consumer drones and most multi-rotors are limited to flight times of <45 min, while fixed-wing drones are limited to <2 h. For monitoring behaviours that may extend beyond battery capacity, drones require battery replacement that interrupt monitoring.

Existing legislation requires line-of-sight operation (up to a maximum horizontal distance of 500 m) which means that the operator would have to launch/operate the drone from multiple locations to cover individual large grey seal breeding colonies that extend over several kilometres. Most of the colonies would only be accessible by boat or helicopter. Biennial grey seal pup production surveys involve 4-5 repeated aerial surveys of around 70 colonies spread out over a large area across Scotland and eastern England. The area requiring coverage has recently increased, both in extent and geographical spread, to incorporate the growing colonies on the east coast of England. The size of the colonies and the distances between areas covered within a single survey campaign are too large to be covered by currently available UAV technology and within existing legislation in a cost-effective manner.

During SMRU harbour seal moult surveys, a few hundred kilometres of coastline are surveyed during a single 4h low tide window on each day. This reduces the potential for movement between haul out

sites during surveys. These surveys are often in remote and hard to reach parts of Scotland, involving convoluted and complex rocky coastlines where seals are found using a thermal imaging camera. It is not currently possible to replicate this approach with drones as this would require transport between areas by vehicle and boat or helicopter and would take many more days.

Despite these limitations, drones have significant potential to provide data to supplement SMRU's regular monitoring and would be a highly effective means to replace ground counts at individual colony level at specific locations. Drones also have potential as a technique for detailed investigation of specific research questions with methods such as photogrammetry-based estimates of body condition and size distribution, photo identification, evidence of entanglement etc.

In conclusion, whilst consumer-grade drones offer significant potential for improving our ability to monitor a number of features at individual colonies or haul-outs, there is not yet sufficient operational ability to replace the current approach of using manned aircraft to achieve the extent and scale of the current UK wide seal monitoring programme. In the foreseeable future, emissions of greenhouse gases could only be reduced by further reducing the frequency of surveys or by having the aircraft used for grey seal pup surveys based at Dundee Airport throughout the season. However, the capabilities of affordable UAVs are continually developing. Therefore, SMRU will continue to review the capabilities of UAVs and other emerging technologies to identify potential future reductions in the environmental impact and in the risk of methods implemented in the current monitoring programme.

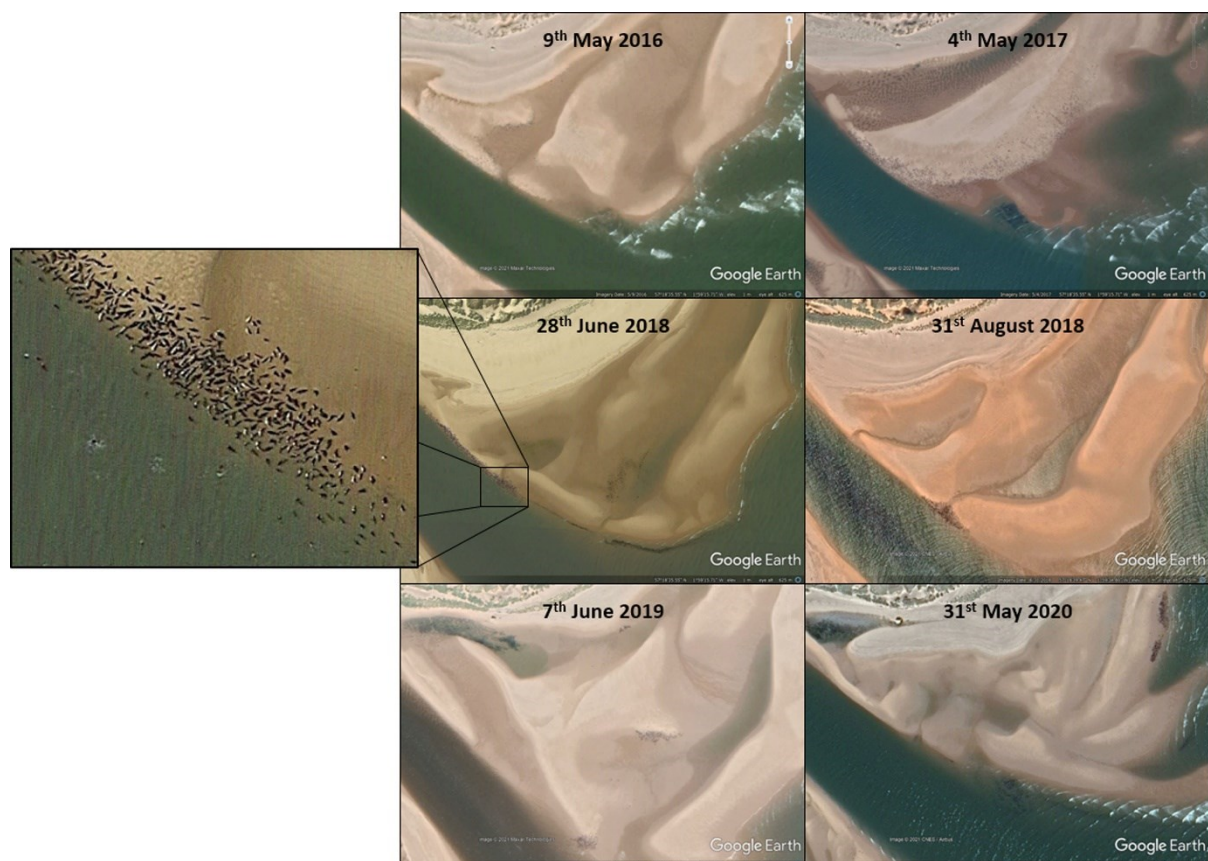


Figure 7. A grey seal haul-out site at the mouth of the Ythan Estuary, north of Aberdeen, shown on the six most recent satellite images available on Google Earth. Large groups of seals are visible on all images, but individual seals are only clearly identifiable on the image taken on 28th June 2018.

Harbour Seal Decline

9. In the 2020 advice, SCOS provided a view on the current potential (major) drivers of the harbour seal decline and their status. Can SCOS provide an updated assessment on these in light of ongoing work.	MS Q10a
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The causal mechanisms of the harbour seal decline have not been identified, but several factors have been rejected as primary causes of the decline, although these may remain as potential secondary causes. Table 9 contains a list of potential factors involved and the current assessment of their importance (modified from SCOS 2020). A few critical factors still remain that require further research, including reduction in prey availability, competition with grey seals for prey resources, predation by grey seals and by killer whales, and exposure to toxins from harmful algal blooms.

The Sea Mammal Research Unit has been funded by the Scottish Government to investigate the causes of the declines. A summary of the progress and initial results of the programme was presented in SCOS-BP 20/06. Previous and recent work conducted during Phase II of this project (which was completed in early 2020) suggest that toxins from HABs may increase harbour seal mortality, based on a bio-energetic model estimating the range of likely daily toxin doses ingested by harbour seals (risk assessment model). A recent publication from Phase II describes concentrations of toxins from HABs in fish species sampled in Scotland, with the highest domoic acid (DA) concentrations measured along the east coast of Scotland and Orkney, and peaks of both DA and Paralytic Shellfish Toxins consistent with phytoplankton bloom timings (see Kershaw *et al.*, 2021). Phase III of the project aims to increase the number of prey samples during HAB events to update the risk assessment approach and compare data on toxin concentrations during and outside HAB events. Phase III of the project also continues to focus on the estimation of survival and fecundity rates at sites of contrasting population trajectories with an extended dataset (2016 to 2022 with a gap year for 2020 due to covid-19 pandemic). Two SUPER DTP funded projects started in 2019 and in 2020, which are addressing inter-species competition and the effect of grey seal predation on regional declines, and killer whale predation on harbour seals.

For information, Table 9 contains a list of potential drivers of decline (proximal and ultimate) and the current assessment of their importance (modified from SCOS 2020). A confidence level (high, medium, low) has been added to each of the potential drivers to reflect uncertainty regarding the assessment of their importance in the observed declines based on the evidence available.

It is recognised that different factors may be implicated in the declines in different SMU populations and that there is no guarantee that the list in Table 9 is comprehensive. Unidentified factors may be important in some SMUs.

Table 9. The current view of the potential proximate causes (9a) and ultimate causes (9b) of the observed declines in harbour seals in some areas (Orkney, East Coast, MF), with an indication of their likely importance as drivers and of the level of confidence in that assessment.

9.a.

	Proximate Causes	Importance status	Confidence level	Evidence	Additional information
1	Reduced adult survival	Likely	High	No direct estimates of adult survival for declining UK harbour seal populations. In some regions, e.g., Orkney and the Firth of Tay and Eden SAC, the decline is too rapid to be solely due to reduced recruitment so adults must have been removed from the population.	Preliminary adult survival estimates for study sites in Orkney and Skye should be available for SCOS_2022
2	Reduced pup survival	Possible	High	Hanson et al (2013) found no difference in survival between stable (West Scotland) and declining (Orkney) populations. However, reduced pup survival/recruitment is thought to be a likely driver of seal populations	
3	Reduced fecundity	Possible	High	No time series of population scale pup production estimates for any declining populations and therefore no evidence to identify changes in fecundity. Preliminary results from ongoing study suggests that pregnancy rates at a site in Orkney were lower than at Skye, Moray Firth or Pentland Firth sites but differences were non-significant ³ .	Preliminary fecundity estimates for study sites in Orkney and Skye should be available for SCOS_2022
4	Increased juvenile dispersal	Possible	Medium	Carroll <i>et al.</i> (2020) suggest significant historical migration from the MFNCO ⁴ local population to Shetland and East Scotland SMUs. Study concluded that migration from Orkney and Moray Firth has reduced since the onset of the decline. Telemetry data showed significant movement of pups from Orkney to adjacent SMUs ³ but no information on temporal trends in such movements.	
5	Increased adult emigration	Possible	Low	Telemetry data have little power to detect emigration. Existing data do not indicate large scale movement between SMUs (Sharples <i>et al.</i> , 2012), although temporary relocation between Orkney, Moray Firth and Shetland SMUs has been observed ⁵ .	

³ Marine Mammal Scientific Support to Scottish Government. HSD 2 Annual report Year 4. <http://www.smru.st-andrews.ac.uk/research-policy/reports-to-scottish-government/>

⁴ MFNCO is Moray Firth and North Coast local seal population and encompasses both the Moray Firth, and the North Coast and Orkney SMUs.

⁵ SMRU unpublished data

9.b.

	Ultimate Causes	Importance status	Confidence level	Evidence	Additional information
1	Nutritional stress	Possible	Medium	Recent analysis of body condition and nutritional health in live captured animals shows no evidence of link to population trends (Kershaw <i>et al.</i> , in press). However, samples represent survivors and may be biased, so power to detect starvation effects could be low.	
1a	Prey quality and availability	Likely	Medium	Coincidence of declines in seabird productivity in N & E Scotland thought to be due to prey abundance or availability.	
1b	Competition with marine predators	Likely	Medium	Competition for prey with the increasing grey seal population and/or other marine predators cannot be ruled out.	Ongoing SMRU based PhD project on grey seal competition.
2	Predation	Likely	Medium	Predation by grey seals (Brownlow <i>et al.</i> , 2016) and killer whales (Deecke <i>et al.</i> , 2011) reported at several locations. Estimates suggest that killer whale predation rates may be high in Shetland. Historical monitoring data insufficient to show causal link.	2 ongoing PhD projects currently investigating predation by grey seals and killer whales.
3	Toxins from harmful algae	Possible	Medium	No direct evidence of large-scale mortality events from strandings in areas of decline. Domoic acid, saxitoxins and okadaic acid continue to be detected in harbour seals (Jensen <i>et al.</i> , 2015) and their prey (Kershaw <i>et al.</i> , 2021). Historical data are insufficient to show correlation with the observed declines, but wide geographical scale and likely severity mean that HABs cannot be ruled out as a contributing factor	
4	Infectious disease and parasites	Possible	Low	No direct evidence of large-scale disease events from strandings or live capture data in areas of decline. Coincident onset of declines and the 2002 PDV epizootic is unexplained but could indicate some chronic effect. Other disease agents (e.g., Mouth rot outbreak in Eastern England) cannot be ruled out as contributing factors. Higher mortality rates among rescued juvenile harbour seals in recent years in the SEE-SMU.	
5	Climate change: indirect effects	Possible	Medium	Changes in prey distribution and/or availability or increases in harmful algal blooms or increased disease prevalence as a consequence of climate change are likely to impact harbour seal populations in future (covered in 1,3 & 4 above)	
6	Climate change: direct effects	Unlikely	High	Observed changes in physical environment in UK waters do not exceed harbour seals' adaptive capabilities. No evidence of major changes in Scotland coincident with the observed declines.	

	Ultimate Causes	Importance status	Confidence level	Evidence	Additional information
7	Fisheries bycatch	Unlikely	High	Data from bycatch observer programmes and absence of major gillnet fisheries in regions of decline suggest that bycatch is unlikely to have been a significant factor in the declines.	
8	Persistent Organic Pollutants	Unlikely	High	Levels of persistent organic pollutants (PCBs, DDTs and PBDEs) are low in the areas of decline and highest in regions where populations have been increasing since 2002 (Hall & Thomas, 2007).	
9	Targeted killing	Unlikely	High	No evidence of targeted killing at levels that could account for observed declines. Prior to the Marine Act 2010 there was no requirement to report shooting of seals in Scotland. Although there are no comprehensive records, legal shooting of seals was not thought to be sufficient to account for the numbers of seals lost during the early stages of the declines. Shooting can be ruled out in some SMUs e.g., East Scotland, but cannot be ruled out as a possible contributory factor in other SMUs.	Licensing under the Marine (Scotland) Act 2010 ensures protection of declining populations from directed takes. As a result of legislative changes in 2021, killing seals to protect fisheries or aquaculture is no longer allowed in the UK.
12	Disturbance at haulout sites	No	High	Possible local re-distribution effects. Most sites are remote and rarely disturbed. Occasional and/or localised disturbance unlikely to have significant population scale effects. Population trends at sites with high levels of tourism/military aircraft activity and offshore renewable energy developments show no signs of negative impacts at the population level.	
10	Loss of habitat	No	High	Data from aerial surveys and telemetry studies show no evidence that foraging, moulting or breeding sites have been lost.	
13	Entanglement in marine debris	No	High	Entanglement in marine debris is not a major recorded cause of mortality in stranded harbour seals in Scotland. There were no known changes in fishing practice likely to have increased entanglement coincident with declines.	
14	Macroplastics and microplastics	No	High	Data from stranded seals and faecal samples indicate that ingestion of macro- and microplastics is currently not a major issue for UK seals at the population level and can be ruled out as a driver of the observed declines.	

<p>10. Can SCOS also advise whether the observed declines occurring in the south east of England could assist with providing answers to the Scottish position?</p>	<p>MS Q10b</p>
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SCOS consider that there are some interesting parallels between the observed declines in southeast England and some areas of Scotland, but also significant differences between regions that provide both opportunities and challenges. The English population decline was noted at an earlier stage and the combination of annual time series of population estimates and pup counts for the southeast England and adjacent Wadden Sea populations may provide additional insights into the changes in trajectories.

In both cases, the magnitude of the declines suggests they cannot be fully explained by a decrease in fecundity and/or juvenile survival alone, and that there must be a decline in adult numbers. Some of the same potential drivers for the declines overlap between the two cases and investigation of these should provide information that will be relevant to the question of harbour seal declines throughout the UK and Europe. However, there are differences between the SMUs in Scotland and in the southeast of England regarding population trends, genetics, ecology and environment which might limit the ability to provide informative answers.

There are some interesting parallels, but also significant differences. It is difficult to predict whether investigations into the apparent decline in the southeast England Seal Management Unit (SEE-SMU) will be able to inform the regional declines in Scotland. The decline in counts of harbour seals in the South East England Seal Management Unit (SEE_SMU) are as yet unexplained. SMRU are currently developing proposals and research projects to investigate the declines and it is hoped that insights derived from such studies will inform the harbour seal decline issue in several Scottish harbour seal populations (Russell *et al.*, 2021). In both cases, the magnitude of the declines suggests they cannot be fully explained by a decrease in fecundity and/or juvenile survival alone, and that there must be a decline in adult numbers. There is some overlap in the potential drivers for the declines in the two regions. Investigation of diet, foraging behaviour, movements, interactions with human activities and interactions with competing predators should provide information that will be relevant to the question of harbour seal declines throughout the UK and Europe. Also, the sudden onset of the decrease in the SEE_SMU after a period of rapid growth may help identify potential drivers and/or make it easier to exclude factors that could not have caused the decrease.

However, there are differences between the SMUs in Scotland and in the southeast of England regarding population trends, genetics, ecology and environment which might either limit the ability to provide informative answers towards explaining the declines in Scotland or provide informative comparisons that help identify or exclude potential drivers.

Genetically, harbour seals in the SEE-SMU are significantly different from all other UK harbour seal SMUs and are considered as part of a different metapopulation together with continental Europe harbour seals (including those in the Wadden Sea) (Carroll *et al.*, 2020). The harbour seals in the SEE-SMU have historically undergone sustained increases in abundance, punctuated by sudden declines during the Phocine Distemper Virus (PDV) outbreaks in 1988 and 2002. These sustained increases contrast with the trends in the SMUs along the east coast of Scotland and the Northern Isles, all of which have recorded declines in harbour seal numbers differing in intensity since 2002 after generally stable population trajectories (Thompson, Duck, Morris, & Russell, 2019). A drop in adult numbers can be caused by increased adult mortality and/or emigration. Emigration is not considered as a major factor contributing to the decline in harbour seal numbers in SMUs in Scotland (SCOS, 2020). but in the case of SEE-SMU, emigration cannot be ruled out as the large size of the adjacent harbour seal population in the Wadden Sea might not allow detection of such potential immigration.

Potential causes of the apparent decline in the SSE-SMU are unknown at this point but include disease, biotoxin exposure and nutritional stress (Russell *et al.*, 2021). Disease outbreak and nutritional stress have been ruled out as main factors driving the decline in Scottish SMUs given that there is no evidence of large scale mortality events from strandings data, live captures show no evidence of disease in areas of decline, and recent analysis of body condition and nutritional health in live captured animals shows no evidence in areas of decline in Scotland either (Hall *et al.*, 2019; Kershaw *et al.*, in press). However, biotoxin exposure from Harmful Algae Blooms (HABs) remains as a potential driver of the declines in SMUs in Scotland too, and is currently being investigated. Russell *et al.* (2021) recommend biological sampling of harbour seal adults and juveniles from the SEE-SMU to investigate health-related drivers in the decline. This would include investigation of HAB toxin concentrations in captured seals. Any results from such investigations might inform the Scottish declines or more generally to get a better overall picture of the exposure of harbour seals to HABs in the UK.

The population trajectories and particularly the monitoring information from the SEE_SMU populations are quite different to those for Western Scotland and the Northern Isles. In the SSE_SMU the main harbour seal population has been surveyed annually since 1988 and there are annual indices of pup production for that population since the 2002 PDV epidemic. Until recently the SEE_SMU harbour seal population has shown continuous rapid growth as it recovered from the effects of severe hunting pressure in the 1960s and early 1970s, and two PDV epizootics in 1988 and 2002. Since 2000 there has been a dramatic, rapid increase in the grey seal population both in terms of the pup production and the summer foraging population throughout the SEE_SMU (16.5% p.a. increase over the last two decades; Russell *et al.*, 2019) until 2019 when it too appears to have levelled off and possibly begun to decline (SCOS-BP 21/03). The role of grey seals in the apparent harbour seal declines warrants consideration (Russell *et al.*, 2021). The high temporal resolution population data available for this region may allow identification of relationships between harbour seal population trends and changes in grey seal population trajectories or changes in other natural or anthropogenic factors.

Around Scotland, regular surveys began in the mid-1990s and are at much lower temporal resolution. Harbour seal populations were relatively stable until several populations began to decline around the turn of the century (SCOS-BP 2019/03; Thompson *et al.* 2019). The grey seal populations around north and west Scotland had either already stabilised, by the mid-1990s as in the Inner and Outer Hebrides, or around 2000 in Orkney, before regular harbour seal monitoring surveys began (SCOS-BP 21/01, Russell, Morris, Duck, Thompson, & Hiby, 2019; Thomas *et al.*, 2019)). There are therefore no systematic or reliable harbour seal population estimates available before the local grey seal populations either reached or approached carrying capacity (SCOS-BP 21/03). If grey seal populations are a major driver of harbour seal dynamics it may be that density related effects were already in place before monitoring began in Scotland. The SEE_SMU time series of population data may provide an opportunity to examine this possibility.

Differences in the timing and scale of natural and anthropogenic changes in the different regions may help identify likely drivers or exclude unlikely factors. For example, the dramatic increase in the construction and operation of offshore wind farms in the SEE_SMU (SCOS-BP 21/03) predates the onset of the decline, whereas large scale construction of wind farms in Scottish waters post-dates the onset of declines in harbour seal populations and there has been no offshore wind development in the areas where the largest declines have occurred. Windfarm developments could have short term impacts on seal distribution during pile driving activity (Russell *et al.*, 2016) and the presence of structures could also impact harbour seals, although these impacts are less clear and may be complex. Assessing the potential impact of changes in the anthropogenic landscape on seal populations in SEE-SMU should benefit the understanding of the potential drivers behind the harbour seal decline.

11. Can SCOS review, present and provide a view on the available evidence on the differences in genetics between the declining and the stable/increasing harbour seal populations.

MS Q11

Carroll *et al.* (2020) reported significant genetic differentiation between most harbour seal SMUs and identified that stable/increasing regions (West Scotland and the Western Isles SMUs) were part of a different metapopulation than declining regions (North Coast and Orkney together with Moray Firth SMU). Carroll *et al.* (2020) also detected a recent metapopulation-wide disruption of migration coincident with the start of the declines and concluded that the northern metapopulation appears to be in decline.

There are no significant differences in heterozygosity levels and inbreeding coefficients between contrasting populations in Orkney and Skye (Bhuta, 2021).

Ongoing work comparing harbour seal genomes globally may shed some further light on differences between populations with contrasting trends.

Carroll *et al.* (2020) used a combination of population trends, telemetry tracking data and UK-wide, multi-generational population genetic data to investigate the dynamics of the UK harbour seal metapopulations. The data comprised microsatellite genotypes from samples collected at UK SMUs between 2003 and 2012, including samples from declining and stable/increasing SMUs, as well as a number of samples from outside the UK and described in Olsen *et al.* (2017). Their results indicated that the UK comprises two distinct metapopulations (northern and southern). The southern group comprises the Southeast England SMU (SEE-SMU) and continental Europe, and the northern group comprises all other SMUs (Northern Ireland and Scottish SMUs). These are in agreement with the two main groups previously identified by Olsen *et al.* (2017). Thus, the harbour seals from the SEE-SMU are genetically distinct from those in the Scottish SMUs, regardless of their population trend.

Within the northern metapopulation, Carroll *et al.* (2020) found significant genetic differentiation between most of the harbour seal SMUs (although not as much as between the two metapopulations). However, not all SMUs were genetically distinct from each other and some could be grouped into local populations, suggesting a total of five local populations: 1) Northern Ireland (declining); 2) Northwest (West Scotland and the Western Isles SMUs, both stable); 3) MFNCO (Moray Firth, where a continued decline is not evident but no signs of recovery to pre-2002 levels, and North Coast and Orkney SMU (declining)); 4) Shetland (continued decline not evident but no signs of recovery to pre-2002 levels); and 5) East Scotland (declining)

Carroll *et al.* (2020) also examined the dynamics of the northern metapopulation before and after the declines in the early 2000s. They identified two putative source populations (Moray Firth and North Coast and Orkney, and Northwest Scotland) apparently supporting three likely sink populations (East Coast, Shetland and Northern Ireland). They also detected a recent metapopulation-wide disruption of migration coincident with the start of the declines and concluded that the northern metapopulation appears to be in decay.

Nikolic *et al.* (2020) reported an analysis of the genetic structure of the Moray Firth harbour seal population. Their analysis suggests that the Moray Firth cluster acts as a one genetic group, with similar levels of genetic diversity across each of the localities sampled. Their estimates of current genetic diversity and effective population size were low, but they concluded that the Moray Firth population has remained at broadly similar levels following the population bottleneck that occurred after post-glacial recolonization of the area.

More recently, samples collected from live-captured harbour seals from Orkney (n = 15, declining site) and Isle of Skye (n = 15, stable/increasing site) in 2016 and 2017 were analysed as part of a Master’s thesis at the University of Auckland, to conduct population genetic analysis using low-coverage whole-genome sequencing (Bhuta, 2021). The results showed that individuals from Orkney and from Isle of Skye had similar heterozygosity levels and that inbreeding coefficients were negative or low, with no significant differences between the two populations. This suggests inbreeding is not the probable cause for decline in abundance in Orkney. The analysis also confirmed the previous genetic population structure results that found the two regions were genetically distinct with a much larger suite (>100K) of genetic markers (single nucleotide polymorphisms). This work is ongoing with some further re-analysis planned.

Liu et al. (2022) compared harbour seal genomes from different world-wide locations. They showed that harbour seals evolved in the Northeast Pacific and the results have implications for harbour seal subspecies delineation, but there are no direct management implications for UK harbour seals.

Seal Protection, Management and Conservation measures

<p>12. Can SCOS advise at what point a decline in grey or common/harbour seal abundance would trigger a change in Natural England’s Conservation Objectives for SAC’s from “maintain” to “restore”?</p>	<p>Defra Q1c</p>
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The appropriate criteria for the magnitude of a decline that would trigger a change in conservation objectives from maintain to restore depends on a variety of factors, therefore it is difficult to determine a ‘one size fits all’ approach that would be applicable across all SACs. Considerations include the regularity of monitoring, the amount of historical data, the variability in previous surveys, and the trends in other parts of the population range.

An examination of the existing monitoring data for any particular site, in combination with trends at other sites within the region, will inform the selection of appropriate trigger points.

In the case of the southeast England harbour seal SAC, there has been an observed decline of 21% between the two most recent counts and counts in the preceding 5-year period (SCOS-BP 21/06). A decline of this magnitude and in light of large declines in other parts of their range, should certainly trigger a change in conservation objectives from maintain to restore.

The overall objective for SACs designated for seals are to provide a coherent network of sites to contribute to the maintenance of the overall favourable status of the population. However, this definition is not particularly helpful in the definition of trigger points for individual SACs. For harbour seals SAC site selection has favoured sites that are important both as general haul-out sites and for moulting and pupping. The largest breeding colonies, based on pup production, have been selected for grey seals. This difference in site selection rationale may require a slightly different approach to defining any triggers for management measures. It is likely that a detectable decrease in abundance, either total abundance in the case of harbour seals or pup production in the case of grey seal sites, below some defined reference value, should trigger concern. This involves the definition of two elements: an appropriate reference value, and the magnitude of the reduction below that reference value that would indicate a concern. The duration over which a decline is observed will also have a bearing. The appropriate magnitude of decline for a trigger depends on the variability in the metric of interest and the ability of any monitoring programme to detect a change over and above levels of background variability. More regularly monitored sites are likely to allow quicker identification of declines of smaller magnitudes than less frequently monitored sites. Considerations relating to the

selection of a reference value include the size at the time of designation as well as any historical trends. For example, many seal populations in the UK are recovering from historical exploitation or from disease outbreaks. It is also important to consider the mobile nature of seals and the fact that reductions in one part of their range may be as a result of distributional shifts and therefore any concerns about local declines, and the need for management measures, must be considered in light of regional trends.

Consideration should also be given to the degree to which the population within a given SAC is genetically distinct, and the absolute size of this population relative to some assumed minimum viable population.

Given these considerations, it is not currently possible to define specific trigger points that should apply appropriately across all SACs for both species. An examination of the existing monitoring data for any particular site, in combination with trends at other sites within the region, will inform decisions to trigger further management actions. However, see the answer to Marine Scotland Q6 below for discussion of generic criteria for the designation of Seal Conservation Areas in Scotland.

In the case of the Wash and North Norfolk coast SAC for which harbour seals are a primary designated feature, there has been an observed decline of 21% between the two most recent counts and counts in the preceding 5-year period (SCOS-BP 21/06). A decline of this magnitude and in light of large declines in other parts of their range, should certainly trigger a change in conservation objectives from maintain to restore.

<p>13. Can SCOS provide advice on current analytical methods being conducted by SMRU to help inform UK led assessments for OSPAR M3 & M5 indicators?</p>	<p>Defra Q3</p>
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The UK is leading the assessment for the OSPAR M3 (Seal Abundance and Distribution) and M5 (Grey seal pup production) indicators for the OSPAR Quality Status Report 2023 (QSR2023). These indicators, which were also considered in an interim assessment (<https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/>), are assessed on two temporal scales: long term (since 1992 or the first year of data thereafter) and short-term (six year rolling baseline; in this case 2014-2019) at the scale of individual Assessment Units (AUs; Figure 8). Assessments for abundance and pup production are made against the following criteria: has there been a decline in seal abundance/pup production of > 25% (long-term) or >6% (short-term)? Changes in distribution are considered as a “surveillance indicator; the metrics are described, but not quantitatively assessed, against an assessment value.

The current assessments (for QSR2023) are led by JNCC, with the analysis being conducted by SMRU and the default approach was to follow the methods used in the interim assessment. For the interim assessment, the methods were developed at an expert workshop (2015), with contributors from most Contracting Parties (CPs), and ultimately agreed by OSPAR. The methods are detailed in Hanson and Hall (2015) and Russell, Hanson & Thomas (2016), and summarised in the assessments (<https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/>). For the current assessment, any noteworthy methodological deviations were made with the aim of increasing robustness; these were proposed by SMRU and discussed with the OSPAR Marine Mammal Expert Group (OMMEG). The methods used are largely sufficient to conduct the assessments. However, there are some key caveats that should be considered particularly with a view to increasing the robustness of future assessments.

Brief Summary of methods

M3 Indicator: abundance

Abundance: Harbour seals

In the previous interim assessment, trends were fitted to a subset of the moult count data covering the period 2009-2014, to provide a short-term assessment. Generalised additive (GAMs) or linear models (GLMs) were fitted for the relevant assessment units (AUs). For the QSR2023, the same method was applied to the same AUs (Figure 8). However, this time the whole time-series of counts (up to and including 2021) was used, maximising power and robustness of the short-term assessment. Minor changes to methods included fitting more than one model type (GLM and GAM) within a single time-series, allowing models to select step changes in abundance, and improved calculation of confidence intervals.

Abundance: grey seals

Grey seal abundance is assessed on a single unit scale – essentially the combined AUs indicated by the colour shaded areas in Figure 8. For the interim assessment the age-structured population dynamics model used for the UK (Thomas *et al.*, 2019) was extended to cover other areas (Russell, Hanson and Thomas 2016). This incorporated a time series on pup production as well as an independent estimate of population size in 2008 (independent from pup production derived estimates). While such population modelling allowed a single assessment to be conducted on the appropriate scale, it had some limitations. Briefly these are: (1) despite increasing grey seal abundance in the UK and Wadden Sea (which account for the majority of Northeast Atlantic grey seal metapopulation) the confidence intervals for the interim assessment were very wide, making robust assessment difficult, in part due to limited sample size for some areas; (2) Lack of knowledge regarding the spatial consistency of the scalar representing the proportion of total population hauled out and thus available to count (derived from UK telemetry data) required to convert summer counts to the independent estimates; (3) Limited appropriateness of the population model structure when applied beyond the UK (specifically with regard to dispersal), and (4) Reliance on the robustness of a scalar to generate pup production estimates from peak pup counts (the only data that are provided by some CPs), which is derived from counts and estimated pup production at Scottish colonies (surveyed by SMRU). These limitations meant that extending the UK pup production model across the whole AU became untenable for the current assessment.

The current assessment considered a larger spatial extent, and since the interim assessment, a jump in estimated pup production in Scotland associated with the switch from film (up to 2010) to digital (from 2012) has become evident. This presented an issue for consistency across the AU, and for generating a scalar between peak count and pup production. Furthermore, only a subset of the CPs provided August data that could be used to generate a time-series for the Independent Estimate of population size. For the current assessment, trends in haul out counts were fitted at the scale of the AUs shown in Figure 8. This was conducted within two GAMs– one for August counts (UK and Ireland) and one for spring moult counts (France, Dutch Delta, and Wadden Sea). This allowed two (one August and one moult) predicted trend lines on which to base assessment against the defined criteria.

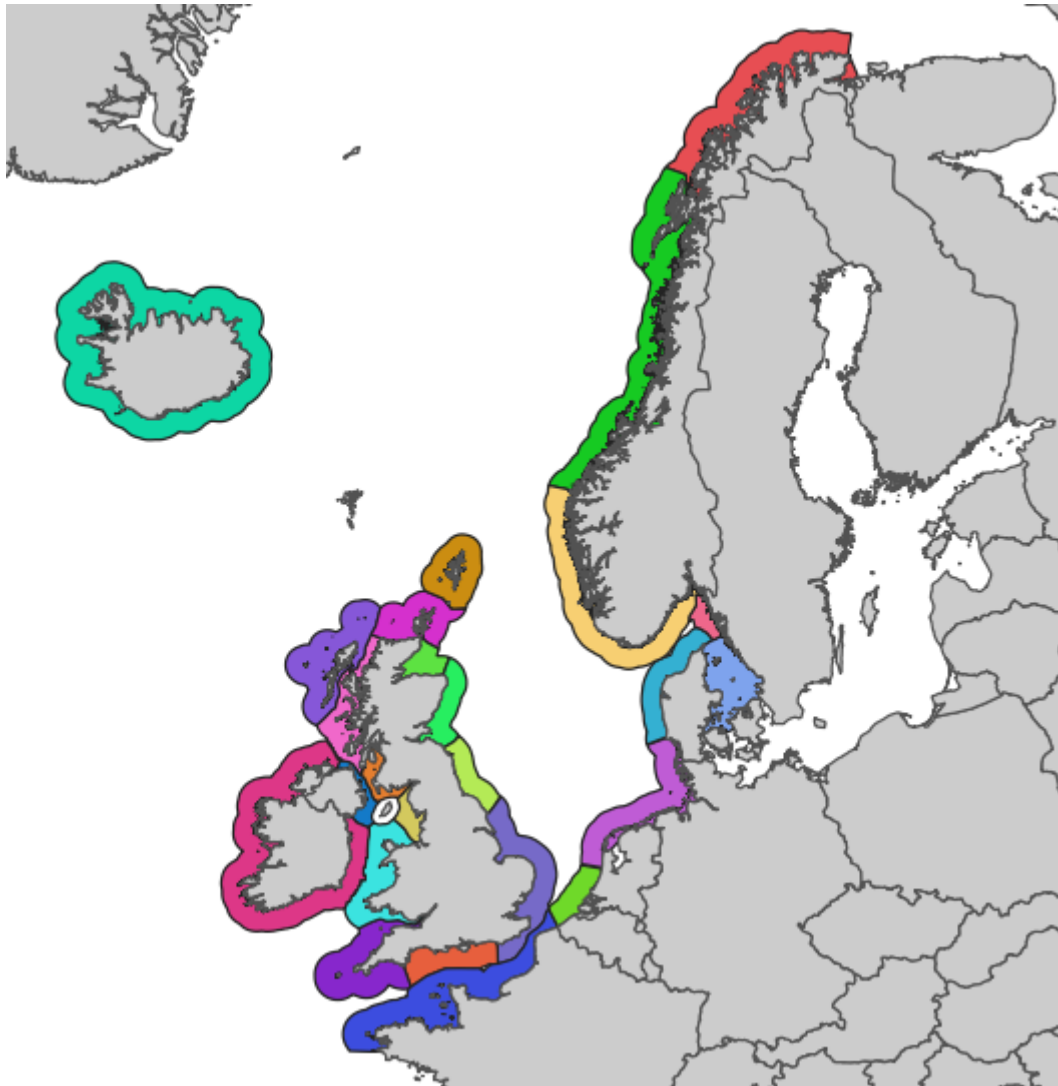


Figure 8. OSPAR Assessment Units used for harbour seal abundance and distribution(M3), grey seal distribution (M3) and grey seal pup production (M5). Grey seal abundance (M3) is assessed on a single spatial unit scale combining all the shaded areas shown.

Distribution: both species

Changes in distribution were considered as a surveillance indicator for both the intermediate and current assessment. A similar set of assessment values as used for seal abundance were suggested for seal distribution, but as meaningful changes in seal distribution are currently difficult to detect and assess from abundance surveys this aspect of the indicator will be considered as a ‘surveillance indicator’. Although the same metrics (see below) were considered in both the intermediate and current assessments, the spatial units considered were different; for the intermediate assessment individual CPs decided the spatial scale at which distribution should be examined for each AU whereas for the current assessment, changes in distribution (presence/absence) were assessed on a 15 km x 15 km grid (where possible). There are two metrics: (i) shift in distribution, representing the change in which specific cells are occupied, and (ii) change in range, which represents the change in the number of occupied cells. These metrics were examined on both a long-term basis with years used being 1992 vs 2019 (long-term assessment) and 2014 vs 2019 (short-term assessment). Where data were not available in the focal years, the closest possible years were used instead.

A number of changes were made to the methods of the intermediate assessment to improve the robustness of the analysis. The key differences were as follows: (1) as far as possible, distribution was considered on the same spatial scale within and across AUs; in the Intermediate assessment, the size of spatial units (i.e., the area to which counts were assigned) varied across AUs, (2) the metrics were only evaluated for years and areas for which there was consistent effort (in terms of coverage) across two periods of interest (which restricted spatial and temporal extent for some AUs); in the intermediate assessment the spatial extent was not always consistent between the two periods, (3) Changes between single years were considered rather than between periods to maximise the temporal separation of periods (e.g., short term assessment: year 1 vs year 6 compared to year 1-3 vs year 4-6 as done in the intermediate assessment).

M5: Grey seal Pup production

In the intermediate assessment, trends were fitted to peak pup counts or pup production (depending on what was provided by CPs), using either generalised additive or linear models. The same method is used for the current assessment except that AUs surveyed by SMRU were fitted within a single GAM (separate smooth for each AU) and a single “jump” in pup production was offered to account for the step change associated with the switch from film to digital surveys.

Considerations

Factors that must be considered for both the interpretation of the current assessments and the development of the methods for the next assessment, are discussed below.

Baselines

The use of both long-term and short-term assessments allow both long-term and rapid declines to be identified. Ideally the baseline year for the long-term assessment would represent the natural level of abundance and distribution. However, in the baseline year (1992), seal populations were recovering from exploitation (and thus potentially reduced compared to historical “natural” levels). Furthermore, a recent Phocine Distemper Virus (PDV) epidemic in 1988 means that some harbour seal populations were suppressed (e.g., Southeast England, Wadden Sea). Finally, many monitoring studies did not extend as far back as 1992 and thus there is considerable variation between AUs in the period that the long-term assessment covers. The long-term assessment for both the intermediate and current assessment compares the latest population estimates with the 1992 baseline estimate. So, the comparison for the current assessment is over a longer time period than the comparison for the intermediate assessment. As a consequence, an assessment of a 25% change equates to a slower rate of change for the current assessment compared to the intermediate assessment. A simple solution would be to compare average annual rates of change.

Consistency of data

Different CPs have different protocols for data collection, and for some CPs there are multiple programs and thus data contributors. This makes combining data within and across areas (e.g., for grey seal population assessment) difficult and limits the comparability of the results across AUs. For example, harbour seal moult count data provided on AU scale varied between raw counts, maximum counts and mean of multiple counts conducted within a year. The inclusion of all raw counts is preferred to appropriately represent the uncertainty around the estimated mean trend. Provision of raw counts on an AU scale is hampered by the misalignment between AUs and the scale on which data are collected; some AUs comprise data from multiple countries and monitoring programs. Furthermore, some CPs collect data more frequently for a subset of an AU which contains the

majority of seals; there is no provision for such indicator counts to be submitted for the abundance assessment.

Assessment Units

The scale of the Assessment Unit varies between species; there is a single unit for grey seal abundance. However, for harbour seals (and grey seal distribution and pup production) the AUs are much finer, and the degree to which unit boundaries were driven by biological relevance versus data availability varies spatially. The aim was to make these units as biologically meaningful as possible (i.e., based on the boundaries between sub-populations or haulout groups), but the area that can be covered within a single survey or year is much finer in many places. A pragmatic approach was developed for grey seal abundance for this assessment; trends were fitted at a finer scale (AUs in Figure 8.), and these trends combined, within a model, at a larger scale. However, the validity of such an approach is reliant on these smaller units representing relatively discrete groups of haulouts with little inter-annual variation in the degree of net movement between them.

SCOS consider that further discussion is needed to assess the appropriateness of a single AU for grey seals. A single AU used without clear consideration of the structure of the metapopulation and systems for monitoring and assessment at scales capable of dealing with localised population change should be re-examined. A briefing paper addressing the effects of combining all AUs for management of the grey seal metapopulation will be presented to SCOS 2022.

Grey Seal abundance

Grey seal abundance was assessed using counts during the harbour seal moult surveys in August (UK and Ireland) and during the grey seal moult in March/April (continental Europe). The majority of the European grey seal population is in the UK where grey seal moult surveys are not routinely conducted; such surveys during winter/spring would not be logistically or financially feasible. Furthermore, the degree to which the clockwise cline observed in breeding applies to the moult is not clear and there is no information on the proportion of seals hauled out at any time during the moult. Thus, in order to generate a single assessment of grey seal trends in abundance, counts during the August harbour seal moult for all AUs would be required. Most countries conduct surveys of harbour seals in August, and also count grey seals. CPs should be encouraged to (1) ensure all grey seal sites are covered during the harbour seal moult surveys, and (2) to report these counts on the smaller AU scale (Figure 8). As well as allowing a single trend to be estimated for all considered OSPAR regions and removing inaccuracies associated with seasonal redistribution, this would also ensure data for grey seal abundance were incorporated for additional units in continental Europe where no dedicated grey seal moult counts are conducted, but August harbour seal moult surveys are (e.g., Kattegat and Limfjord).

Distribution

There are multiple considerations when examining changes in distribution including: (1) minimising the impact of day-to-day variability in haulout locations, (2) variation in spatial survey effort between years, (3) size of distributional units, (4) number of spatial units, and (5) density within units. All of these also need to be considered when interpreting the metrics. For example, due to variation in the spatial extent of AUs, a change of -10% in one AU may represent an absence at just one haulout site (which may be a site not frequently used), whereas the same result may represent variation in > 10 haulout sites in another AU. Thus, any qualitative assessment should be based by reviewing the metric value and the associated maps underlying the metrics in combination. In future assessments, a weighting of distribution by density would potentially result in a more robust indicator than the current presence/absence approach. Furthermore, the size of the spatial units

used in reporting should be consistent between AUs. In particular, this analysis would be more robust if data were provided on a standardised grid, including the spatial extent of surveys where possible, such that survey effort could be appropriately accounted for. In the current assessment, the extent to which all data could be used was limited by variation in the format and spatial resolution of survey effort reported by CPs. Inconsistencies between counts and reported survey coverage meant that some data could not be used, as it was not possible to get a robust indication of presence/absence.

Indicator Areas

In some AUs, the whole unit is routinely covered during each survey whereas in others (many UK AUs) the complex coastline and the distribution of sites means a subset (often the most populous) gets surveyed more frequently. For AUs surveyed by SMRU in the UK, for which the time series pertaining to the whole AU was restricted in temporal extent or number of data points, a time series of a subset of the AU was generated. Trends were then fitted to this subset as an indicator of likely trends in the whole AU. Generating these subset time-series was a complex process and thus only possible because the raw data and expertise were available in-house. Although SMRU also developed AU subset time-series for France and the Dutch Delta, it was not possible for other areas (due to lack of availability of the required fine scale data or the complexity of the data). For some AUs, there were too few data points to conduct any analyses (e.g., Norway). In the next assessment, the data call should encourage submission of time-series of subsets of AUs where appropriate.

<p>14. Can SCOS review and comment on the biological management perspective of seal management units proposed by the Inter-Agency Marine Mammal Working Group (IAMMWG)?</p> <p>Discussions are ongoing within the Inter Agency Marine Mammal Working Group (IAMMWG) to define the structure of spatial reference areas within which management might be applied to UK harbour seals and grey seals. For harbour seals, we suggest that the MU's should be based on those assessment units that are presently used by SCOS for monitoring the UK population.</p> <p>There are broadscale challenges associated with the connectivity of grey seal metapopulations around the UK coast when it comes to considering the species for management advice and decisions. To best account for this, and still enable a reasonable level of appropriate management to be advised upon, two units reflecting an east/west divide across the Celtic and Great North Sea are proposed for grey seals. The units largely represent a reference population and management of activities within either unit may require reference to refined spatial scales using evidence on local/regional dynamics where available. What is SCOS's view on this.</p>	<p>Defra Q4</p>
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The 14 current UK seal SMUs were defined initially for harbour seals on the basis of distribution of haulout sites and for pragmatic reasons including the ability to survey an SMU within one season. For grey seals the same arguments apply, without inferring discrete populations. So, for pragmatic reasons the structure was accepted for both species.

The IAMMWG does not propose to change the structure for harbour seals but has proposed revision of those assessment units for grey seals, splitting the UK population into two, an eastern

UK unit (North Sea, and Northern Isles) and west (Hebrides, West Scotland, Irish Sea, Wales and Southwest England).

SCOS consider that there is no strong justification for these proposed units and are concerned that this division implies that the grey seals on either side of the UK can be considered as distinct, isolated populations. There is a clear danger that such a split would provide justification for including the large, combined population for an area in assessment of or justification for activities that may have only a local effect. Although there is clear evidence of wide-ranging movements by some individual grey seals, many grey seals remain within single SMU throughout the years, so local concentrations persist and may not recover rapidly from local effects.

For any management issue or potential impact, the correct procedure would be to identify the SMU populations involved/at risk and use the combined population estimates available for those SMUs, weighted in an appropriate fashion.

The terminology used may be a factor in this debate. The terms Seal Management Unit or Seal Management Area may imply that those groupings should be managed on a stand-alone basis. This was never the intention. The false impression that these are explicit management divisions could easily be solved by renaming them as Seal Monitoring Units, with a clear understanding that these practical monitoring units can and should be combined in appropriate ways in response to the management question being addressed.

The UK Statutory Nature Conservation Bodies (SNCBs) require an understanding of the geographical range of populations and subpopulations, in order to provide advice on the assessment of impacts and management at the most appropriate spatial scale. As part of the need to meet such requests, Natural Resources Wales commissioned work on defining management units in Welsh waters (Evans, 2012) and the Scottish Government commissioned similar work by Sea Mammal Research Unit (SMRU) for Scottish waters (Northridge, 2012). The 14 current SMUs in the UK were defined initially for harbour seals on the basis of distribution of haulout sites and for pragmatic reasons including the ability to survey an SMU within one season and the locations of jurisdictional boundaries. For grey seals the same arguments apply, without inferring discrete populations. So, for pragmatic reasons the structure was accepted for both species. These SMUs are used as a subset of the 21 SMUs used for harbour seals in the OSPAR assessments of seal management in the northeast Atlantic.

The IAMMWG have not proposed any alteration to the management structure for harbour seals. However, for grey seals there is a perception that wide ranging movements of individual seals makes the current structure unnecessarily fine structured.

For grey seals, ICES (2014) proposed two assessment units: (1) North Sea (Region II) and (2) western Britain, Ireland and western France (Regions III and part of IV). These were issued as ICES Advice to OSPAR (ICES 2014b) noting that grey seals range widely at sea such that these two units were not independent, and that grey seals visit multiple distant haul-out sites, although mature seals of both sexes are usually faithful to particular breeding sites (ICES, 2014). For the OSPAR intermediate assessment of seal abundance and distribution in 2017, a single assessment unit for the entire European area was used. This has been adopted for the UK and is the structure used in the forthcoming OSPAR assessment (see answer to Defra Q3) although sub-units equivalent to those for harbour seals are also recognised. However, it should be noted that these assessments are not designed to be used for developing specific management actions. It is not immediately apparent what management function these assessment units address other than for general/comparative assessment of population status on large geographical scales.

The IAMMWG have proposed a revision of the assessment units for grey seals based on current understanding of the presence and structure of biological populations, with divisions proposed based on ecological evidence and/or divisions used for the management of human activities. The proposed units therefore comprise partially artificial divisions of biological populations., splitting the UK population into two, an eastern UK unit (North Sea, and Northern Isles) and west (Hebrides, West Scotland, Irish Sea, Wales and Southwest England). The division, which would apply to waters within the UK EEZ, would be delineated in the north by a line running north from Cape Wrath and, in the south, by a line running across the English Channel to the east of Normandy. These units have been based on IAMMWG's understanding of the presence and structure of biological populations, with divisions proposed based on ecological evidence and/or divisions used for the management of human activities and therefore comprise partially artificial divisions of biological populations. Again, it is not immediately apparent what management function these assessment units address.

SCOS consider that there is no clear justification on grounds of biological population structure for splitting UK grey seal populations in any particular way. Where telemetry data are available, they show substantial movement between adjacent and less frequent movement between distant SMUs. The proportion of seals moving between non-adjacent SMUs is small and the majority of tracked seals remained within one SMU for the duration of the tracking. SCOS consider that there is a danger that because local concentrations of seals will likely persist, and that movement from distant SMUs is limited, anthropogenic effects may have disproportionate impacts on the local populations. Assessing such effects against pooled populations from much larger AUs is likely to underestimate their impacts on local populations.

However, if there is a pressing need to define two separate management units, then the proposed division is justifiable on the basis that movements of seals between the west (West Scotland and Western Isles SMUs) and the east (North Coast and Orkney SMU) are apparently less frequent than movements between those SMUs and adjacent SMUs to their south. However, there is likely to be significant mixing and movement between west and east SMUs around northern Scotland. Again, in the English Channel there is limited evidence of east-west movement in the limited telemetry data available.

SCOS are, however, concerned that presenting such a structure gives the impression to interested parties that the grey seals on either side of the UK can be considered as distinct, isolated populations. There is a clear danger that such a split would provide justification for including the large, combined population for an area in assessment of or justification for activities that may have a significant local effect.

SCOS do not see any particular advantage in splitting the UK population in this way. Presumably the drive for such large management units is the need to manage issues/interactions such as the bycatch of grey seals off Southwest Britain, where a mismatch between the scale of the bycatch and the available information on seal population size suggests that immigration from distant SMUs must be occurring (see answer to Q22). In this and any other case where a wider management issue is involved the correct procedure would be to identify the SMU populations involved/at risk and use the combined population estimates available for those SMUs, weighted in an appropriate fashion.

The terminology here may be a factor in this debate. The terms Seal Management Unit or Seal Management Area may imply that those groupings should be managed on a stand-alone basis. This was never the intention. The false impression that these are explicit management divisions could easily be solved by renaming the current Seal Management Units as Seal Monitoring Units, with a clear understanding that these practical monitoring units can and should be combined in appropriate ways in response to the management question being addressed.

<p>15. Could SCOS please advise on the locations of the top two breeding sites and top two haul out sites for both harbour seal and grey seal in each Seal Management Unit? Could SCOS also comment on whether the top two sites have been consistent over the last 5 years, or whether there is interannual variability?</p>	<p>Defra Q5a</p>
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There is a lack of appropriate data to allow the identification of the top two breeding sites and haul out sites for both species of seal in each SMU. Whilst the available data will allow moult sites to be identified for harbour seals, breeding sites are not comprehensively monitored with counts only available for limited areas. For grey seals, breeding sites are comprehensively monitored in the main breeding sites in Scotland and on the east coast of England, but less so in the SW of England and Wales. There are no data on the moult distribution of grey seals.

Additional difficulties relate to the definition of 'site'. The sites covered by monitoring range in size from small groups of haulout sites through to substantial sections of coast lines.

An approach similar to that used to designate seal haul outs in Scotland could be used to identify the largest haulout groups in England where data exist. However, the individual sites identified by such a method do not generally match up to the scale of the current site designations for Sites of Special Scientific Interest (SSSI) or SACs, which, in general, cover much larger areas. Notwithstanding these caveats, a review of the main sites in each SMU is presented below.

SCOS cannot answer this question in its current form for two reasons: lack of appropriate data and a clear definition of what constitutes a site. These issues are described below, together with a brief description of the current protected site designations that apply to breeding and haulout site/groups/colonies for both species.

Insufficient data.

Harbour seals

The data available for harbour seals comprises time series of counts of seals hauled out during the annual moult around the UK coast. All of the coast is covered with the exception of SW England and Wales where very small numbers of harbour seals are reported. The temporal resolution of these data range from annual surveys of the South East England SMU (SEE_SMU), the Moray Firth SMU (MF_SMU), the Firth of Tay and Eden Estuary SAC, and the Tees estuary, to a five-yearly cycle for the remaining SMUs around the coast of Scotland. Some additional surveys have been carried out where local population changes required attention (details of survey programme and the counts are presented annually to SCOS with the most recent data for South East England in SCOS-BP 21/06 and SCOS-BP 21/07 and for the whole of the UK in SCOS-BP 20/03. Notwithstanding issues with definition of 'site', these data would allow identification of and ranking of harbour seal moulting sites as requested.

Harbour seal breeding season data are limited. Counts are available for the SEE_SMU, a time series of data exist for the coast between Donna Nook and Scroby Sands for the period 2001 to 2018 (SCOS-BP 19/04) and there are two pup counts from 2011 and 2018 for the rest of the SEE_SMU. Time series of counts are also available for the MF_SMU, and the Tees Estuary, which is the only harbour seal breeding site in the NEE_SMU, for similar periods.

Surveys to assess the breeding distribution of harbour seals have not been carried out around the rocky shore coastlines of Scotland because of the cost and the practical difficulties of detecting and identifying pups on such haulout substrates. There are therefore no pup production estimates or pup counts for any other harbour seal SMU populations. It is not therefore possible to identify the most important harbour seal breeding sites anywhere outside the Moray Firth and the Wash.

Grey seals

There are detailed time series of pup production estimates for all the main breeding sites in the Inner and Outer Hebrides, Orkney, the Firth of Forth colonies, the mainland colonies in North Scotland and Berwickshire and colonies along the east coast of England. In Scotland, surveys were carried out annually from 1987 to 2010 and biennially since 2010. Colonies in England are surveyed annually. Sporadic counts are available for Shetland colonies. Some sites in Wales are surveyed annually, e.g., Skomer and the Marlowes MCZ, Bardsey Island, and parts of Ramsey Island, but the rest of Wales and the south-west England SMU are surveyed infrequently. Any identification of the top sites for breeding would be restricted to these regularly surveyed areas.

There are no data on distribution or abundance of grey seals during their annual moult, except for a single survey of part of Orkney in 2014 and observation data from individual sites scattered around the UK. There are detailed data on the distribution and abundance of grey seals in all areas covered during the harbour seal moult surveys in August (SCOS-BP 20/03). A series of surveys in late winter/early spring around the whole coast would be required to ascertain the distribution of moulting grey seals. It is therefore not possible to define the most important moulting sites for grey seals.

Definition of a site.

There is no clear or agreed definition of a "site". At present there are two types of haulout site definition; ad hoc sites for the purposes of survey reporting and a more rigorous site definition for designating protected haulout sites in Scotland. In addition, SACs and SSSIs have been designated in all SMUs, but the sites covered range in size from small groups of haulout site through to substantial sections of coast lines.

SMRU survey sites

For the purposes of the summer surveys for harbour and grey seals a site is roughly defined as one or more discrete haulout groups in a small area. However, sites range from individual sandbanks or skerries, to small archipelagos such as the Southeast Islay SAC, up to substantial sections of coastline such as Donna Nook or the Berwick and North Northumberland Coast European Marine Site (BNNC_SAC). In practice, how haulout groups are combined is to some extent arbitrary and often based on tradition.

Helicopter and fixed wing surveys around Scotland and NE England have always assigned specific geographical coordinates to all groups of seals counted. However, for sites in the large tidal estuaries where mapping is less precise this was not possible for fixed wing surveys until recently. For example, in the Wash site names and designations were based on historical surveys from the 1960s and '70s when seal hunting was targeted at named sites. We have recently abandoned the allocation of seals to these named sites, and instead identify the location of each group by recording geographical co-ordinates. However, there are still arbitrary decisions on when to split groups that are found on the same sand bank or along the same tidal creek and location accuracy depends on the flight path.

Designated sites in Scotland

For the purposes of designating haulout sites for protection from harassment under the Marine (Scotland) Act 2010 (SCOS-BP 12/07), sites in Scotland were identified and classified using a fine scale distribution map. Virtual Observation Points (VOPs) were placed at 100m intervals along the coast, and sighting histories of both species for each individual VOP were calculated as the sum of all sightings that lie within 300m radii around each VOP. 300m was chosen as an appropriate buffer radius to ensure that all recorded seal sightings are contained within at least one buffer area and to account for some error in positioning of seals and sites in the GIS mapping process. To a limited extent, this also helps deal with the fact that seals don't always haul out at exactly the same spot. A Time Weighted Average (TWA) of each species for each VOP was calculated.

Sites were then defined by drawing a polygon shape by eye around parts of the coast, small islands and skerries that contained seal sightings. Again, this is a somewhat arbitrary process, and the extent of individual sites was influenced by the local distribution. E.g., in some cases, sections of coast with scattered small groups were combined into one site.

That process could be used to identify the largest haulout groups in England where data exist. However, the individual sites identified by such a method do not generally match up to the scale of the current site designations for SSSIs or SACs (European Marine Sites) where seals are qualifying features. In general, such sites in the marine and coastal habitats cover much larger areas.

Existing protected sites

Many sites in the UK are already designated as European Marine Sites/SACs (Figure 9) and/or as SSSIs for seals. There are 9 SACs where harbour seals are a primary feature and seven where grey seals are a primary feature. There are a further 26 where seals are a secondary feature/species of interest. For information we include a description of the currently designated sites in each SMU.

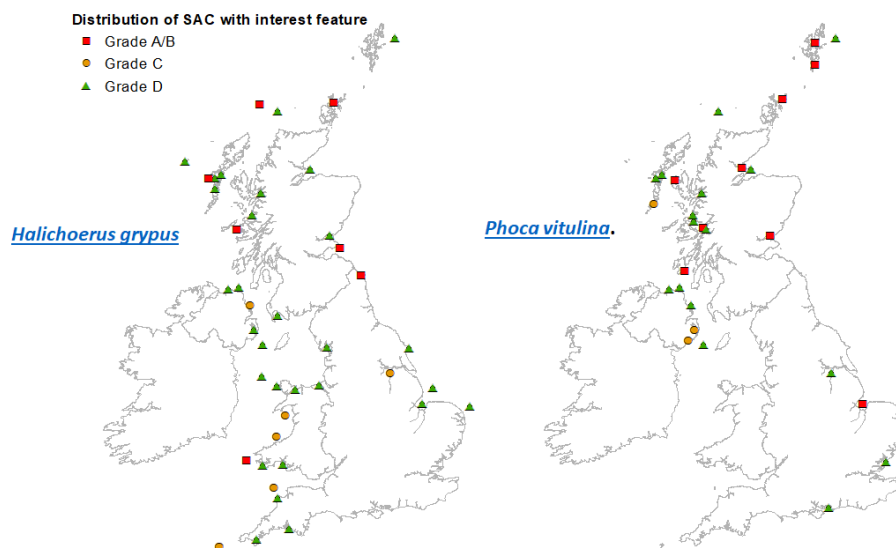


Figure 9. Distribution of SACs/EMSs around the UK that have seals as qualifying or additional features of interest. (JNCC 2021). Site classifications: Grade A - Outstanding examples of the feature in a European context; Grade B - Excellent examples of the feature, significantly above the threshold for SSSI/ASSI notification but of somewhat lower value than grade A sites; Grade C - Examples of the feature which are of at least national importance; Grade D - These features are not the primary reason for SACs being selected.

The following descriptions are based on the most recent surveys described in detail in SCOS_BPs 21/01 & 21/06.21/07,20/04 and 20/05

England

Northeast England SMU (NEE_SMU)

Grey seals: there is only one large grey seal breeding population in the NEE_SMU, at the Farne Islands. This lies within the BNNC SAC and is also itself designated as a National Nature Reserve (NNR) and an SSSI. There are no other significant grey seal breeding sites in the NEE_SMU.

Outside the breeding season there are major haulouts at the Farne Islands and in the Lindisfarne NNR, both of which lie within the BNNC_SAC. The only other large grey seal haulout site in the NEE_SMU is on Coquet Island. This is an RSPB managed bird reserve with no public access and is designated as an SSSI, but seals are not listed as qualifying features.

Harbour seals: apart from a small group at Lindisfarne, numbering less than 5 seals in recent surveys, the only significant haulout group is at Seal Sands in the Tees estuary. Harbour seals are a primary feature of the Teesmouth and Cleveland Coast SSSI.

Southeast England SMU (SEE_SMU)

Grey seals: there are large and rapidly increasing grey seal breeding populations at Donna Nook, Lincolnshire, and Blakeney Point and Horsey Sands in Norfolk.

- Donna Nook is a NNR, and is part of the Humber Estuary SSSI and Humber Estuary SAC.
- Blakeney is a NNR, and within the Wash and North Norfolk Coast SAC, and the North Norfolk Coast SSSI.
- Horsey Sands is an SSSI, but seals do not feature in the citation, probably because of the recent very rapid growth of the colony.

Outside the breeding season (based on summer surveys) the two haulout sites that hold the largest numbers of grey seals in the SEE_SMU are:

- Donna Nook, currently the largest grey seal haulout group in the NE Atlantic population which held around 60% of the SEE_SMU 2020 summer haulout count for grey seals.
- Scroby Sands (SCOS-BP 21/06), which has grown rapidly over the past decade. Scroby Sands is not designated as far as we are aware. The haulout groups are adjacent to and have recently spread into Scroby Sands wind farm as extensive new drying sandbanks have appeared within the farm
- Other large haulout groups occur at Blakeney Point and on sand banks in the northeast corner of the Wash, close to Gibraltar Point NNR. These sites are all within the Wash and North Norfolk coast SAC.
- There is a large grey seal haulout comprising several groups of grey seals on Goodwin Sands, off the Kent coast. The haulout sites are within the Goodwin Sands

Marine Conservation Zone (MCZ), although seals are not a qualifying feature or listed in the citation for the MCZ designation.

Harbour seals: If the Wash is taken as a single site, it is by far the largest harbour seal breeding site. If the Wash population is subdivided, it is likely that two of the subdivisions will be the largest pupping sites in the SEE_SMU. Only small numbers of pups are counted at Donna Nook, Blakeney or Scroby Sands. All haulout sites in the Wash and at Blakeney are within the Wash and North Norfolk Coast SAC

The most recent pup survey of the greater Thames estuary (ZSL 2019), which covers the remainder of the SEE_SMU, produced a count of 128 pups in 2018. These were scattered throughout the inshore banks and tidal creeks. The two largest groups were in Hamford Water (27 pups) and on Buxey Sands (44 pups).

Hamford Water NNR is designated as an SAC and an SSSI although seals do not appear in the citations as features or species of interest. Buxey Sands is adjacent to Foulness SSSI but does not appear to be designated.

South West England SMU (SWE_SMU).

Grey seals: the two largest breeding sites in the SWE_SMU are the Isles of Scilly and Lundy Island. In total the Cornish mainland produces a greater number of pups than Lundy, so the relative importance of Lundy and Cornish mainland sites will depend on the degree to which sites are combined. Grey seals are a designated feature of the Isles of Scilly SAC and much of the archipelago is within one of the 12 MCZs designated in and around the Isles of Scilly. Grey seals are a designated feature of the Lundy SAC and the island is designated as a SSSI and is part of the Lundy MCZ. Grey seals are also listed in the citations for Godrevy to St Agnes and Boscastle to Widemouth SSSIs.

Leeney *et al.* (2012) published the results of a single boat survey of the Cornish coast and Isles of Scilly. Approximately 80% of the hauled-out seals were in the Isles of Scilly. The largest groups on the mainland were recorded at Boscastle, Godrevy and Longships, but only small numbers were recorded, less than 30 seals in each group.

Harbour seals: we are not aware of any significant harbour seal breeding or haulout sites in the SWE_SMU.

North West England SMU (NWE_SMU).

Grey seals: The only known grey seal breeding site is on the mainland in the South Walney in Cumbria. Grey seals began pupping there in the mid-2010s and numbers are increasing. The two largest, and effectively the only large haulout groups in the NWE_SMU are at West Hoyle Bank (often referred to as Hilbre Island) in the Dee estuary, Cheshire, and at South Walney in Cumbria. South Walney is a local nature reserve and the breeding and haulout sites are within the South Walney and Piel Channel Flats SSSI, but seals are not mentioned in the citation. The Hilbre/West Hoyle site straddles the border between England and Wales and lies within the Dee Estuary SAC, and is an SSSI designated for grey seals. *Harbour seals:* we are not aware of any significant harbour seal breeding or haulout sites in the NWE_SMU.

Scotland

Grey seals: In Scotland all significant breeding sites are protected from disturbance and classed as designated haulout sites under the Marine Scotland Act 2010. In addition, some breeding sites in each SMU are within SACs or are specifically designated as SSSIs.

Outside the breeding season a large proportion of the grey seal population is protected at haulout sites that lie within SACs or SSSIs and/or are designated protected haulout sites. These sites have not been monitored during the grey seal moult but were designated on the basis of summer haulout distributions as described above. The array of protected sites has been established using several more nuanced or more flexible criteria than simply selecting the largest sites. However, notwithstanding the differences in criteria, the largest sites in each SMU are included in the listings for at least one of the categories.

Harbour seals: SACs where harbour seals are either the primary reason for designation or are listed as species of interest have been designated throughout Scotland: three in West Scotland_SMU, two in Shetland SMU; one each in the Western Isles SMU, South-West Scotland_SMU, Orkney and North Coast SMU, Moray Firth SMU and East Scotland SMU.

All relatively large haulout sites are designated as protected haulouts under the Marine (Scotland) Act 2010. In addition, Seal Conservation Areas have been designated for Shetland, Orkney and North Coast, Western Isles, Moray Firth, and East Scotland SMUs.

Wales

Grey seals: The largest grey seal breeding sites in Wales are on Ramsey Island, and on Skomer and the adjacent Marloes Peninsula. Both sites are in the Pembrokeshire Marine SAC. Ramsey Island is a NNR, and designated as a SSSI. Skomer and the Marloes form the Skomer MCZ. The next largest site is on Bardsey Island, which is in the Lleyn Peninsula and the Sarnau SAC, of which grey seals are listed as a qualifying feature. The large grey seal haulout site at Hilbre/West Hoyle straddles the border between England and Wales, and lies within the Dee Estuary SAC, and is an SSSI designated for grey seals.

Harbour seals: we are not aware of any significant harbour seal breeding or haulout sites in Wales.

<p>16. Could SCOS advise whether they consider the current guidance on designating the top two sites (as SSSIs) appropriate? Are there any SMUs where this approach would not be appropriate? If this is the case, what approach to protect seals through designated sites would SCOS recommend for these SMUs?</p>	<p>Defra Q5b(i)</p>
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As discussed above, the data are lacking to allow the designation of the top two breeding and moulting sites for each species. Possible solutions include assuming harbour seal breeding season distribution is similar to the moult and for grey seals to assume that moult distribution is similar to distribution during August. This approach will be more robust if protected areas are large enough to cover several haulout sites. In practice, the network of protected sites already appears to cover the required locations.

The current JNCC Guidance document (JNCC, 2021) states that the top two breeding and moulting sites can be designated to protect seals primarily from disturbance during the breeding and moulting seasons. However, that does not take account of the available data. The extensive surveys required to robustly assess the distribution of moulting grey seals and the breeding distribution of harbour seals would require significant additional resource and likely be prohibitively expensive.

In the absence of breeding data for most harbour seal SMUs, a possible compromise for harbour seals would be to assume that the distribution during the breeding season is similar to that during the moult. Observations in the Wash and Moray Firth suggest that pup numbers are lower on the exposed outer banks, but at a reasonably large scale the distributions of breeding and moulting sites are similar.

In the absence of moult distribution data for grey seals a possible compromise would be to assume that the distribution during the moult is similar to the distribution during August, which represents the main foraging season between moult and breeding. A survey of grey seal distribution during the moult in Ireland indicated that seals were concentrated on sheltered sites some of which were not used during the summer. A single grey seal moult survey in Orkney showed substantial changes in relative importance of sites compared to the summer distribution. However, absence of moult data for most SMUs makes this a moot point because the summer distribution is the only available information on distribution of seals on haulout sites outside the breeding season.

These assumptions will be less likely to be violated if the protected areas are large enough to incorporate several haulout sites. In practice, the network of protected sites already appears to cover the required areas/locations, but seals are not currently named as protected features of all of these sites. If possible, adding seals to the designations of these existing sites would increase the protection afforded to seals over their most populous areas.

<p>17. Does SCOS believe that notifying further SSSIs for seal populations at risk will aid in their overall protection? Does SCOS have any recommendations of other approaches to improve overall protection for populations at risk?</p>	<p>Defra Q5b(ii)</p>
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SSSI designations may provide more easily targeted management of threats to seals on those specific haulout sites.

All grey seal populations for which there are comprehensive population monitoring data are either increasing or are at historical maximum population sizes. It is therefore not clear that any grey seal populations in the UK would be considered at risk.

Most UK harbour seal populations of concern are already afforded protection at various levels. e.g., the majority of the SEE_SMU harbour seal population is already protected as a qualifying feature of the Wash and North Norfolk Coast SAC and populations in the northern Isles and along the east coast of Scotland are in designated Seal Conservation Areas and important haulout sites in Scotland are protected from harassment.

Consideration should be given to developing individual based protection, which would avoid some of the problems with identifying appropriate site protection measures.

SCOS considered that SSSI designations may provide more easily targeted management of threats to seals on those specific haulout sites. However, at least in Scotland, SSSIs are terrestrial site designations and of limited value in addressing marine threats. In addition, as they provide

protection at fixed locations any designated sites would have to be large enough to encompass potential local redistribution of seals.

It is not clear that any grey seal populations in the UK would currently be considered at risk. At present, all grey seal populations for which there are comprehensive population monitoring data, are either increasing or the current estimates represent all-time highs.

Harbour seal populations in the northern Isles and along the east coast of Scotland have declined or are continuing to decline. Seal Conservation Areas covering all of these SMUs have been designated and important haulout sites in Scotland are protected from harassment. It is not clear what increased protection would be provided by designating additional SSSIs within the conservation areas.

The recent declines in the SEE_SMU population may indicate that this population is at risk, but there is no clear indication of what is driving this decline. As this population is already protected as a qualifying feature of the Wash and North Norfolk Coast SAC and the majority of seals haulout within the Wash SSSI, it is not clear that notifying additional SSSIs would provide additional protection.

If additional protection was deemed necessary, considering individual based protection would avoid the problems associated with site protection such as relocation or short-term variability in use of haulout sites. Individual based protection is already afforded to seals in Northern Ireland.

18. Can SCOS please advise on how best to determine the “vulnerability of sites” for seals? (with specific reference to SSSI designation).	Defra Q5c
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SCOS have difficulty in interpreting the meaning of “vulnerability of sites” in the context of SSSI designation. The current guidance does not define vulnerability. If it is an important factor in the justification for designation of SSSIs it needs to be more clearly defined.

It’s difficult to determine what is meant by the term “vulnerability of sites” in the context of SSSI designation. The concept of vulnerability is commonly used in species protection and defining conservation status but less commonly reference to site designation. “Vulnerable” is a specific threat category in the IUCN Red list which means that the species is at a high risk of extinction, so it could be interpreted as meaning that additional sites should be designated for populations at risk of extinction or decline.

In the guidance for SSSI designation it states that sites should ideally contain viable populations of the species they support but given the scale of seal populations relative to the scale of SSSI sites, this is clearly not possible. In the case of large or mobile species it is therefore recommended that the overall network of SSSIs should protect a viable population and in this context some sites may be more vulnerable than others in relation to specific threats. For example, some areas have been identified where seals hauled out are particularly vulnerable to disturbance (e.g., certain sites in Cornwall). This could lead to rationale for protecting more sites, especially if specific sites are identified as being vulnerable to a particular threat and therefore in more need of the protection offered by designation. However, this is little more than guesswork in the absence of any further guidance.

Assessment of vulnerability will require information on likelihood and potential severity of hazards as well as likelihood and time course of both response and recovery. Assessment of vulnerability should also recognise geographical and site-specific variation in the degree to which different sites

may be affected by particular threats and provide a mechanism to identify and protect additional sites which have been characterised as vulnerable.

<p>19. In 2019, SCOS advised that scientifically informed criteria were required to establish whether seal conservation areas should be introduced or revoked. Can SCOS advise on what such criteria should consist of? In the absence of such criteria, but noting current population trends, can SCOS advise whether the threat to seal populations still remains in current seal conservation areas, particularly the Western Isles.</p>	<p>MS Q6</p>
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To date SCAs have been introduced for three different scenarios: response to a rapid decline with a clear related anthropogenic threat (Moray Firth and seal shooting for fisheries management); response to a rapid decline with unknown cause (Orkney and North Coast SMU, Shetland SMU and East Scotland SMU); and a response to a protracted decline with unknown cause (The Western Isles SMU).

It is clear that criteria should differ depending on the frequency of monitoring. Proposed criteria for introducing and revoking SCAs are presented below.

The causes of declines in the Northern Isles and along the east coast of Scotland have not been identified and there is therefore no evidence of threats having been removed. While that uncertainty remains, and there is potential vulnerability to a future PDV outbreak, SCOS recommends that existing Conservation Area designations remain in place in the Northern Isles, Moray Firth and East Scotland SMUs.

SCOS considers that the Conservation Area designation for the Western Isles SMU harbour seal population could be removed.

Historical criteria for designating Seal Conservation Areas (SCA)

Under the Marine (Scotland) Act 2010, Scottish Ministers may designate a “seal conservation area” where they consider it necessary to do so in order to ensure the proper conservation of seals. The primary effect of such a designation is that “The Scottish Ministers must not grant a seal licence authorising the killing or taking of seals in a seal conservation area unless they are satisfied that the killing or taking authorised by the licence will not be detrimental to the maintenance of the population of any species of seal at a favourable conservation status in their natural range (within the meaning of Article 1(e) of the Habitats Directive)”.

The definition of favourable conservation status is not particularly helpful in defining scientific criteria for establishing or revoking Seal Conservation Area (SCA) status. It is “defined” with respect to species by Article 1 (i) of the Directive as: “conservation status will be taken as ‘favourable’ when: population dynamics data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitats, and the natural range of the species is neither being reduced nor is likely to be reduced for the foreseeable future, and there is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis.” It is therefore left to the regulator to decide on the relevant criteria for determining favourable conservation status.

Clearly the designation of an SCA is a response to a perceived decline or threat of decline in a population, that may bring it into unfavourable conservation status. To date these have been designated for three different types of decline.

- The Moray Firth order was a response to a rapid decrease in population estimates in a situation where a clear anthropogenic threat was thought to be driving the decline, in that case the threat was shooting as part of fisheries protection measures.
- The Orkney and North Coast SMU, Shetland SMU and East Scotland SMU populations were declining rapidly due to unknown causes.
- The Western Isles SMU population had shown a slow but protracted period of decline due to unknown causes.

Each of these declines was accepted as an appropriate reason for designation and the SCAs have continued in place to date. However, the question remains of what levels of declines would be needed to trigger action.

To a large extent the magnitude of decline required will or should depend on the monitoring programme in each area. In some areas such as the Moray Firth, regular frequent (annual) surveys allowed identification of a rapid decline within the first few years of the problem developing. In the rest of Scotland, the survey frequency is much lower, with surveys approximately every five years. SCOS recommends that different criteria should be flexibly applied to populations that are subjected to different monitoring programmes.

The following section describes suggested criteria for establishing and revoking SCA designations on the basis of discussion at SCOS. However, SCOS believe that the choice of values used to fix the criteria should be a matter for regulators and will depend on the level of risk that is deemed acceptable, and on the ability of the monitoring programmes to detect changes in population dynamics.

Criteria for considering designation of Seal Conservation Areas

SCOS recommends that a formal decision process should be adopted for designating or revoking Seal Conservation Areas. In the absence of any specific anthropogenic activity or natural factors that a decline can be attributed to, SCOS propose instigating conservation measures when observed declines exceed threshold rates of decline. This approach takes into account the frequency of the monitoring and the extent of the observed decline. Here SCOS outlines the proposed decision process but refrains from advising on specific threshold rates of decline.

The decision to designate a Seal Conservation Area should be based on the following:

1. In any situation where an identified anthropogenic activity or natural factor such as a disease event can be shown to be causing or likely to cause a decline in a population, mitigation measures should be established as quickly as possible. Such mitigation may include designation of SCAs.
2. Where there is no a priori reason to suspect a particular anthropogenic or natural threat, conservation measures should be considered when:
 - In populations with frequent/regular surveys, a decline of X% per annum maintained over a three-year period or a decline of Y% over a five-to-ten-year period has been observed.
 - Investigations to identify the cause of the decline should be instigated and SCA designation should be considered.
 - In populations with a five-year survey cycle, successive counts drop by Z%.
 - Additional survey(s) should be carried out as soon as practical to confirm the decline

- and SCA designation should be considered.
- In populations with a five-year survey cycle, counts drop by W% over 3 surveys.
 - SCA designation should be considered.

The values for rates of decline that would trigger SCA designation will depend on the frequency and quality of population monitoring (i.e. on the level of confidence in our ability to detect declines), and on the acceptable level of risk for seal population management. Further discussion between scientists and regulators/managers will be required to determine appropriate values.

Criteria for revoking/removing Seal Conservation Areas

SCAs could be revoked or removed when a population is considered to have recovered, e.g., returned to at least its pre-decline level, and where causes of the decline have been identified as anthropogenic effects, those causes have been removed.

Reasons for existing Seal Conservation Areas

In 2004, in response to local declines in harbour seal numbers, the Scottish Government introduced a Seal Conservation Order (SCO) under the Conservation of Seals Act 1970, to cover harbour and grey seals in the Moray Firth SMU in response to rapid decline thought likely to be the result of shooting to control seal interactions with salmon fisheries. In 2007 additional SCOs were applied to harbour seals in the Shetland, Orkney and North Coast and the East Scotland SMUs. The latter covering the Scottish east coast between Stonehaven and Torness, including the Firths of Tay and Forth. These were in response to large scale apparently rapid declines in populations.

The Conservation of Seals Act was superseded by the Marine (Scotland) Act 2010, and existing seal Conservation Orders were converted to Seal Conservation Areas (SCA) under the Marine (Scotland) Act 2010.

In 2009 SCOS noted a long-term decline (35%) between 1996 and 2008 in the population of harbour seals in the Western Isles SMU, equivalent to a 3% p.a. decline. In response, and after consultation with stakeholders, the Minister designated a Seal Conservation Area for harbour seals in the Western Isles in 2011, with the intention that the order would remain in place until concerns about this local population are resolved.

Assessment of the continuing requirement for existing Seal Conservation Areas

Declines in Orkney and North Coast SMU, and in the Tay and Eden SAC have continued with no sign of recovery. Counts in Shetland fell sharply in the early 2000s and have been relatively stable since, but with no sign of recovery to pre-2000 levels. In the Moray Firth SMU, counts were decreasing at a rate of 5.6% p.a. (95% CIs: 2.5, 8.5) between 1994 and 2000, followed by a drop of c.28% occurring between 2000 and 2003. There is no significant trend in counts from 2003 to the most recent count in 2019 indicating a stable but depleted population.

There is no clear evidence that the threats to those populations have been removed and SCOS therefore recommends that existing conservation orders remain in place in the Northern Isles, Moray Firth and East Scotland SMUs, at least until new count data are available to reassess their status.

A complete survey of the Western Isles SMU, carried out in 2017, produced a count of 3,533 which was the highest recorded count for the Western Isles. The counts decreased between 1996 and 2008 at approximately 3% p.a., but the 2011 count was similar to the 1996 count and the 2017 count was

29.0% higher than the previous 2011 and was 25% higher than the count in 1996. Model selection based on AIC (SCOS BP 21/03) suggested that the survey data are best described by a GAM that indicates a decline from the late 1990s until approximately 2006 followed by a rapid increase until 2017 (Figure 10).

The SCA was designated in response to a gradual decline of 3% p.a. between the mid-1990s and 2008. Since reaching a minimum around 2006-2008 the survey counts have increased by approximately 90%, equivalent to a 7% p.a. rate of increase between 2008 and 2017. This rapid increase clearly indicates that the factors that caused the decline are no longer driving the population down.

Bearing in mind that the main protection bestowed by the Conservation Area designation is an effective ban on licenced removals, and that such removals for fisheries protection have now been banned throughout the UK, SCOS considers that the Conservation Area designation for the Western Isles SMU harbour seal population could be removed without serious risk to the harbour seal population.

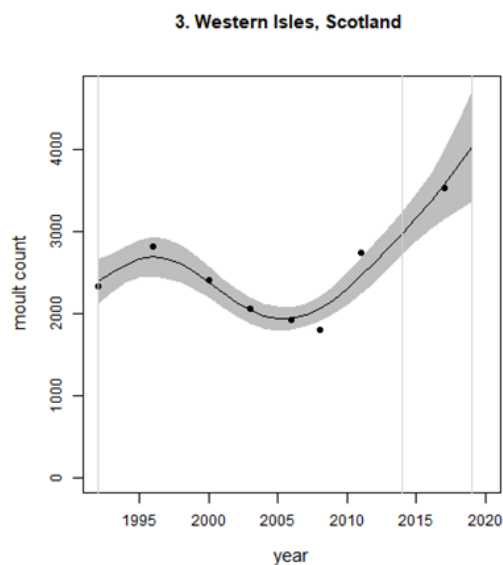


Figure 10. GAM fitted to harbour seal haulout counts between 1992 and 2017.

<p>20. SCOS previously advised a five yearly review cycle for designated seal haulout sites. Does SCOS consider that this is the most appropriate time frame for reviewing seal haul sites based on the survey data and rate of change in the population?</p>	<p>MS Q17</p>
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Given the five-year cycle for whole of Scotland census it would not be possible to carry out a full reassessment more frequently than every five years. Counts are variable so there is a danger of changing designations as a result of survey to survey variability rather than true changes in distribution. SCOS recommends a comprehensive reassessment every ten years, but with flexibility to respond to major changes between survey cycles.

The current monitoring programme aims to survey the entire Scottish harbour seal population every five years. Some sites have been surveyed more often; additional surveys have been carried out to assess the rate of change in the rapidly declining Orkney and N Coast SMU, and the estuarine sites in the Moray Firth SMU and the Tay and Eden SAC in the East Scotland SMU are surveyed annually. Given the 5-year cycle for whole of Scotland census it would not be possible to revise the designations, based on the overall population distribution, on anything less than a five-year schedule.

However, the counts at haulout sites are inherently variable, so a comprehensive re-assessment based on the original criteria, after each five-year survey round means that there is a danger of changing designations in response to that variability rather than to meaningful changes in distribution. SCOS recommends a comprehensive reassessment every ten years, i.e., after two survey rounds, but with inspection of the counts at designated sites after each is surveyed. This would retain the flexibility to respond to major localised changes in distribution in the shortest feasible time while avoiding over interpretation of the variability in count data. The criteria for triggering such a change would need to be defined in advance.

Seal Licensing and PBRs

<p>21. Can SCOS provide updated Potential Biological Removals (PBRs) figures for 2021?</p>	<p>MS Q12</p>
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Due to Covid related restrictions there are no additional surveys in 2020. The harbour seal PBR estimates reported in SCOS 2020 are therefore the most up to date estimates. The revised analysis of proportion of grey seals hauled out during the surveys has changed the scalar between counts and Nmin for grey seals (SCOS-BP 21/02). This has reduced the PBR estimates by approximately 3.5%

PBR estimates for both harbour and grey seals for each SMU in Scotland, together with a description of the calculations and the rationale for selection of SMU specific Recovery Factors (F_R) are presented in SCOS-BP 21/08. PBR values for the grey and harbour seal “populations” that haul out in each of the seven SMUs in Scotland are presented here (Tables 10 & 11), based on suggested values for the recovery factor and the latest confirmed counts in each management area.

Information on the alternative PBR estimates posted for UK grey and harbour seals in the NOAA data portal are provided below.

Table 10. Potential Biological Removal (PBR) values for harbour seals in Scotland by SMU for 2021. The most recent population data, estimates of N_{min} and the recommended F_R values are shown.

Seal Management Unit	2016-2019		selected	
	count	N_{min}	F_R	PBR
1 Southwest Scotland	1709	1709	0.7	71
2 West Scotland	15600	15600	1.0	936
3 Western Isles	3532	3532	0.5	105
4 North Coast & Orkney	1405	1405	0.1	8
5 Shetland	3180	3180	0.1	19
6 Moray Firth	1077	1077	0.1	6
7 East Scotland	343	343	0.1	2
SCOTLAND TOTAL	26846	26846		1147

Table 11. Potential Biological Removal (PBR) values for grey seals in Scotland by SMU for 2021. The most recent population data, estimates of N_{min} and the recommended F_R values are shown.

Seal Management Unit	2016-2019		selected	
	count	N_{min}	F_R	PBR
1 Southwest Scotland	517	1927	1.0	116
2 West Scotland	4174	15554	1.0	933
3 Western Isles	5773	21512	1.0	1291
4 North Coast & Orkney	8599	32043	1.0	1923
5 Shetland	1009	3760	1.0	226
6 Moray Firth	1657	6175	1.0	370
7 East Scotland	3683	13724	1.0	823
SCOTLAND TOTAL	25412	94695		5682

Alternative PBR estimates

In addition to the PBR estimates for Scottish SMUs presented above, the JNCC entered population data and a set of UK-wide PBR estimates into the NOAA bycatch portal to comply with requirements under the USA Marine Mammal Protection Act. The values posted to NOAA differ from those presented in this and previous SCOS reports. Given that there are now two different sets of PBR calculations in the public domain it is important that the differences and the justifications for the two sets are clearly understood.

The calculations for grey and harbour seals are described separately.

GREY SEAL

There are significant differences between the method used to generate a grey seal PBR in the NOAA portal and the method that is currently used to generate PBRs for Scottish Seal Management Units.

SCOS SMU specific PBRs

The PBRs in the SCOS reports are estimated for each individual Scottish SMU and are based on the most recent summer counts of grey seals hauled out in each SMU. Several SMUs hold substantial populations during the summer foraging season but do not have large grey seal breeding sites. As most interactions with human activities and management actions are likely to occur while seals are dispersed outside the breeding season, there is a need to allocate management targets (in this case PBR estimates) appropriately across all SMUs. The best estimate of the number of seals in an SMU is the number counted, corrected for the proportion that are not hauled out and are unavailable to be counted. The grey seal counts from the August surveys are multiplied by a factor derived from telemetry data which showed that around 25.15% (95% CI: 19.2 - 28.6%) were hauled out during the survey windows (Russell *et al.*, 2016 SCOS-BP 16/03). These data suggest that the N_{min} (the lower 20th percentile of the estimated population size) should be 3.73 x count.

UK-wide PBR

The PBR estimates entered in the NOAA portal are calculated for a single UK wide grey seal population. The population value used is the most recent estimate derived from a population dynamics model fitted to the grey seal pup production data (Thomas *et al.*, 2019; SCOS BP 20/01). This number is augmented to account for pup production in a small number of areas that are not included in the regular surveys. The mean estimate and the approximate standard errors from the model are then used to derive an N_{min} value.

The existing data in the NOAA portal for the UK wide PBR estimate are based on the overall UK population estimate in the SCOS 2020 report:

$$N_{best} = 149700 \text{ (CI: } 129000 - 174900)$$

$$SE = CI/1.96 \text{ The CIs are not symmetrical, but here we have used the lower CI}$$

$$CV = 0.072$$

$$N_{min} = 140776 \text{ the lower 20}^{th} \text{ percentile of the mean estimate } N_{min} = N_{best} - 0.845 * SE$$

$$R = 0.12$$

$F_R = 1.0$ As the regional populations of grey seals are all either at historical highs or are growing, the recommended F_R value grey seals in all SMUs is 1.0, so it seems sensible to use that value for the combined UK population as well.

$$PBR = 8447$$

As recovery factors are all set to 1.0 in both methods the result of pooling all of the individual SMU PBRs should sum to the same as the single UK-wide estimate. Any discrepancy will be due to variability in the predictions of the population dynamics model and the pup production estimates on which they are based, and variability in the summer survey counts.

HARBOUR SEAL

Again, there are differences between the methods used to generate a single UK-wide harbour seal PBR in the NOAA portal and the method that is currently used to generate PBRs for individual Seal Management Units in Scotland.

SMU specific PBRs

The PBRs in the SCOS reports are estimated for each individual Scottish SMU and are based on the most recent summer counts of harbour seals hauled out in each SMU. When the PBR method was first applied to harbour seals in Scotland SCOS were concerned that the conversion factor was based on only a small sample of adult seals in one particular year (Lonergan *et al.*, 2013). Given the declines in several harbour seal populations around Scotland, SCOS recommended taking a more precautionary approach that involved using the moult count as a proxy for N_{\min} rather than estimating the lower 20th percentile of the population estimate. This means that the PBRs presented in the SCOS report are approximately 28% lower than estimates based on the lower 20th percentile. Given the continued declines in Orkney and North coast SMU and the Tay and Eden SAC as well as the absence of any recovery in Shetland or the Moray Firth SMUs this policy has remained in place.

UK-wide PBR

In the NOAA portal, the values entered are the population estimate, i.e., a composite of the most recent counts from for all SMUs corrected for the estimated proportion of seals hauled out (0.72; Cis 0.54-0.88). The confidence intervals on the proportion are used to estimate the N_{\min} .

Based on the population estimates for harbour seals published in SCOS 2020, the values put into the NOAA portal are:

$$N_{\text{best}} = 44100 \text{ (CIs } 36100 - 58800)$$

$$CV = 0.129$$

$$N_{\min} = 40632$$

F_R ... selecting an appropriate recovery factor for the overall population is not a simple matter. The value of 0.5 entered in the NOAA portal is derived from the separate F_R values for each SMU, but it is not clear how the UK-wide recovery factor should be calculated. The UK-wide PBR estimate using F_R of 0.5 would be 1,220. The values used in individual SMUs range from 0.1 for the SMUs in the Northern Isles and along the east coast of Scotland, up to 1.0 for the West Scotland SMU. The Wash population has undergone a large drop since 2018 so it may be sensible to reduce the FR for that SMU. Some form of weighted average would seem to be most appropriate. Depending on the averaging method chosen the F_R could be set between 0.34 and 0.39. Replacing the existing value of 0.5 with values of 0.39 or 0.34 would reduce the PBR estimate by 22% or 32% respectively.

The use of the actual counts for harbour seals rather than the estimated N_{\min} is a more precautionary approach. As a result, the PBRs presented in the SCOS report are approximately 28% lower than estimates based on the lower 20th percentile. As a consequence, the UK-wide PBR estimate posted in the NOAA portal will be substantially larger than the sum of the individual SMU PBR estimates.

The PBR method was developed to manage anthropogenic impacts on discrete functional population units. The individual SMU approach violates the assumption that the populations are discrete/closed, particularly for grey seals. This is taken into account when deciding on the appropriate F_R for harbour seals and is simply avoided by setting the grey seal F_R to 1 in all SMUs. As widely discussed in SCOS 2020 there can be difficulties in managing wide-scale issues using the individual SMU approach. However, pooling groups of SMUs to address specific wide-scale issues should address such problems. On the other hand, using a single UK-wide PBR approach precludes fine scale management of localised issues or at least requires that the national PBR be subdivided in some appropriate way.

Seal Bycatch

<p>22. What is the latest understanding on levels of seal bycatch across the UK? Where is seal bycatch considered to predominantly occur by region and gear type and is there any data to show any bias by seal species, sex or specific age groups?</p>	<p>Defra Q8</p>
<p>What are the latest bycatch estimates for grey seals in the UK, especially Southwestern British Isles, including Ireland?</p>	<p>NRW Q2</p>
<p>What are the latest estimates of seal (grey and harbours) bycatch across fisheries in Scotland and the wider UK? Are there particular seasonal and / or geographical hot spots of high seal bycatch? Are there any areas where it has not been possible to collect seal bycatch data?</p>	<p>MS Q16</p>

The most recent estimated bycatch of seals in UK fisheries was in 2019. The total estimate was 488 animals (95% CI 375-872). This estimate is based on bycatch in gill net/tangle net fisheries; rare and sporadic captures in trawl fisheries are discussed below. The estimated bycatch was very close to the 2018 estimate. Bycatch estimates for ICES Divisions are presented in table 12.

Statistical analyses have not found any strong seasonal signal to seal bycatch rate.

There are no data to show any bias in species; all recorded species IDs in the SW are of grey seals, as there are few harbour seals west of the Solent area. Most bycaught animals are small. Species ID is uncertain for quite a few especially where they cannot be brought on deck. SCOS recommend that effort should be directed towards identifying the species and if possible, the sex and age structure and genetic information from the bycaught seals. This could be achieved by obtaining photographs of the animals and taking a skin sample.

Approximately 81% of the bycatch estimate occurs in the south-west, in ICES area VII, where the UK gillnet/trammel net fishery is concentrated. The remainder occurs in area IV which covers the North Sea and waters around Shetland and Orkney with less than 1% occurring in area VI around the Hebrides and Northwest Scotland.

SCOS are not aware of any reasons why specific areas have not been sampled, all sampling is simply constrained by resources. A Marine Scotland funded study is currently underway examining the distribution of bycatch monitoring effort.

Seal bycatch estimates

Seal bycatch estimates for the UK are made for both species of seal (grey and common/harbour) combined (Kingston *et al.*, 2021). Most seals that have been examined were young grey seals which can be hard to differentiate from harbour seals. All seals taken in gillnets were thought to be grey seals and were taken in the southwest where harbour seals are rare. The numbers of harbour seals recorded are too low to generate a useful bycatch estimate, so for expedience a single combined seal bycatch total is calculated. Although it is reasonable to assume that the majority of these bycaught animals are grey seals, for bycatch in the North Sea at least, some proportion will likely be harbour seals. There are no data to show any bias in species; all recorded species IDs in the SW are of grey seals, as there are few harbour seals west of the Solent area. Most bycaught animals are small. Species ID is uncertain for quite a few especially where they cannot be brought on deck.

SCOS recommend that effort should be directed towards identifying the species and if possible, the sex and age structure and population of origin of the bycaught seals. This could be achieved by obtaining photographs and skin samples from the animals.

The total seal bycatch estimate for UK waters in 2019 is 488 animals (CV = 0.07; 95% confidence limits 375-872) which is very close to the previous year (474). Estimates of seal bycatch have fluctuated year to year but are generally in the region of 400-600 seals per year, with no clear trend (Table 12).

Statistical analyses have not found any strong seasonal signal to seal bycatch rate. No specific hot spots have been identified in UK fisheries.

Table 12. Recent estimates of annual seal bycatch in UK gillnet fisheries with 95% confidence limits

Year	Estimated number	95% confidence interval
2013	469	285-1369
2014	417	255-1312
2015	580	423-1297
2016	610	449-1262
2017	572	429-1077
2018	474	354-911
2019	488	375-872

Recent analysis of data from the Irish EEZ (Luck *et al.*, 2020) shows that bycatch rates are related to proximity to areas of high seal density, around haulout sites and in inshore waters in particular. That analysis suggests that bycatch estimates can be significantly biased by the distribution of sampling effort. Increased marine mammal bycatch monitoring on French, Irish and other EU registered vessels fishing in this region would be helpful. UK sampling has covered all vessel categories (inshore and offshore) in this region, though sampling from Welsh ports and in the Bristol Channel has been limited and could be increased. The potentially large takes in these fisheries mean that the bycatch rates presented above may significantly under-estimate the scale of the problem.

Distribution of bycatch

The published data are not presented at sufficiently high resolution to ascertain whether there are any particular local hotspots of by-catch within particular ICES areas, but we are not aware of any such persistent hotspots. Table 13 shows the estimates by ICES Division and general area. Approximately 81% of the bycatch (394 seals) was estimated to have occurred in ICES area VII, around the south and south-west of the UK and Ireland. The majority of this occurred in the Western Channel and Celtic Sea, (around 300 seals per year), largely due to the overlap of high levels of fishing effort and relatively high seal densities. Bycatch rates in the Eastern Channel are estimated at around 88 seals per year.

Gear type

Most of the seal bycatch recorded in 2019 was in large mesh tangle nets and trammel nets, which accounted for 91% of the estimated bycatch. Effort in these fisheries is highly focused in area VIId, e & f (61% of UK tangle net effort). Sampling has been focused mainly in VIId-g. Other areas that are under-sampled and where there is a large amount of effort, or a high density of seals, could benefit from further observational data. These would include IVa (northern North Sea), IVc (southern North Sea), VIId (eastern Channel) and VIIf (North Devon and Cornwall and South Wales).

No seal bycatch was reported from trawl fisheries in 2019. In 2018 six grey seals were reported caught in sandeel trawls. Seal bycatch records in trawl fisheries are clumped, often involving several individuals in one location, but the overall recorded mean bycatch rate is very small and will have extremely wide confidence intervals, so no estimate of trawl fishery bycatch is included in the annual bycatch estimates.

Sampling is not strictly apportioned according to effort or to gear type, and it is possible that there may be additional sources of bycatch mortality that remain unknown. Sampling under the Protected Species Bycatch Monitoring Programme is focused on static gear in those areas where effort is generally highest, notable in the SW of Britain. No formal assessment of potential biases in the sampling programme has yet been made.

Potential consequences of bycatch

Although the total bycatch estimate of 488 is not large compared to the entire UK grey seal population of over 150,000 animals, the local populations around the Celtic Sea, where most bycatch is known to occur are much lower. The current estimate for the combined pup production in SW England, Wales and Ireland was approximately 4800 in 2019, but has a high level of uncertainty see Q 5 above. With the same assumptions as used to derive a PBR for the Welsh grey seal populations, (i.e., that $N_{\min} = 2.2 * \text{pup production}$; $FR = 0.5$; $r = 0.12$ (NRW Q2, and SCOS 2016 answer to Q9)) this pup production produces a PBR of 283 grey seals. The current estimated bycatch for UK registered vessels in ICES areas VIIa-c, e-j, was 303 (Table 13), approximately 7% greater than the conservative PBR.

The bycatch totals in table 13 are the estimates for just the UK registered vessels. This is likely to grossly underestimate the total bycatch in the Southwest. Bycatches (of unknown extent) by Irish, French, and Spanish vessels working the same areas will add to the total. Luck *et al.* (2020) estimated total bycatches of between 202 and 349 seals per year between 2011 and 2016 by all vessels within the Irish EEZ. Unfortunately, these cannot be simply added to the UK vessel bycatches as the Irish EEZ figures will include some of the UK registered vessel bycatch. Although bycatch was not broken down by country of registration, the fishing effort by French vessels (43%) was similar to the combined effort by Irish (21%) and UK (23%) registered vessels in the Irish EEZ. In addition, some

French and Irish vessels fish in UK waters and will also likely take seals as bycatch but are not included in either the Kingston et al. (2021) or Luck et al. (2020) estimates.

Table13. Seal bycatch estimates by ICES Division 2019 (from Kingston et al., 2021)

Region	ICES Division	Estimated total bycatch	Two-Sided 95% LCL	Two-Sided 95% UCL	One-sided 90% UCL
North Sea	IVa	29	24	35	33
	IVb	3	2	3	3
	IVc	47	39	68	63
West Scotland offshore	VIb	10	9	12	12
Irish Sea	VIIa	3	2	7	5
	VIIc	4	3	5	5
Eastern Channel	VIIId	91	66	178	148
	VIIe	151	123	207	191
Western Channel and Celtic Sea	VIIIf	125	104	154	145
	VIIg	10	8	18	15
	VIIh	7	6	10	9
	VIIj	3	2	3	3
Biscay	VIIIabcd	6	5	9	8

Despite the fact that the recorded bycatch levels are high relative to local population estimates, the grey seal pup production in the region is thought to be increasing. For example, regularly monitored colonies in Pembrokeshire are increasing by around 6% p.a. (Bull *et al.*, 2017 a, b, Lock *et al.*, 2017, Morgan *et al.*, 2018). A large proportion of the bycaught seals were assessed to be first- or second-year animals and first-year mortality is thought to be high in grey seals (SCOS-BP 20/02). If the bycatch mortality pre-dates this enhanced pup mortality it may have a relatively small effect on the dynamics of the populations. Notwithstanding such effects, the bycatch seems unlikely to be sustainable by local populations alone. That they continue to increase suggests that the removals may include or are being compensated for by immigrants. The most likely source of immigrants would be the large breeding colonies in the Hebrides where the population has been relatively stable and where post weaning juvenile survival rates are estimated to be low (SCOS-BP 21/05). As the bycatch is almost exclusively young grey seals a sample of 50 weaned grey seal pups on the Monach Isles were tagged with satellite transmitters in November 2021 to investigate early dispersal and estimate migration rates to the southwest region.

In addition to these movement studies, SCOS would recommend additional efforts to recover samples from bycaught animals to allow the analysis of genetic material to indicate the origin of these animals.

At present there are no indications that the declines in harbour seals in some seal management regions in Scotland and in southeast England are related to bycatch. English harbour seal populations have, until recently, been increasing and there do not appear to be conservation concerns associated with the observed bycatch rates of grey seals, as yet. However, given the scale of static net fisheries in the southwest, the amount of depredation that is being recorded during bycatch monitoring, the estimate of UK vessel bycatch and the existence of an unknown, but likely large foreign vessel, bycatch in the region, the western channel and Celtic Sea would seem to be an appropriate area for additional work.

SCOS are not aware of any reasons why specific areas have not been sampled, all sampling is simply constrained by resources. Observer effort is concentrated in SW and there may be requirements for increased and wider effort, work to assess these requirements is ongoing.

Seals and Fisheries and Aquaculture

<p>23. Non-lethal seal mitigation measures in commercial fisheries:</p> <p>Can SCOS provide recommendations on what the latest non-lethal mitigation devices, gear modifications and measures are to minimise seal depredation in commercial fisheries?</p>	<p>Defra Q7</p>
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There has been limited progress in the development or demonstration of any measures to mitigate seal depredation in commercial fisheries since SCOS 2020.

There have been some additional trials of the Targeted Acoustic Startle Technology (TAST) device in handline mackerel fisheries in the northeast of Scotland with evidence of strong deterrent effects. More work is necessary to determine effects on catch weight.

There have been no reported developments in gear modifications or other measures.

There have been very few additional studies on potential mitigation methods for minimizing seal depredation in commercial fisheries since SCOS 2020.

A pilot study was carried out in 2020 to assess the effectiveness of a GenusWave Targeted Acoustic Startle Technology (TAST) device (Götz & Janik 2015, 2016) in deterring seals from depredation of mackerel from handlines off northeast Scotland (Whyte *et al.*, 2020b). The study revealed a strong deterrence effect of TAST on seal activity directly around fishing vessels, in which seal detections on the vessels' fish finder (sonar) decreased by 97%. Fishing metrics such as 'fishing stop duration' i.e., the length of an individual fishing bout from stopping at a site to moving on to the next site, and 'catch weights' were primarily influenced by time-of-year (seasonality). However, fishing stop duration was almost twice as long when TAST was used. As fishermen usually terminate a fishing bout when the mackerel shoal below them disperses, the authors suggest that this increased duration may be the result of a reduction in shoal dispersal caused by seals. There were insufficient data to assess whether TAST had a significant effect on catch weight. Additional trials in net and line fisheries in England are expected to start in January 2022.

SCOS is not aware of any further progress in relation to gear modifications or other approaches to reduce depredation since those reported in SCOS 2020. Practical measures applied to date include reduced net soak time and avoiding areas where previous high rates of depredation have been encountered. The effectiveness of gear modification and other approaches will vary with fishery and target species. As highlighted by Cronin *et al.* (2014) for Irish waters, a detailed review of seal control measures used internationally along with case studies to test their effectiveness in UK fisheries is required.

<p>24. Seal Depredation in commercial fisheries:</p> <p>Can SCOS advise on the latest information available to provide evidence of seal depredation in the UK?</p> <p>Can SCOS advise on new research that could be undertaken to best to collect robust data on this important issue of concern within UK commercial fisheries?</p>	<p>Defra Q9</p>
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SCOS is not aware of any new published quantitative information on the extent, frequency, intensity, or geographical pattern of interactions between seals and fishing operations and no quantitative information on rates of removals or frequency of seal damage to fish in gear.

SCOS recommends that a UK wide workshop involving fisheries managers, local and national fisheries organisations and marine mammal scientists be convened to design a study, with the aim of defining the specific issues and identifying locations and timings of interactions that warrant further investigation. This would allow data requirements to be assessed, and appropriate structured monitoring programmes to be developed.

Examination of existing data from the UK Protected Species Bycatch Monitoring Scheme should be prioritised.

In 2018, 2019 and 2020 Defra/MMO reported that there are increasing numbers of anecdotal accounts of seals causing considerable damage to fish that have been caught in nets and on lines at various locations on the English coast. It is clearly felt strongly by the fishing industry that impacts of seals on fishing operations has increased in recent years and that effective solutions are necessary.

A similar question was answered in the SCOS (2020) Advice and SCOS advised that an MMO sponsored workshop had discussed local seal fishery interactions, but had not resulted in the development of a formal programme of research or monitoring (MMO, 2020a,b). SCOS was not aware of any structured programme to log and assess the validity of these reports, to quantify the scale of removals or estimate the economic cost or to identify trends in these metrics. This remains the case in 2021.

As advised in SCOS (2020), SCOS recommends that a workshop involving fisheries managers and scientists, local and national fisheries organisations from the whole UK and both marine mammal and fisheries scientists would be a useful first step in defining the specific issues, locations and timings of interactions, and identifying research opportunities and potential solutions that warrant further investigation. It is likely that a structured monitoring programme using an integrated approach involving the industry is required to progress the collection and collation of robust quantitative information on the scale and extent of damage to catch and fishing gear.

The UK Protected Species Bycatch Monitoring Scheme has collected data for 20 years on the bycatch of marine mammals through on-board observations, some of which is associated with depredation. It has also collected information on seal-damaged fish recovered from nets. SCOS recommend that additional resources should be allocated to conduct a quantitative assessment of these data.

Standardised post-mortem examinations of stranded seals and recovery of bycaught seals for examination may also provide some evidence for the extent of this issue.

25. Are there any parts of the wider ecosystem that are likely to experience significant impacts as a result of an increasing Scottish seal population? What are these impacts, and would they be positive or negative?

MS Q7

SCOS noted that grey seal population increases over the past decade have been confined to the Central and Southern North Sea. So, in most of Scotland, other than the North Sea coast, grey seal populations are thought to be currently stable.

Harbour seal numbers have declined in some regions of Scotland, such as the Northern Isles and along the East coast, but have increased in others, such as the West of Scotland, the Western Isles and SW Scotland.

Overall, there has been no general increase in seal numbers in Scottish waters, although trends for both species vary regionally.

It is likely therefore that there will be regional differences in the level of interactions between seals and the wider ecosystem. The effects of increasing seal populations on fish prey populations were considered in detail in SCOS 2019, therefore this answer focuses more on other impacts rather than repeat that answer.

It is important to note that seal predation can have significant impacts on particular fish stocks, but this can vary considerably between stocks. In some areas/ecosystems seal predation has been identified as having a significant impact on recovery of specific fish stocks, whereas in others, increasing seal populations appear to have had minimal impacts. As highlighted in SCOS 2019, predicting ecosystem effects of changes in predator population size is complex and difficult and requires a multispecies ecosystem modelling approach. This requires information on fish abundance and distribution, spatial and temporal patterns of seal predation, spatial and temporal distribution of fishing effort and an understanding of multispecies functional responses. Work is underway to fill several of the data gaps highlighted in SCOS 2019.

Seals of both species are known to interact with aquaculture developments to prey on farmed salmonids and both species are also known to prey on wild salmonids in rivers. It has been estimated that $\leq 1\%$ of the general seal population specialise in predating wild salmonids in rivers, while a small but unknown proportion of seals depredate salmonids at fish farms.

Even where interactions are known to occur there is limited information on current or historical predation rates at either aquaculture installations or in rivers. This limits our ability to predict the effects of increasing seal populations in areas where they overlap. Previous analyses have not been able to demonstrate a clear link between seal abundance close to rivers and levels of predation in rivers.

Other potentially significant effects of population increase in either or both seal species include: increased competitive interactions between the two seal species, increased predation of grey seals on both harbour seals and harbour porpoises, and increased availability of seals as prey for killer whales.

A number of data gaps are identified, which if were filled, would improve the ability to answer this question in future.

Some of the issues raised in this question were addressed in SCOS 2019, in relation to effects on wild fish populations (including salmonids in rivers) and fish stocks. As a result, this answer does not

repeat the information provided previously, but focuses on the potential impacts of increasing seal populations on other aspects of the marine environment including effects on other mammal populations and aquaculture.

Although there has been continued increase in the overall UK grey seal population in terms of both pup production (SCOS-BP 21/01) and total population (SCOS-BP 21/03), the majority of the increase in pup production over the past 20 years has been at colonies in the North Sea and in the past 10 years that has been concentrated at colonies in the southern North Sea. Based on the distribution of hauled out seals during the summer, the numbers of grey seals foraging around Scotland have remained relatively stable, while in the central and southern North Sea the numbers of grey seals foraging in summer have increased at sites along the east coast of England and particularly in the southern North Sea. Harbour seal populations around the north and east of Scotland have undergone dramatic declines, whilst those on the west coast have increased. Overall, there has been no general increase in the population of seals foraging around Scotland in the past decade, although predation levels have likely increased in the areas where harbour seals are increasing on the west coast of Scotland and declined in Orkney where harbour seal populations have declined dramatically.

Understanding seal diet is key to being able to predict ecosystem effects of increasing populations and as detailed in SCOS (2019), the results of previous major studies of seal diet in the UK are described in detail in a series of recent reports to Scottish Government (Hammond & Wilson, 2016; Wilson *et al.*, 2016; Wilson & Hammond, 2016 a, b). The results of the most recent study (2010/11) are summarised in Wilson and Hammond (2019), in the context of regional variation in trends in population size of both species of seal. Overall, sandeels and large gadids were the two main prey types, but results showed considerable seasonal and regional variability. SCOS note that these data are now more than 10 years old and may not provide an accurate description of seal diets in areas where fish stocks and seal populations have changed.

In terms of diet composition, in the southern North Sea, sandeel dominates grey seal diet, whereas flatfish, gadids and sandy benthic species are more important for harbour seals. In the Moray Firth, the diet of both species is dominated by sandeel. In the Northern Isles, sandeel and gadids are important in both species' diets, with pelagic prey also important for harbour seals. Gadids are the main prey of both species in the Inner Hebrides. In the Outer Hebrides, sandeel and gadids are the main prey of grey seals and pelagic species and gadid featuring in harbour seal diet (Wilson and Hammond, 2019).

UK seal population trends should be seen against a background of major long-term changes in the productivity of key ecosystem components of the North Sea, Celtic Sea and adjacent waters. The ecological changes resulting from predator population increases are likely to be highly complex and difficult to predict. Clearly predation by seals is large enough to be a potential factor in the dynamics of some fish populations (e.g., grey seal predation has been shown to be an important factor in the failure of cod stock recovery on the Scotian Shelf (Neuenhoff *et al.*, 2019), although in other cases, seals have minimal impact, e.g., harp seal consumption of cod off Newfoundland was found not to be an important driver of the northern cod stock (Buren *et al.*, 2014), and in the Gulf of St Lawrence although harp seal consumption did affect cod dynamics it was not as important a driver as fishing or water temperature (Bousquet *et al.*, 2014). However, uncertainties in several factors, e.g., fine scale variation in seal diet composition, the spatial and temporal overlap between seals and fisheries at sea and overlap between the size distribution of prey eaten by seals and selectivity of the fisheries all combine to mean that confidence in predictions of effect levels will be low. Determining the ecosystem-level impacts of an increasing seal population will require an integrated ecosystem modelling approach with inputs on the drivers of distribution for key components of the ecosystem.

A number of data gaps were identified in SCOS 2019, and work is underway on a number of projects to address these (e.g., the EcoSTAR project under the INSITE II programme is developing multispecies functional response models for seals and porpoises and integrating outputs within a North Sea ecosystem model which will allow future scenarios of change to be modelled.

The impact of increasing seal populations on the level of interactions with aquaculture is also difficult to predict, although it is well known that both harbour and grey seals depredate on salmon at fish farms, there is very little robust quantitative evidence of the nature and scale of such depredation and therefore a limited ability to understand how this will scale with any increases in population size. Northridge *et al.* (2013) found that the proximity to the nearest harbour seal haulout site made no difference to the amount of depredation occurring on fish farms, though all sites in their study were within 10 km of a harbour seal haul out. The number of harbour seals counted within 3, 5, 10 or 20 km of a fish farm site also made no difference to the amount of depredation. Northridge *et al.* (2013) also reported an unexpected positive relationship between the amount and frequency of depredation and the distance to the closest grey seal haul out. They also found that farms with grey seal hauls outs closest recorded less damage than those for which grey seal haul out sites were further away (up to 11 km). There was also less frequent damage at farm sites where there were larger numbers of grey seals counted at haul outs within a 20 km radius during August surveys than farms with fewer grey seals. Given these findings, how depredation at fish farms might scale with changes in local seal abundance and distribution is hard to predict. To predict how depredation may increase in future with further increases in seal population, a detailed study of the spatial and temporal nature of current levels of seal depredation is required. This would ideally include an updated analysis of the relationship between levels of depredation and local seal abundance. If effective physical protection can be achieved at fish farms, then an increasing seal population will have a limited effect on aquaculture.

Increases in wild salmon predation by growing seal populations has been blamed by fisheries managers for declines in salmon stocks (e.g., as detailed in Butler *et al.*, 2008 in the Moray Firth) and recovering pinniped populations have been identified as a factor affecting the recovery of endangered salmon stocks in the US Pacific Northwest (see Chasco *et al.*, 2017). However, direct evidence linking seal predation with declines in salmon stocks in Scotland and other parts of the world is lacking. SCOS 2019 concluded that there was unlikely to be a direct link between seal population size and the observed decline in rod and line caught salmon. Salmon are consumed by several predators including other fish, birds, seals and cetaceans and predation is one of 12 identified threats to Scottish wild salmon populations (Scottish Wild Salmon Strategy, 2021⁶). With salmon numbers in decline, and over half of assessed rivers being in poor conservation status⁷, any threat is likely considered important. e.g., Photo identification and telemetry studies have indicated that individual seals representing a small proportion of the population ($\leq 1\%$) specialise in using rivers (Graham *et al.*, 2011). How these individuals learn and develop these predation strategies is uncertain and therefore how this proportion may scale with increasing local population size is also uncertain. Graham *et al.* (2011) concluded that the proximity to breeding and moulting sites for each species of seal may influence the observed patterns of seals in rivers. Following this logic, local increases in seal population may result in increases in the numbers of seals using rivers, although no monitoring has been in place across relevant timescales to determine this. Bioenergetic modelling by Butler *et al.*, (2006) predicted that seal removals would result in increased catches of salmon in rivers but did not predict the result of seal increases on salmon numbers in rivers. Such modelling

⁶ <https://www.gov.scot/publications/scottish-wild-salmon-strategy/documents/>

⁷ <https://www.gov.scot/publications/salmon-fishing-proposed-river-gradings-for-2022-season/>

could be carried out, but it would require assumptions to be made about how river predation would scale with population size.

Understanding the relationship between seal population size and the numbers of seals involved in depredation of salmonids at fish farms or in rivers is severely limited by a lack of quantitative historical information on levels of depredation or levels of seal presence or activity in either situation.

If fish farms or salmon rivers are, as often assumed, highly attractive foraging locations and/or the seals involved are specialists that represent a small proportion of the population, it is unlikely that there will be a simple relationship between population size and predation levels. SMRU (1984) compared time series of salmon smolt survival estimates for the river North Esk and grey seal population trends. Despite the fact that the data covered a period of rapid seal population growth there was no detectable reduction in smolt survival rates. They also analysed the time series of daily reports from fixed net salmon fishing stations and found no relationships between grey seal population sizes and seal sightings rates nor reported levels of seal damage. Although these represented different situations, the results indicate that even with detailed records the relationships between overall seal population sizes and predation activity levels are unlikely to be easily identified.

Increases in seal populations of one species may also have impacts on the dynamics of the other species. Increasing grey seal populations have been hypothesised as being at least partly responsible for declines in harbour seal populations in some regions (Wilson and Hammond, 2019). This could be mediated through competition for prey, given regional similarities in prey preferences of the two species (Wilson and Hammond, 2019). Impacts could also occur as a result of direct predation by grey seals on harbour seals (Brownlow *et al.*, 2016, van Neer *et al.*, 2019). A PhD project at SMRU is investigating these interactions between grey seals and harbour seals.

Grey seals are also known to predate harbour porpoises (Leopold *et al.*, 2015), so increasing grey seal populations could have implications for harbour porpoise populations. However, the extent of this behaviour and the potential for it to lead to a significant impact on harbour porpoise populations is unknown.

Killer whales are known to prey on both seal species in Scottish waters with reports of predation from the Northern and Western Isles. Such predation has been suggested as a driver of harbour seal population declines, but conversely, increased seal populations may increase the prey resource and potentially increase the reliance of killer whales on seals. Interestingly, such an increase could lead to different predation mortality rates for the two seal species depending on their regional population dynamics.

Data gaps

Data required to develop an understanding of the implications of increasing seal populations on fish prey populations and fisheries were outlined in SCOS (2019). Here we outline the work required to develop our understanding of how seal population increases might impact on other aspects of the marine environment, including aquaculture and salmon predation. This would require further investigation of:

- Grey seal/harbour seal interactions including competition and predation (PhD project at SMRU underway)
- Extent of grey seal predation on harbour porpoises
- Develop quantitative predictive models of factors influencing seal depredation at fish farms,

including the effect of local and regional seal abundance and distribution on levels of depredation

- Develop a better understanding of the relationships between levels of seal predation on wild salmon in rivers, and local or regional seal abundance and distribution.
- Develop models of killer whale predation on harbour seals (work underway on EcoPREDS project at SMRU and associated PhD project).

<p>26. Based on distribution and demographics of seal populations, can SCOS advise whether it would be possible predict times and locations where there may be a greater chance of interactions with the aquaculture industry? Please can SCOS advise what work would be required to achieve this.</p>	<p>MS Q8</p>
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There are a number of analyses that could inform the potential for interactions between seals and aquaculture, including ‘risk mapping’ approaches on the basis of overlap of seal predicted density and fish farm locations, and a detailed examination of existing telemetry data to look for evidence of specific interactions at fish farm locations.

However, such spatial overlap analyses will only provide a crude estimate of the potential for future interactions because spatial overlap does not necessarily imply direct interactions. Research into the spatial and temporal patterns of the occurrence and magnitude of seal depredation, and the relationships with environmental covariates, farm activities and cage characteristics is required to fully understand and to develop an ability to predict the potential for future interactions.

SCOS recognise that there are two potential direct impacts of seal activity around aquaculture sites: predation and stress effects on fish. More information is required on both to allow assessment and predict the effects of seals on aquaculture.

There is considerable overlap between seal distribution and areas of aquaculture production around Scotland’s west coast and northern isles. Northridge *et al.* (2013) found that the proximity to the nearest harbour seal haulout site made no difference to the amount of depredation occurring on fish farms, though all sites in their study were within 10 km of a harbour seal haul out. The number of harbour seals counted with 3, 5, 10 or 20 km of a fish farm site made no difference to the amount of depredation. Northridge *et al.* (2013) also reported an unexpected positive relationship between the amount and frequency of depredation and the distance to the closest grey seal haul out. There was an unexpected positive relationship between the amount and frequency of depredation and the distance to the closest grey seal haul out site. They found that farms with grey seal hauls outs closest recorded less damage than those where grey seals haul out sites were further away (up to 11 km). There was also less frequent damage at farm sites where there were larger numbers of grey seals counted at haul outs within a 20 km radius during August surveys than farms with fewer seals.

The absence of a close link between proximity of haulout sites and predation levels may indicate that only a small proportion of the local population is involved in attacking cages, or that seals specialising in depredation at farms are not necessarily hauling out locally.

These analyses could be repeated with contemporary data from farm sites and the levels of depredation experienced, and up to date seal survey data.

It may be that at-sea density is a better predictor of interactions and an analysis of depredation in relation to predicted seal at-sea density could be carried out using the predicted seal density maps provided in Carter *et al.* (2020). Such analyses would require data on depredation events, which are often collected by fish farms but are not routinely made available or are not collected at a sufficient temporal resolution to enable analyses.

Even in the absence of detailed data on depredation, a simple 'risk mapping' approach could allow identification of the areas of highest overlap between seal distribution and aquaculture activity. This would involve the predicted seal density maps being overlain with a map of all operating fish farms. Each 5 x 5 km grid square could be applied a risk score which is derived from a combination of the predicted seal density and the number of fish farm operations within it. Although this may indicate a crude potential for the locations where interactions may occur, this may not be a reliable indication of the actual level of interactions and will not allow any prediction.

Other possibilities include an examination of existing seal telemetry data for direct overlap of seal activity with fish farm locations. There are datasets from a large number of deployments on both species of seals using GPS GSM telemetry devices in areas around Scotland where there are active fish farms. The tracks from these deployments could be examined in detail for evidence of interactions with fish farms. A recent PhD project used telemetry data from harbour seals tagged in Skye to estimate acoustic exposure of seals from ADDs at fish farms (Findlay *et al.*, in review). This study combined tracking data with maps of predicted ADD noise to quantify sound exposure and estimate the potential for auditory impairment. A similar approach could be taken with a larger sample of tracking data across a wider geographical area and could incorporate the investigation of behavioural metrics that would indicate association with fish farms.

However, these approaches may only provide crude estimates of the potential for interactions and will allow a limited predictive ability as the factors that drive levels of seal depredation at fish farms are still poorly understood. Research into the spatial and temporal patterns of the occurrence and magnitude of seal depredation, and their relationships with environmental covariates is required to fully understand and to develop an ability to predict the potential for future interactions.

An understanding of how factors such as cage/net design and operational practices influence depredation is also crucial. Better information on the residence times of seals around farm sites, the species and age classes involved, the degree to which individuals associate with specific farms, and the numbers of individuals that associate with specific farms may help to understand the motivations and behaviour of seals that habitually target farm sites and improve our predictive ability, as well as allowing the tailoring of preventative measures. These research recommendations have been made in a number of previously published reviews and should be a priority (e.g., Northridge *et al.*, 2013; Coram *et al.*, 2014, 2016, 2017). Attempts to make use of the data available from the industry has revealed that the data on depredation is not often recorded at sufficient temporal resolution to allow analysis to inform this question (e.g., Coram *et al.*, in press).

One pressing issue identified by SCOS is the need for information on the indirect effects of seal presence in the vicinity of cages, particularly on stress in farmed fish. This is important and will to a large extent determine the types of protection or seal deterrence required. If seal presence causes unacceptable stress to fish it will be necessary to exclude seals from the entire site. If seal presence does not induce high levels of stress, it will only be necessary to prevent seals gaining direct access to the fish. The former will require wide area deterrence, which may have important negative impacts on nontarget wildlife, whereas the latter will only require defence of the cages themselves.

<p>27. SCOS provided advice in 2020 on non-lethal options to address seal – fisheries / fish farm interactions. Since the 2020 advice (and in light of ongoing efforts globally to address such interactions), are SCOS aware of any further developments in other countries or emerging technologies that could be consider/applied to Scotland.</p>	<p>MS Q15</p>
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The only additional work that SCOS is aware of in this area are additional studies with the TAST system including trials on fishing vessels and on fish ladders in the USA, including trials at Ballard Locks where TAST use resulted in a significant increase in fish passes relative to control periods.

SCOS are not aware of any further developments or emerging technologies that could be considered or applied to Scotland that were not discussed in earlier advice. More studies have been carried out in the past year using the TAST in relation to fisheries and these have been described in the answer to Defra Q7 above. In addition to these TAST systems were carried out near fish ladders that suffered from seal predation in the USA. A five-week deployment of a TAST system outside the entrance to the Ballard Locks fish ladder in Seattle in 2020 resulted in an increase in fish passes by 4419 animals, an increase of 46% over control periods. Similar results were found in three other locations in Washington State and Alaska in 2020 and 2021 (Unpublished data).

Climate change

<p>28. Can SCOS review latest scientific information available on current environmental impacts seals face due to climate change, such as acidification, sea level changes and coastal collapses and changing prey distributions.</p>	<p>Defra Q12</p>
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The effects of climate change were reviewed in SCOS 2020. SCOS are not aware of any significant recent developments. The answer from SCOS 2020 is repeated below with modification where new published information is available.

Climate change is already having a range of effects in UK waters, including changes to water temperature and salinity and is likely to change timing and intensity of stratification and locations and timings of fronts. Such changes will influence patterns of productivity and fish distributions and will affect prey availability to seals. These changes could have either positive or negative effects on seals in the UK. Changes in air temperature may have impacts on seal behaviour and reproductive performance during time on land.

Predicting the population consequences of climate change for seals is difficult. There is little information on the relationships between environmental drivers and seal population dynamics. It is therefore unlikely that cause and effect will be reliably assigned to specific aspects of climate change with respect to changes in seal population dynamics. Observed trends in UK seal populations show growth mainly in southern parts of their range despite indications that distributions of currently preferred prey are shifting northwards.

There is uncertainty in the predicted effects of climate change on frequency and intensity of Harmful Algal Blooms (HABs) or on the effects of HABs on seals. However, the potential severity of HAB effects highlights the need for further research into HAB effects on seals.

Changes in sea level and resulting increased wave action on breeding beaches may reduce breeding and haulout site availability in some areas. Increased storminess in terms of maximum and average wind speeds and frequency of storm systems may lead to increased wave action on breeding sites which can increase pup mortality. Seals may be able to accommodate by moving breeding sites if alternative sites are available.

The seas around the British Isles, have warmed faster than the global average over the past 50 years. Sea surface temperatures (SST) in the North-east Atlantic and North Sea have risen by between 0.1 and 0.5°C per decade over the past century, and the rate of warming has been particularly rapid since the 1980s (Dye *et al.*, 2013). There are a wide range of interacting factors driving population change so it is extremely difficult to disentangle their effects and identify specific causes. Albouy *et al.*, (2020) carried out an assessment of the vulnerability of all marine mammal species to global warming. They produced a ranked list of species by vulnerability to climate change effects. Grey seals (16) and harbour seals (20) appeared on a list of the top twenty most vulnerable species of marine mammals to climate change extinction risk. However, the model was driven by an index of temperature sensitivity, but the fact that none of the Atlantic ice associated seals or Antarctic seal species are listed suggest that this approach may have limited value for predicting climate effects for temperate water seals like grey and harbour seals.

Most of the research on the impact of climate change on marine mammals has focused on the Arctic, where dramatic changes in ice volume and extent are already having profound effects on habitat availability. Changes in ice availability, and timing of freeze up and ice break up are already having direct impacts on ice breeding seals., In the Gulf of St Lawrence in eastern Canada grey seals are increasingly breeding on land and the distribution of breeding sites is shifting northwards. In the Baltic, changes in timing of freeze up and ice break up are changing the breeding habitat availability and forcing seals to breed on land, causing either direct mortality or reducing lactation efficiency and pup growth rates potentially as a result of water balance issues (Jüssi *et al.*, 2008; Hammill *et al.*, 2013). Shuert *et al.* (2020) showed that high temperature and lack of access to water can reduce pup weaning mass and increase likelihood of pup abandonment in grey seals breeding at temperate sites such as the Isle of May. Bull *et al.* (2021) associated lagged SST indices with changes in pupping dates of grey seals on Skomer MCZ. A temperature increase of 2°C was associated with an advance in pupping date of approximately seven days. They concluded that the temperature index was related to transient changes in age distribution due to “immigration” of older mothers (older mothers tend to give birth earlier in the season).

Changes in cold temperate waters, such as the seas around the UK, may also be profound and will likely impact on continental shelf marine predators such as seals. However, in UK waters, the projected changes in the physical environment, such as air and water temperatures, water depth and salinity, are not predicted to exceed the homeostatic ranges for seals. E.g., harbour seals occur in temperate coastal waters as far south as San Diego, California, and Brittany and the Wadden Sea in Europe where summer water and air temperature exceed those currently experienced by seals in southern England. Existing conditions at the southern limit of existing ranges are generally higher than projected temperatures in the UK over the next century even under high warming scenario predictions, but although harbour seals in other parts of their range experience higher summer temperatures, it is unclear what effects increased summer temperatures may have on terrestrial breeding behaviour and breeding success of harbour seals in the southern UK.

Prediction from status quo

Species distributions are not usually determined by physical capabilities alone. The distributions of both prey and competing predator species will influence the distribution of predators such as seals. So, the consequences of changes in the physical environment will be difficult to predict. If we could assume that competitors, prey, and other factors would maintain their current relation to variables such as water temperature and depth, we could use the current distribution patterns to predict future distributions under different climate change scenarios.

Boehme *et al.* (2012) and Zicos *et al.* (SCOS-BP 17/07) used location fixes and water temperature records from the extensive telemetry datasets for UK harbour seals, and grey seals in both the UK and Canada to derive predicted distributions based entirely on water depth and sea surface temperature in the North Atlantic. Zicos *et al.* then explored potential habitat shifts across the entire Atlantic ranges of both species under two scenarios of climate change, the lowest and highest scenarios of warming as determined for the IPCC's 2014 report.

The low warming scenario predicted an overall compression of core habitat, with slight loss of habitat in the northern and extensive habitat loss in the southern edges of distribution in the North Atlantic. In the high warming scenario, there was a general northward shift in predicted core habitat for both species. In geographical terms the predicted northern expansion of habitat would exceed the southern contraction so that both species would be predicted to have larger foraging habitat extents in the future.

Changing prey distributions.

The effects of climate change on prey distributions and changing patterns of fishing activity will both likely impact the distribution and population dynamics of seals. North Sea stocks of cod, plaice and haddock have shown northward shifts (Engelhardt *et al.*, 2011 & 2014; Skinner 2009). Recently, Baudron *et al.* (2020) published an analysis of scientific survey data that provides an overview of changes in distribution for 19 northeast Atlantic fish species encompassing 73 commercial stocks over 30 years. All species experienced changes in distribution. Two thirds of the shifts in centre of gravity (CoG) displayed by northern species were northward. Baudron *et al.* (2020) concluded that the overall northward direction of the changes in distribution together with observed range contraction for northern species, and expansion of southern species ranges into UK waters, e.g., solenette (*Buglossidium luteum*), were consistent with the poleward distribution shifts expected from warming sea temperatures.

Atlantic populations of grey and harbour seals however have not followed this general northward trend. For grey seals on both sides of the Atlantic the numbers of seals in the southern parts of the range are increasing rapidly while populations in the central and northern parts of the range have stabilised leading to a southward trend in CoG. Similarly, for harbour seals in Europe, a southward shift in the CoG of the population has been recorded over the past 30 years despite the disproportionate effects of PDV epizootics in the southern North Sea.

The drivers of this redistribution are not known, but the changes in seal distribution do not simply map directly to changes in distribution of their existing prey species. Nor do they conform to the broad scale northward movement of increased air and water temperature associated with climate change.

Boveng *et al.* (2020) recently reported preliminary results of a study of Arctic seals that included harbour seals on the Aleutian Islands, in environmental conditions similar to northern Scotland. Though harbour seal data were limited to three sampling events during 2014–2016, they observed a striking decline in body condition: an estimated annual decrease of about 45g of body mass per centimetre of length. Harbour seal populations have undergone a long-term decline in the Aleutian

Islands. The population dropped precipitously between 1980 and 1999. The decline was most dramatic in the western Aleutians, where counts dropped by 86 percent, to about 5,500 individuals. The population has not recovered since, and the cause is unknown. The estimates of recent declines in body condition represent almost a 20% decrease in body mass in two years' time. Such decreases would have serious consequences for individual and population fitness if not followed by recovery of body mass. The researchers consider that the recent declines in body condition are likely an acute response to the recent very strong North Pacific marine heat wave, presumably mediated through reduced prey availability, rather than a continued chronic response to whatever has caused the long-term decline in numbers.

Harmful Algal Blooms (HABs)

There is some debate about the likely future patterns of HABs in UK waters (Bresnan *et al.*, 2020). Increased water temperature will have different effects on different species, but experimental studies of growth and survival rates of a range of species have suggested that HABs are likely to increase rather than decrease in the North Sea (Peperzak, 2003). Projections of sea surface temperature also suggest that the habitat of most species will shift north and may lead to more frequent harmful blooms in the central and northern North Sea (Townhill *et al.*, 2018) and increased temperature may increase toxin production (e.g., Aquino-Cruz *et al.*, 2018). Gobler *et al.* (2017) investigated potential changes on a larger scale and came to similar conclusion, that increasing ocean temperatures have already facilitated the intensification of certain HABs.

However, Edwards *et al.* (2006) used long term data from the northeast Atlantic and North Sea (1960s to early 2000s) to investigate spatial variability in the frequency of HABs. Significant increases were restricted to the waters off Norway and there was a general decrease along the eastern coast of the United Kingdom. The most prominent feature in the interannual bloom frequencies over the preceding four decades was anomalously high values in the late 1980s in the northern and central North Sea areas. Dees *et al.* (2017) examined long term data sets from the Northeast Atlantic and North Sea for one toxic algal genus, *Dinophysis* and found that over the modelled period (1982–2015) and the whole Continuous Plankton Recorder time series (1958–2015), there was no statistically significant positive relationship between abundance and sea-surface temperature. They also showed that periods of large *Dinophysis* blooms in the 1970s and 1980s, were followed by a period of briefer bloom events lasting until 2014. Dees *et al.* concluded that there was no increasing trend in number or annual duration of blooms.

Given this lack of consensus on the likely patterns of HABs and the uncertainty in the rates of consumption and likely levels of toxicity in seal diets, it is not possible to reliably predict the potential effects of climate related HAB changes on UK seal populations. However, the potential for such events to cause large scale mortality events means that further investigation is warranted.

Local oceanographic changes

Earlier stratification of warmer water and changes in the timing of plankton blooms and secondary production blooms will likely have effects throughout the food chain (e.g., Wiltshire and Manly, 2004). Such changes have already been detected in the North Sea at several levels of the food chain. This may have knock on effects on the timing of prey availability that may impact on seal condition. Changes in flow patterns and locations of frontal systems may also impact seal foraging habitat quality. None of these possible effects have been studied in terms of their potential impacts on seals in UK waters.

Large scale oceanographic changes

Future predictions of marine climates around the UK will be heavily influenced by what happens to the Atlantic Meridional Overturning Circulation (AMOC). The AMOC significantly warms the northeast Atlantic and drives the general climate of northwest Europe partly through its influence on the track of the jet stream. Both direct observations (2004–2017) and sea surface temperature reconstructions, show that the AMOC has weakened since 1900 (IPCC, 2019). The data timeseries are too short to confirm that the weakening is due to anthropogenic forcing, but CMIP5 model simulations show similar weakening of AMOC as a result of anthropogenic forcing.

The AMOC is projected to weaken in the 21st century, although a collapse is very unlikely. Weakening of the AMOC is projected to cause a decrease in marine productivity in the North Atlantic and an increase in storms in Northern Europe (IPCC, 2019). Both reduced productivity and increased storminess could have potential population scale effects on UK seal populations.

Competition with fisheries

The climate driven changes will not only affect natural predators. The patterns of fisheries exploitation will also be affected. Current quota allocation structures will need to adapt to changes. How these changes are implemented is likely to have major implications in terms of prey availability for seals and other predators, and changes or re-distribution of fishing practices may affect issues such as bycatch.

Ocean Acidification and Low Oxygen

Increased atmospheric CO₂ is absorbed by sea water which causes a reduction in pH and may have already lowered global ocean pH by 0.1 pH units since the industrial revolution (Orr *et al.*, 2005). North Sea pH has decreased at a rate of around 0.0035 pH units per year (Williamson *et al.*, 2017).

Ocean acidification may have direct and indirect impacts for the recruitment, growth and survival of exploited species. Effects are likely to be more important for shellfish (Pinnegar *et al.*, 2017) but changes to larval fish behaviour and reduced survival and recruitment have been reported (Munday *et al.*, 2010); for example, projected ocean acidification levels (from IPCC RCP 8.5) have been shown to double daily mortality rates of cod larvae (Stiasny *et al.*, 2016). The potential impacts of ocean acidification are an active field of research and the effects on future prey availability for seals are, as yet, unknown.

Reduced oxygen concentrations in marine waters have been cited as a major cause for concern globally (Diaz & Rosenberg, 2008), and there is evidence (Queste *et al.*, 2012) that areas of low oxygen saturation have started to proliferate in the North Sea. However, the European Environment Agency (2019) suggested that hypoxic or reduced oxygen levels were mainly restricted to Scandinavian fjord waters with some reduced oxygen levels recorded on the North Sea near the Oyster grounds. To what extent these are the result of long-term climate change remains unclear and it is also unknown whether such changes will impact upon fish populations (Pinnegar *et al.*, 2017).

Breeding habitat changes.

Predicted increases in sea level are small compared to the changes that grey and harbour seal populations have experienced due to sea level rise and iso-static rebound of the coastline since the last ice age. However, there is no reason to suspect that the availability of offshore islands, skerries, rocky shore or intertidal sand banks has decreased over that time or that availability will decrease, in the medium to long term, under projected sea level changes.

However, seal responses to previous sea level rises were not influenced by human activity patterns. In the face of future sea level rise it is likely that coastal defences will be maintained along large sections of coastline and particularly in estuaries. In such cases, because the upper tidal limit is fixed by sea defences, any increase in mean sea level is likely to reduce the amount of suitable intertidal habitat available to seals as haulout sites. This would affect both species, but the effects on harbour seals would be more pronounced because a substantial proportion of the UK harbour seal population pup on intertidal banks in estuaries.

The UK State of the Climate Report 2019 (Kendon *et al.*, 2019) states that there are no compelling trends in storminess when considering maximum gust speeds over the last four decades. As there are no detectable trends there have been no studies that have so far shown a link between changes in UK storminess and climate change (Kendon *et al.*, 2019). However, in the short term, rising sea levels mean that storm surges and storm waves will increase the frequency and severity of wave action on breeding beaches. This will likely lead to increased mortality as observed in Welsh grey seal pupping colonies in 2017 (Buche & Stubbings, 2017; 2019). Such mortality events will likely increase in frequency and severity as sea levels rise.

Coastal erosion leading to mortality due to landslides are rare events, we have been unable to locate any published accounts. They are also unlikely to be greatly increased by projected climate change scenarios. The majority of coastal erosion concerns are along the south and east coasts of England. We are not aware of any sites where seals haulout beneath rapidly eroding cliffs in that region. In other areas there may be particular concerns about cliff beaches, but we are not aware of any information on changes in the rates/frequencies of land slips associated with seal haulout areas.

Novel diseases.

An additional concern is the spread of infection into regions where organisms may not have previously been exposed or where their capacity to survive may previously have been compromised due to unfavourable environmental conditions. With climate change, marine pathogens that were previously restricted to warmer, more southerly waters might be able to become established in UK waters (Baker-Austin *et al.*, 2017). It should be noted that mass mortality events are not all related to novel infectious disease.

Sanderson & Alexander (2020) reviewed occurrence of infectious disease-induced mass mortality (ID MME) events in marine mammals between 1955 and 2018. They conclude that extrinsic factors significantly influenced ID MMEs, with seasonality linked to their frequency and severity of these events. Importantly, they showed that global yearly SST anomalies were positively correlated with occurrence of ID MMEs. With climate change forecasted to increase SSTs and the frequency of extreme seasonal weather events Sanderson & Alexander concluded that epizootics causing MMEs are likely to intensify with significant consequences for marine mammal survival.

Renewable energy

<p>29. Scottish Government are aware of (recent) incidents involving seals becoming trapped and drowning in structures associated with fixed offshore wind developments. Are SCOS aware of such events, and if so, what structures were the cause, and can SCOS provide any information on the prevalence of these events?</p> <p>Furthermore, based on what we know from these events, what other marine structures could pose a similar risk to seals? Can any lessons be learned from other offshore industries or other regions outside of the UK with respect to mitigating and monitoring such events?</p>	<p>MS Q13</p>
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SCOS are aware of recent reports in which at least three seals have become trapped and drowned in subsea cable conduits associated with offshore wind turbines. As far as SCOS is aware there have been very few similar incidents at any other developments, although there is a single report of a seal accessing the central space inside a monopile structure via a subsea cable hole in the wall of the turbine.

Given the paucity of events and knowledge surrounding their occurrence, it is difficult for SCOS to recommend specific mitigation measures. It may be prudent to consider capping subsea openings that would allow seals to enter or minimising the time they are exposed.

SCOS are aware of an incident in which 3 dead seals were found inside subsea cable conduits (J-tubes) during the subsea preparation prior to inter array cable pull-in to substructures. Following the event, the Marine Scotland Licensing Operations Team wrote to offshore renewable energy developers with a series of questions to determine whether there have been any other similar incidents at other developments. Of the responses received (15), none had recorded similar incidents at their developments.

Other reports of seals becoming trapped within offshore wind development structures are sparse. In 2016, a grey seal was observed inside one of the monopiles during grouting procedures after pile driving had taken place (Gardline 2016). After investigations, it was concluded that the most likely entrance into the monopile was a subsea cable hole through the wall of the pile; this was 340mm in diameter and was located 3.8m from the sea floor after the pile had been driven in. The seal was observed within the pile over a period of approximately 2 hours. After this, no further sightings were made and it was assumed that the seal had either managed to exit back through the cable hole or had died (Gardline, 2016).

Seals are curious and are likely to investigate any novel structures in their environment. There are frequent reports of pinnipeds entering dam races, fish ladders and power plant cooling water system (CWS) intake pipes in North America (NMFS 2008). For example, between 1989 and 2006, a total of 69 California sea lions and five harbour seals were entrained by the cooling water system at the Scattergood Generating Station, Los Angeles, US; between 1978 and 2006, a total of 11 California sea lions and five harbour seals were entrained by the cooling water system at the El Segundo Generating Station, Los Angeles, US (NMFS 2008). However, it is important to highlight that CWS pipes are generally relatively large diameter (~700mm) and exert significant negative pressure due to the pump system, which will increase the likelihood of animals being drawn into the duct.

Table 14: Summary of the measured axial girths (cm) of seals captured as part of research by SMRU between 1988 and 2019. It is important to highlight that the estimates of diameter presented here are approximations based on the axial girth measurements and an assumed spherical cross section.

Species	Age class	Number of seals	Median axial girth \pm 95% CIs (cm)	Estimated diameter (cm)
Harbour seal	Pup	90	67 (50 – 89)	21 (16-28)
	Juvenile	60	80 (71 – 93)	26 (23-30)
	Adult	807	106 (92 – 120)	34 (29-38)
Grey seal	Pup	122	93 (82 – 101)	30 (26-32)
	Juvenile	202	90 (76 - 118)	29 (24-38)
	Adult	615	133 (104 - 153)	42 (33-49)

Given the relative paucity of reports of similar incidents across the renewables industry, it is difficult to recommend specific mitigation measures. However, it may be prudent to consider, where appropriate, capping subsea openings which have dimensions that would allow seals to enter, or minimising the time when these are exposed. For reference, a summary of the measured axial girths of captured harbour and grey seals from the SMRU capture database is provided in Table 14. It should be noted that seals are capable of forcing their heads through smaller holes as evidenced by cases of seals with frisbees and packing bands caught around their necks.

<p>30. There are known knowledge gaps associated with seals with respect to potential impacts in relation to underwater noise and collision risk with tidal turbines, for example. With these and other knowledge gaps in mind, can SCOS provide an update on emerging technologies they are aware of that could be used for quantifying seal behaviour and/or physiology (e.g., developments in animal borne sensors such as fNIRS).</p>	<p>MS Q14</p>
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There are a number of emerging technologies that may be useful for measuring the behaviour and physiology of seals. These include novel seal tag developments currently being developed to track the physiology and energetics of seals; these are likely to be important tools for measuring physiological and energetic consequences of interactions with anthropogenic activities, an important knowledge gap in being able to predict population consequences. Other developing technologies include remote and/or autonomous imaging monitoring techniques. A summary of these technologies, their applications and their current stage of development is provided.

In response to this question, we have assumed that the knowledge gaps relate primarily to behavioural and physiological responses by seals to offshore renewable energy developments and their associated activities.

There are a number of emerging technologies that may be useful for measuring the behaviour and physiology of seals and quantifying how these may be affected by interactions with offshore renewable energy activities. Broadly, these can be divided into technologies that are deployed on the seals (tags) and those that are remote or autonomous. The seal tag technology has been further divided into those that require retrieval to access data (archival) and those that transmit data via a communications system such as the GSM or satellite network (telemetry). It is important to consider

some broad benefits and constraints associated with each of these. In particular, the use of telemetry systems mean that data can generally be retrieved safely throughout the tag deployment. However, data resolution may be limited by effective bandwidths of the telemetry systems, often resulting in relatively low-resolution data, which may make them unsuitable for investigating some renewables research questions. In contrast, data collected by archival tags is generally high resolution but the need to retrieve the tags to access the data means that they may not be a practical solution for some species and applications. A high-level summary of emerging technologies that we are aware of is provided in Table 15; it may be useful to carry out a more detailed assessment of potential effectiveness of specific technologies to address specified priorities.

Table 15. Summary of emerging animal-borne technologies for measuring behaviour and physiology of seals in relation to anthropogenic activities and estimated Technology Readiness Levels (TRLs).

Technology	Type	Description	Potential applications	TRL	Development stage details
Sound and movement tags (e.g., DTAG)	Archival tag	High resolution sound and body movement archival tag for measuring received sound levels and behaviour.	Proven tool for measuring acoustic exposure, high resolution changes in 3D movements and dive behaviour, and foraging attempts in relation to anthropogenic activities.	9	Proven with a range of free-ranging pinniped species including grey and harbour seals (Mikkelsen <i>et al.</i> 2019; Goulet <i>et al.</i> 2020; Vance <i>et al.</i> 2021)
Sonar tags	Archival tag	A miniature sonar and movement archival tag to study the biotic environment and predator-prey interactions in aquatic animals.	Potentially valuable for measuring behavioural responses and changes in foraging behaviour in relation to anthropogenic disturbance.	9	Proven with a range of free-ranging pinniped species (Goulet <i>et al.</i> 2019).
NIRS phone tag	Telemetry tag	GPS Phone telemetry tag with integrated near-infrared spectroscopy (NIRS: non-invasive biomedical imaging technique) sensors that measures movements and dive behaviour, together with tissue-specific blood oxygen saturation, heart rate, and cerebral metabolic rate.	Potentially valuable tool for assessing the short-term behavioural responses and energetic costs (e.g., dive-by-dive) of anthropogenic disturbance.	6	Existing phone tag technology is proven with a range of free-ranging pinniped species, and NIRS sensor technology has been proven in free-swimming seals in captivity (McKnight <i>et al.</i> 2019). Integration of the two systems is currently underway at SMRU and is expected to be complete by 2024.
Body density phone tag	Telemetry tag	GPS Phone telemetry tag with integrated accelerometers to measure changes in at-sea body lipid stores (through changes in their buoyancy).	Potentially valuable tool for estimating the medium-long term (days-weeks) behaviour, foraging success, and changes in body condition as a result of anthropogenic disturbance.	6	Existing phone tag technology is proven with a range of free-ranging pinniped species. Use of accelerometer data to track body density has been validated in elephant seals (Aoki <i>et al.</i> 2011). Investigation of its effectiveness for shallow divers (e.g., harbour and grey seals) is currently underway.

Radar transponder tag	Telemetry tag	Radar transponder detected using XBAND radar to track seal surface locations and ID within a localised area of the radar (~100's m)	Potentially valuable low-cost tool for measuring localised seal interactions with anthropogenic structures/activities (e.g., radar mounted on tidal or wind turbines).	2	Radar transponder tags used in military applications (Pan & Narayanan 2011) and successfully used to track terrestrial species (Dore <i>et al.</i> 2020). Currently early concept only for seals.
ABR tag	Archival tag	Archival DTAG with an electroencephalogram (EEG) data stream to measure Auditory Brainstem Responses (ABRs)	Potentially valuable tool for measuring detection of anthropogenic sounds in combination with high resolution movements and dive behaviour.	4	ABRs have been measured from a stationary harbour porpoise using the prototype tag (Smith <i>et al.</i> 2021).
Asset recovery device	NA	Device that allows users to locate and release archival tags from seals. Emerging systems utilise a hand-held radio transceiver to trigger the recovery device.	Useful tool for the retrieval of archival tag technologies described above.	7	Technology proven in the lab and used successfully in a small number of pinniped studies (pers comm, Wildlife Computers).
Acoustic dosimeter phone tag	Telemetry tag	GPS Phone telemetry tag with integrated acoustic processing capabilities to measure long term acoustic exposure and behaviour.	Potentially valuable tool for estimating the medium-long term (days-weeks) behaviour and summary acoustic exposure from anthropogenic activities.	4	Existing phone tag technology is proven with a range of free-ranging pinniped species. Integration of acoustic processing capabilities is at an early design stage.
Electrical Impedance Tomography (EIT)	NA	Non-invasive medical imaging tool which uses surface electrodes to measure electrical conductivity, permittivity, and impedance, and create tomographic images of a localised region of the body. Provides measures of lung and cardiac function.	Potentially valuable tool for measuring the cardio-respiratory regulation of seals over days and assess the physiological responses to anthropogenic disturbance.	3	Has been used extensively in humans and terrestrial animals (e.g., Crivellari <i>et al.</i> 2021). There has been one validation study in diving humans (Magnani <i>et al.</i> 2018), but no application yet in marine mammals. Needs development to integrate with tags to work on free-living animals
Sub-THz Radar system	Remote	Radar system to provide automated detection, classification, and high-resolution tracking of seals at the water surface up to ranges of ~200m.	Potentially valuable low-cost tool for measuring localised seal interactions with anthropogenic	5	Radar system well proven technology. Application to detecting small marine mammals is currently underway through a collaboration

			structures/activities (e.g., tidal or wind turbines).		between University of St Andrews School of Physics and Astronomy and the University of Birmingham Microwave Integrated Systems Laboratory.
Remote camera systems	Remote	New generation of remotely accessible, autonomous HD video/infra-red to provide images of seals at remote locations.	Potentially valuable tool for measuring the abundance of seals at key locations (e.g., designated haul-outs), and movements and life history of individuals (Photo ID).	9	Technology well proven with a range of species. Recent increases in video and infrared image resolution, and advances in automated image processing make this a potentially attractive monitoring tool. An increasing number of seal applications.
High frequency imaging sonar	Remote	New generation of high frequency sonars to provide automated detection, classification, and high-resolution tracking of seals underwater up to ranges of ~50m	Useful tool for measuring the occurrence and behaviour of seals in close vicinity to infrastructure (e.g., tidal turbines or aquaculture facilities)	9	Technology well proven with a range of species (Hastie <i>et al.</i> 2019a; Hastie <i>et al.</i> 2019b). Recent advances in automated image processing make this a useful monitoring tool.

TRL	Description
1	Basic principles observed
2	Technology concept formulated
3	Experimental proof of concept
4	Technology validated in lab
5	Technology validated in relevant environment (industrially relevant environment in the case of key enabling technologies)
6	Technology demonstrated in relevant environment (industrially relevant environment in the case of key enabling technologies)
7	System prototype demonstration in operational environment
8	System complete and qualified
9	Actual system proven in operational environment (competitive manufacturing in the case of key enabling technologies; or in space)

<p>31. What is the current state of knowledge on grey seal interactions with tidal energy devices?</p> <p>Can SCOS recommend what the most appropriate avoidance rates should be in collision risk models or encounter rate models for grey seals and tidal turbines?</p>	<p>NRW Q4 & Q5</p>
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There is currently no information available on grey seal interactions with tidal energy devices. All of the research to date has been on harbour seals. Evidence from these harbour seal studies indicate some avoidance of operational tidal turbines at scales of 100s to 1000s of metres. Information on fine scale behaviour and the ability to evade collisions is still lacking.

There is little information on grey seal behaviour in tidally energetic waters, and SCOS recommend caution is extrapolating from harbour seal studies to grey seals.

There is currently no information available on grey seal interactions with tidal energy devices. This is a key data gap for assessing the impacts of tidal turbines on grey seals. However, as reported previously to SCOS, there are now a number of studies that report changes in harbour seals distributions in response to operational tidal turbines, including to the Strangford Lough turbine (Joy *et al.*, 2018), to playbacks of tidal turbine sounds (Hastie *et al.*, 2017; Robertson *et al.*, 2018), and to the MeyGen turbine array (Onoufriou *et al.*, 2021). Care should be taken when extrapolating from harbour seal observations as interspecific difference in foraging patterns and foraging ranges mean that potential barrier effects are likely to have less impact on grey seal. Published data on grey seal diving in tidally energetic environments is limited to a small sample of pups in a planned turbine array site in the Pentland Firth (Evers *et al.*, 2017).

Joy *et al.* (2018) analysed GPS/GSM location data from tagged harbour seals (*Phoca vitulina*) and used a Brownian Bridge movement model to develop fine scale probability density surfaces for seal density in the 3x3 km² region centred at the SeaGen tidal turbine before deployment and after installation of the turbine. Results suggested a mean spatial avoidance of 68% (95% C.I., 37%, 83%) by seals within 200 meters of the turbine, i.e., seals were 68% less likely to occupy the area within 200m of the turbine.

Hastie *et al.* (2017) carried out a series of acoustic playbacks of tidal turbine sounds (SeaGen turbine) in a narrow, tidally energetic channel on the west coast of Scotland. Results showed there was a localised impact of the turbine signal; tagged harbour seals exhibited significant spatial avoidance of the sound that resulted in a mean reduction in the usage by seals of 27% (95% C.I., 11%, 41%) at the playback location.

Robertson *et al.* (2018) studied the surface behaviour of harbour seals (measured from a land-based observation station) in response to acoustic playbacks of a tidal turbine (RivGen turbine) in Admiralty Inlet off the west coast of the US. The study reports that there were no significant differences in seal abundance or proximity to the sound source in response to the playbacks. However, the authors highlight that, due to markedly lower acoustic source levels compared to those used by Hastie *et al.* (2018), seals in their study would need to have been within 10 m of the playback location to experience similar received levels. Consequently, the authors suggest that the two studies (Hastie *et al.*, 2017; Robertson *et al.*, 2018) may actually be in agreement

More recently, Onoufriou *et al.* (2021) carried out a study of the behavioural responses by tagged harbour seals to the presence and operation of the MeyGen array of four tidal turbines in the Pentland Firth, Scotland. Distributions of seals were compared before and after installation of the array, and

between periods when the turbines were operating or stationary. The results showed that the presence of the turbine array did not significantly influence at-sea distribution but that the operational status of the array did. Model predictions suggested that seal presence decreased significantly up to 2 km from the turbine array during operational periods; mean change in usage within 2 km of the turbine was -27.6% (mean 95% CIs: -11% and -49%).

In practice, these empirical changes in abundance (Hastie *et al.*,2018; Joy *et al.*,2018; Onoufriou *et al.*,2021) could be most appropriately used to scale the underlying density estimates in encounter or collision risk models. It is also important to highlight that the observed responses were to a single point source or small array and may not be appropriate for estimating the effects of large operational tidal arrays. Further, recent evidence suggests that avoidance responses to tidal turbine noise are likely to be highly context-dependent (Hastie *et al.*,2021).

Although good progress has been made in understanding how harbour seals behave in response to operating turbine at scales of 100's to 1,000's of metres, information on the fine scale underwater movements (at a scale of metres) of individual seals around operating turbines remains the critical research gap with respect to deriving avoidance/evasion rates and understanding the potential impacts of tidal devices. However, a NERC and Scottish Government funded research project is due to deploy a combined active sonar and passive acoustic tracking system alongside an operating tidal turbine in 2022. This aims to track individual seals at high resolution (metres) within 30 m of the turbine and quantify movements around the turbine. The combination of this and the results of the previous studies (Hastie *et al.*,2017; Joy *et al.*,2018; Robertson *et al.*,2018) should provide information on behaviour of seals at the range of spatial scales required to effectively derive empirical avoidance rates to operating turbines.

In summary, there is a complete lack of information on close range evasion of turbine blades by any seal species and a general lack of information on interactions between grey seals and tidal turbines. Although data exist for harbour seals, their responses appear variable (Table 16) and there does not appear to be a scientific basis on which to move away from the 'present a range of potential avoidance rates' currently recommended for estimating collision risk (Scottish Natural Heritage, 2016).

Table 16. Summary of the previous studies to measure the avoidance of operating turbines, or their sounds, by harbour seals. The table shows the mean change in abundance (%), the tidal turbine and location of the study, the scale that a response was measured at, and the reference for the study.

Mean % change in abundance	Source	Scale	Reference
-68% (95% CIs: -37%, -83%)	SeaGen turbine (Strangford Lough)	Within 200m	Joy <i>et al.</i> (2018)
-27% (95% CIs: -11%, -41%)	Acoustic playback of turbine sounds (Kyle Rhea, Skye)	Within 500m	Hastie <i>et al.</i> (2018)
No significant change	Acoustic playback of turbine sounds (Puget Sound, U.S.)	Within 1000m	Robertson <i>et al.</i> (2018)
-28% (95% CIs: -11%, -49%)	MeyGen turbine array (Pentland Firth)	Within 2000m	Onoufriou <i>et al.</i> (2021)

32. Please could SCOS recommend the most appropriate at sea abundance and distribution data source for use in licensing applications and planning activities (both renewables and major infrastructure). Where such data sources provide relative density, could SCOS please advise on an appropriate method to convert to absolute density.

MS Q18

The most appropriate at-sea abundance and distribution estimates for informing licencing and planning decisions are those derived from habitat preference modelling (Carter *et al.*,2020). These are more up to date, in terms of both telemetry and haulout count data, than previous maps (Russell *et al.*,2017) and do not rely on null usage (decaying use with distance from haul out sites) for areas which lack sufficient telemetry data. However, the limitations associated with the respective methods (discussed in Russell and Carter 2020) should be considered during interpretation. Critically, for both the usage maps (Russell *et al.*,2017) and the habitat preference maps (Carter *et al.*,2020), the confidence intervals are calculated on a cell-by-cell (5 x 5 km cell) basis and thus should not be summed over multiple cells to generate lower or upper confidence intervals for a wider area (e.g., a windfarm footprint).

The habitat preference maps present at-sea seal density values as relative abundance (i.e., percentage of the at-sea population of the study area estimated to be in a cell at any one time), rather than absolute abundance (i.e., number of animals per cell). This is because the conversion process from relative to absolute abundance involves certain assumptions and caveats (discussed below). Thus, relative density maps (rather than absolute) should be used whenever possible. Nevertheless, absolute abundance estimates are required for certain applications. The process for estimating absolute density is detailed below. The at-sea abundance estimates used the most recent available haulout count data up to 2018 but can be updated in the future with more up-to-date counts.

Currently, uncertainty around the size of the at-sea population (at individual haulout sites or overall) cannot be incorporated into the maps; the lower and upper confidence intervals for absolute density maps only represent uncertainty in the habitat preference relationships, and therefore relate to uncertainty in the spatial distribution of a fixed number of seals emanating from each haulout area.

The predicted at-sea abundances are derived from combining the haulout counts which were used to generate the relative densities, the estimated proportion of the population hauled out and thus available to count during surveys, and the estimated proportion of the total population at sea during the main foraging season (i.e., excluding breeding and moulting). The latest at-sea maps of seal distribution (Carter *et al.*,2020) provide a relative index of density (the percentage of the total at-sea abundance, i.e., the mean maps will sum to 100% across all grid cells). Separate maps of 95% upper and lower confidence intervals associated with these mean relative density values are also provided. These confidence intervals encompass only the uncertainty in the habitat preference relationships (i.e., the latest haulout count was considered for each 5 x 5 km cell; no uncertainty in the relative weighting of haulout counts was incorporated). The density estimates (percentage of total at-sea population) presented in these maps were based on weighting the predicted at-sea distribution emanating from each 5 x 5 km haulout grid cell by its most recent August count. To convert these relative estimates to absolute estimates, the first step is to convert the total from the above-mentioned August haulout counts (36,982 and 46,763 for harbour and grey seals, respectively) into a population estimate, accounting for the seals that were at sea during the surveys. This was done using the mean estimated proportion of the population hauled out during the survey window, and thus available to count, from telemetry data: 0.72 for harbour seals (Lonergan *et al.*,2013) and 0.2515 for grey seals (SCOS-BP 21/02).

The second step is to estimate the mean total at-sea abundance during the months over which the maps represent (i.e., excluding breeding and moulting) using the proportion of the population estimated to be at sea; estimated to be is 0.8236 for harbour seals (October to May; Russell *et al.*, 2015) and 0.8616 for grey seals (May to August; Russell *et al.*, 2015). This results in an estimated at-sea total of 42,303 harbour and 160,203 grey seals⁸. These values could be used to calculate mean predicted absolute abundance over any number of grid cells by multiplying the percentage value in each cell of by the estimated total at-sea abundance for the species and summing this value over all grid cells of interest. Note that the proportion of the population estimated to be at sea is averaged across days and years, and thus does not account for variation in the proportion of time spent at-sea with season and state of tide. Moreover, lower and upper confidence limits for absolute density maps do not capture uncertainty related to variation in the proportion of time spent at-sea throughout the year, thus relative density maps should be used where possible.

Other Impacts and Emerging Issues

<p>33. a. Can SCOS review and analyse whether repeated disturbance to seals (such as repeated flushing into the water) could lead to localised behavioural or welfare implications up to a wider population-level effect?</p> <p>b. Can SCOS review current guidance for anthropogenic related seal disturbance and determine whether different categorised thresholds for land (public at beach haul outs), sea (by boat and water sports) and air (use of aerial drones), could be usefully calculated from NGO monitoring data and implemented to help reduce disturbance.</p> <p>c. Could SCOS please advise what data should be collected, at a minimum, on disturbance events? This would help to inform a standardised approach should a nationwide reporting and threshold system for recording disturbance events be developed.</p>	<p>Defra Q6</p>
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33a. The potential for individual and population-level consequences of disturbance to seals on land

Disturbance to hauled out seals has the potential for a range of effects from increased vigilance through to flushing seals into the water which may disrupt important rest, moult and breeding activities. Repeated disturbance is likely to exacerbate such effects and could lead to abandonment of pups, or possibly to desertion of haulout sites. Interspecific differences in sensitivity to disturbance could potentially exacerbate competition between grey and harbour seals. Little is known about the potential for human-induced disturbance of seals on land to adversely affect their ability to reproduce and survive, and therefore no information to allow estimation of population consequences. However, while disturbance can clearly affect individual animal welfare, there is no evidence that disturbance at haulout sites is currently a concern at the population level.

Observed responses to disturbance are very site and context specific and the impact of responses are likely to vary significantly depending on the species, time of year and life history stage of the animals involved. There are also well documented examples of both species habituating to disturbance from

⁸ Due to a review of the scalars associated with converting haulout counts into at-sea abundance estimates, these totals are different to those presented in Carter *et al.* (2020).

land-based tourism, boats and low-flying aircraft. Lower-level stress responses may occur with no visible behavioural response.

Although there are concerns about localised effects of repeated disturbance in specific areas, and legitimate welfare concerns where seals have been injured when flushing from haulout sites on rocky shores, there are no known examples where current levels of disturbance have led to population level consequences at regional or national scales for UK seals.

A frequently expressed concern over the energetic effects on seals of being flushed into the water is unlikely to be important. Both grey and harbour seals are thermo-neutral in the water temperatures experienced in UK coastal waters and little energy will be expended in running to the water. Disturbance during the moult and breeding seasons may have more important impacts, disrupting the skin and hair renewal during the moult and potentially breaking maternal bonds and suckling behaviour during the breeding season.

33b. Guidance for seal disturbance and thresholds

There is already sufficient information to show that disturbance threshold distances are location specific. For example, seals at some sites allow very close approaches by pedestrians or boats without showing overt signs of disturbance while at other sites seals respond to the presence of observers at ranges of several hundred metres. It is also clear that types of vessels or familiarity with specific vessels can alter the reaction threshold distances. Anecdotal evidence suggests that time of year also has a large effect on the sensitivity to disturbance.

All existing guidance documents share a number of commonalities. They all acknowledge that the likelihood and level of response is very variable and context dependent and that at some times of year seals are more sensitive than at others. They also all put a considerable emphasis on the use of careful observation, and several provide useful information on the signs to watch out for to indicate seals are being disturbed. Clearly defined distance thresholds or buffers are rare.

Most published guidance relates to disturbance by land-based activities, while a smaller number address boat-based disturbances, particularly for recreational boating and wildlife tourism. Drone activity around haulout sites and resulting disturbance events are increasing but few specific guidance notes address drone flying. There have been two published studies on the responses of seals to drone overflights which could be used to develop guidance.

33c. Advice on data that should be collected on disturbance events

To understand the potential for disturbance to significantly affect UK seal populations, at local, regional and national levels, more information is required on the levels and severity of disturbance events and the behavioural responses of seals, as well as information on the potential effects of these on individual health, energetics, breeding success and survival. Of particular importance are species specific disturbance effects/responses that have the potential to influence both the frequency and the consequences of interactions between seal species. Focused effort on documenting changes in breeding success or the health/energetic status of disturbed seals will be required to predict how disturbance on land could translate to population-level effects.

To develop distance thresholds requires information on seal responses together with detailed information on the type, intensity and proximity of the disturbing stimulus. NGO monitoring data on seal disturbance made available to SCOS did not include records of the distances of activities to which recorded responses occurred, therefore SCOS cannot use these data to derive such thresholds.

Data collection requirements depend on the specific question being addressed. Monitoring the presence and severity of response in relation to different activities and approach distances at a local level could allow the development of specific localised guidance for boat operators, tourists and recreational users of the coast.

Detecting population level effects of disturbance using observations at haulout sites would be difficult and would require monitoring effort to be focused on understanding the extent to which disturbance could affect the survival and reproduction of individuals. This could involve monitoring of a range of metrics related to human activity and numbers of seals hauled out, metrics related to breeding – suckling behaviour, weaning mass, pup counts, also individual measures of health and condition. It will not be possible to estimate some of these metrics from simple observations, and a nationwide study of such metrics would be extremely expensive. These could better be addressed through targeted research.

However, if co-ordinated and standardised visual observations are done at a sufficiently large number and representative range of sites, and over several years it might enable a nationwide meta-analysis of the potential nature and extent of human disturbance to seal populations. Some of this information may already be being routinely collected by local and regional groups, e.g., NGO monitoring of haulout numbers may allow an analysis of haulout patterns to investigate possible large-scale effects of human disturbance by comparing haulout counts times when human activity is higher, e.g., at weekends, with period of generally lower activity.

33a. The potential for individual and population-level consequences of disturbance to seals on land

In order to have a population-level effect, a stressor must affect the ability of individuals to survive and/or reproduce and enough individuals must be thus impacted to alter the trajectory of the population. Little is known about the potential for human-induced disturbance to seals on land to affect vital rates and therefore lead to population-level consequences. It is possible that in some circumstances, individual survival can be affected if disturbance results directly in injuries to individuals. For example, a disturbance event leading to a seal falling from height onto rocks whilst trying to reach the water resulting in severe injury or mortality, or as in a recent well-publicised case, severe injuries caused by dogs attacking seals⁹. Although these types of events are known to occur, and are a clear animal welfare concern, it is unlikely that they are currently occurring to the extent that population vital rates will be affected, and disturbance is not at present included in the list of potential population threats (see Answer 36 below).

Assessing the sub-lethal effects for individuals and the resulting population-level consequences of any stressor is a significant challenge because it requires detailed knowledge of the nature, extent and magnitude of individual responses to the stressor in question, as well as baseline knowledge of behavioural patterns, life history and demography of the population(s) in question. One common approach to the assessment of the population consequences is the Population Consequences of Disturbance Framework (PCoD) originally developed as a conceptual framework for acoustic disturbance by US National Academies of Sciences National Resource Council in 2005 (National Research Council (2005)) to evaluate how changes in behaviour caused by acoustic disturbance, may result in population effects by affecting the critical life functions of marine mammals. It was later generalised to all types of disturbance and describes a process, progressing from changes in individual behaviour and/or physiology, to changes in individual health, then vital rates, and finally to population-level effects. Much effort has been focused over the past decade on parameterising parts of this framework for a range of species and stressors (e.g., see Pirodda *et al.*, 2018).

⁹ <https://www.bbc.co.uk/news/uk-england-london-56489147>

Disturbance that occurs at sea is often assumed to affect energy balance of individual marine mammals through reduced foraging, for example as a result of displacement from foraging grounds, or as a result of increased travel cost due to avoidance of areas of disturbance and therefore the potential energetic consequences can be predicted on the basis of effects on energy balance and then on the basis of the links between energy balance and survival and reproduction. These individual based consequences can then be scaled up to population effects based on an estimate of the numbers of individuals affected. The consequences of disturbance to seals on land is harder to predict. This is partly because the drivers for seals to haul out are variable and therefore the consequences of disrupted haul out will be variable and context specific. Seals are thought to haul out for a variety of reasons; for rest, to carry out necessary physiological processes (e.g., e.g., moult, digestion), for predator avoidance and for breeding and provisioning pups. Therefore, any disturbance disrupting these activities has the potential to have a wide range of consequences which will be very context specific. There are a number of mechanisms by which chronic disturbance could be hypothesised to affect behaviour, physiology and health in a manner that could affect vital rates.

Behavioural responses of seals on land to human disturbance, such as increased alertness, movement towards water and flushing into the water, have been documented in many studies globally (e.g., e.g., Renouf & Lawson 1986, da Silva & Terhune 1988, Suryan & Harvey 1999, Strong & Morris 2010, Johnson & Acevedo-Gutierrez 2007). Changes to haul out numbers in response to disturbance and time taken for these numbers to recover to pre-disturbance metrics have also been documented in many studies (e.g., e.g., Henry & Hammill, 2001; Mathews, 2016; Paterson *et al.*, 2019). Documented responses are very variable and depend on the type of disturbance (pedestrians, dogs, kayak, powerboat, cruise ship, aerial etc.), distance of approach and location. However, there are also some clear UK examples where obvious human presence, in some cases involving close approaches to seals, is not acting as a deterrent to hauling out (e.g., e.g., Horsey) or breeding (e.g., e.g., Blakeney and Donna Nook) by grey seals. It is also apparent that hauled out seals of both species can habituate to the presence of, and tolerate close approaches by tourist boats, e.g., e.g., tourist boats at Dunvegan, the Farne Islands and Blakeney Point now regularly approach to within 20-30m of seals on haulout sites without causing apparent disturbance response. The likelihood of behavioural responses and their potential to lead to individual welfare, health and energetic consequences for individuals is clearly very location and context specific. In some areas in the UK (e.g., e.g., Cornwall, the Ythan Estuary) there are regular reports of repeated disturbance of seal haulout sites and growing concern among NGO groups that such disturbance will negatively impact individual seals and pose potential threats to the continued use of sites for hauling out and/or breeding (e.g., e.g., Cornwall Wildlife Trust, 2021).

However, the consequences of these responses for individuals are not well understood. A small number of seal telemetry studies have examined individual responses to disturbance in detail and these may be informative about the potential for such consequences. Andersen *et al.* (2014) and Paterson *et al.* (2019) indicate that harbour seals show strong site fidelity even when subject to repeated disturbance. Tagged harbour seals in Islay, Scotland would either haul out again shortly after the disturbance or would head off to sea on what appeared to be normal foraging trips. Similarly, Andersen *et al.* (2014) reported that tagged harbour seals at the Anholt seal reserve in Denmark would forage after being disturbed instead of resuming hauling out, which perhaps enabled them to minimise the cost of disturbance. Although pedestrian disturbances caused longer at sea trips than undisturbed trips, in general the extent and areas used during disturbed and undisturbed trips were comparable. Paterson *et al.* (2019) found there was no change in haul out use of harbour seals in terms of preferred sites, despite the availability of alternative nearby haul out sites so disturbed seals did not incur additional travel costs by moving to other sites, nor did abandonment of preferred sites appear to be a risk under conditions of repeated disturbance and flushing. Paterson *et al.* (2019) also found that the number of harbour seals on the haul out returned to 94% (95% CI 55–132%) of pre-disturbance numbers within 4 hr.

Data from such telemetry studies may provide suitable data on individual responses to disturbance events over appropriate time scales to use in bio-energetic simulation-based modelling approaches. Results from these simulations could be incorporated with models to predict future population trajectories and to determine whether population-level impacts are possible from observed and future projections of the extent of disturbance. However, as noted above, such an approach requires a detailed understanding of baseline abundance and demographics at appropriate scales which is not often available for many seal populations. There are no equivalent data on the individual responses of grey seals to disturbance on land to currently enable this approach for grey seals.

The effects of disturbance are also likely to vary significantly depending on the time of year or life history stage of the animals involved. For example, Andersen *et al.* (2012) observed that Anholt harbour seals left haul out sites to enter the water during the pre-breeding and post-breeding periods but were reluctant to leave the haulout during the breeding period. In addition, return times depended on the time of year with seals coming back during the hours of darkness during pre and post breeding periods but came back immediately after disturbance during the breeding period. This could indicate that the probability of a behavioural response is negatively correlated to the potential consequences of response. It is important to note therefore that a lack of response does not equal a lack of impact and conversely a response may not indicate an impact. This, of course, makes interpreting observed responses (or lack of them) very difficult.

Repeated flushing into the water could have more significant consequences during the annual moult. Both harbour and grey seals spend more time hauled out at this time to circulate blood to their skin, allowing for efficient regrowth of hair avoiding excessive heat loss to water (Ling, 1970). Repeated immersion during this period may slightly increase heat loss, but more importantly it may impede the growth of new hair, extend the moult duration, and affect the longer-term energy balance of individual seals. The magnitude of disturbance required to result in an effect on survival or breeding success as a result of this pathway is unknown.

During breeding on land, disturbance has the potential to affect survival and reproduction directly. Pups forced into the water may suffer thermoregulatory impacts and smaller pups with less insulation are at risk of hypothermia. Energy balance could be affected which might lead to lower weaning mass. Similarly, if suckling is disrupted this could result in a reduction in the energy transfer from mothers to pups during lactation also resulting in lower weaning mass. Pup weaning mass correlates with suckle bout durations during early and mid-lactation in elephant seals (Engelhard *et al.*, 2002). Weaning mass and condition correlates with post-weaning survival in a number of seal species (e.g., McMahon *et al.*, 2000; Hall *et al.*, 2001; Harding *et al.*, 2005) so this provides a potential mechanism for disturbance-induced impacts on the survival of pups.

It is unlikely that disturbance of individual suckling bouts or even repeated, short duration disruptions would have a detectable effect on overall energy transfer, as short delays in suckling are not important. Indeed, Engelhard *et al.* (2002) found that in spite of the relationship between suckling and weaning mass, there was no evidence that the presence of disturbance directly affected weaning mass in southern elephant seals. Engelhard *et al.* (2001) reported that although mothers and their pups were smaller in an area of higher human activity, in proportion to their own size, females in areas of higher disturbance produced weaners of similar mass. This pattern of smaller mothers being present in more disturbed sites may have been as a result of site selection by larger, more experienced females selecting less disturbed sites. Similarly, Wilkinson and Bester (1998) found that direct onshore human disturbance (which was described as frequent and considerable) was not a factor in the decline of elephant seal numbers on Marion Island. In addition, despite frequent visits by tour boats to grey seal breeding beaches on Ramsey Island, Wales, and documented behavioural responses to the presence of human activity, no reduction in reproductive rate was recorded (Strong & Morris, 2010).

Direct disturbance could result in mother-pup separation leading to pup abandonment, however complete pup separation as a result of flushing is unlikely unless it happens very early in the lactation period due to close coordination between mums and pups and the role that vocal behaviour plays in the maintenance of the bond (McCulloch & Boness, 2000; Sauv   *et al.*, 2015). Male aggressive charges at human intruders, or adult seals fleeing from human disturbance has the potential to cause direct mortality to pups.

The large numbers of harbour seal pups taken into rescue centres in the UK and the rest of Europe include a proportion of pre-weaned pups, suggesting that harbour seal mother pup bonds are susceptible to disturbance. There is little information on the levels of disturbance required to sever these bonds. Disturbance during catching and handling of harbour seal pups for studies of pup survival and pre- and post-weaning foraging patterns suggest that single disturbance events, even those involving protracted separation of the pair and extensive disturbance on the haulout site did not lead to breakdown of the mother pup bond (Bekkby, Bjorge & Bryant 2000; Bekkby & Bjorge, 2003; Hanson *et al.*, 2014). Repeated captures to track the mass changes of harbour seal pups during late lactation and 40 days after weaning (Muelbert & Bowen, 1993) did not cause any pup mortality. Previous studies have shown that harbour seals can be displaced from haulout sites when exposure to anthropogenic activity is continued over several years (Becker, Press, & Allen, 2009; Becker, Press, & Allen, 2011).

There are also a number of other potential impacts as a result of disturbance including increased predation risk and stress. Disturbance leading to flushing could result in increased risk of predation. This is only likely in a limited number of areas in the UK, for example in Shetland, where seal predation by killer whales is known to occur. Effects on physiological parameters as a result of stress is much harder to determine, but it is possible that in some circumstances, high levels of chronic disturbance could lead to levels of physiological stress that could affect health status of individuals.

Although there are no examples of human disturbance leading to population level effects in the UK, there is evidence linking human disturbance to effects on vital rates and/or declines in other phocid populations. Although not the only factor, human disturbance is one of the factors thought to be responsible for the decline of both the Hawaiian and Mediterranean monk seals to critical levels. In some cases, human disturbance led to Hawaiian monk seals abandoning core habitat and moving to sub-optimal habitats where breeding success was lower (Gerodette & Gilmartin 1990). Liukkonen *et al.* (2017) showed that perinatal mortality in Saimaa seals increases significantly in areas in which the nearest building is within 800 m of a birth lair.

In the UK, although observed disturbance is clearly an animal welfare concern, and measures to reduce disturbance are a sensible approach, particularly as many local seal populations are increasing in areas where there is much human activity and interactions are likely to increase, there is currently no evidence that disturbance is affecting the numbers of seals present in any areas in a local or regional context or is affecting breeding success. Areas where the largest declines in UK harbour seal populations have been observed are the areas likely to be the least disturbed (e.g., Orkney and the North coast of Scotland) and grey seal numbers continue to increase in some of the most heavily populated areas of coast.

To conclude, while there are a number of potential mechanisms for human disturbance to affect individuals to the extent that their ability to breed successfully or survive might be reduced, to understand the potential for disturbance to significantly affect UK seal populations, at local, regional and national levels, more information is required on the levels and severity of behavioural responses and the potential effects of these on individual health, energetics and breeding success and ultimately survival. Focused effort on documenting changes in breeding success or the health/energetic status of disturbed seals will be required to predict how disturbance on land could translate to population-level effects.

33b. Guidance for seal disturbance and thresholds

Specific guidance on the offence of intentional or reckless harassment at designated seal haul-out sites in Scotland has been published¹⁰. This guidance details that the sensitivity of seals at haul-outs can be site specific and can highly variable. It also highlights that mothers with pups are more sensitive than other seals and that pups on land can be separated from their mothers. Sensitive times are described as breeding and moult seasons and greater caution is required during these times. Although careful to highlight that it is up to the courts to decide what might constitute an offence, the document offers guidance on the terms ‘intentional’, ‘reckless’ and ‘harassment’. Under the definition of ‘harassment’ the following is stated: “it would include any action that causes a significant proportion of seals on a haul-out site to leave that site either more than once or repeatedly or, in the worst cases, to abandon it permanently.” A number of examples are provided that Marine Scotland may consider would constitute intentional or reckless harassment. These include “approaching too close to a designated seal haul-out from seaward (particularly in a kayak, jet ski or speed boat) that causes a significant number of seals on a designated haul-out to stampede into the water.” Also: “Any other intentional or reckless action that causes a significant number of seals on a designated haul-out to stampede into the water.” Note in these two cases the emphasis on the reactions of the seals rather than the actions themselves.

There are also a number of actions that are not linked to any specific consequences: “Intentionally or recklessly “buzzing” seals on a designated haul-out by repeated overflight in a fixed wing aircraft or helicopter at low level (i.e., less than 1,000 feet). Intentionally or recklessly approaching or sneaking up on seals on designated haul-outs from the landward side. Intentionally or recklessly crowding or encircling seals on designated haulouts.” Specific guidance is also given on how to determine the response of seals and it is emphasised that it is important to allow the animals to decide how close is acceptable.

The same concerns of aerial disturbance will apply to drones. Recreational use of drones is expanding rapidly and there are many press reports and social media examples of disturbance of seals at haulout sites, often involving flushing of animals from haulout sites. There is little guidance directly targeted at drone flying over seal haulouts, but there is published information on the effects of drone flights on hauled out grey seals during the breeding and moulting seasons (Pomeroy *et al.*, 2015; Arona *et al.*, 2018)) that could be used to generate guidance.

There has been no specific monitoring of the success of the designation of seal haul outs in reducing deliberate harassment at haul outs. However, it is clear that the legal protection afforded by the seal haul out designation provides a framework for activities causing disturbance at designated haul outs to be reported and subsequently managed. The public awareness of this protection may be contributing significantly to seal protection in some locations.

Although such restrictions do not apply in the rest of the UK, guidance on general seal watching has been published by the Marine Management Organisation¹¹ and information notes on the evidence relating to the effects of wildlife watching and a range of different activities in relation to disturbance on seals at haulout sites in marine protected areas have been published by Natural England¹². Guidance notes to provide advice on best practice for wildlife watchers and wildlife tourism operators have been

¹⁰ https://consult.gov.scot/marine-environment/possible-designation-of-a-seal-haul-out-site/user_uploads/guidance-on-the-offence-of-harassment-at-seal-haul-out-sites.pdf-1

¹¹ <https://marinedevelopments.blog.gov.uk/2016/08/11/seals-protected-illegal-touch-feed/>

¹² Natural England Evidence Information Notes EIN025-37

published by both government and voluntary organisations (e.g., e.g., NatureScot¹³, National Trust¹⁴ and The Seal Alliance¹⁵).

All of the available guidance documents share a number of commonalities. They all acknowledge that the likelihood and level of response is very variable and context dependent, and that some times of year are more sensitive than others. They also all put a considerable emphasis on the use of careful observation, and several provide useful information on the signs to watch out for to indicate seals are being disturbed. Clearly defined distance thresholds or buffers are rare.

NGO monitoring data on seal disturbance made available to SCOS did not include records of the distances at which the reported responses occurred therefore SCOS cannot use these data to derive such thresholds. There exists a wide variety of published data on the distance at which seals respond to various activities at a range of locations but as highlighted above, these are highly variable and context specific and therefore it is difficult to determine generalised thresholds or buffer zones.

33c. Advice on data that should be collected at disturbance events

It is generally accepted that minimising disturbance of wildlife is beneficial, and that disturbance should be avoided wherever possible. This is particularly true for some haul-out sites along rocky coasts with high tidal range, where disturbance can directly lead to injuries. However, the majority of disturbance events do not lead to injury, and as detailed above, there is little information available for seals to assess the consequences of disturbance for individuals or populations.

Although the variation in responsiveness of seals to disturbance limits the usefulness of setting generalised threshold distances, monitoring the presence and severity of response in relation to different activities and approach distances at a local level could allow the development of specific localised guidance for boat operators, tourists and recreational users of the coast. This would require the adoption of standard definitions of response type and severity, or restricting the definition of a response to a very specific outcome, such as animals leaving the haul out, as well as a reliable way of estimating the distance of approach of each activity. Non-responses to the presence of activity are equally as important to record as responses. Carrying out behavioural observations during non-disturbed periods would also allow 'normal' activity budgets to be determined, which would allow useful comparison with behaviour in the presence of human activity and develop an understanding of the impact of the observed responses. Regular counts of haul outs in a region at an appropriate spatial scale (to be able to detect movement between haul outs in response to disturbance) would also be required to link disturbance to an effect on haul out use and local population size.

As discussed above, to address concerns about population level effects of disturbance, information about the extent to which disturbance could affect the survival and reproduction of individuals is required. This information is challenging to collect using observation alone and might be better tackled with targeted research. However, if co-ordinated and standardised visual observations are done at a sufficiently large number of representative sites, and over a sufficient period of time, the resulting information could enable a nationwide meta-analysis of the potential nature and extent of disturbance. It is likely that some of this information (for example haul out counts, activity budgets and occurrence of disturbance events) is already being collected by local groups and a co-ordinated effort to standardise and collate these datasets may be useful, e.g., e.g., NGO monitoring of haulout numbers may allow an analysis of haulout patterns to investigate possible large-scale effects of human disturbance by

¹³ <https://www.nature.scot/professional-advice/land-and-sea-management/managing-coasts-and-seas/scottish-marine-wildlife-watching-code>

¹⁴ <https://nt.global.ssl.fastly.net/godrevy/documents/how-to-watch-seals-responsibly-without-disturbing-them.pdf>

¹⁵ <https://www.gov.uk/government/news/public-urged-to-give-seals-space>

comparing haulout counts times when human activity is higher, e.g., e.g., at weekends, with period of generally lower activity. Although this would not provide information on specific disturbance events it may shed light on the scale of disturbance effects.

Two studies have examined the effects of controlled disturbance of harbour seals on haul-out sites on their short and medium-term behaviours (Andersen *et al.*, 2014; Paterson *et al.*, 2019). In both cases telemetry data showed that post disturbance movements and swimming behaviour were similar to behaviour after normally terminated haul-out periods. As stated above, responses will likely be context specific and different types or levels of disturbance may produce different responses. To date there have been no specific studies of the movements or swimming behaviour of grey seals in response to disturbance at haulout sites.

<p>34. If funding became available to undertake post-mortems on a limited number of seals in England, could SCOS please advise on which strandings should be the top priority to investigate? For example, which apparent causes of death, which species, age class, location etc. Could additional post-mortems be of benefit to our understanding of wider issues e.g., e.g., on the decline in The Wash harbour seal population, for example?</p>	<p>Defra Q10a</p>
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There are several current policy-related research questions relating to the status of English seal populations and their management that could be usefully informed by post-mortems of seals. Examination of the cause of death and associated ecological and life history information for any stranded harbour seals in the southeast England Management Unit may help inform our understanding of drivers of the current observed decline.

Other areas that could be informed by seal post-mortems in general include disease surveillance, ecological factors such as diet, exposure to marine pollution (including entanglement) and evidence of interactions with fishing gear. Collection of material from bycaught grey seals for genetic analysis may help elucidate the population source of bycaught animals.

Stranding schemes can provide a useful means of surveillance of wildlife health and disease and SCOS would support the inclusion of seals in the national stranding scheme for England and Wales (they are already part of the SMASS in Scotland).

In addition to the relatively small number of seals that can be subjected to full post-mortem examination, information from detailed photographs with associated location and date/time information can provide useful data. Promoting such data collection and establishing systems for gathering and collating such reports may be a cost-effective approach to understanding the patterns and trends of seal strandings. Notwithstanding the research priorities, various physical aspects of individual stranding events will need to be taken into account when deciding whether or not to collect a specific carcass for post-mortem examination, including freshness of the carcass, location/accessibility and presence or absence of a clear cause of death, e.g., net entanglement or severe trauma.

In terms of structuring a general strandings sampling programme and prioritising cases, the existing strandings schemes in the UK and internationally, have developed best practices. Those schemes should inform any proposed increase in sampling effort in the UK. However, sampling schemes need to be flexible enough to respond to new and emerging problems and there are some current policy related research questions relating to the status of English seal populations that may be usefully informed by post-mortems of seals.

It is difficult to determine which strandings (in terms of location, species and age) should be prioritised without an understanding of strandings patterns and trends – the utility of post-mortems in providing information to inform the management of a species will increase with increasing sample sizes and durations of monitoring, and it is difficult to determine this *a priori*. In terms of structuring a general strandings sampling programme, these issues have already been addressed by the existing strandings schemes in the UK and Europe, and the best practices of those schemes should inform any proposed increase in sampling effort in the UK. However, sampling schemes need to be flexible enough to respond to new and emerging problems and there are some current policy related research questions relating to the status of English seal populations and their management that may be usefully informed by post-mortems of seals.

The first of these is the recently observed decline in the harbour seal population in the southeast of England (SCOS-BP 2021/06). The drivers behind this decline are currently unknown but any recording, recovery, and post-mortem examination of dead harbour seals in this region may inform our understanding of the reasons for the decline and provide information/evidence to allow us to rule potential drivers in or out of contention. Although it is important to note the inherent bias in strandings data, in that few seals that die beyond a few km from the shore will be likely to strand, specific causes of death that may occur more offshore will likely be underrepresented in strandings. Notwithstanding this bias, identifying the major causes of death of any strandings in this region may still allow us to determine the degree to which any diseases or particular conservation threats may be having an effect, and whether they could be occurring at a magnitude that could be responsible for the observed decline. The demographics and locations of stranded animals may also be informative.

In addition to determining any emerging patterns of specific causes of death, a more detailed examination of stranded harbour seals may be useful to inform our understanding of potential ecological drivers of observed population trends. This includes investigations such as the examination of stomach contents and/or analysis of tissues for stable isotope signatures, providing information on diet, as well as the characterisation of contaminant and toxin exposure, any evidence of interspecific predation (e.g., spiral wounds typical of grey seal predation) and information on the age and reproductive status of stranded individuals. Potential biases in the animals available for sampling and problems of small sample sizes must be considered when analysing such datasets.

The UK harbour seal population experienced significant mortality from outbreaks of the Phocine Distemper Virus (PDV) in 1998 and in 2002. It is possible that another PDV epidemic will occur and given the already declining status of the southeast England harbour seal population, its effects could be catastrophic. Screening for PDV in stranded individuals may provide an early warning system for a future outbreak.

There are a number of other general policy areas that could be informed by investigations of stranded seals if it involved collation of evidence over a relatively long timescale. These include the presence and incidence of evidence of interactions with fisheries and fishing gear, ship and propeller strikes and evidence of entanglement and ingestion of micro and macro plastics. If carcasses were sufficiently fresh, the structures of the ears can be examined for evidence of hearing damage that might have been caused by anthropogenic noise.

Notwithstanding the research priorities, various physical aspects of individual stranding events will need to be taken into account when deciding whether or not to collect a specific carcass for post-mortem examination, including freshness of the carcass, location/accessibility and presence or absence of a clear cause of death e.g., net entanglement or severe trauma.

<p>35. Can SCOS advise on recent observations of ‘mouth rot’ (e.g., swollen muzzles; open wounds and oral ulcerations that can lead to bone exposure, bone necrosis and potentially septicaemia and death), an unknown disease that appears to be affecting harbour seal pups on the east coast of England? Specifically, what data should be recorded to enable and enhance further investigations? Do SCOS consider that this disease should be taken into account during the investigation of the harbour seal decline in the Wash?</p>	<p>Defra Q10b</p>
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In order to evaluate the prevalence of mouth rot in harbour seal pups and the extent to which it poses a threat for conservation, including its potential as a contributory factor in observed regional harbour seal declines, a robust quantitative analysis of the incidence and circumstances of the disease is required. To enable this requires records of each case observed, with ancillary information recorded to provide useful covariate information (see below). Ideally this information should be provided in the context of all cases of rescue/stranded harbour seal pups to allow an evaluation. This will allow an evaluation of trends and indicate any potential overall increase in pup mortality rates. This will likely involve collating data from a range of sources including rescue centres, RSPCA, SSPCA, SMASS, BDLMR and CSIP.

The causal agent and the extent of the problem are currently unknown, but SCOS understands that detailed investigations of the pathology, bacteriology and virology of the disease are underway by researchers at Teesside University together with British Divers Marine Life Rescue.

SCOS are aware of recent observations of ‘mouth rot’ in harbour seal pups on the east coast of England because of discussions at recent Defra led-seal network meetings attended by SMRU. The causal agent and the extent of the problem are currently unknown, but SCOS understands that detailed investigations of the pathology, bacteriology and virology of the disease are underway by researchers at Teesside University together with British Divers Marine Life Rescue (BDMLR) veterinary staff, and SCOS look forward to the results of these investigations.

Based on photographs of the lesions it is apparent that ‘mouth rot’ is not a new issue. In 2013 several seals with similar mouth lesions were recorded in southeast England, and similar lesions were recorded in at least two harbour seals in the same region in 2002 collected during the PDV epidemic. There are also unconfirmed reports that similar ‘mouth rot’ cases have been observed in harbour seals on the European mainland coast.

In order to evaluate the current prevalence of this illness, and its potential population level effects, including the potential for this to be a contributory factor in the declines observed in the southeast England SMU, information is required on a number of metrics. This includes the number of cases observed, with care taken to ensure double reporting of cases is minimised or at least identifying where double reporting cannot be ruled out. The geographical location of each observed case and the outcome of each case (survival or recovery) should also be recorded. The sex, estimated age, mass and condition (length and girth measurements) of each affected seal will also provide useful covariate information in further investigation of patterns of incidence and help identify risk factors.

Data on the total numbers of seal pup rescues/call outs and their locations and outcomes will also be useful in order to place the numbers of mouth rot cases in the context of total cases. Evaluation of potential biases in reporting and recovery will also be required to assess whether the collated data are representative of the likely level of prevalence of the illness in the wider population. Details of pup stranding/rescue data in previous years on a UK wide basis will also allow an evaluation of trends and

indicate any potential overall increase in pup mortality rates. This will likely involve collating data from a range of sources including rescue centres, RSPCA, SSPCA, SMASS, BDLMR and CSIP.

<p>36. Can SCOS review and provide an update on any new studies looking into how macroplastics, microplastics, chemical pollution (including but not exclusively pharmaceutical drugs flushed into water systems), Abandoned, Lost or otherwise Discarded Fishing Gear (ALDFG), other marine pollution and Harmful Algal Blooms are affecting seal populations? What research is specifically required to help fill data gaps and evidence base in this area? How could impacts of plastic pollution be usefully picked up in part under reporting of strandings and post-mortem work by CSIP?</p>	<p>Defra Q11</p>
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SCOS are not aware of any significant new information published since SCOS 2019, on the effects of macroplastics, microplastics, abandoned (ghost) fishing gear or other plastic pollution on seal populations.

The number of studies investigating the effect of microplastics, macroplastics, abandoned fishing gear and other forms of plastic pollution on seals is limited. There have been studies on discarded fishing gear and on the trophic transfer, retention, and excretion of microplastics and there is ongoing research on the impact of plastic contaminants and plasticizers on UK seals. However, the population consequences of these forms of marine debris have not been quantified so we do not know whether they are of concern. There are significant information gaps and current research will help shape future studies.

Both the CSIP and SMASS are collaborators and co-authors on recent publications on frequency of occurrence of plastics in seals. The strandings recovery and post-mortem work carried under these schemes is an essential part of ongoing studies.

Studies of POPs, plasticizers, and antimicrobial resistance (AMR) are continuing, with indications of widespread AMR organisms in seals in UK, Ireland, and Canada.

The environmental effects of pharmaceuticals entering the ocean either directly down rivers or through sewage treatment systems is a major concern. However, with the exception of recent studies on AMR, there are no published reports on the effects of pharmaceuticals on seals.

Some of the issues raised in this question were addressed in SCOS 2019 and updated in SCOS 2020. The relevant parts of the 2020 answer are included here for completeness. Although there have been a number of published reviews and reports, SCOS is not aware of any significant developments in the field since the previous report that would materially change the general conclusions.

Nelms *et al.* (2021) reviewed conservation threats to marine mammals and included plastic pollution, chemical contaminants, and pathogen pollution as key threats to marine mammals in general. However, at present there is insufficient information to assess the population-level effects of interactions of seals with plastic (e.g., ingestion and entanglement) or other forms of pollution.

Microplastics

The potential impact on seals of different types of plastic marine debris at the individual and population level varies depending on their sources and physical characteristics, e.g., different size ranges. Microplastics (defined as plastic particles <5mm long) can be translocated across the gastro-intestinal membranes via endocytosis-like mechanisms (Alimba & Faggio, 2019) in invertebrates. They are also

capable of adsorbing organic contaminants (such as persistent organic pollutants (POPs)), metals and pathogens, which will add to their toxicological profile as these will be in addition to their inherent plasticizer compounds.

Nelms *et al.* (2019a) investigated the occurrence of microplastics in the gastrointestinal tracts of 50 marine mammals of 10 different species that stranded around the UK coast. Microplastics were ubiquitous: they were found in every animal examined but at relatively low numbers per animal (mean = 5.5) suggesting the particles were transitory. Stomachs contained a greater number than intestines, indicating possible temporary retention. However, only 3 grey seals and 4 harbour seals were included in this study. Nelms *et al.* (2019b) also found microplastics (1-5 pieces per gram of faeces) in 8 out of 15 grey seal scats (53%). The samples were all collected during the breeding season on Skomer Island off the Welsh coast, so they may only represent near-shore exposure.

Hernandez-Milian *et al.* (2019) recorded microplastics in 12 out of 13 grey seals drowned in bottom set trammel nets in a monkfish fishery off the south coast of Ireland and Philipp *et al.* (2020) found microplastics in all ten intestine samples and all nine faecal samples from stranded harbour and grey seals in Germany. Bravo Rebolledo *et al.* (2013) analysed 107 stomachs, 100 intestines and 125 scats of harbour seals from the Netherlands for the presence of plastics. They reported the occurrence of plastic in 11% of the stomachs, 1% of the intestines, and 0% of the scats. Hudak & Sette (2019) found anthropogenic micro debris (<500 µm) including cellophane, alkyd resin and EPDM rubber in 6% of harbour seal and 1% of grey seal scats collected at haulout sites on Cape Cod, Massachusetts, USA.

Nelms *et al.* (2018) showed that grey seals readily excrete microplastics in their faeces and feeding studies using polystyrene balls (3 mm) to determine fish otolith recovery rates, suggest that they all pass through the GIT within 6 days (Grellier and Hammond, 2006). Zantis *et al.* (2021) reviewed the literature on microplastics in marine mammals. All relevant published information from that review is included above.

Whilst microplastics may be readily excreted by seals, retention in the stomach and intestine prior to passage may facilitate the release of chemical compounds such as plasticizers during the digestive process. Toxicological impacts of microplastics for seals have not been reported in the literature at either the individual or population levels.

Senko *et al.* (2020) recently reviewed the published information on individual and population-level effects of plastic pollution on marine megafauna. They conclude that, despite increased reporting of the extent and intensity of plastic pollution in the marine environment, and the well-documented effects on individuals, the extent and magnitude of demographic impacts on marine megafauna have not been addressed.

Microplastic ingestion is unlikely to cause immediate or direct issues for animal health but may lead to sub-lethal effects. Greater understanding of what happens to ingested microplastics is needed. Nelms *et al.* (2021) identified three key knowledge gaps with respect to plastic pollution:

- Potential for sub-micron sized plastic particles to pass through the gut wall and into the blood stream, and reach organs, such as the liver or the lymphatic system.
- Extent to which plastic ingestion exposes marine mammals to chemicals on or within them compared to their usual dietary and environmental exposure.
- Effects of plastic ingestion on animal health and exposure to disease.

Ingestion of macroplastics

The ingestion of larger plastic debris, the macroplastics, may cause blockage in the gastrointestinal tract and injury to the gut mucosa. Macroplastic ingestion by marine mammals has been reported to have a range of effects such as causing obstruction/blockage/damage to the gastrointestinal tract (e.g., Baird and Hooker, 2000; De Stephanis *et al.*, 2013; Jacobsen *et al.*, 2010), and inflammation that can reduce feeding rates and potentially lead to malnutrition (Kühn *et al.*, 2015). However, to date these studies have all addressed effects of macroplastic ingestion by cetaceans. The prevalence of this as a cause of morbidity or mortality in UK seals is not known. It is rarely reported as a proximate or ultimate cause of death in seals by the Scottish Marine Animal Strandings Scheme (<http://www.strandings.org/smass/>).

Kühn and van Franeker (2020) reviewed marine mammal interaction with marine debris by either ingesting it or by getting entangled in debris. Out of 123 marine mammal species, 69 were recorded with ingested plastics. The incidence of macroplastic ingestion was reported as 4.4% in studied phocid seals, but it has not been possible to identify the relevant primary publications. It appears that plastic ingestion varies widely between marine mammal taxa, but in general seals appear less prone to plastic ingestion than cetaceans.

Ingestion of macroplastics, and subsequent impacts on health, seem to be less common in seals than other marine mammals but more data from post-mortems would be beneficial. To date, ingestion of macroplastics has not been recorded as cause of death in UK seals. However, sometimes the cause of death may be listed as an infection, but the cause of infection was an obstruction caused by a synthetic item. Making such links more explicit in reporting would help us better understand the extent of the problem.

Abandoned, Lost or otherwise Discarded Fishing Gear (ALDFG)

Entanglement is likely to be a key concern due to the propensity of seals to become entangled in netting, the immediate risk to health caused by entanglement-related injuries, and the welfare issues relating to these injuries (i.e., protracted and painful death). The fact that entanglements are often highly visible and distressing sights makes this a more pressing issue in public opinion.

Jepsen & de Bryn (2019) reviewed the available literature on entanglement of marine wildlife in general. They concluded that entanglement in oceanic plastic pollution poses a threat for at least 243 marine species and that most of the plastics that cause entanglements appear to be monofilament lines, ropes, and other fishing related gear.

Entanglement of seals in marine and plastic debris, particularly discarded fishing gear may increase the risk of drowning but perhaps more commonly, may restrict feeding or cause deep blubber and skin cuts and abrasions (particularly around the head and neck). Allen *et al.* (2012) used sightings records and a photo identification catalogue from a haul out site in southwest England to investigate the prevalence of entanglement in grey seals. Between 2004 and 2008 the annual mean entanglement rates varied from 3.6% to 5% (n= between 83 and 112 animals). Of the 58 entangled cases in the catalogue, 64% had injuries that were deemed serious. Of the 15 cases where the entangling debris was visible, 14 were entangled in fisheries materials.

Butterworth (2016) concluded that globally pinnipeds are at the visible end of the spectrum of animals which become entangled, snared, trapped or caught in marine debris, particularly plastics in the form of net, rope, monofilament line and packing bands, with severe consequences. This is in line with a study by Unger and Harrison (2016) who characterised beach litter based on a data set established by the Marine Conservation Society (MSC) beach-watch weekends. Debris collected around the UK was divided into three main types of debris: (1) plastic, (2) fishing, and (3) fishing related plastic and rubber on a

total of 1023 beaches. Debris attributable to fishing was identified on clusters of beaches mainly located on the coasts of Scotland and along the English Channel. They concluded that the fishing industry is responsible for a large proportion of the marine debris on UK beaches, particularly in areas with adjacent fishing grounds.

While individual effects of entanglement have been widely reported, extrapolating from such observations to estimate population scale mortality rates has not been possible. Sightings of entangled individuals, or seals with serious injuries, may not be representative of the frequency of occurrence in the population as the sightings could potentially be biased in either direction depending on whether entangled seals are more or less likely to come ashore. Likewise, strandings of seals killed by entanglement will be under-represented as seals killed more than a few kilometres offshore are unlikely likely to strand and entangled seals may be more likely to sink due to the weight of negatively buoyant netting. Although it is not clear what the population scale effects of entanglement are, there are examples where entanglement in discarded nets may have had significant effects on local populations e.g., significant pup mortality in a single ghost net at the Orkney study site of the HSD project.

In order to assess the extent and importance at a population scale we would require a large-scale monitoring programme. Allen *et al.* 2012 showed that valuable information can be collected by regular observation at specific haulout sites. Coordinating reports and images from volunteer observers and expanding such programmes through volunteer networks such as the UK seal alliance could potentially provide useful information. A structured and consistent recording methodology would need to be developed. Drone surveys of haul outs could provide an effective way to monitor entanglement rates.

Retrospective analysis of aerial survey images may provide some additional information. However, images collected to date have been for a specific purpose, i.e., to count and identify seals to species level and to identify harbour seal pups. Thus, most images will not be of sufficient resolution to reliably identify the less obvious examples of entanglement. Improvements in camera and lens technology means that it is now feasible to collect suitable images at some sites during routine survey flights. Calibration of the detection rates from aerial surveys for different types of entanglement would be required.

Persistent organic pollutants (POPs)

POPs are endocrine disruptors that can alter adipose tissue development, regulation and function, in addition to their well-established effects on reproductive, immune and thyroid function. Top marine predators are particularly vulnerable because they possess large fat stores that accumulate POPs. Recent results on the concentrations of organochlorine pollutants in grey seal pups from the Isle of May (SCOS BP-17/06) suggested a modest but significant decrease in polychlorinated biphenyls (PCBs) occurred between 2002 and 2015, whereas levels of the organochlorine pesticide - DDT and its metabolites (DDX) increased over the same period. In both cases, the concentrations measured were below the limits that cause immediate negative health effects in seals. Bennett *et al.* (2021) examined the impact of alterations to blubber metabolic characteristics and circulating thyroid hormone (TH) levels associated with PCBs, polybrominated diphenyl ethers (PBDEs), and DDX on suckling mass gain and weaning mass in wild grey seal pups on the Isle of May. PCBs and PBDEs appear to act antagonistically, with PCBs reducing blubber glucose uptake while PBDEs were associated with mass gain during suckling. POP impacts on whole-animal energy balance in grey seal pups appear to partially offset each other through opposing effects on different mechanisms. POP effects were generally minor, but the largest POP-induced reductions in weaning mass occurred in small pups. Since weaning mass is positively related to first-year survival, POPs may disproportionately affect smaller individuals, and could have population-level impacts even when levels are relatively low compared to historical values. The predictive power of the models in this study was low, so that although findings from these studies could inform risk assessments to estimate low level POP effects on populations, more information is

needed on how different POP classes alter fat accumulation. Blubber and liver expression of genes involved in energy balance are altered by POPs in other seal species (Brown *et al.*, 2014; 2017), but the whole animal consequences of this type of metabolic disruption, particularly for young animals, are not well understood.

Although the conditions, e.g., salinity, pollutant burdens and seal species may be different, it may be informative to examine trends in effects of POPs on seals in other regions. Sonne *et al.* (2020) recently reviewed the available information on contaminant exposure and health effects on a range of marine mammal and bird species in the Baltic during the period of general reductions in POP exposure.

Roos *et al.* (2012) showed that pregnancy rates in Baltic grey seals increased over the period 1990–2010, while the prevalence of uterine occlusions and stenosis and uterine tumours decreased. This is an ongoing tendency supported by findings that the reproductive rate of grey seals is normal at present and that birth rate in Finnish waters is 88% (mean for 2013–2018, no uterine occlusions observed) (Kauhala *et al.*, 2017). This implies that reduction in POPs has led to a decrease in negative effects, and further implies that levels of POPs in UK waters may not pose a direct threat of reduced fecundity.

The prevalence of skull lesions and skull asymmetry in Danish harbour seals increased between 1981 and 2014 (Pertoldi *et al.*, 2018). The authors hypothesise that increases could be linked to immune suppression from cumulative stress of multiple factors such as increasing PFAS (per and polyfluoroalkyl substances) concentrations and decreases in the quality and quantity of food resources (Sonne, 2010).

Colonic ulceration has continued to increase in the Baltic, but is not an issue in the North Sea or UK waters where only one case in a grey seal has been documented (Baker, 1980), and none were found in later studies (Bäcklin *et al.*, 2013, ÓNeill & Whelan, 2002).

Triosi *et al.* (2020) examined the relationships between PCB burdens and a range of sex hormones (progesterone; P4, 17 α -hydroxy progesterone; 17 α -OH-P4, testosterone; T4, 17 β -oestradiol; E2, estrone; E3) in plasma samples from grey and ringed seals in the Baltic and at Sable Island and Svalbard. PCB concentrations were significantly higher in Baltic seals than other sampling locations and mean hormone concentrations in Baltic seals were lower than Svalbard and Sable Island seals. Regression analysis indicated significant correlations between levels of PCBs and several sex hormones. As the authors state, correlations are not necessarily evidence of cause and effect, but the fact that these reductions were detected at PCB concentrations found throughout the species ranges warrants further investigation and monitoring.

Plasticizers

A joint project involving Abertay University and SMRU is investigating the effects of a group of plasticisers; the phthalates (in the form of benzyl butyl phthalate or BBP) on the insulin signalling pathway, an important regulator of fat metabolism in seals that inhibits lipid release from storage (Bennett *et al.*, 2015), and expression of key fat metabolism genes in blubber using a novel *in vitro* approach (Bennett *et al.*, 2017; Robinson *et al.*, 2018). The project is currently using a novel *in vitro* approach to test whether activation of one of the key enzymes in insulin signalling, known as Akt, is affected by BBP exposure. Changes to Akt levels or its activation in the presence of insulin will imply disruption of insulin signalling. Differences in fat metabolism gene expression between BBP treated and control blubber explants will indicate disrupted fat tissue function

Pharmaceuticals in the marine environment

Pharmaceuticals represent a major group of emerging pollutants found in freshwater and coastal waters. The occurrence of pharmaceutical substances such as contraceptives, antidepressants

(Sehonova *et al.*, 2018) and potential endocrine disruptors such as metformin (Tao *et al.*, 2018) in the marine environment is of global concern and the scale of the problem and extent of their risks and impacts on human health and biota is largely unknown (Branchet *et al.*, 2021; UNESCO, 2017). So far, this topic is under reported and we are unaware of any relevant publications on the direct effects of pharmaceuticals on seals.

Anti-microbial resistance (AMR) in seals

AMR does not pose any significant, direct threat to individual seals in the wild and cannot therefore pose any population level threat. However, the potential role of seals as a reservoir of AMR organisms may be important in the future spread of AMR through the environment. The spread of AMR poses an existential threat to human health, and possible direct transmission of AMR organisms from seals to humans during seal rescue and rehabilitation is a potential risk.

It has long been argued that the widespread and intensive use of antibiotics in human medicine, veterinary medicine, and agriculture means that sewage (both treated and untreated), hospital waste and agricultural run-off can cause the spread of AMR to marine ecosystems (e.g., Young, 1993). AMR bacteria can be ingested with prey and the possibility of gene transfer between bacteria in the seal gut may allow AMR genes to move between harmless and disease-causing bacteria.

An ongoing PhD project at Abertay university is combining AMR information from faecal samples with tracking data from 120 seals tagged at sites around the UK by SMRU. Preliminary results show that approximately 30% of the samples exhibited presence of resistance to tetracycline, a commonly used prophylactic antibiotic in aquaculture.

Two recently published studies have documented AMR in harbour seals and harbour porpoises.

Vale *et al.* (2021) reported AMR in *E. coli* from faecal swabs taken from 25 rescued seals (23 harbour and 2 grey seals) in Ireland. All *E. coli* isolates investigated in this study ($n = 39$) were ampicillin resistant while 26 (66.6%) were multi-drug resistant (MDR).

Norman *et al.* (2021) recorded antibiotic-resistant bacteria in dead stranded harbour seals in the Salish Sea, British Columbia. Of the 67 harbour seals sampled successfully, 37% showed resistance to one of the 15 antibiotics tested, and 24% showed multi-drug resistance. Porpoises were significantly more likely to carry resistant organisms compared to seals. Multiple antibiotic resistance (MAR) indices suggested that the AMR results from exposure to anthropogenic pollution.

AMR may be an important issue for seals in rehabilitation/rescue centres. Stoddard *et al.* (2009) showed that duration of residence in a rehab facility increased the level of AMR in rehabilitated northern elephant seals (*Mirounga angustirostris*), even for animals that had not been treated directly with antibiotics. Interestingly they also showed that 34% of the intake from the wild carried AMR bacteria. Tight control of antibiotic use in captive animal/rehab facilities is essential to minimise the spread of AMR to wild populations.

Harmful Algal Blooms

Toxin exposure from harmful algal blooms (HABs) has resulted in widespread morbidity and mortality in marine life, including top marine predators. Kershaw *et al.* (2021) reported concentrations of domoic acid (DA) and saxitoxin (including Paralytic Shellfish Toxin (PST) analogues), in the viscera of 40 different fish species caught in Scotland between February and November 2012 to 2019. DA concentrations peaked in the summer/autumn months and were highest in pelagic species including Atlantic mackerel and herring, key forage fish for marine predators including seals, cetaceans, and seabirds. The highest DA concentrations were measured along the east coast of Scotland and in Orkney. PSTs showed highest

concentrations in early summer, consistent with phytoplankton bloom timings. The detection of multiple toxins in such a range of demersal, pelagic and benthic fish prey species suggests that both the fish, and by extension, piscivorous marine predators, experience multiple routes of toxin exposure. Risk assessment models to understand the impacts of exposure to HAB toxins on marine predators therefore need to consider how chronic, low-dose exposure to multiple toxins, as well as acute exposure during a bloom, could lead to potential long-term health effects ultimately contributing to mortalities.

The potential synergistic, neurotoxic, and physiological effects of long-term exposure to multiple toxins require investigation in order to appropriately assess the risks of HAB toxins to fish as well as their predators. Studies of presence and levels of harmful algae in fish from coastal waters in the east of Scotland are ongoing as part of the Marine Scotland MMSS programme.

<p>37. Can SCOS review and collate the latest scientific information available on current environmental impacts seals face with a best assessment of the relative levels of risk posed by each impact?</p>	<p>Defra Q13</p>
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There are multiple potential threats to seal populations in the UK, although there is no evidence that grey seal populations are currently at significant risk from any threats at current levels of exposure. Several regional harbour seal populations are in decline, however, and are likely to be more vulnerable to pressures. Many of the specific threats are detailed elsewhere in this advice but an overview is provided here.

The principal environmental impacts with the potential to affect UK seal populations are considered below, with reference to other parts of this Advice for more detailed information on specific threats. These include: Competition between seal species; Direct predation; Fisheries interactions (direct mortality and impact on prey resource); Climate change (direct and indirect effects); Infectious diseases; Harmful Algal Blooms; Marine Pollution (entanglement, plastic ingestion, persistent organic pollution); Underwater Noise; Physical disturbance; Interaction with marine renewable energy industry (Collision with tidal turbine blades).

Some of these threats are local and some global, and the scales of the potential impacts and necessary interventions are also at different scales ranging from local to national and international. It is not clear what priority

SCOS do not consider that ranking these threats is within the scope of the meeting and will require a more extensive analysis taking into account the policy drivers that determine the priorities, e.g., the importance of specific threats in terms of national and or local/regional conservation goals, natural versus anthropogenic threats and likelihood versus severity of threats.

For most potential impacts, there is some information on the nature and extent of individual level effects, but studies to understand the potential for population level effects are generally lacking which makes ranking of relative risk difficult.

The marine environment is subject to a number of pressures and many of these have the potential to impact the individual and population health of seals. A comprehensive review of all the potential impacts facing seal populations and a robust ranking of relative risk levels is a significant undertaking that has not been carried out in the time available. Not least because there is a lack of definitive quantitative information on the extent and nature of most of these threats, and a lack of understanding of vulnerability of the various seal populations to these threats. Any assessment of relative risk ideally would involve a detailed analysis of the extent of such threats (exposure potential) combined with an

understanding of the sensitivity of each species to each threat. SCOS/SMRU and Defra will discuss the requirement for a qualitative ranking of potential threats considering both the significance of possible population scale effects and the potential for management interventions to mitigate those risks. SMRU will report back to SCOS 2023.

A detailed attempt to robustly quantify these effects and rank them definitively in order of level of concern would require significant additional resource but an overview of potential current threats is provided. An assessment of relative levels of risk is critically hampered by a lack of data and knowledge on the extent and nature of most of these threats but a qualitative ranking might be possible taking account of both the level of threat/impact and the capacity to address the issues. The most prevalent threats are considered in more detail in other answers (9,22,28-33,35,36) and these are referred to below where relevant.

SCOS are also aware of an ongoing global project, the MegaMove initiative (www.megamove.org), which is currently inviting contributions from the marine megafauna research community to develop an index to evaluate the global risk of anthropogenic threats to marine megafauna. This uses the IUCN threat scoring system, which involves applying scores relating to the timing, scope of severity of particular threats. The list of threats and sub-threats being assessed in this process include many of the threats considered here. It is possible that a similar approach could be applied at a UK level, but it is possible that a lack of knowledge would also hamper such an approach.

As highlighted in the answer to Defra Q5, there is no evidence that the UK grey seal population is currently at risk of significant decline as a result of current levels of exposure to any pressures, although in some areas impacts have the potential to have localised effects. Harbour seal populations in the northern Isles and along the east coast of Scotland and in the southeast of England have declined or are continuing to decline, and therefore clearly already being impacted, although the specific drivers for these declines are unclear. Discussion around these declines and likely drivers can be found in the answers to Marine Scotland Q10 and Defra Q1.

Strandings data can be informative in understanding the relative risks posed by various threats. For example, strandings data from CSIP and SMASS provided a key source of evidence for assessing levels of vulnerability to porpoise and dolphin species to various threats during the development of the UK Dolphin and Porpoise Conservation Strategy currently under consultation. Only SMASS routinely carry out investigations into cause of death of seals and a detailed examination of results from post-mortems in recent years would help inform an assessment of current threats in Scotland. Similarly, should seal post-mortems be carried out by CSIP in future, this would provide valuable information on the incidence of various causes of death in England and Wales (see Defra Q10). In 2019, only 9.1% of all SMASS seal post-mortems were directly attributable to anthropogenic impacts (SMASS, 2019). Although indirect impacts or mortality due to cumulative effects, for example due to prey depletion or disturbance, are more difficult to ascertain. The highest proportion of deaths in 2019 were reported as being due to a variety of causes including starvation/hypothermia, maternal separation/starvation, live stranding, (possible) grey seal attack, bottlenose dolphin attack, and metabolic disease (SMASS, 2019).

The principal environmental impacts with the potential to affect UK seal populations are considered further in turn, below. For most potential impacts, there is some information on the nature and extent of individual level effects, but studies to understand the potential for population level effects are generally lacking which makes ranking of relative risk difficult. These have not been ranked and are not in any particular order.

Competition between seal species

Competition for prey between grey and harbour seals has been suggested as a potential driver of observed harbour seal declines. This is currently under investigation as a driver for the Scottish regional harbour seal declines. Analysis of body composition and nutritional status of adult harbour seals in regions of decline shows no evidence of nutritional stress. However, it is likely that these live caught animals represent a biased sample of survivors, which are less likely to show signs of nutritional stress. Competition with grey seals is also a putative driver for the harbour seal decline in the southeast of England, although further research is needed to investigate this.

Direct predation

Killer whales predate on seals in parts of the UK (Deecke *et al.*, 2011). The rates of killer whale predation on seals may be locally important in some areas, e.g., the Shetland Isles. Research on the interactions between killer whales and their seal prey in the UK is currently underway.¹⁶

There is considerable evidence for grey seal predation on harbour seals in several areas around the UK (Brownlow *et al.*, 2016) and increasing numbers of cases have been reported to SMASS each year with a total of eighty-nine seals with trauma consistent with spiral or corkscrew injuries recorded in 2019. This makes grey seal predation the most commonly identifiable reason for harbour seal mortality in the strandings records in Scotland. Research on this is ongoing.

Fisheries interactions – direct mortality

Globally, fisheries interactions are recognised as the biggest threat to seal populations (Kovacs *et al.*, 2012). The levels of seal bycatch in fishing gear are reported in answer 22 above. In the UK the largest reported bycatch rate occurs in the southwest region, with the levels of recorded grey seal bycatch likely underestimating the scale of the problem due to the presence of several additional unmonitored fisheries. This is of particular concern due to the fact that reported levels of bycatch already exceed the calculated PBR for the regional grey seal population. The regional population is not thought to be in decline, therefore there must be immigration from elsewhere and this is currently under investigation. Mitigation efforts for reducing seal bycatch have had little attention, but programmes involving stakeholder participation are being developed, e.g., the Clean Catch UK initiative (<https://www.cleancatchuk.com/>).

Prior to recent legislative changes, licenced (and possibly unlicensed) shooting of seals interacting with aquaculture and river fisheries was a commonly reported cause of death for Scottish seals. How much unlicensed direct mortality occurs now or may occur in the future as a result of increased interactions remains unknown.

Fisheries interactions - Change in prey availability due to fishing pressure

There is considerable overlap in seal diet composition and fish species targeted in commercial fisheries so there is the potential for fishing induced changes in prey availability to impact on seal populations, although most research effort in this area has focussed on the impacts of seal predation on commercial fish catches. This issue is discussed further in the response to Marine Scotland Q7.

Climate change (direct)

As discussed in the answer to Defra Q12, projected changes in the physical environment in the UK are not expected to exceed the homeostatic ranges for UK seal species. Changes in sea level may reduce

¹⁶ <https://ecopreds.com/>

breeding and haulout sites in some areas and lead to increased wave action on breeding sites which can increase pup mortality, but such changes will be gradual, and seals should be able to accommodate by moving breeding sites if alternative sites are available.

Climate change (indirect)

Changes in prey availability as a result of climate change could significantly affect seal populations. There is some evidence that warming is responsible for disrupting the food web and altering distributions of prey species and affecting recruitment in the North Sea (Engelhard *et al.*, 2011 & 2014; Skinner 2009; Regnier *et al.*, 2019). Climate induced changes in prey availability is thought to be a driver for observed seabird declines (e.g., Mitchell *et al.*, 2020) but there is very limited evidence for effects on seals to date as seal distributional changes appear to be in the opposite direction to observed prey shifts (see Defra Q12). However, prey climate induced changes in prey availability cannot be ruled out as a potential driver of regional harbour seal declines. Increases in harmful algal blooms and increases in infectious diseases and the emergence of new diseases are also potential indirect effects of climate change that could significantly affect UK seal populations in future.

Infectious disease

Major epizootics of Phocine Distemper Virus (PDV) have occurred in 1988 and 2002, significantly affecting the North Sea harbour seal population. Infrequent cross-species transmission and waning immunity are believed to contribute to periodic outbreaks (Puryear *et al.*, 2021). The first documented PDV outbreak in 1988 in Europe was strongly correlated to an unusual harp seal invasion from the Arctic into the North Sea, and harp seals are thought to be a likely reservoir of the virus (Puryear *et al.*, 2021). Closely related PDV strains are thought to be circulating in multiple seal species along the coastlines of North America and Greenland and therefore further outbreaks are considered likely (Daoust *et al.*, 2020). Due to likely very low levels of immunity PDV re-introduction in European harbour seal populations are likely to cause a major epizootic with high infection rates and mortality. A further PDV outbreak at a time when harbour seal populations are already in decline may be catastrophic.

A major outbreak of H10N7 avian influenza in 2014 killed 500 harbour seals in western Sweden and eastern Denmark (Krog *et al.*, 2015; Zohari *et al.*, 2014), and 1,500–2,000 seals in western Denmark and in Germany and Dutch waters (Bodewes *et al.*, 2015). More recently Venkatesh *et al.* (2020) reported the discovery of H3N8 influenza A virus in a rescued grey seal pup. The IAV had a particular mutation indicative of mammalian adaptation of an avian virus. There is clearly an ongoing risk of further outbreaks of avian flu in UK seal populations.

Phocid herpesvirus 1 (PhHV-1) is known to infect grey seals *Halichoerus grypus*, Baily *et al.* (2019) found PhHV-1 in approximately 60% of 119 live grey seal pups and 56% of dead pups at the Isle of May. PhHV-1 was associated with hepatic necrosis, thymic atrophy and buccal ulceration in the dead pups. The high prevalence of PhHV-1 in grey seal pups and juveniles and the increased mixing of grey and harbour seal populations, particularly in the southern North Sea is a cause for concern for the depleted harbour seal population.

Toxins from Harmful Algal Blooms (HABs)

Toxin exposure from harmful algal blooms (HABs) has resulted in widespread morbidity and mortality in marine life, including top marine predators. This threat is discussed in more detail in the answer to Defra Q11. High concentrations of domoic acid (DA) and saxitoxin (including Paralytic Shellfish Toxin (PST) analogues) have been reported from 40 different fish species caught in Scotland, including key forage fish for seals (Kershaw *et al.*, 2021). The detection of multiple toxins in such a range of demersal, pelagic and benthic fish prey species suggests that both the fish, and by extension, piscivorous marine

predators, experience multiple routes of toxin exposure. The potential effects of long-term exposure to multiple toxins require investigation in order to appropriately assess the risks of HAB toxins to seal populations.

Marine Pollution – entanglement

Both species of seals have been recorded with evidence of entanglement in marine debris, including fishing nets and plastic hoops. This type of entanglement is common, and animals can remain entangled for many years before succumbing to the physical effects of the constriction or secondary infection. More details on entanglement in marine debris can be found in the response to Defra Q11. Entanglement of seals in marine and plastic debris, particularly discarded fishing gear may increase the risk of drowning, but, perhaps more commonly, may restrict feeding or cause deep blubber and skin cuts and abrasions (particularly around the head and neck) and lead to secondary infections. There is the possibility that strandings of seals killed by entanglement will be under-represented as seals killed more than a few kilometres offshore are unlikely to strand and entangled seals may be more likely to sink. More work is required to assess the extent and importance at a population scale.

Marine Pollution – plastic ingestion

Microplastic ingestion is unlikely to cause immediate or direct issues for animal health but may lead to sub-lethal effects. Greater understanding of what happens to ingested microplastics is needed. The ingestion of larger plastic debris, the macroplastics, may cause blockage in the gastrointestinal tract and injury to the gut mucosa. As discussed in the response to Defra Q11, it appears that plastic ingestion varies widely between marine mammal taxa, but in general seals appear less prone to plastic ingestion than cetaceans. Ingestion of macroplastics, and subsequent impacts on health, seem to be less common in seals than other marine mammals but more data from post-mortems would be beneficial.

Marine Pollution – Persistent organic pollution (POPs)

The evidence describing the potential effects of POPs such as polychlorinated biphenyls (PCBs) and the organochlorine pesticide DDT and metabolites (DDX) is summarised in the response to Defra Q11. There is clear evidence of individual level effects of exposure to these pollutants at concentrations encountered in the environment. Understanding the potential for population level effects will need additional work. Data from individual studies can be used in risk assessment frameworks to predict potential effects on populations.

Physical disturbance

See answer 33 above for more detail on this issue. There are concerns about the effects of human activity causing disturbance reactions by hauled out seals and the impacts this may have on the welfare and health of individuals that may be experiencing repeated disturbance. Whilst this is a concern in a number of locations and can clearly affect individual animal welfare, there is no evidence that this is currently a concern at the population level.

Underwater Noise

Underwater noise from a variety of sources is known to affect the local distribution and behaviour of seals. Noise from pile driving during construction of offshore wind farms results in localised avoidance and behaviour change (Russell *et al.*, 2016; Whyte *et al.*, 2020a). Levels of predicted exposure also has the potential to cause changes in auditory sensitivity (Hastie *et al.*, 2019; Whyte *et al.*, 2020a). Exposure to noise from vessels is also potentially a concern, with a small number of studies documenting exposure to noise from vessels and behavioural responses to vessel noise (Jones *et al.*, 2017; Trigg *et*

al., 2020; Mikkelsen *et al.*, 2019). Underwater explosions (e.g., from clearance of unexploded ordnance) and seismic activity have the potential to impact seals, but no data is available on this.

The operation of tidal turbines is likely to be audible to seals. Risch *et al.* (2020) recently estimated that the Atlantis AR1500 tidal turbine at the MeyGen array was likely audible to ~2km. There is also some evidence of local avoidance of turbine noise and of the MeyGen array (Hastie *et al.*, 2018; Onoufriou *et al.*, 2021). This degree of avoidance is not of concern at the current scale of tidal energy development but could increase as developments scale up to large commercial arrays.

Even if single sources of underwater noise do not result in any significant population level concerns, when multiple activities occur at the same time and over an extended period of time, and extended areas, the impact is likely to be greater; the ranges at which behavioural or physiological responses to noise occur, and the ranges at which significant masking of seal calls, predator calls and acoustic foraging cues occur will increase as source levels and numbers of sources increase.

Evidence on the cumulative population level impacts of noise is lacking. Population Consequences of Disturbance (PCoD) models have been developed to combine available information on population processes and both behavioural and physiological responses to noise, to address these uncertainties, identify important knowledge gaps and derive estimates of population consequences using best available information. Pirotta *et al.* (2018) provide a helpful overview of the process and Dunlop *et al.* (2021) describe the application of a PCoD model to investigate the effects of seismic survey noise on developed for humpback whales (*Megaptera novaeangliae*).

Other sources of direct mortality

There are concerns about the potential for collisions with marine renewable energy devices, although currently there are very few devices installed around the UK and limited potential for interactions. However, this could increase as the tidal energy industry scales up to large arrays, and there is a potential for impacts to be locally significant. There is some evidence of avoidance (Sparling *et al.*, 2017; Joy *et al.*, 2018; Hastie *et al.*, 2018; Onoufriou *et al.*, 2021) which may reduce the risk of collisions but detailed information on the fine scale behaviour of seals around tidal turbines is currently lacking. Although likely at a small scale, there are reports of seals becoming trapped in underwater structures (see Q 29 above). Underwater explosions related to military activity such as the bombing at coastal and offshore sites and the destruction of unexploded ordnance during clearance of offshore marine renewable sites could also cause mortality.

<p>38. at is the evidence that seals can contract COVID 19 (e.g., likely routes of transmission for wild and captive animals as well as physiological and immunological susceptibility), and can act as symptomatic or asymptomatic carriers and thus wildlife reservoirs of this disease?</p>	<p>Submitted by K. Bennett, Abertay University</p>
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UK seals are likely susceptible to Covid-19. No testing of any seals has been carried out to our knowledge and no significant reports of morbidity or mortality related to respiratory illness. Despite the likely animal origin, no wildlife reservoir for the SARS-CoV-2 virus has been found and the role of wild mammals in natural transmission and reservoir capacity is speculative.

Covid-19 can be transmitted to the marine environment via untreated sewage, but this is unlikely to be occurring to any significant extent in the UK due to widespread secondary sewage treatment.

Other potential routes for transmission include when humans are handling seals. The two settings where this will occur is in scientific research and in rescue/rehab. Precautions should be taken in these settings to limit the risk of transmission. Samples could be taken from seals in these settings and analysed to provide further data on the prevalence of the virus in the wild seal population.

UK seals are thought to be susceptible to Covid-19 (Mathavarajah *et al.*, 2021) given ACE2 gene conservation in marine mammals. ACE2 is the host receptor targeted by the virus SARS-CoV-2 (the causative agent of Covid-19), and variability in the receptor contributes to why some species are susceptible and not others (Mathavaraja & Dellaire, 2020). An examination of ACE2 genetic variability in a range of marine mammal species indicated that harbour seals are predicted to be highly susceptible. Aligning regions for the genome sequence for grey seals were not available, so grey seal susceptibility was not explicitly predicted in this study. However, it was noted that many of their mutations resemble that of the other seal species that were predicted to be highly susceptible (Mathavarajah *et al.*, 2021). This suggests that UK seal species may contract the virus if encountered in their environment. SCOS are not aware of any UK seals having been tested for SARS-CoV-2. The USGS Wildlife Health Centre has a programme of testing a range of pinniped species for SARS-CoV-2 (USGS, 2021) but at time of writing there no results were available. There have been no recent reports of morbidity of seals in captivity or in the wild in relation to any respiratory illnesses.

SARS-CoV-2 has been detected animals exposed to infected humans or challenged experimentally. These include domesticated cats, dogs, and ferrets, and captive-managed mink, lions, and tigers (Mahdy *et al.*, 2020, O'Connor *et al.*, 2020, Shi *et al.*, 2021). In addition, there is clear evidence that SARS-CoV-2 is widespread in wild deer populations in the USA and that several transmission events have occurred (Chandler *et al.*, 2021). These studies do suggest that there is a possibility of the involvement of multiple species in SARS-CoV-2 circulation and persistence, but few studies have been completed thus far and no confirmed cases of natural transmission from animals to humans have been confirmed. The role of wild mammals in general in Covid-19 transmission and reservoir capacity is speculative. Comparative genomic analysis has suggested that SARS-CoV-2 evolved naturally with bats as the likely origin, being closely related to two SARS-like CoV sequences that were isolated in bats during 2015-2017 (Zhang *et al.*, 2020) with the human SARS-CoV-2 sharing a recent common ancestor. So far, to our knowledge, no natural animal reservoir for SARS-CoV-2 has been identified (Haider *et al.*, 2020), although pangolins, mink and ferrets have all been suggested as the most likely intermediate hosts for SARS-CoV-2 (Fenollar *et al.*, 2021, Royce, 2021).

It has been proposed that the virus can be transmitted to the marine environment via sewage effluent, and this could provide a pathway for transmissions to seals (Mathavarajah *et al.*, 2021). Although the RNA of SARS-CoV-2 has been detected in untreated sewage in the UK¹⁷, most UK sewage treatment involves secondary treatment which significantly reduces the possibility of virus exposure via treated effluent (Peccia *et al.*, 2020). However, there are a very small number of areas in the UK where only primary treatment occurs (including Kirkwall, Lerwick and Stornoway) and any problematic sewage overflow could lead to exposure in the marine environment for vulnerable species.

The other potential routes of transmission between humans and seals include situations where humans handle seals, including rehabilitation and research. SMRU is the only research group in the UK with a Home Office licence to capture and handle wild seals and there have been very limited fieldwork activities involving seal handling since the beginning of the pandemic. The only seal handling that has been carried out since the beginning of the pandemic was a recent trip to tag grey seal weaned pups at the Monachs in the Outer Hebrides at the end of October 2021. All SMRU personnel underwent regular testing before the field trip to ensure they were negative for the virus and no members of the team

¹⁷ <https://informatics.sepa.org.uk/RNAmonitoring/>

were knowingly exposed to coronavirus during the trip. Samples were taken from the tagged seals (n=50) for other purposes that could allow screening for the presence of SARS-CoV-2. Similarly, samples could be taken in future seal catching trips planned in 2022.

Several animal welfare organisations routinely bring seals into rehabilitation centres, particularly during and following the breeding season when seal pups are found by members of the public. The historical risk of influenza transmission means that such sites should already have protocols in place to prevent transmission of respiratory viruses. It may be prudent to ensure that protocols are in place to reduce the risk of transmission and to swab any handled animals for subsequent testing to provide further data on the prevalence of the virus in the wild seal population.

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ANNEX I Terms of Reference

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 C.E. Sparling	Sea Mammal Research Unit, University of St Andrews.
Dr J. Wilson	Marine Scotland, Science, Aberdeen.
Dr M. Biuw	Institute of Marine Research in Norway. Tromso.
Dr G. Engelhard	Centre for Environment Fisheries and Aquaculture Science, Lowestoft
Professor B. Wilson	Scottish Association for Marine Science, Dunstaffnage, Oban
Dr J. London	National Marine Mammal Lab. Seattle.
Dr K. Bennett	Abertay University, Dundee.
Dr O. Ó Cadhla	National Parks and Wildlife Service, Ireland.
Dr K. Frior (Secretary)	UKRI Natural Environment Research Council, Swindon.

ANNEX II Questions to SCOS.

Questions from Marine Scotland

Organisation: *Scottish Government*

Scottish Government Questions – Special Committee on Seals – 2021

Question No.	Question	Driver/rational behind question(1-2 sentences)
	Advice on the status of seal populations in Scotland	
1	What are the latest estimates of the number of grey and harbour seals in Scottish waters?	General update on the estimated numbers of grey seals and harbour seals in Scottish waters.
2	What is the latest understanding about the population structure, including survival, reproduction and age structure, of grey and harbour seals in European and Scottish waters?	Information about the structure or make up of these populations that might assist management measures.
3	What are the latest SAC relevant count/pup production estimates for the harbour and grey seal SACs, together with an assessment of trends within the SAC relative to trends in the wider seal management unit/pup production area?	To provide current SAC specific estimates/trends for consideration in HRA assessments.
4	The frequency of grey seal surveys in some areas of Scotland are likely to be reduced in future years. Can SCOS advise on what a reduction in survey effort would mean in terms of the confidence of population estimates?	Information on what a reduction in grey seal surveys will mean for population estimates.
5	Are there any technologies (existing or new/emerging) that could be considered as an alternative to aerial surveys that could help meet Net Zero aspirations, or does the method currently used remain the most appropriate vehicle?	Considering whether lower impact vehicles could be used to survey seal populations.
6	In 2019, SCOS advised that scientifically informed criteria were required to establish whether seal conservation areas should be introduced or revoked. Can SCOS advise on what such criteria should consist?	Scientific information to review the current protection measures for harbour seals (seal conservation areas).

	In the absence of such criteria, but noting current population trends, can SCOS advise whether the threat to seal populations still remains in current seal conservation areas, particularly the Western Isles.	
7	Are there any parts of the wider ecosystem that are likely to experience significant impacts as a result of an increasing Scottish seal population? What are these impacts and would they be positive or negative?	To inform consideration of the potential impacts of grey and harbour seals on the wider ecosystem, and marine industries including aquaculture.
8	Based on distribution and demographics of seal populations, can SCOS advise whether it would be possible predict times and locations where there may be a greater chance of interactions with the aquaculture industry? Please can SCOS advise what work would be required to achieve this.	Seal populations in Scotland are increasing, resulting in greater interactions with marine users. It is possible predict where the greatest issues (interactions) may occur?
	Harbour seal decline	
9	Please could SCOS provide an update on the regional harbour seal declines, including current and projected trends.	Information on the latest trends in local harbour seal populations around Scotland to inform management measures.
10	In the 2020 advice, SCOS provide a view on the current potential (major) drivers of the harbour seal decline and their status. Can SCOS provide an updated assessment in light of ongoing work? Furthermore, could SCOS provide a view on whether the observed declines occurring in the south east of England could assist with providing answers to the situation in Scotland?	Seeking clarity on the potential drivers that require further effort, in order to consider the need for any conservation and management measures
11	Can SCOS review, present and provide a view on the available evidence on the differences in genetics between the declining and the stable/increasing harbour seal populations.	As above.
	Potential Biological Removal	
12	Please provide updated Potential Biological Removals (PBRs) figures for 2021?	This seeks an update on the PBR figures to inform licensing decisions.
	Marine Renewable Energy	
13	Scottish Government are aware of (recent) incidents involving seals becoming trapped and drowning in structures associated with fixed offshore wind developments. Are SCOS aware of such events, and if so,	To determine the occurrence and potential cause of seal mortality in tubes at offshore renewable installations and the potential for other marine installations (e.g., oil and gas and renewable energy structures) to pose a similar risk.

	<p>what structures were the cause, and can SCOS provide any information on the prevalence of these events?</p> <p>Furthermore, based on what we know from these events, what other marine structures could pose a similar risk to seals? Can any lessons be learned from other offshore industries or other regions outside of the UK with respect to mitigating and monitoring such events?</p>	
14	<p>There are known knowledge gaps associated with seals with respect to potential impacts in relation to underwater noise and collision risk with tidal turbines, for example. With these and other knowledge gaps in mind, can SCOS provide an update on emerging technologies they are aware of that could be used for quantifying seal behaviour and/or physiology (e.g., developments in animal borne sensors such as fNIRS).</p>	<p>Whether there are any other emerging technologies that could be used to quantify seal behaviour. Information should include a summary of the aims and capabilities of the technology, Technology Readiness Levels and timelines for commercialisation if not already on the market, estimated costs and constraints, and logistical considerations.</p>
	Seal – fisheries interactions	
15	<p>SCOS provided advice in 2020 on non-lethal options to address seal – fisheries / fish farm interactions. Since the 2020 advice (and in light of ongoing efforts globally to address such interactions), are SCOS aware of any further developments in other countries or emerging technologies that could be consider/applied to Scotland.</p>	<p>Seeking an update on whether there are any emerging seal control technologies since the 2020 advice that we can advise industry to consider using for seal control in the absence of lethal measures.</p>
16	<p>What are the latest estimates of seal (grey and harbours) bycatch across fisheries in Scotland and the wider UK? Are there particular seasonal and / or geographical hot spots of high seal bycatch? Are there any areas where it has not been possible to collect seal population/bycatch data, but where there is a potential risk?</p>	<p>To understand the potential for impact of bycatch on seal populations and inform any future requirement for monitoring.</p>
	Seal haul out sites	
17	<p>SCOS previously advised a five year cycle for reviewing the list of designated haul out sites. Does SCOS consider that this is the most appropriate time frame for reviewing seal haul sites based on the survey data and rate of change in the population?</p>	<p>To assist with reviewing the seal haul out sites, we need to consider the most appropriate time frame which will be determined by the survey data.</p>
	Seal usage maps	
18	<p>Please could SCOS recommend the most appropriate at sea abundance and distribution data source for use in licensing applications and planning activities (both renewables and major infrastructure). Where such data</p>	<p>The updated usage maps only provide relative densities, but absolute densities are required for use in licensing casework.</p>

	sources provide relative density, could SCOS please advise on an appropriate method to convert to absolute density.	
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Questions supplied by Defra & compiled by Lara Turtle & Victoria Bendall (Defra), Ophelie Humphrey & Claire Ludgate (NE), Roma Banga (JNCC) and Rachel Wright (MMO).

Question No.	Seal Population & Management Questions: Required by policy and conservation advisors to be reviewed, summarised & updated annually if new information available.	Policy Driver/rational behind question:
1	<p>Seal Population Estimates: What are the latest estimates of the number of grey and common/harbour seals in English waters and what are the latest figures based upon (e.g., level of coverage, timings, and occurrences of aerial surveys)?</p> <p>Are trends in common/harbour seal abundance considered to be declining in English waters and if so, what are the potential influencing factors and where is further research needed?</p> <p>Can SCOS advise at what point a decline in grey or common/harbour seal abundance would trigger a change in Natural England's Conservation Objectives for SAC's from "maintain" to "restore"?</p>	<p>General update on information regularly provided by the Committee in previous years but relating to seals in English waters.</p> <p>Counts of the harbour seal population in The Wash from 2019-2020 have revealed a 20-30% decline. Natural England need to ensure that their advice reflects the most recent scientific evidence on the population.</p>
2	<p>Seal Population Structure: What is the latest information about the population structure, including mortality, age and sex structure, and carrying capacity of grey and common/harbour seals in English waters? Is there any new evidence of grey or common/harbour seal populations or sub-populations specific to localised/regional areas?</p>	<p>General update on information regularly provided by the Committee in previous years relating to seals in English waters. There is a need for greater understanding in occurrences of localised/regional populations, to allow for more targeted conservation and management measures.</p> <p>Defra would therefore be interested to learn more on recent population DNA research studies to highlight Conservation Management Areas (e.g., PhD research project by Kristina Steinmetz)</p>
3	<p>OSPAR Population Indicators: Can SCOS provide advice on current analytical methods being conducted by SMRU to help inform UK assessments for OSPAR M3 & M5 indicators?</p>	<p>For the UK, the assessments require monitoring grey seal pup production, estimating total grey seal population size from the</p>

		<p>pup production data and monitoring harbour seal abundance during the annual moult.</p> <p>Defra and JNCC are aware that SMRU are currently analysing these assessments which will be reviewed by OSPAR's Biodiversity Committee (BDC) in 2022 before being integrated into the OSPAR Quality Status Report (QSR2023) and subsequent MSFD Biodiversity Descriptor reporting.</p> <p>We are therefore keen to gain understanding by SCOS on current analytical methods being used to help inform future assessments.</p>
4	<p>Seal Management Units (MUs): Can SCOS review and comment on the biological management perspective of seal management units proposed by the Inter-Agency Marine Mammal Working Group (IAMMWG)?</p>	<p>JNCC are undertaking a review of cetacean and seal MUs in 2020-2021. These units are presently being reviewed internally by JNCC based on previous 2019 SCOS advice and are due to be finalised for the IAMMWG at the end of June 2021.</p> <p>As far as possible, the management units defined have been based on the presence of known populations, with divisions proposed on the basis of ecological evidence and/or divisions used for the management of human activities. Therefore, whilst being consistent with the best biological understanding, a MU refers to the animals of a particular species in a geographical area to which management of human activities is also applied. As such, these MUs comprise partially artificial divisions of biological populations.</p>
Question No.	Seal Protection & Conservation Questions: Required by policy and conservation advisors to be reviewed, summarised & updated annually if new information available.	Policy Driver/rational behind question:
5	<p>Seal SSSI Guidance: Could SCOS please advise on the locations of the top two breeding sites and top two haul out sites for both harbour seal and grey seal in each Seal Management Unit? Could SCOS also comment on whether the top two sites have been consistent over the last 5 years, or whether there is interannual variability?</p>	<p>The current guidance for notifying SSSIs for seals states that the top two breeding sites and the top two haulout sites in each 'stock' (now SMU) can be notified as a SSSI.</p> <p>Defra and Natural England are currently reviewing the possibility of notifying further SSSIs for seals, to improve seal protection and reduce disturbance at important seal sites.</p>

	<p>Does SCOS have any recommendations of other approaches to improve overall protection for populations at risk through the use of SSSI's? Can SCOS please advise on how best to determine the "vulnerability of sites" for seals?</p>	<p>We appreciate that there may be caveats to the answer, for example where SMUs populations are too small, where local organisations may hold the data, or colonies where counts have not been counted annually etc.</p>
6	<p>Population-level consequences of disturbance in seals Can SCOS advise whether repeated disturbance to seals (such as repeated flushing into the water) could potentially lead to a population-level effect?</p> <p>Can SCOS review current guidance for anthropogenic related seal disturbance and determine whether different categorised thresholds for land (public at beach haul outs), sea (by boat and water sports) and air (use of aerial drones), could be usefully calculated from NGO monitoring data and implemented to help reduce disturbance.</p> <p>Could SCOS please advise what data should be collected, at a minimum, on disturbance events? This would help to inform a standardised approach should a nationwide reporting and threshold system for recording disturbance events be developed.</p>	<p>Defra and Natural England are aware of datasets held by some regional NGO's on the frequency of disturbance events to seals.</p> <p>Similar data may be held by other organisations and could be collated and analysed. Having thresholds would help to determine whether disturbance was an issue and required further attention.</p> <p>[In response to a request in 2020, SCOS noted that there was no formal or co-ordinated nationwide reporting system for recording disturbance events. They then note that local site managers and NGOs have developed their own guidance and, in some cases, monitor disturbance events.]</p>
Question No.	Fisheries Interaction Questions: Required by policy and conservation advisors based upon latest emerging issues for seals	Policy Driver/rational behind question:
7	<p>Non-lethal seal mitigation measures in commercial fisheries: Can SCOS provide recommendations on what the latest non-lethal mitigation devices, gear modifications and measures are to minimise seal depredation in commercial fisheries?</p>	<p>Based upon recent government action to prevent the intentional or reckless killing of seals in English, Welsh and Northern Irish waters as a result of commercial fishing under Fisheries Act 2020, which became effective from 1st March 2021.</p> <p>Defra and MMO are looking to work with industry on non-lethal seal deterrents which warrant further research and development for UK fisheries during 2021 - 2022.</p> <p>Defra and MMO are proposing to extend previous studies undertaken in 2019 (MMO report on non-lethal seal</p>

		<p>deterrents) by testing Acoustic Startle Devices in other areas/with other gears.</p> <p>We therefore require SCOS to help identify any new device(s) and/or practises that we can advise industry to use for non-lethal seal control.</p>
8	<p>Seal Bycatch monitoring requirements:</p> <p>What is the latest understanding on levels of seal bycatch across the UK? Where is seal bycatch considered to predominantly occur by region and gear type and is there any data to show any bias be seal species, sex or specific age groups?</p>	<p>Understanding levels of incidental wildlife bycatch in commercial fisheries is vital for improved clean catch fisheries management measures. It is important that we understand the scale and distribution of the problem so we can look at appropriate mitigating measures, if needed, particularly in light of recent amendments under Fisheries Act 2020.</p> <p>Defra are currently working with industry, scientists and eNGOs on “Clean Catch UK: Joint Action to Reduce Wildlife Bycatch”, a forward-looking national approach to monitoring and mitigating bycatch in the UK – driven by the Fisheries Act 2020 and new National Plans of Action for reducing bycatch of sensitive species.</p> <p>As of 1st April 2021, Defra also let a new 10-year contract to the Cetacean Strandings Investigation Programme (CSIP) to annually report on threats to cetaceans through carrying out post-mortems with the aim of broadening it’s the scope to other vulnerable marine species such as grey and common/harbour seals.</p> <p>We therefore require SCOS to help identify what the current gaps in scientific knowledge are for seal bycatch and how best to collect additional information to provide valuable evidence of the current issue in commercial fisheries.</p>
9	<p>Seal Depredation in commercial fisheries:</p> <p>Can SCOS advise on the latest information available to provide evidence of seal depredation in the UK?</p> <p>Can SCOS advise on new research that could be undertaken to best to collect robust data on this important issue of concern within UK commercial fisheries?</p>	<p>We have seen increasing complaints from the fishing industry of seal depredation for large percentages of catch reported.</p> <p>There are now heightened animal welfare concerns around such interactions between fishers and seals and any intentional or reckless killing of seals by fishers, in light of recent Fisheries</p>

		Act 2020 amendments to remove the 'Netsman's Defence' and ability to apply for a licence to shoot seals.
Question No.	Emerging Issues Questions: Required by policy and conservation advisors based upon latest emerging issues for seals	Policy Driver/rational behind question:
10	<p>Post-mortems of seals: If funding became available to undertake post-mortems on a limited number of seals in England, could SCOS please advise on which strandings should be the top priority to investigate? For example, which apparent causes of death, which species, age class, location etc. Could additional post-mortems be of benefit to our understanding of wider issues e.g., on the decline in The Wash harbour seal population, for example?</p> <p>Can SCOS advice on recent observations of 'mouth rot' (e.g., swollen muzzles; open wounds and oral ulcerations that can lead to bone exposure, bone necrosis and potentially septicaemia and death), an unknown disease that appears to be affecting harbour seal pups on the east coast of England? Specifically, what data should be recorded to enable and enhance further investigations? Do SCOS consider that this disease should be taken into account during the investigation of the harbour seal decline in the Wash?</p>	<p>Defra and Natural England have received a proposal outlining the indicative costs of undertaking a limited number of post-mortems on seals. Necropsies could be a useful source of information on wider issues e.g., the decline in The Wash population of harbour seals, or physiological effects of repeated disturbance in the southwest.</p> <p>Data on cases of the 'mouth rot' disease have not been routinely collected to date and it would be beneficial to ensure the right data is collected going forward to ensure appropriate investigations can be undertaken. It would also be useful to know if this may be a contributory factor to the decline of harbour seals in the Wash.</p>
11	<p>Impacts on seals from plastic and other marine pollution: Can SCOS review and provide an update on any new studies looking into how macroplastics, microplastics, chemical pollution (including but not exclusively pharmaceutical drugs flushed into water systems), Abandoned, Lost or otherwise Discarded Fishing Gear (ALDFG) and other marine pollution are affecting seal populations? What research is specifically required to help fill data gaps and evidence base in this area? How could impacts of plastic pollution be usefully picked up in part under reporting of strandings and post-mortem work by CSIP?</p>	<p>Due to various microplastics, macroplastics, chemical and other pollutants having a significant negative effect on marine life, it is important to understand how such pollution has and is affecting seal populations.</p> <p>Defra policy requests SCOS recommendations on how to increase our understanding and improve monitoring within this area.</p>
12	<p>Impacts on Seals through climate change: Can SCOS review latest scientific information available on current environmental impacts seals face due to climate change, such as acidification, sea level changes and coastal collapses and changing prey distributions.</p>	<p>Due to climate change having a significant negative effect on marine life, it would be important to understand how climate change has and is affecting seal populations.</p>

		Defra policy requests SCOS recommendations on how to increase our understanding and improve monitoring within this area.
13	<p>Holistic review of factors impacting health and welfare of seals: Can SCOS review and collate the latest scientific information available on current environmental impacts seals face with a best assessment of the relative levels of risk posed by each impact?</p>	<p>As the marine environment is being impacted by multiple issues that also act cumulatively, it is important to be aware of the big picture context within which seals exist and are impacted. Understanding the relative importance of each impact can help drive future policy priorities.</p> <p>Defra policy requests SCOS recommendations on how to increase our understanding and improve monitoring within this area.</p>

Questions supplied by Natural Resources Wales

Question No.	Question	Driver/rational behind question(1-2 sentences)
1	What are the latest estimates and trends for grey seals in the UK?	Please provide the estimated pup production by region and the resulting population size for grey seals, including in Wales and south western British Isles, including Ireland. Also see question 3.
2	What are the latest bycatch estimates for grey seals in the UK, especially Southwestern British Isles, including Ireland?	Understanding the level of bycatch is necessary for NRW to provide up-to-date advice to marine planning authorities and developers on the likely effects of potential seal collisions and other anthropogenic removals, in relation to the PBR for grey seals in SW British Isles. NRW use a wide spatial scale to represent a biologically and management appropriate grey seal management unit and encompasses the Celtic Sea, Irish Sea and English Channel and includes Southwestern and eastern Irish waters and the sea area off North West France. Knowledge of bycatch estimates in these areas is required.
3	Could SCOS provide advice on the most appropriate multiplier to use when estimating an all age population size from pup production in the Southwestern British Isles (including Ireland) region.	<p>It is often desirable to estimate total population size of grey seals in the region (SW British Isles and adjacent Seas – inc. Ireland) from pup production estimates using a simple multiplier eg Hewer (1964) life tables, ratio of pups to adults in monitored colonies/well parameterised models etc. However, the multipliers used in the literature related to this geographical region ranges from a 2.5 to 4.5. (see Baines <i>et al</i> 1995; Cronin <i>et al</i> 2007; O’Cadhla <i>et al</i> 2013; Stringell <i>et al</i> 2014)</p> <p>Baines M.E., Earl S.J., Pierpoint C.J.L. and Poole J. (1995) The west Wales grey seal census. CCW Contract Science Report, no. 131, 238 pp.</p> <p>Cronin M.A., Duck C.D. and O’Cadhla O. (2007) Aerial surveying of grey seal breeding colonies on the Blasket Islands, Co. Kerry, the Inishkea Group, Co. Mayo and the Donegal Coast, Ireland. <i>Journal for Nature Conservation</i> 15, 73–83.</p> <p>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. <i>Irish Wildlife Manuals</i>, No. 74. National Parks and Wildlife Service, Department of the Arts, Heritage and the Gaeltacht, Dublin, Ireland.</p> <p>Stringell TB, Millar CP, Sanderson WG, Westcott SM, McMath MJ. (2014). When aerial surveys will not do: grey seal pup production in cryptic habitats of Wales. <i>Journal of the Marine Biological Association of the United Kingdom</i>. 94 (6): 1155-1159.</p>

4	What is the current state of knowledge on grey seal interactions with tidal energy devices?	Knowledge of the latest information about interactions and behaviours of grey seals around operational tidal stream installations is key to assessing consenting risk for the tidal industry in Wales. Please can SCOS highlight any new information and summarise the status of present empirical knowledge on grey seal interactions with tidal turbines.
5	Can SCOS recommend what the most appropriate avoidance rates should be in collision risk models or encounter rate models for grey seals and tidal turbines?	<p>When assessing the predicted risk of collisions with tidal turbines through encounter rate or collision risk modelling, a single avoidance rate/factor is applied, which ranges from 0 to 100%. This single factor typically incorporates near-field evasion and far-field avoidance. For marine birds, an avoidance rate of 98% is often used. Given the lack of empirical information on avoidance rates in marine mammals, existing guidance (SNH 2016) recommends a range of avoidance rates are used to generate a range of estimates. Can SCOS recommend what the most appropriate avoidance rate should be for grey seals around tidal turbines?</p> <p>Scottish Natural Heritage (2016) 'Assessing collision risk between underwater turbines and marine wildlife'. SNH guidance note. Guidance Note - Assessing collision risk between underwater turbines and marine wildlife.pdf (nature.scot)</p>

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.

List of briefing papers

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21/02	Estimating the proportion of grey seals hauled out during August surveys. Russell DJF and Carter MID	185
21/03	Trends in seal abundance and grey seal pup production Russell Debbie JF, Duck CD, Morris CD, Riddoch NG, Thompson D	195
21/04	2018 Annual review of priors for grey seal population model. Russell, D.J.F., Thompson, D. and Thomas, L.	218
21/05	Estimating the size of the UK grey seal population between 1984 and 2019. Thomas, L.	227
21/06	Recent changes in status of harbour seals in the Wash and North Norfolk SAC and adjacent sites. Dave Thompson & Debbie Russell	238
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Grey seal pup production in Britain in 2019

Chris D. Morris, Nick Riddoch and Callan D. Duck

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Abstract

In the 2019 grey seal breeding season, SMRU successfully surveyed over 65 of the 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.

Using the standard pup production model run (0.9 for proportion of moulters correctly classified, 23.0 days for mean time to fully moulted and 31.5 days for mean time to leave), pup production at the Inner Hebrides colonies was estimated to be **4,455**, slightly lower than the 2016 estimate of 4,541. Pup production at colonies in the Outer Hebrides was **16,083** (15,732 in 2016), in Orkney production was **22,153** (23,849 in 2016), in the Firth of Forth production was **7,261** (6,426 in 2016). Total pup production at all of these regularly monitored colonies in Scotland was **49,952** (50,548 in 2016).

At the four main English North Sea colonies, pup production in 2019 was **10,725** compared with 8,213 in 2016 and 6,795 in 2014. Pup production at Blakeney Point continued to increase with an estimated **3,399** pups born in 2019 compared with 2,403 born in 2016. Production at Horsey, East Norfolk has also increased with **2,316** born in 2019 compared with 1,526 born in 2016.

Combining with an estimated additional **4,592** pups born at other colonies in Scotland and England, an estimated **2,250** pups born in Wales, and an estimated **250** pups born in Northern Ireland, the total grey seal pup production for the UK in 2019 was estimated to be **67,789**.

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 80%) 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 north-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. Reductions in funding, combined with increasing aerial survey costs, have resulted in SMRU reducing monitoring the main Scottish grey seal breeding colonies from an annual to a biennial regime. The number of pups born at colonies along the east coast of England has been monitored annually through ground counting by different organisations: National Trust staff count pups born at the Farne Islands (Northumberland) and at Blakeney Point (Norfolk); staff from Lincolnshire Wildlife Trust count pups born at Donna Nook and staff from Natural England (plus volunteers) count pups born at Horsey/Winterton, on the east Norfolk coast.

Due to the increasing in size of these colonies making ground counting more difficult, they were surveyed aerially by SMRU in 2018 and again in 2021. Scottish Natural Heritage (SNH) staff ground counted grey seal pups born in Shetland.

Restrictions due to COVID-19 precluded any surveying in 2020.

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 the estimated pup production in 2019 at the main grey seal breeding colonies in the UK.

Materials and Methods

SMRU aerially surveys the main breeding colonies around Scotland. Grey seal pups born at colonies in England and Shetland 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 are estimated from a series of 3 to 6 counts derived from aerial images, using a model of the birth process and the development of pups (Russell *et al.*, 2019). The method used to obtain pup production estimates for 2019 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.

SMRU successfully surveyed all the main grey seal breeding colonies in Scotland (excl. Shetland) between September and December 2016. Four to six 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.

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. Images on 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 v1.4.4. 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. The images were scanned visually in Manifold by an expert and individual pups were marked on a separate layer and classified as whitecoat or moulted pup (Figures 5 and 6).

The pup production model allows different misclassification proportions to be incorporated. Previously, because there was a significant risk of misclassifying moulted pups as whitecoats on the large format film photographs, 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 generally less likely.

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). Since 2012, the digital images were of sufficient quality to reduce the probability of misclassification, so a proportion of 90% was used as standard for all production estimates since 2012. In line with previous years, the standard mean time to moult of 23.0 days and mean time to leave of 31.5 days were also incorporated into the pup production model.

Results & Discussion

The locations of the main grey seal breeding colonies in the UK are shown in Figure 1. In 2019, pup production at the main biennially monitored breeding colonies in Scotland was estimated to be **49,952** compared with 50,548 in 2016, an average annual decline of -0.4% (Table 1).

In 2019, pup production at the annually monitored colonies in England was estimated to be **10,725** compared with 8,213 in 2016, an average annual increase of +9.3% (Table 1). Total pup production estimates since 1960, for the four regions used in the grey seal population model, are given in Table 2 and are plotted in Figure 7.

Including **4,112** pups born at other colonies in Scotland (Table 3), an estimated **450** pups born in south-west England, an estimated **50** pups at smaller sites in east and north-west England, an estimated **2,250** pup born in Wales, and an estimated **250** pups born in Northern Ireland, the total grey seal pup production for the UK in 2019 was estimated to be **67,789** (Table 1).

The plots shown for the Scottish colonies monitored by aerial surveys indicate that there has been a general step increase in the pup production estimates since 2012 when the large format film camera was replaced by two digital cameras.

Pup production at colonies in the Inner Hebrides

In 2019, grey seal pup production at 13 colonies the Inner Hebrides was estimated to be **4,455** compared with 4,541 in 2016, an average annual decline of -0.6% (Table 1). Grouped colonies from different parts of the Inner Hebrides show slightly different production trajectories (Figure 8). Breeding colonies in the Inner Hebrides have only been surveyed since the late 1980s, so it is not possible to group them by age of colony.

Pup production at colonies in the Outer Hebrides

At 16 colonies in the Outer Hebrides, pup production in 2019 was **16,083** compared with 15,732 in 2016, an average annual increase of +0.7% (Table 1). Grouping colonies in the Outer Hebrides by location and age, reveals different pup production trajectories (Figure 9). Production at older, long

established colonies around the Sound of Harris is declining while production at colonies in the Monach Isles and new colonies at the southern end of the Outer Hebrides has increased.

Pup production at colonies in Orkney

At 28 colonies in Orkney, pup production was **22,153** in 2019 compared with 23,849 in 2016, an average annual decline of -2.4% (Table 1). Grouping colonies of similar ages showed that production at the long-established colonies is slowly declining, while several colonies established much later are still increasing slowly (Figure 10).

Pup production at colonies in the Firth of Forth

At 4 colonies in the Firth of Forth, pup production in 2019 was **7,261** compared with 6,426 in 2016, an average annual increase of +4.2% (Table 1). Production at Fast Castle continues to increase and it is now the biggest colony in the North Sea (Figure 11). This rapid increase is due to expansion to the south-east towards St Abbs Head and westwards towards Siccar Point.

Pup production at colonies on the north and north-east coast of Scotland

At 6 colonies on the north mainland coast of Scotland, pup production in 2019 was **2,465**, compared with an estimated 2,665 born in 2016 (included in 4,192 for other colonies, Table 1). These colonies lie between Helmsdale and Duncansby head in the Moray Firth and at Loch Eriboll and Eilean nan Ron on the north coast of Scotland (Figure 1). The latter two are very close to an active RAF bombing range and access for aerial survey can be restricted when the range is busy.

Pup production at colonies in east England

In England in 2019, **10,725** pups were born at the annually monitored colonies on the east coast compared with 8,213 born in 2016, an average annual increase of +9.3% (Table 1). All four colonies have been increasing over the past years, and especially rapidly at Horsey and at Blakeney Point, which remains the biggest colony in England (Figure 12).

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Table 1. Grey seal pup production estimates from 2019 compared with production estimates from 2016.

Location	Pup production in 2019	Pup production in 2016	Average annual change 2016 to 2019
Inner Hebrides	4,455	4,541	- 0.6%
Outer Hebrides	16,083	15,732	+ 0.7%
Orkney	22,153	23,849	- 2.4%
Firth of Forth	7,261	6,426	+ 4.2%
Regularly monitored Scottish colonies	49,952	50,548	- 0.4%
Other Scottish colonies ¹ (incl. N & NE mainland & Shetland)	4,112	4,193	- 0.6%
Total Scotland	54,064	54,741	- 0.4%
Farne Islands	2,823	2,295	+ 7.1%
Donna Nook, Blakeney, Horsey	7,902	5,918	+10.1%
Annually monitored colonies in eastern England	10,725	8,213	+ 9.3%
SW England ^{1,2}	450	250	
Small sites in E and NW England ^{1,3}	50	50	
Total England	11,225	8,513	+ 9.7%
Wales ^{1,4}	2,250	1,650	
Northern Ireland ¹	250	150	
Total UK	67,789	65,054	+ 1.4%
Isle of Man	69	84	

¹ Includes estimated production for colonies that are rarely monitored from different years

² Includes estimates for Scilly Isles, Lundy, various sites in Devon & Cornwall

³ Includes Coquet Island, Ravenscar, Scroby Sands, South Walney

⁴ Multiplier derived from indicator colonies surveyed in 2004 and 2005 and applied to other colonies last monitored in 1994

Table 2. Estimates of grey seal pup production from annually surveyed colonies in the Inner and Outer Hebrides, Orkney and in the North Sea between 1960 and 2016.

YEAR	Inner Hebrides	Outer Hebrides	Orkney	North Sea	Total
1960			2,048	1,020	
1961		3,142	1,846	1,141	
1962				1,118	
1963				1,259	
1964			2,048	1,439	
1965			2,191	1,404	
1966		3,311	2,287	1,728	7,326
1967		3,265	2,390	1,779	7,434
1968		3,421	2,570	1,800	7,791
1969			2,316	1,919	
1970		5,070	2,535	2,002	9,607
1971			2,766	2,042	
1972		4,933		1,617	
1973			2,581	1,678	
1974		6,173	2,700	1,668	10,541
1975		6,946	2,679	1,617	11,242
1976		7,147	3,247	1,426	11,820
1977			3,364	1,243	
1978		6,243	3,778	1,162	11,183
1979		6,670	3,971	1,620	12,261
1980		8,026	4,476	1,617	14,119
1981		8,086	5,064	1,531	14,681
1982		7,763	5,241	1,637	14,641
1983				1,238	
1984	1,332	7,594	4,741	1,325	14,992
1985	1,190	8,165	5,199	1,711	16,265
1986	1,711	8,455	5,796	1,834	17,796
1987	2,002	8,777	6,389	1,867	19,035
1988	1,960	8,689	5,948	1,474	18,071
1989	1,956	9,275	6,773	1,922	19,926
1990	2,032	9,801	6,982	2,278	21,093
1991	2,411	10,617	8,653	2,375	24,056
1992	2,816	12,215	9,854	2,436	27,321
1993	2,923	11,915	11,034	2,710	28,582
1994	2,719	12,054	11,851	2,652	29,276
1995	3,050	12,713	12,670	2,757	31,190
1996	3,117	13,176	14,531	2,938	33,762
1997	3,076	11,946	14,395	3,698	33,115
1998	3,087	12,434	16,625	3,989	36,135
1999	2,787	11,759	15,720	3,380	33,646
2000	3,223	13,472	16,546	4,303	37,544
2001	3,032	12,427	18,196	4,159	37,814
2002	3,096	11,248	17,952	4,520	36,816

YEAR	Inner Hebrides	Outer Hebrides	Orkney	North Sea	Total
2003	3,386	12,741	18,652	4,870	39,649
2004	3,385	12,319	19,123	5,015	39,842
2005	3,427	12,397	18,126	5,232	39,182
2006	3,501	11,719	19,335	5,463	40,018
2007	3,118	11,342	19,184	5,780	39,424
2008	3,317	12,279	17,813	6,501	39,910
2009		11,887*	18,548	7,384	41,136
2010	3,108	11,831	18,562	8,160	41,661
2011					
2012	4,088	14,134	22,920	10,205	51,347
2013					
2014	4,054	14,331	23,777	12,655	54,817
2015					
2016	4,541	15,732	23,849	14,639	58,761
2017					
2018				16,845	
2019	4,455	16,083	22,153	17,986	60,677

*2008 production estimates were used as a proxy for seven colonies in the Outer Hebrides for which new production estimates could not be derived in 2009.

The following new colonies were first included in the regional total in the year given in parentheses:

Inner Hebrides: Oronsay (2001); Oronsay Strand (2005); Soa (Coll) (2012)

Outer Hebrides: Berneray & Fiaray (1998); Mingulay (2003); Pabbay (2005); Sandray W (2010); Sandray NE&SE (2019)

Orkney: South Ronaldsay E&W (1991); Calf of Eday & Copinsay (1993); Stronsay Sty Taing (1994); Calf of Flotta (1996); Sule Skerry (1997); Fara (1999); N Flotta & Westray S (2003); Rothiesholm Head (2005); NE Hoy (2008); Hacks Ness (2016)

North Sea: Fast Castle (1997); Inchkeith (2003); Craigleith (2004)

Table 3. Estimates of grey seal pup production from irregularly surveyed colonies around Scotland.

Region	Location	Survey method	Last surveyed	Most recent count
Inner Hebrides	Loch Tarbert, Jura	SMRU visual	2007	4
	Treshnish small islands & Dutchman's	SMRU photo & vis	2010	~20
	Staffa	SMRU visual	2008	~5
	Little Colonsay, by Ulva	SMRU visual	2008	6
	Meisgeir, Mull	SMRU visual	2008	1
	Craig Inish, Tiree	SMRU photo	2005	2
	Cairns of Coll	SMRU photo	2008	10
	Muck	SMRU photo	2005	18
	Rum	SNH ground	2013	15
	Canna	SMRU photo	2005	25
	Ascrib Islands, Skye	SMRU photo	2008	64
	Fladda Chuain, North Skye	SMRU photo	2019	262
	Trodday, NE Skye	SMRU photo	2008	55
	Summer Isles	SMRU photo	2010	~60
	Islands close to Handa	SMRU visual	2009	10
Outer Hebrides	Sound of Harris islands	SMRU photo	2008	296
	St Kilda	NTS reports	rare	~5
North Mainland	Loch Eriboll & Whiten Head	SMRU photo	2019	536
	Eilean nan Ron, Tongue	SMRU photo	2019	73
Orkney	Fers Ness, Eday	SMRU photo	2019	21
Shetland	Various sites	SNH ground	2012	761
NE Mainland	Duncansby Head to Helmsdale	SMRU photo	2019	1,856
Firth of Forth	Inchcolm	Fife Seal Group	2019	7
Total	Other Scottish colonies		to 2019	4,112

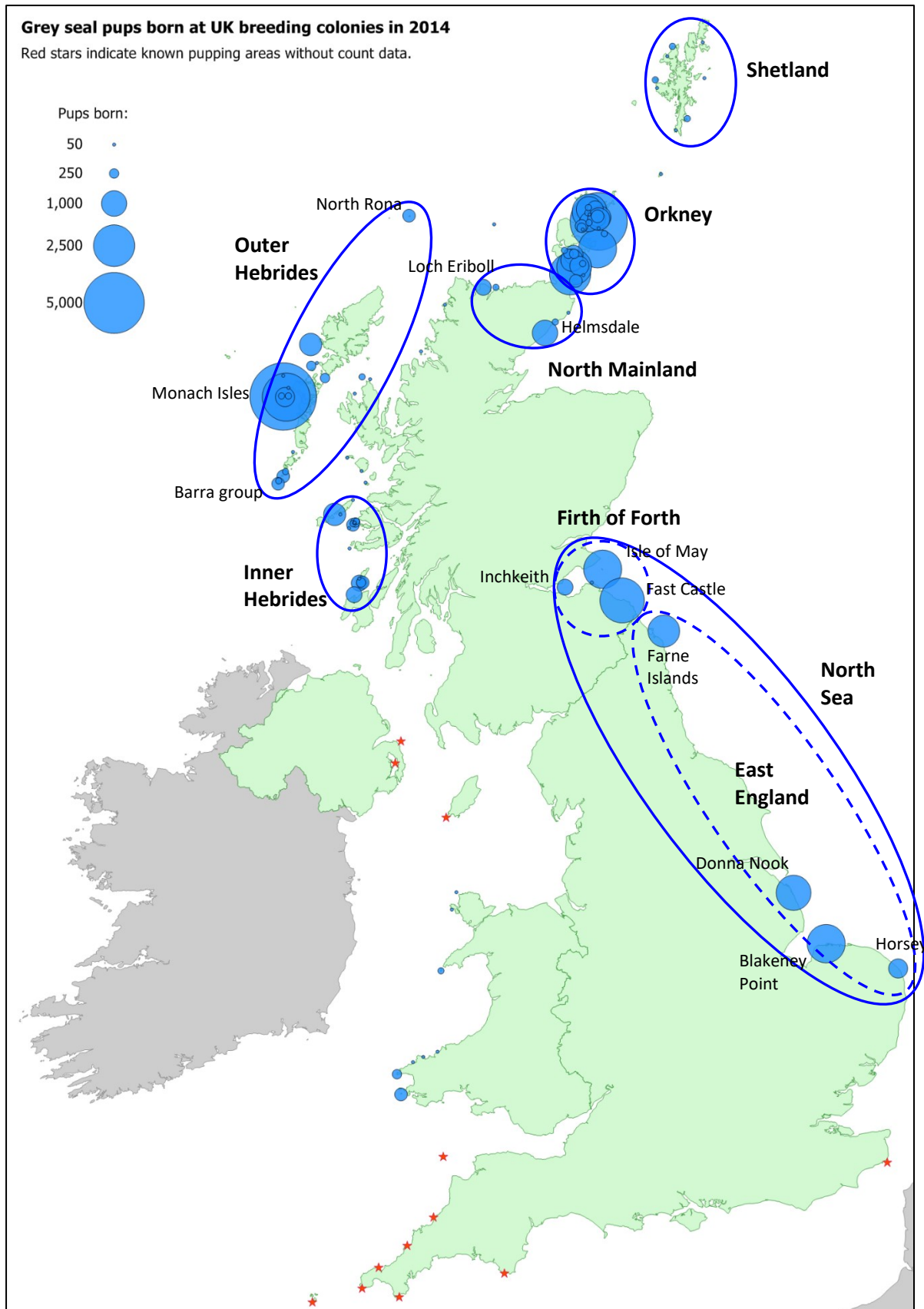


Figure 1. Pup production at the main grey seal breeding colonies in the UK in 2014. Smaller numbers of grey seals will breed at locations other than those indicated here, including in caves.

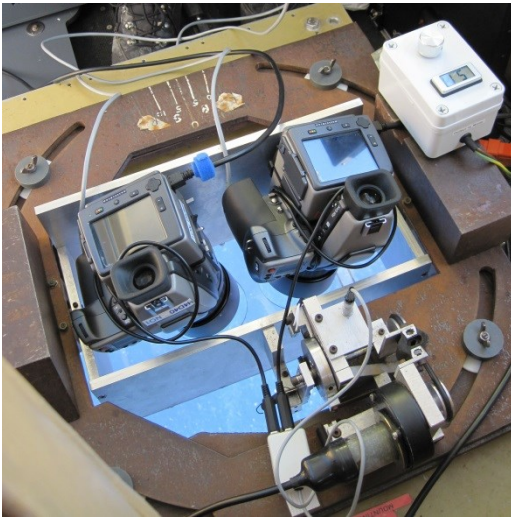


Figure 2. Two Hasselblad H4D-40 medium format cameras fitted in SMRU's Image Motion Compensation (IMC) mount. Each camera is set at an angle of 12 degrees to increase strip width. The cradle holding the cameras rocks backwards and forwards during photo runs. Rocking speed is set depending on the altitude 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 130kts at 1100ft (providing 20% overlap between consecutive frames). The resulting ground sampling distance is approximately 2.5 cm/pixel.

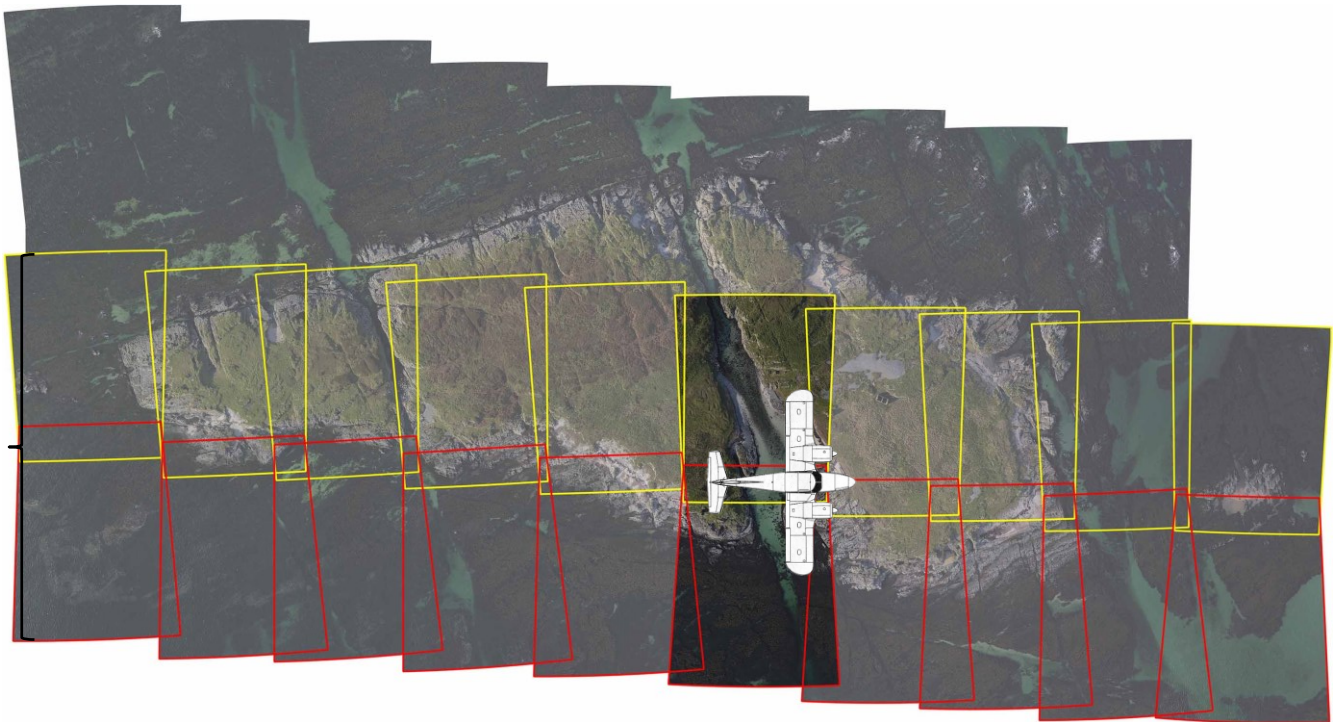


Figure 3. The individual footprints of each pair of photographs taken on a run over Eilean nan Ron, off Oronsay in the Inner Hebrides, flying at 1,100ft (red: left-hand camera; yellow: right-hand camera).

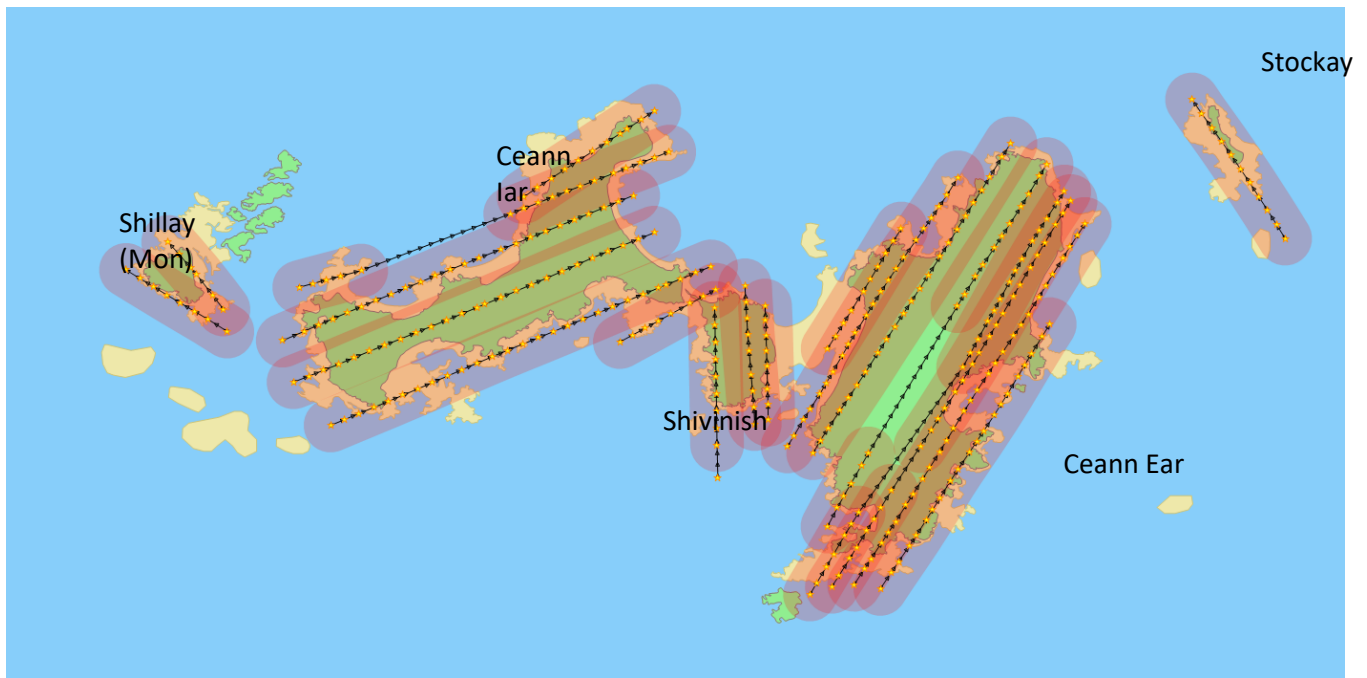


Figure 4. Survey runs and approximate camera trigger locations (yellow dots) for five colonies in the Monach Isles in the Outer Hebrides on 26 October 2012.

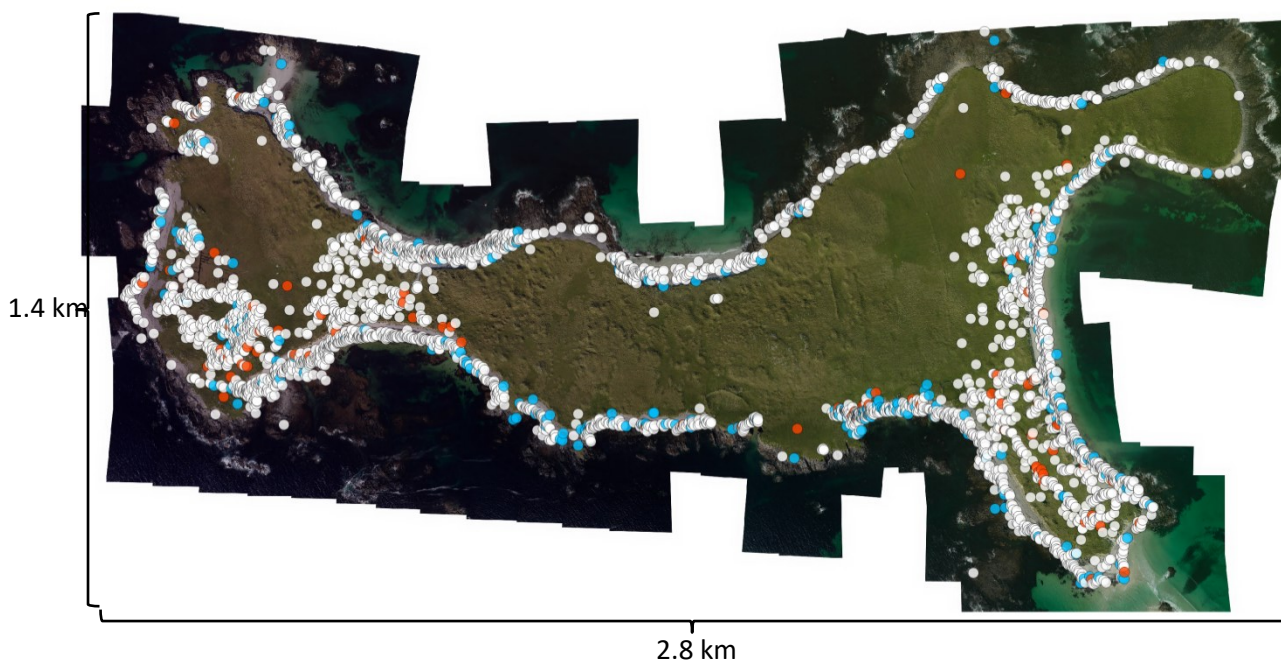


Figure 5. Ceann Iar, the second biggest of the Monach Isles in the Outer Hebrides, is the largest grey seal breeding colony in Europe (ca. 6,000 pups are born each year). This screenshot shows white-coated (white), moulted (blue) and dead pups (red) counted from approximately 200 stitched photographs 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.

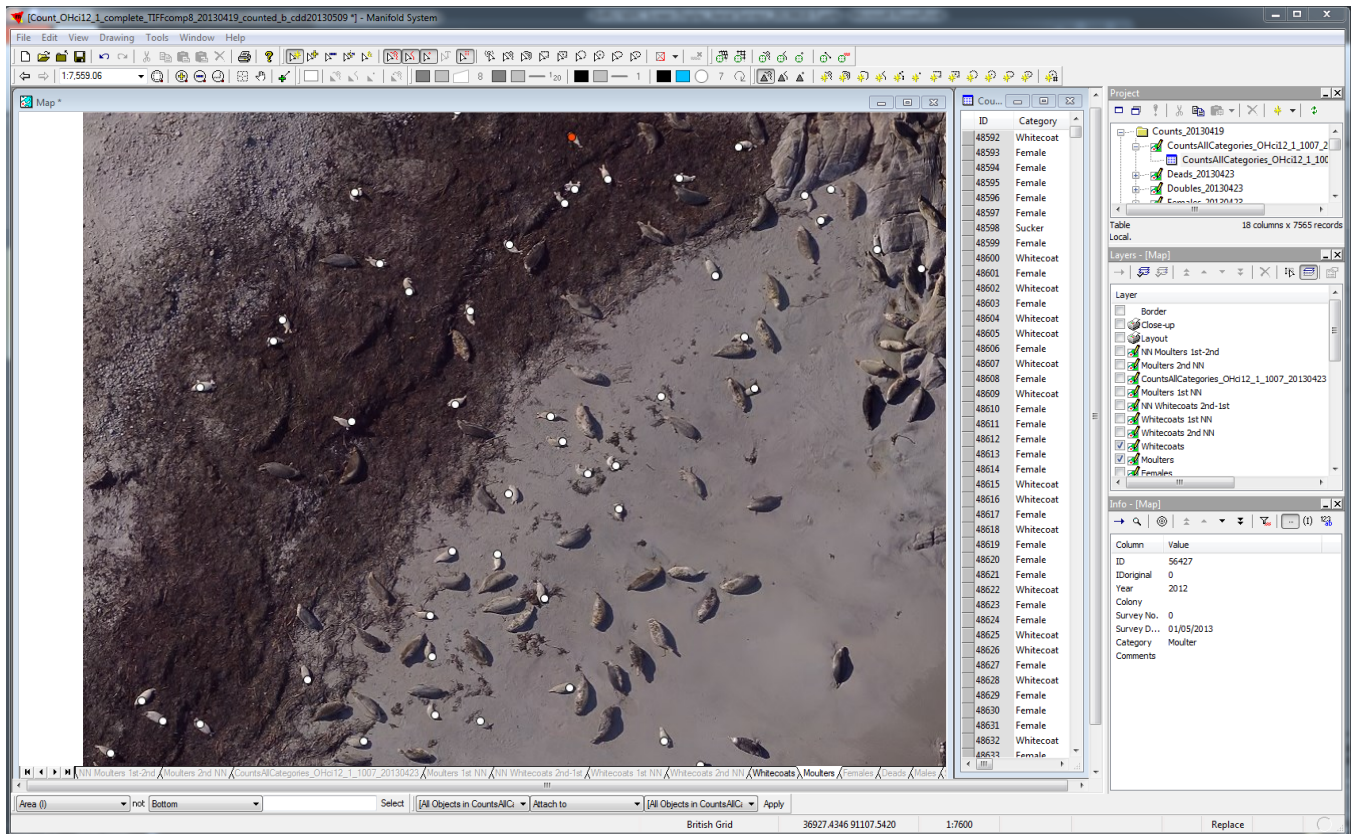


Figure 6. Manifold GIS 8.0® screenshot showing grey seal pups counted on Ceann Iar. Pups are marked and classified as whitecoats or moulted pups (and as dead if evident). The images are not geo-referenced but there is the potential for further processing, thus obtaining approximate coordinates for every pup counted on a small number of images.

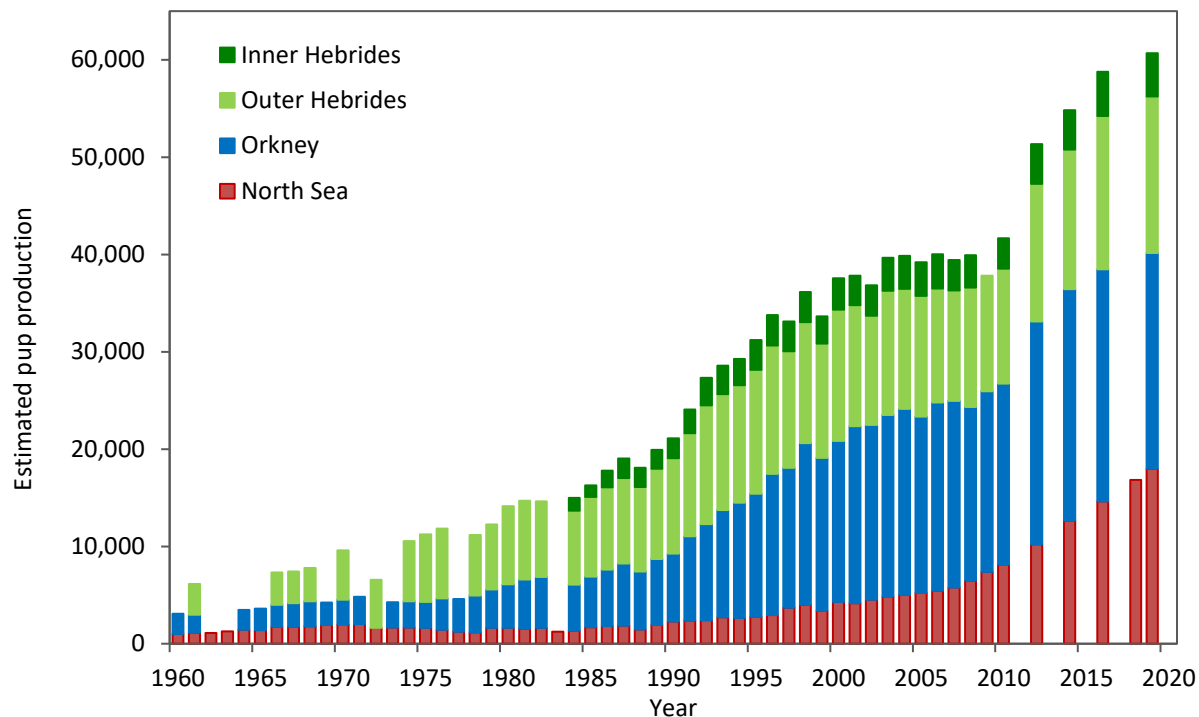


Figure 7. Grey seal pup production at routinely surveyed breeding colonies in Scotland and England from 1960 to 2019. These four regions are used in the grey seal total population model.

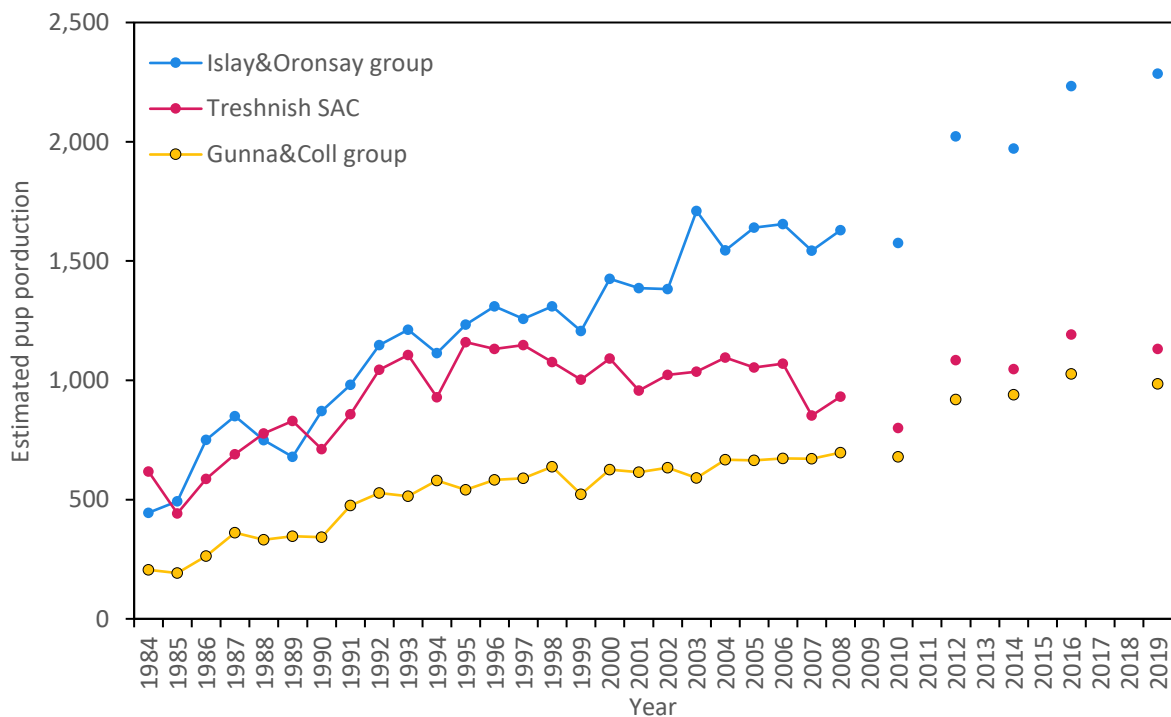


Figure 8. Grey seal pup production in the Inner Hebrides, grouped by location. The change in methodology from film to digital is likely to be responsible for a step increase between 2010 and 2012.

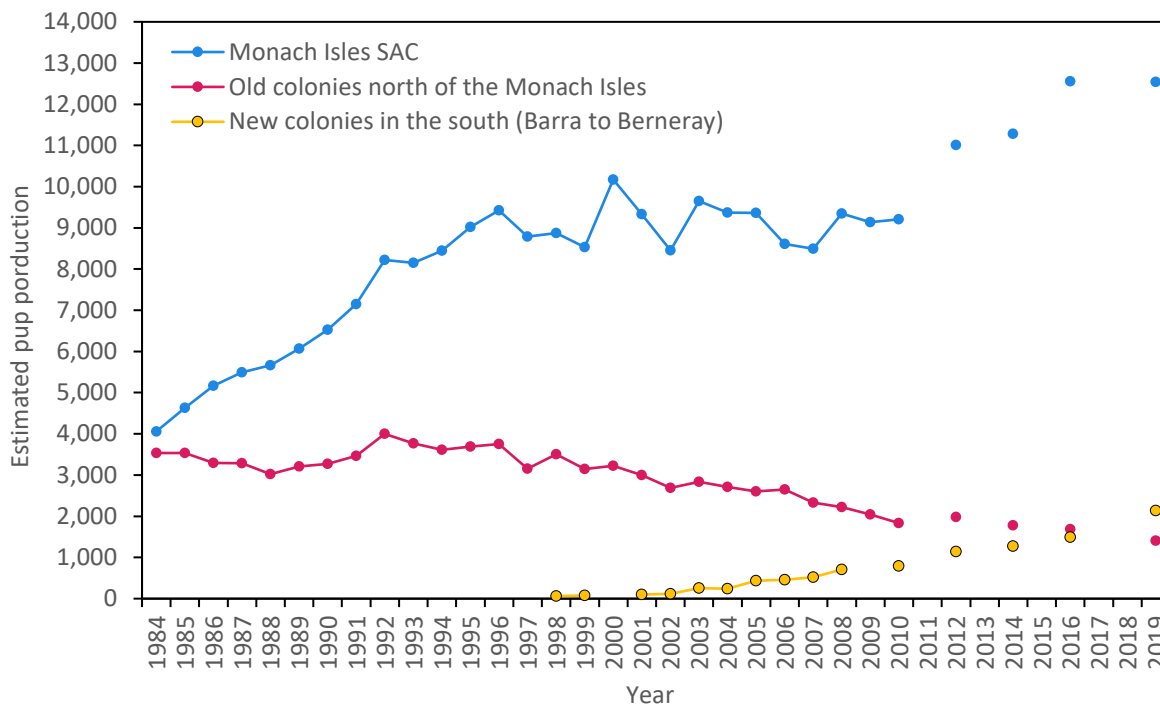


Figure 9. Grey seal pup production in the Outer Hebrides, comparing breeding colonies on the Monach Isles, long established (old) colonies, and newly established colonies. The change in methodology from film to digital is likely to be responsible for a step increase between 2010 and 2012.

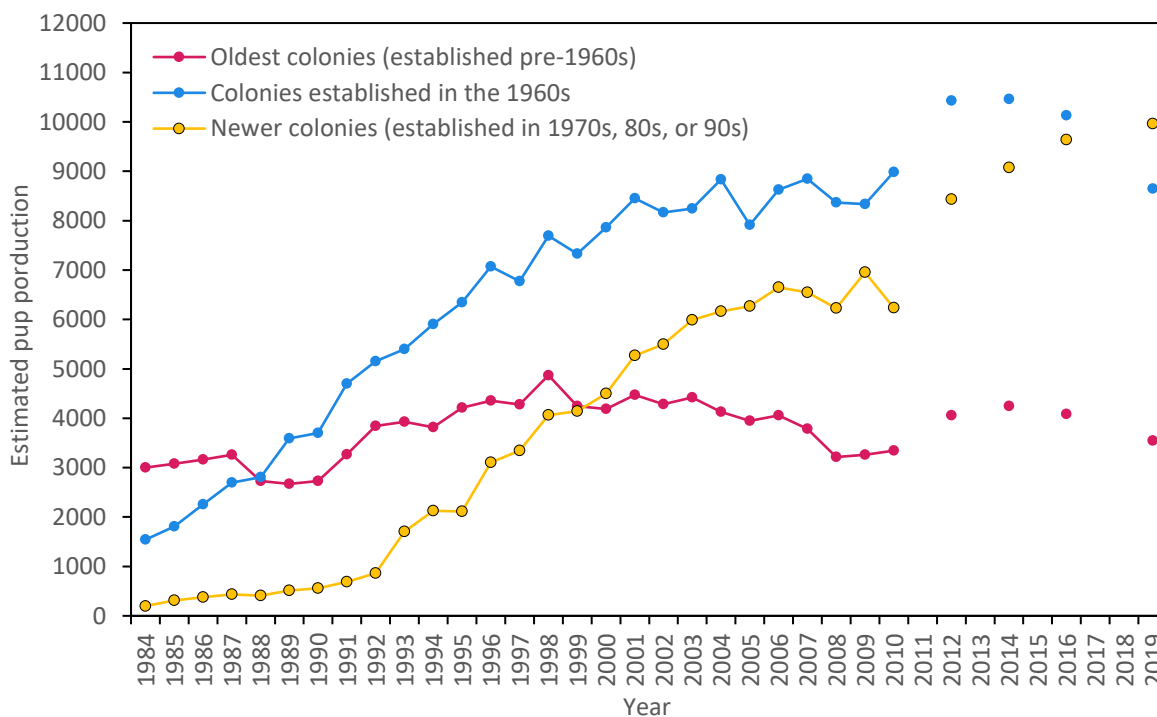


Figure 10. Grey seal pup production at colonies in Orkney, comparing colonies well established before the 1960s, colonies established during the 1960s and colonies established more recently. The change in methodology from film to digital is likely to be responsible for a step increase between 2010 and 2012.

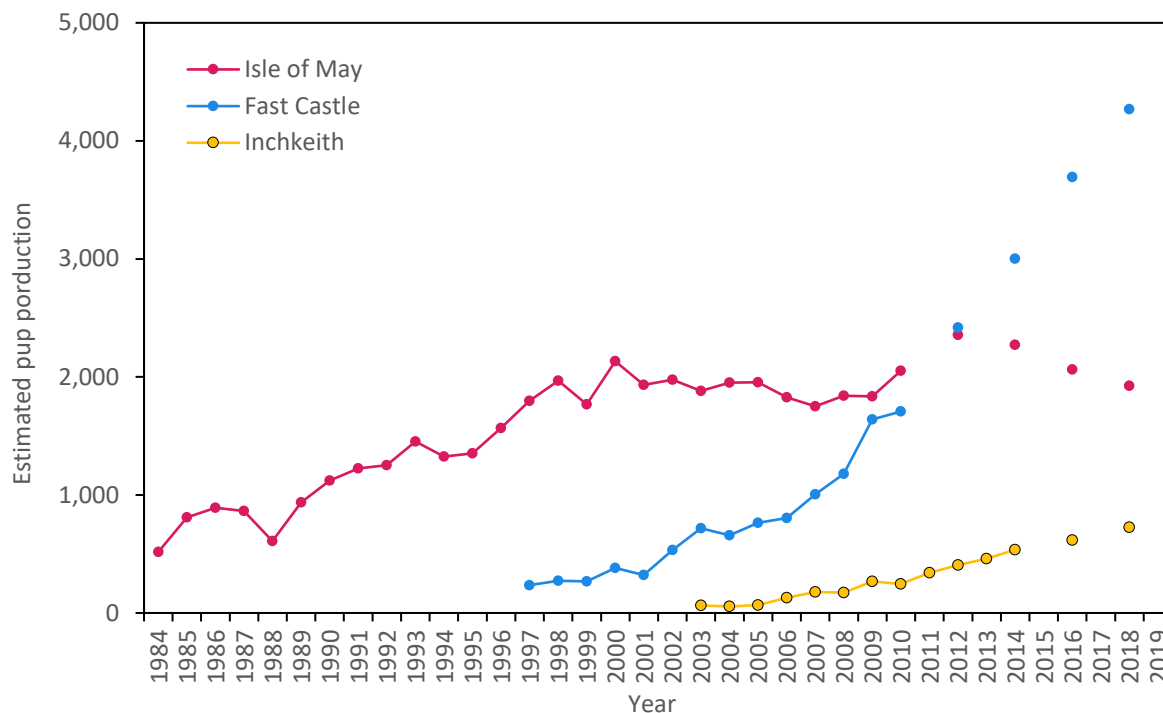


Figure 11. Grey seal pup production at the main colonies in the Firth of Forth. The change in methodology from film to digital is likely to be responsible for a step increase between 2010 and 2012.

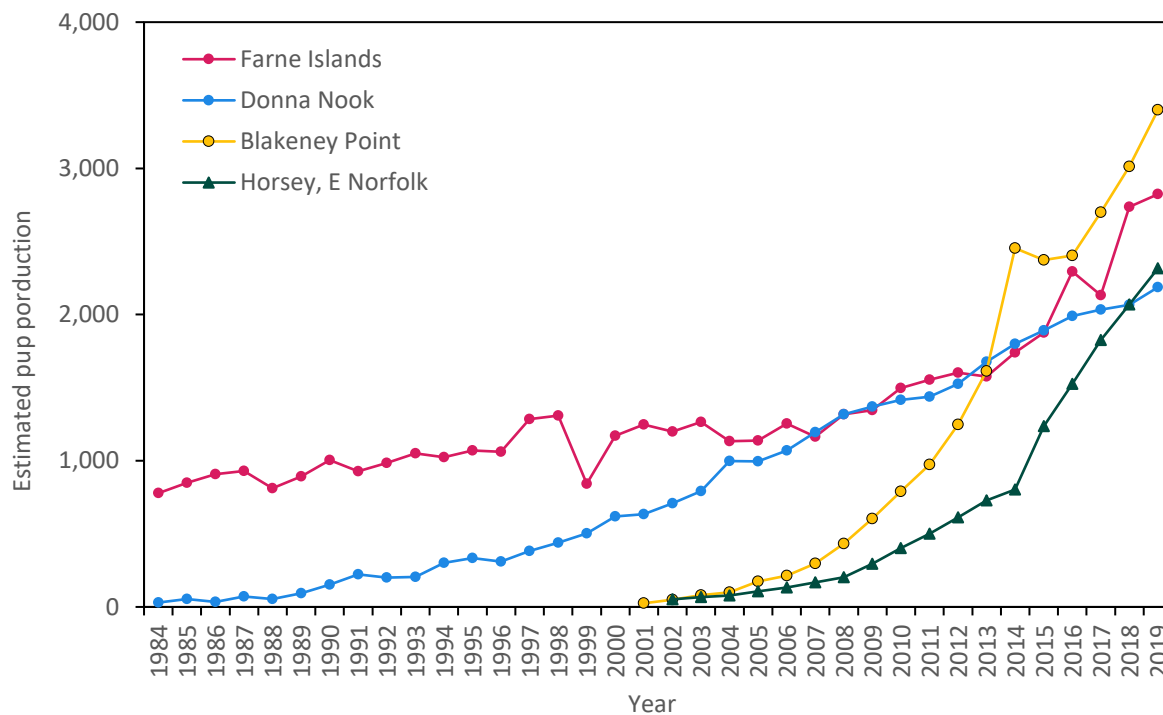


Figure 12. Grey seal pup production at colonies in East England. These colonies have been ground counted by the National Trust (Farne Islands and Blakeney Point), the Lincolnshire Wildlife Trust (Donna Nook), Natural England (Horsey, up to 2011), and Friends of Horsey Seals (Horsey, since 2012).

Grey seal independent estimate scalar: converting counts to population estimates

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Abstract

A key component of the grey seal population model is an estimate of population size based on counts of grey seals hauled out during the harbour seal moult surveys in August. These counts are converted to an estimate of total population size using a scalar based on the estimated proportion of time grey seals spend hauled out during the aerial survey window (i.e. 2 h either side of low tide in August between 10:00 – 18:00), derived from telemetry data. Previous research using low resolution Argos tags gave a mean estimate of 31% (95% CI: 15-50%) (Lonergan et al., 2011). Subsequent preliminary analysis using high resolution data from 25 GPS tags showed that Argos data is unlikely to be appropriate, and gave a revised estimate of 23.9% (95% CI: 19.2-28.6%), resulting in a change in the 2008 population estimate of ~30% (Russell et al. 2016). Since that preliminary analysis, a large grey seal tagging programme has resulted in a dramatic increase in sample size (n=60), allowing the analysis to be revisited. This study provides a new mean estimate of the percentage of the population hauled out of 25.15% (95% CI: 21.45-29.07%). In addition to the revision of the scalar, this study examined the influence of biotic (body length, sex) and abiotic (region, days from spring tide, day of August, time of low tide, weekend vs weekday, weather, quarter of survey window) covariates on the proportion of time spent hauled out during the aerial survey windows. A small effect of quarter of survey window was detected, but confidence intervals overlapped, and it was not deemed appropriate to incorporate this into the scalar estimate. None of the other covariates were found to influence the probability of seals being hauled out. A qualitative examination of the width of the scalar confidence intervals and the variation in counts for a constant population size (i.e. multiple August counts in the same area and year) indicated that although the confidence intervals likely encompass the mean scalar over the month of August there is substantial day-to-day variation in the mean proportion hauled out which the confidence intervals do not incorporate. The reasons and implications of this are discussed.

Introduction

A robust estimate of grey seal population size and trends is fundamental for their effective management. For the UK, estimates are generated using a Bayesian state-space model (SCOS-BP 21/05), incorporating: (i) a time-series of pup production estimates, (ii) knowledge of life-history parameters, and (iii) estimates of population size (2008, 2014, 2017) which are independent from the pup production data (hereafter independent estimates). These independent estimates of UK population size are derived from counts of grey seals hauled out on land during the harbour seal moult surveys in August. Translating these counts into an independent estimate of grey seal population size requires estimates of proportion of the overall population expected to be hauled out during the aerial survey window (2h either side of low tide in August, where low tide falls between 10:00 – 18:00), and thus available for count. The reliability of population estimates is dependent on the reliability of the scalar (inverse of the proportion of the population hauled out). Proportion of time hauled out is estimated from locational and behavioural data (e.g. haulout information) from animal-borne tags which are glued to the fur on the back of the neck (falling off by or during the annual moult).

There have been two previous estimates of the proportion of the population hauled out during the survey window. Lonergan *et al.* (2011) estimated that 31% (95% CI: 15-50%) of the population would be hauled out during a survey window, based on analysis of the available telemetry data

(predominantly low-resolution Argos tags with <12 locations per day and spatial error probabilities often exceeding 2.5 km). Subsequent analysis comparing the Argos data with high-resolution data from GPS/GSM tags (>50 locations per day with spatial errors typically <50 m) revealed that the spatial and temporal resolution of Argos data is likely to be inadequate to estimate a robust scalar (Russell *et al.* 2016). Indeed, based on GPS/GSM data, Russell *et al.* (2016) generated a lower estimate of the proportion of time spent hauled out during the survey window: 23.9% (95% CI: 19.2-28.6%). This resulted in an increase in the 2008 population estimate of >13,000 seals (~30%) over that of Lonergan *et al.* (2011). However, Russell *et al.* (2016) comprised a preliminary study based on a relatively small sample size ($n = 25$) and spatial extent (seals tagged in East Scotland and Southeast England).

Since Russell *et al.* (2016), a large-scale deployment of GPS/GSM tags on grey seals, funded by the UK Government Department for Business, Energy and Industrial Strategy (BEIS), has generated a comprehensive dataset of grey seal movements from haulout sites around the UK (Carter *et al.*, 2020). Such a dataset, combined with the GPS/GSM data used in Russell *et al.* (2016), provided an opportunity to re-examine the scalar and also the impact of biotic and abiotic drivers on the probability of seals being hauled out during the survey window. As well as sex and length of the tagged individuals, the following abiotic covariates were considered: region, weekday/weekend, time of day, day of August, days from spring tide, weather (daily mean rainfall and windspeed, maximum daily temperature), quarter of the 4 h tidal window. These covariates were chosen because of a perceived potential influence on grey seal haul out behaviour, in line with Lonergan *et al.* (2011). No evidence of influence of these covariates on the probability of seals hauling out was found in Lonergan *et al.* (2011), though as discussed above there are concerns regarding the robustness of the conclusions of that study given the data resolution. Of the above covariates, sex, region and time of day were considered in a study of factors influencing seal activity budgets (Russell *et al.*, 2015). Significant effects on the proportion of time hauled out were found for grey seals, but this was at a comparatively coarse temporal resolution (six-hours) and was not specific to August.

Quantifying the impact of covariates on the proportion of time hauled out during the survey window is critical in generating a robust scalar. Any covariate effects would require the following considerations to be made: (1) count-specific scalars could be applied where appropriate (e.g. by region, week, time of day, day of August); (2) any sex/age impacts would require an adjustment of the scalar to reflect the estimated age/sex composition of the population (compared to the composition of tagged seals); (3) environmental conditions in the August tagging data should approximate conditions during the August survey season, but surveys are not conducted in adverse weather (rain, high winds), thus the tagging data may encapsulate conditions that are not reflected in the count data; (4) although generating a specific scalar for different weather conditions is unlikely to be feasible, an understanding of any impact of weather would allow the generation of robust standard errors that take into account variation in probability of hauling out resulting from the condition-mediated non-independence of individuals.

Methods

Telemetry Data

Individuals Considered

Telemetry data were restricted to tags that transmitted data throughout the whole of August (Russell *et al.*, 2016). Seal behaviour may be anomalous for a short time after tagging (e.g. a week), thus tags that were deployed in late July or any time in August ($n=4$) were excluded. Tags may stop transmitting for a number of reasons related to device failure, or animal death. Individuals may exhibit anomalous behaviour prior to death, thus tags that stopped transmitting during August ($n=4$)

were also excluded. Telemetry data were also restricted to haulout events in Seal Management Units (SMUs) that are covered by UK aerial surveys (Scotland and Eastern England). Finally, the remaining tag data were quality checked to ensure adequate data resolution for the analysis. This process resulted in a final sample size for analysis of 60 tags (Table 1); the original 25 tags (2005-2015) analysed by Russell et al. (2016) and 35 from the more recent tag deployments (2017-2019). The spatial distribution of haulout events recorded from these tags is shown in Figure 1.

Table 1: Number of GPS tags used for this analysis by deployment year, location, Seal Management Unit and sex.

Deployment	Year	Deployment Location	SMU	M	F
pv14	2005	Abertay Sands /	East Scotland	1	1
gp13	2008	Tentsmuir		2	4
hg48/hg48a	2015	Blakeney Point / Donna Nook	Southeast England	6	11
hg53	2017	Orkney	North Coast &	2	3
hg59	2018		Orkney	2	0
hg54	2017	Islay / Oronsay	West Scotland	1	4
hg65	2019			2	1
hg55	2017	Monach Isles	Western Isles	2	5
hg64	2019			2	3
hg60	2018	Findhorn / Dornoch	Moray Firth	2	6
TOTAL				22	38
				60	

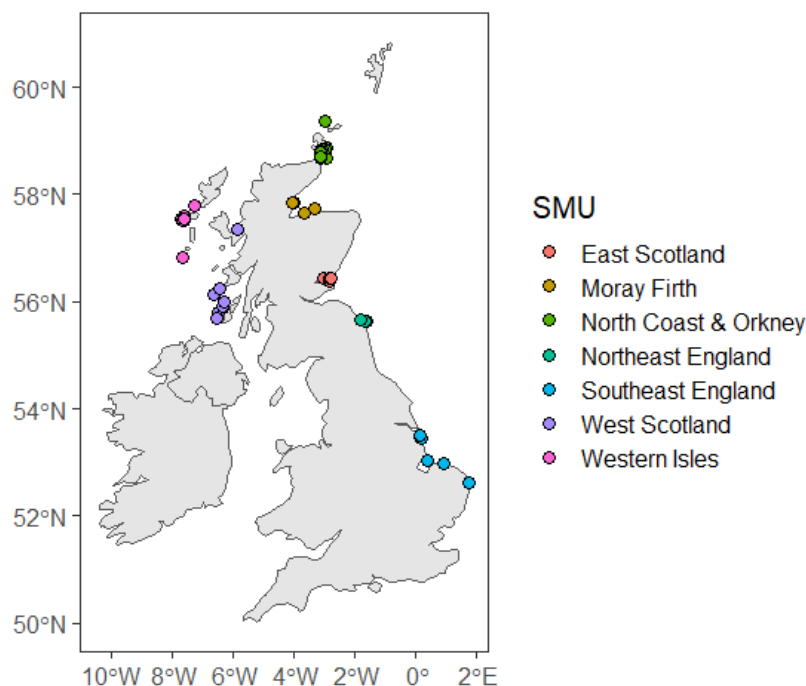


Figure 1: Locations associated with grey seal haulout events during the August aerial survey window, colour-coded by Seal Management Unit.

Defining haulout events

The GPS/GSM tags record and transmit haulout events; these occur if the tag has been dry for over 10 minutes (with the start then adjusted to incorporate these 10 minutes) and end if the tag has been wet for 40 seconds. There was an issue with the definition of haulout events on tags deployed prior to 2017; haulout data from these tags had to be manually adjusted (see Russell et al. 2016 for details). Tags occasionally record a haulout event even though the seal is in the water. This likely happens when the seal is at the surface with part, or all, of the tag exposed to the air, and the wet/dry sensor (located on the leading edge of the tag) remains below the “wetness” threshold for long enough to trigger a haulout event. Seals often spend prolonged periods at the surface, both offshore during foraging trips and nearshore while waiting for a tidal haulout site to be exposed. If possible, such haulout events should be removed; including at-sea haulouts within the analysis would artificially inflate the proportion of time seals are estimated to be hauled out during the survey windows.

As a first step, a protocol was developed to assign haulout events, as reported by the tags, (n = 3491) to land or sea. Haulout events with concurrent location data within the limits of a georeferenced mean low tide layer were assumed to be onshore haulouts (true haulouts; n = 2279; 65%), as were those with no concurrent location data (i.e. interpolated locations; n=142; 4%). Haulout events occurring offshore (>1 km from the mean low tide boundary) with concurrent location data were classed as at-sea haulouts and were removed (n = 544; 16%). However, determining the status (onshore vs at-sea) of haulout events that were nearshore (outwith the mean low tide boundary but < 1 km of the coast; n = 362 observed and 70 interpolated (combined = 12%)); or were offshore but had interpolated locations (n = 94; 3%) was not straightforward. This was complicated by location error (for observed locations and resulting from interpolation) and error in the low tide maps (due to limited spatial resolution and changes in the shape and distribution of sandbars through time).

The status of nearshore haulout events was investigated further using (a) additional wet-dry data transmitted by a subset of tags, and (b) through manual exploration of the data. To investigate how to distinguish between onshore and at-sea haulouts, a subset of the tags deployed during the BEIS project (n=23) were programmed to transmit additional data from the wet/dry sensor (number of 4 s intervals above the “wetness” threshold), and mean duration of wet periods (successive 4 s intervals above the “wetness” threshold). Using these two parameters, and the start and end times of haulout events, these data were examined for a signal that might allow inference of which haulout events occur at-sea versus on land. This investigation indicated that such data may be useful in distinguishing at-sea haulout events from true haulouts using a threshold of 30% of 4 sec intervals during the event above the “wetness” threshold (< 30% being classed as onshore, and > 30% as at-sea). However, a greater sample size is required to determine the robustness of this finding. To address the issues of lack of accuracy in the low tide maps, nearshore haulout events within 100 m of low tide for which there were concurrent location data (n = 284) were examined in Google Earth using historical satellite images taken at, or close to low tide, and, for areas with dynamic coastlines (e.g. sandbars), if possible from years concurrent with seal telemetry data. As a result, 211 (74%) of these 284 nearshore haulout events could be confidently assigned to the intertidal zone (i.e. true haulouts). This resulted in 2632 true haulouts, and 315 for which the status was uncertain (nearshore with concurrent locations or without observed locations). Using this corrected dataset, a sensitivity analysis was conducted to investigate the impact on the scalar of various different treatment rules for classifying haulout events of unknown status.

In the analysis conducted by Russell et al. (2016), a protocol for assigning haulouts as on land or at-sea was developed. Haulouts with concurrent location data >1 km from low tide were treated as at-sea haulouts and those <1 km from low tide were treated as true haulouts. Haulouts without concurrent location data were treated as true haulouts if the interpolated location fell within 1 km of low tide. Haulouts without concurrent location data were treated as at-sea or “unknown” (they did not contribute to analysis of proportion of time hauled out vs at-sea) on the basis of the distance

from the interpolated location to the coast and the amount of time between the surrounding observed locations; i.e. at-sea if the seal could not have feasibly been hauled out and have such an interpolated location, and “unknown” otherwise. Specifically haulout events were flagged as at-sea if any of the following applied in terms of the distance of the interpolated location from the mean low tide and time between the surrounding observed locations: >10 km & <2 h; >5 km & <1 h, or >1 km & <0.5 h. Haulout events were flagged as unknown if >10 km & >2 h; >5 km & <10 km & >1 h; and >1 km & <5 km & >0.5 h. Applying these rules from Russell et al. (2016) to the dataset used here gave a mean estimated proportion of time hauled out of 0.2609 (95% CIs 0.2248-0.299). We examined the impact of six realistic alternative treatments for haulouts of uncertain status (onshore vs at-sea) on the mean estimated proportion of time hauled out. In general, there was little impact of different treatments (range of mean estimate 0.2453 - 0.2609). Based on these findings, and close scrutiny of these nearshore haulout events with concurrent location data, a threshold of 20 m was selected above which haulouts with concurrent location data were treated as at-sea. There was no change in the treatment of haulouts without concurrent location data compared to that of Russell et al. (2016) described above. For the final dataset 2744 haulouts were considered. This approach minimises the risk of artificially inflating the proportion of time hauled out due to at-sea haulouts, while still allowing some margin for GPS positional error and variation in spatial extent of haulout area due to spring low tide.

Covariate data

Covariate data were sourced as follows. Low tide data were extracted for each haulout location from Poltips (The Proudman Oceanographic Laboratory, National Oceanography Centre). Weather-related covariates (wind speed, rainfall and temperature) were extracted from the Met Office Integrated Data Archive System (MIDAS) for UK land surface stations <https://catalogue.ceda.ac.uk/uuid/dbd451271eb04662beade68da43546e1>, which provides daily mean, maximum and minimum estimates. Values were extracted for the nearest weather station to the seal location data (mean distance = 18 km +/- 13.5 km SD). Days from spring tide was calculated in R (R Core Team, 2020) using the “lunar” package (Lazaridis, 2014).

Modelling framework

The response variable (proportion of 4 h survey window spent hauled-out) was modelled as a function of abiotic and biotic covariates (binomial error distribution) using generalised additive models (GAMs) in a generalised estimating equation (GEE) framework (GEE-GAMs) using the packages “geepack” and “splines” (Halekoh et al., 2006) in R. Covariates (see Introduction) were input as factors (categorical) and smooth (continuous) terms. The GAM approach allowed the inclusion of smoothed terms to investigate non-linear relationships with the response variable while the GEE framework ensured prediction of the population mean with associated standard errors robust to any residual non-independence within individuals (Zuur et al. 2009). The most parsimonious model was found by backwards selection using model information criterion score from a full model (containing all possible covariates). Quasi Information Criterion (QIC) was used, as maximum likelihood based alternatives (e.g. Akaike Information Criterion) are not applicable to GEEs (Cui and Qian, 2007). Threshold for covariate removal $\Delta QIC < 2$ (Burnham and Anderson, 2002). Within-individual non-independence is a potential feature of this dataset, as the probability of a seal being hauled out in a given survey window is likely to be dependent on the activity of the seal in previous survey windows. The use of GEEs with individual seal ID as a blocking factor allows residual correlation within an individual and standard errors to be adjusted accordingly (Zuur et al., 2009). Two individuals recorded few (<10) observations (i.e. known haulout status data during a 4 h low tide survey window) due to missing data. However, removing these individuals from the dataset had no impact on model selection results, or subsequent plots of model output, so they were kept in the dataset.

Three analyses were conducted: (i) testing of weather covariates (rainfall, windspeed and temperature), (ii) testing of the effect of quarter of the survey window, and (iii) testing of key biotic and abiotic (excluding weather) covariates. This three-phase approach was taken as, for investigation of weather effects, the data had to be restricted to exclude observations (4 h tidal survey windows for which there are haulout data for an individual seal) that were >10 km of the coast (n=57), since conditions at the haulout will presumably not affect the probability of hauling-out if the seal is far offshore. Furthermore, observations where windspeed was > 40 kmh were excluded as there were very few data points (n=20; 1.9% of data) and aerial surveys are not conducted in such conditions. This resulted in a final dataset of 1043 observations. To investigate the impact of quarter of survey window, the data needed to be considered at a 1 h resolution (compared to a survey window (4 h) resolution). The main analysis (phase iii) was conducted using all individuals and survey windows in the dataset (n=1153 observations). In addition to the models, a non-parametric bootstrap by individual (with replacement; N=500,000) was used to estimate the uncertainty around the population mean.

Results

(i) *Weather covariates*

There was no evidence of an effect of weather on the probability of seals being hauled out during the survey window in this analysis; none of the weather covariates considered (daily rainfall, maximum daily temperature, or mean windspeed) were retained in the minimal adequate model.

(i) *Quarter of survey window*

There was evidence of an effect of quarter of survey window on the proportion of time seals spent hauled out. Quarter of the survey window was retained in the minimal adequate model. The probability of being hauled out was greatest closer to low tide (Q2 and Q3), but confidence intervals overlapped (Fig. 2).

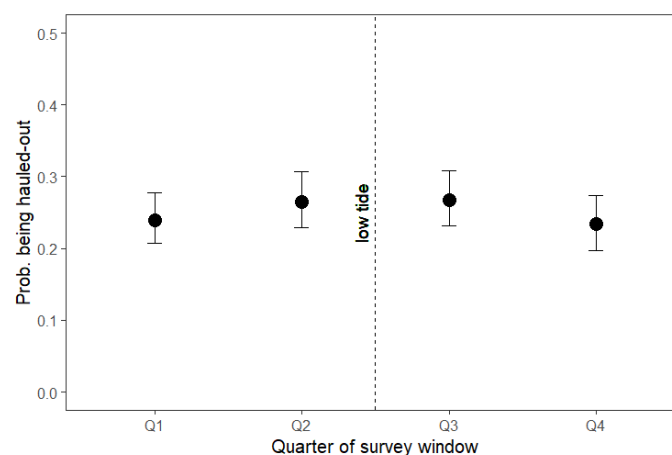


Figure 2. Model-predicted effect of quarter of the survey window on probability of being hauled out during the August survey window. Dots show the population mean, lines reflect the upper and lower 95% confidence limits.

(ii) *Key biotic and abiotic covariates*

None of the covariates tested were retained in the minimal adequate model. Day of August had a small effect on probability of being hauled-out, but the delta QIC value (1.03) was not enough to

justify retaining it in the final model (see Discussion). Eight seals in the database were missing seal length records (17 observations). Model selection was initially performed excluding these individuals. However, seal length was dropped on the first round of model selection ($\Delta\text{QIC} = -10.01$), so the full dataset was used for the rest of model selection. The results from the GEE (intercept only) revealed a mean proportion of 0.2514 (0.2171-0.2907 lower and upper 95% confidence intervals) which was similar to that generated from bootstrapping: mean: 0.2515.

Discussion

The revised proportion of the population hauled out (0.2515; 95% CI 0.2145 – 0.2907), resulting in a population scalar (i.e. $\frac{1}{prop.hauled-out}$) of 3.98 (95% CI 3.44 – 4.66), is slightly higher than that reported in Russell et al. (2016); 0.238, giving a scalar of 4.18 (95% CI: 3.50 – 5.21), which was derived from data from just under half of the tags analysed here. This result provides further support for a higher mean scalar than was derived from the Argos tags (3.23; Lonergan *et al.*, 2011).

At-sea haulout treatment protocol

The sensitivity analysis conducted here suggests that a threshold of 20 m from mean low tide is appropriate to classify nearshore haulouts with concurrent location data as true haulouts. This threshold allows margin for GPS location error, and variation in the available haulout area due to spring tides, while minimising the risk of artificially inflating the proportion of time hauled out through inclusion of at-sea haulouts. The assumption made here is that haulouts occurring within 20 m of mean low tide are not the result of seals resting at the surface. This behaviour is unlikely to occur so close to shore during low tide. However, given the fact that changing the data treatment protocol from that of Russell et al (2016), where the threshold was set at 1 km, resulted in a reduction in the mean estimate of 0.0094 (equivalent to an increase in the 2014 population estimate of 12,395 seals), and that the status (onshore vs at-sea) of haulouts without concurrent location data (9% of all haulouts; $n = 306$) still remains uncertain, the problem of defining at-sea haulouts warrants further research. Ideally, concurrent accelerometer data is required to determine the body orientation of the seal during such haulout events, and distinguish between a seal lying prone on a haulout or resting at the surface in a vertical position (i.e. bottling). However, accelerometer data are not currently transmitted by the GPS/GSM tags due to the large size of associated data packets. With further research, an algorithm could be developed to abstract these data into a simple indicator of body position during haulout events which could be readily transmitted alongside the haulout records.

Impact of covariates

The only covariate retained in model selection was quarter of survey window. This covariate was not considered in the overall analyses (iii) or when generating the bootstrapped estimates for proportion of time hauled out because (1) the effect size was relatively small and the confidence intervals overlapped, (2) conducting the main analyses on that scale would have likely caused complex correlation relationships within the residuals (within survey window, within individual), (3) there were not enough data to determine whether or not this relationship was temporally and spatially constant (in terms of the impact of tidal extent or region), (4) generating and combining quarter-specific scaled population estimates (with associated confidence intervals) for the independent estimates would be challenging.

Although quarter of survey window was the only covariate retained during model selection, this does not allow us to conclude that the other covariates have no impact. The sample size of 60, although relatively large in the context of studies using telemetry data, meant that there was limited

power to detect impacts, especially given there was such high variability in haulout patterns, and a maximum of 31 data points per individual. There may be drivers of haulout behaviour acting on multiple spatial scales associated with habitat (e.g. availability of haulout sites, trip duration and weather conditions), and thus detecting the impact of individual covariates is difficult and interactions between variables are precluded by limited sample size. Furthermore, the weather data was amalgamated over the whole day and was from the nearest weather station which was between 1 and 62 km from the haulout site. Both in the UK and on the continent, observers have highlighted the link between very hot weather and relatively low August aerial survey counts. Such weather conditions are not common and would not be detectable within the telemetry data, but can have a considerable impact on the counts, and thus population estimates. For example, a count of 196 on a particularly hot day (with >100 visible in shallow waters adjacent to the haulout site) was recorded for the Monach Isles in 2011, whereas in surrounding years the counts were $\geq 1,350$. Moreover, the impact of weather is dependent on both recent and current conditions. For example, larger haulout numbers are often associated with a dry day that follows a long period of wet weather compared to a one within a period of dry weather.

Scalar uncertainty

The scalar uncertainty does take into account inter-individual variation in haul out patterns (via nonparametric bootstrapping). There is considerable variation among individuals; some individuals make short foraging trips, hauling-out every day, while others make prolonged trips offshore, then return to haul out on land for multiple days at a time. For example, one individual in the dataset spent a mean proportion of time hauled out >0.5 across all survey windows. There was no evidence for anomalous data within the track (e.g. tag issues or early breeding behaviour). The individual hauled out in 15 out of 20 survey windows, making frequent short foraging trips within Scapa Flow, Orkney, throughout the whole duration of the track (including August). In contrast, some (but not all) individuals tagged in the Western Isles travelled to the self-edge (Carter et al. 2020), spending only 17% of time hauled out on average during the August windows. Such individuals often spend multiple successive survey windows hauled out, followed by many days (often weeks) at-sea. Such long trips punctuated by long haulout events may have a disproportionate impact on the analysis, depending on where the individuals are in their trip - haulout cycle at the start of the time series (e.g. an individual already on a long foraging trip at the start of August may only record one haulout event during the time series, but an individual hauled out at the start of the time series may record three haulout events). Furthermore, the proportion of time individuals haul out for may impact the probability that they are encountered for tagging in the first place (i.e. individuals that make long trips may be less likely to be tagged), resulting in an overestimate of the mean proportion of time spent hauled out for that site.

The confidence intervals surrounding the scalar pertain to the population mean for all August survey windows. It was necessary to temporally aggregate all the data to generate the average scalar across the whole month of August. As such, the uncertainty does not incorporate day-to-day variation in the proportion of time hauled out during the survey window. Caution should be used when applying this scalar to survey data from individual haulout counts to generate abundance estimates. The scalar does not account for movement between haulout sites; such movement, especially for haulout sites used by a limited number of individuals will result in additional variation in counts. Stochasticity also becomes important when considering small counts. For a relatively large spatial scale (minimal influence of movement) and large counts, if the probability of a seal hauling out is independent of other seals and abiotic/biotic covariates then the confidence intervals generated should encompass the true scalar. However, given our limited ability to detect impacts of covariates (see above) we cannot be confident that the confidence intervals are appropriate. Influences such as weather, or a propensity to haul out with others would likely be acting on a relatively fine spatial scale. Thus, ideally we would explore the day-to-day variation in the proportion of tagged individuals

hauled out within an SMU. However, the limited sample size and the limited number of windows meant that the lower confidence interval then incorporated 0.

Other sources of information on uncertainty

The survey data provide another opportunity to examine variation in haul out probability. However, most areas are surveyed infrequently (at most annually), and examination of variation in haulout probability is often confounded by the trends in abundance and limited spatial extent of individual surveys (see above regarding seal movements). The Southeast England SMU provides a unique opportunity to examine variation in counts. Surveys are often conducted more than once a year which allows examination of the variation for a given abundance and although there is some interchange with northern UK and also the continent, day-to-day variation in the proportion of seals moving to or from these other regions is assumed to be minimal (see Russell (2016)). The surveys often cover only a proportion of the Southeast England SMU and thus we examined the variation on multiple spatial scales with a focus on the three largest haulout sites (Donna Nook, Wash and Blakeney). These haulouts represented 74% of the SMU in 2019 (Russell et al. BP *seal trends bp), and in ten years there have been two counts (three in 2021) in August. Excluding one of those years (one of the counts was so low, it was assumed they had just been disturbed), the coefficient of variation (CV) of the two (three in 2021) counts in each year ranged from 0.003 to 0.72 (median 0.067). In three of the nine years (including 2021), the confidence intervals surrounding the resulting population estimates from each count did not overlap. In 2021, the confidence intervals from the two later counts overlapped, but not with the first count. Although these haulouts do not represent a closed population, high counts in one haulout area (e.g. Donna Nook) are generally associated with high counts in neighbouring areas indicating that high numbers are not due to local redistribution. Indeed, in 2021, of the two surveys covering five key haulout areas (Donna Nook, The Wash, Blakeney Point, Horsey and Scroby Sands), the count was higher in the second survey for all sites. In 2021, the first Donna Nook count was over 60% higher than the mean of the other two surveys resulting in a population estimate of 20,867 (95% CI: 18,059 – 24,455) compared to estimates from the other two surveys of 12,346 (95% CI: 10,685 – 14,469) and 13,276 (95% CI: 11,490 – 15,559). These findings indicate that there is substantial day-to-day variation in the proportion of the population hauled out, and that the confidence intervals of the scalar generated from telemetry data are not representative of the true variation in the proportion of the population hauled out in a given survey window.

Overall, the estimated proportion of the population hauled out is likely to be a reasonable estimate of the August-wide mean proportion of the population hauled out during survey windows. However, as discussed above the apparent day-to-day variability in haul out probability means the width of the confidence intervals is underestimated. The scalars and associated uncertainty are applied to aerial survey counts to generate population estimates independent from the pup production estimates (hereafter independent estimates). Realistic confidence limits surrounding the scalars is important for robust estimates of population size and trends; the relative CVs surrounding the independent and pup production estimates essentially weight the importance of the estimates (small CV, higher weight). Each independent estimate includes counts for multiple years, with the majority of counts collected within a three-year period. Within these periods, where more than one count is available, the counts are combined to generate a mean count. In areas like the Southeast England SMU where a minimum of three counts are used to provide an averaged count, the confidence intervals around the scalar will be more representative than in areas where a single count is available. An alternative to the three independent estimates would be to scale estimated mean counts from fitted trends (SCOS BP 21/03) to abundance estimates which would incorporate both uncertainty from the scalar and the uncertainty in the mean count prediction. This alternative should be considered and, if appropriate, applied in future years.

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Trends in seal abundance and grey seal pup production

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Abstract

Scotland and eastern England (SMUs 1-9) hold the majority of the UK populations of grey and harbour seals. The key method of monitoring harbour seal trends in these areas are through aerial survey counts during their August moult (covering all areas in a 5-year cycle). Estimates of harbour trends are essential for effective conservation and management. Such estimates based on data up until 2017 were provided in Thompson *et al.* (2019). An update of these estimates is required, especially given the recent low counts in harbour seals in the Southeast England Sea Management Unit (SMU) which had previously been the only large SMU showing a sustained increases in abundance.

For grey seals, population estimates and trends in grey seal abundance are estimated within an age-specific population dynamics model (Thomas 2021) using data from four regions: Inner & Outer Hebrides, Orkney, and North Sea; the first three regions are equivalent to the West Scotland, Western Isles and North Coast & Orkney SMUs respectively, and the North Sea region is an aggregation of East Scotland, Northeast & Southeast England SMUs. The data considered in the population model are pup production estimates from regularly monitored breeding colonies and independent estimates of population size. These independent estimates are generated using grey seal count data from August surveys and a telemetry-derived scalar to account for seals not hauled out during surveys, and are termed independent because they are independent from those derived from pup production data.

The population model provides population estimates on the scale of the four regions and is based on the distribution during the breeding season. It is critical to understand spatial variation in abundance and trends therein, on an SMU scale, during the foraging season (the majority of the year) which is when seals are most likely to be impacted at-sea processes (e.g. anthropogenic activities, prey availability), and also when they are most likely to impact harbour seals. In addition, an analyses of pup production data is required for an understanding of trends for SMUs and trends in the context of SACs while accounting for, and quantifying, the jump in pup production associated with the change in survey methods (film to digital).

Here we fit trends to the available data for the above-described three metrics (harbour and grey seal August counts, and grey seal pup production) by Seal Management Unit (SMU). As well as illustrating these trends, we overlay the relevant counts/production estimates for Special Areas of Conservation (SACs). In addition, we use the August grey seal count data to generate a third independent estimate for use in the population model (Thomas *et al.* 2021), as well as adjusting the second and third estimate (using an updated scalar; Russell & Carter 2021). Finally, we combine trends in August counts of grey seals across SMUs to provide a single trend in counts for SMUs 1-9, and highlight the potential future utility of such a prediction.

The results in this BP are a preliminary extension of the analyses currently being conducted for the upcoming OSPAR Assessment.

Introduction

Scotland and eastern England (SMUs 1-9) hold the majority of the UK populations of grey and harbour seals. The key method of monitoring harbour seal trends in these areas are through aerial survey counts during their August moult (covering all areas in a 5-year cycle). Estimates of harbour seal trends are essential for effective conservation and management. Such estimates based on data up until 2017 were provided in Thompson *et al.* (2019). An update of these estimates is required, especially given the recent low counts in harbour seals in the Southeast England Sea Management Unit (SMU) which had previously been the only large SMU showing sustained increases in abundance.

For grey seals, population estimates and trends in abundance are estimated within an age-specific population dynamics model (Thomas *et al.* 2021) using pup production data from four regions: Inner & Outer Hebrides, Orkney, and North Sea; the first three regions are equivalent to the West Scotland, Western Isles and North Coast & Orkney SMUs respectively, and the North Sea region is an aggregation of East Scotland, Northeast & Southeast England SMUs. The data considered in the population model are pup production estimates from regularly monitored breeding colonies and independent estimates of population size. These independent estimates are generated using grey seal count data from August surveys and a telemetry-derived scalar to account for seals not hauled out during surveys. They are termed independent because they are independent from those derived from pup production data.

The population model provides population estimates on the scale of the four regions, and is based on the distribution during the breeding season. It is critical to understand spatial variation in abundance and trends on an SMU scale, during the foraging season (the majority of the year) which is when seals are most likely to be impacted by at-sea processes (e.g. anthropogenic activities, prey availability), and also when they are most likely to have an effect on harbour seals. In addition, an analysis of pup production data that accounts for, and quantifies, the jump in pup production associated with the change in survey methods (film to digital), is required for an understanding of trends for SMUs and trends in SACs.

Methods

August surveys

All data were based on counts made during the annual harbour seal moult in August (2 hours either side of low tide). Almost all data are from aerial surveys conducted by SMRU, augmented by data from fixed wing aerial surveys of the Thames estuary, conducted by Zoological Society London (aerial survey; Cox *et al.* 2020; SCOS-BP 21/07) and ground surveys in the Tees estuary, conducted by Industry Nature Conservation Association (Bond 2020). Surveys of rocky shores were conducted by helicopter using a thermal imaging camera whereas surveys of sandbanks (much of the UK east coast) were predominantly conducted by fixed-wing aircraft. For details on survey methods, refer to Thompson *et al.* (2019). Where possible, entire SMUs were surveyed synoptically (i.e. within a single August survey season). However, in some cases that was not possible and so counts had to be combined across multiple years; the resulting count was assigned to the year that encompassed the majority of the total (focal year). Furthermore, some areas, particularly the offshore islands (e.g. North Rona and Sula Sgeir) which grey seals haul out on, are surveyed less frequently and thus their associated counts are used in multiple years (trend analyses) and multiple independent estimates.

For the trend analyses, where the limited number of years with counts prohibited robust model fitting for a particular SMU, the largest subset of sites within it (i.e. the subset of haulout sites with the largest proportion of the SMU total), for which the monitoring was frequent enough to allow

Table 1. For each SMU (and any associated subsets/proxies) the latest count/pup production estimates (year; percentage of the SMU for proxies) are shown. Note that non-named subsets may not be consistent areas between metrics.

SMU number	Name	Harbour seal August counts	Grey seal August counts	Grey seal pup production
1	Southwest Scotland	1709 (2018)	517 (2018)	No known colonies
2	West Scotland	15600 (2018)	4174 (2018)	c. 5030
	West Scotland subset	NA	NA	4455 (2019; 87%)
3	Western Isles	3532 (2017)	4038 (2011)	c. 16400
	Western Isles subset	NA	5478 (2017; 93% in 2011)	16083 (2019; 98%)
4	North Coast & Orkney	1405 (2019)	8599 (2019)	22944 (2019)
	North Coast & Orkney subset	NA	NA	22153 (2019; 97%)
5	Shetland	3180 (2019)	1009 (2019)	761
	Shetland subset	NA	NA	495 (2018; peak count; 56%)
6	Moray Firth	1077 (2019)	2513 (2019)	1865 (2019)
	Moray Firth subset: Loch Fleet to Findhorn	1008 (2019; 94% of SMU)	NA	NA
7	East Scotland	356 (2016)	3782 (2016)	7261* (2019)
	East Scotland subset: Firth of Tay and Eden Estuary SAC	37 (2020; 14% of SMU in 2016)	NA	NA
8	Northeast England	79 (2019)	4660 (2020)	2823* (2019)
	Northeast England subset	76 (2019; 96% of SMU)	NA	NA
9	Southeast England	3752 (2019)	8667 (2019)	7902 (2019)
	Southeast England subset: The Wash and North Norfolk Coast SAC	2859, 2626, 3058 (2021; 73% of SMU in 2019)	6605, 4176, 4787 (2021; 74% in 2019)	NA

* Excludes between 10 and 20 pups estimated to be born at other sites within the SMU.

model fitting, was used as a proxy for the SMU. Table 1 reports the latest count for each SMU and subset. For some SMUs, trends for the whole SMU and a proxy were fitted (if the proxy represented a higher sample size). The relationship between the SMU and subset counts in years when the whole area was surveyed can be used to assess how representative the subset trends are of the regional trends.

Analyses

All analyses were conducted in R (R Core Team 2021).

Harbour seal analyses

Counts were modelled as a function of year assuming negative binomial errors broadly following methods described in Thompson *et al.* 2019. Updated counts were available for all but one SMU (Western Isles; Table 1). For some SMUs, the limited number of data points resulted in problems estimating the theta parameter for the negative binomial distribution. In these cases, a Poisson distribution was assumed. Please note that, in contrast to Lonergan *et al.* (2013) and Thompson *et al.* (2019), AIC rather than AICc was used for model selection. For all datasets, at least three models were fitted: an intercept-only GLM (null model; i.e. a stable trend), an exponential (linear on the link scale) year effect within a GLM, and a nonlinear smooth year effect within a GAM (restricted to 5 knots).

Phocine Distemper Virus (PDV) caused sudden declines in the Northeast and Southeast England SMUs in 1988 and 2002, and thus additional models were fitted with a step change in abundance and/or trends associated with 2002 (PDV epidemic; data were not available for the entire SMUs prior to the 1988 PDV epidemic). Although the declines in north and east Scotland SMUs were not thought to be due to PDV, there were sudden drops or declines in Shetland and North Coast & Orkney SMUs during multi-year gaps in surveys that spanned 2002 and a sudden change in trend around 2002 in East Scotland SMU. Because of the unknown nature of these declines, additional models were also fitted for these SMUs. Specifically, additional models were fitted for SMUs 4 – 9 that allowed any combination of stable/exponential trends prior to and following 2002 (including the same trend across the time-series) and with/out a step change associated with 2002. In some SMUs there was evidence of a non-linear trend in the final period (2002 onwards), thus for this final period GAMs (smooth trends) were used, if preferred by AIC.

Grey seal analyses (August counts)

Changes in grey seal August counts were examined at two temporal and spatial scales. The coarse scale refers to the independent population estimates (Russell *et al.* 2016; Table 2) for Scotland and eastern England (SMUs 1-9). The underlying counts are surveys conducted over multiple years; most of the counts are from a block of three years (survey block), with the population estimate assigned to the middle year (focal year). For the 2008 population estimate, 97% of seals were counted between 2007 and 2009; the remaining 3% were counted in 2005 and 2006. For 2014 estimate, 93% were counted between 2013 and 2015; with the remaining 7% counted in 2011 and 2016. Here we generated counts for the third independent (2017) estimate using the same protocol as for the first two (Table 1 in Russell *et al.* 2016). The focal year was 2017; 96 % were counted within the three-year survey block (2016-2018), with 1% from 2014 and 3% from 2019. Where multiple surveys were conducted within the survey block (e.g. Southeast England), the means of these counts are used to minimise day-to-day variation in counts (see Russell and Carter 2021). The updated scalar (SCOS-BP 21/02), based on the telemetry-derived estimates of the proportion of the population hauled out during survey windows, was used to generate estimations of population size from the three counts.

On the scale of individual year, counts for each SMU were modelled as a function of year assuming negative binomial errors within a single GAM. The models allowed a different temporal trend for each SMU. A combined trend was predicted (with confidence intervals) using parametric bootstrapping.

Grey seals - pup production

Pup production estimates (SCOS-BP 21/01) used for SMUs 2-4 and 6-7 (see Russell *et al.* 2019) were almost entirely derived from aerial survey data; these were estimated using probabilities of correctly classifying a moulted pup (PMoult) values of 0.5 and 0.9 for film and digital surveys, respectively; all other parameters were kept constant throughout the time series and as reported in Russell *et al.* (2019). Some counts of Inchkeith, East Scotland, were provided by Fife Seal Group. The values used for the remaining SMUs (5, 8 – 9) were based on ground counts: Shetland (peak counts; NatureScot), Northeast England (production; National Trust) and Southeast England (production; National Trust, Lincs Wildlife Trust, and Friends of Horsey Seals). Note there are no known established breeding colonies in the Southwest Scotland SMU. The latest pup production estimates for each SMU and any proxies are reported in Table 1.

The production estimates used here as proxies for West Scotland, Western Isles and North Coast & Orkney match those used in the population model (regularly monitored colonies in Inner Hebrides, Outer Hebrides, and Orkney, respectively), and represent c. 87, 98 and 97% of production in those SMUs (Table 1). The estimates for East Scotland, Northeast England and Southeast England sum to the totals used for the North Sea region in SCOS-BP 21/05. Shetland and Moray Firth SMU data are not incorporated in the population model.

Pup production (peak count for Shetland; SCOS-BP 21/01) was modelled as a function of year assuming negative binomial errors (see Russell *et al.* 2019 for details). For Scottish SMUs surveyed by SMRU (all except Shetland), a step increase in pup abundance was offered between 2010 (the last film survey) and 2012 (the first digital survey) to account for any artificial increase in pups associated with the change in aerial survey method, thus allowing the trends to be examined excluding this jump. To maximise the data available to fit this jump, all applicable SMUs were modelled within a single GAM (limited to $k=5$), allowing a different temporal trend for each SMU but a single adjustment for the change in survey methods.

For SMUs where the data were derived from ground surveys, three models were fitted: an intercept-only GLM (null model), an exponential (linear on the link scale) year effect within a GLM, and a nonlinear smooth year effect within a GAM (restricted to $K=5$).

Limited flexibility for the smooths represented a pragmatic approach aimed to estimate trends on the appropriate temporal scale. For consistency the same approach was used across SMUs; occasionally this resulted in a potentially suboptimal fit for periods of time (i.e. Moray Firth; Fig 6).

Results & Discussion

Harbour seals

There are a number of key differences compared to the results of Thompson *et al.* (2019). An increasing trend (Fig 1a – 3a) was fitted to the three western SMUs (stable trend in Thompson *et al.* 2019). There was one additional data point for Southwest Scotland and West Scotland but for Western Isles the change was driven solely by the change in selection method (AIC vs AICc). The estimated trend for the Western Isles (Fig 3a) shows a decline to c. 2005 followed by an increase. The data points for the Sound of Barra SAC (not included in Thompson *et al.* (2019) because harbour

seals are not a primary feature) indicates that the abundance in the SAC is depleted (compared to in the 1990s). The component areas of West Scotland SMU show the same trend as in Thompson *et al.* (2019): a stable trend in the southern area of the SMU (Fig 2ii a) but increases in the central (Fig 2iii a) and north (no additional data; Fig 2iv a) areas. Although trends in SACs were not fitted here, the additional data point for the SACs in the southern area (Fig 2ii a; Eileanan agus Sgeiran Lios mor and Southeast Islay SACs) did not suggest a deviation from the stable trend reported for the SACs in Thompson *et al.* (2019). There were no additional data for the Ascrib, Isay and Dunvegan SAC in the central region; in contrast to the area as whole, this SAC shows a stable trend (Thompson *et al.* 2019). For the northern SMUs there was an additional data point from a survey in 2019 but no discernible change in the estimated trends compared to Thompson *et al.* (2019). For North Coast & Orkney (Fig 4a), counts were stable until 2001; the next count in 2006 was c. 45% lower and counts have been declining ever since. However, the most recent count (2019) for the North Coast and Orkney was higher (1405) than the previous count in 2016 (1349) and thus the decline may be slowing. The 2019 count for the Sanday SAC was also slightly higher than in 2016; the Sanday SAC showed a similar trend to the SMU (Thompson *et al.* 2019). Shetland shows a stable trend either side of a drop of c.40% between 2001 and 2005 (Fig 5a) The 2019 counts for the Shetland SACs appear to follow the trends estimated in Thompson *et al.* (2019): stable and declining for Yell Sound Coast and Mousa SACs, respectively.

The trends in the Moray Firth SMU (Fig 6a; represented by Loch Fleet to Findhorn - c. 94% of harbour seals in the SMU), which included two additional counts, were similar to that fitted in Thompson *et al.* (2019) but a GAM, rather than a decline to 2002 and stable thereafter, was preferred by model selection. A declining trend was fitted in Thompson *et al.* (2019) for the Dornoch Firth and Morrich More SAC and given the high variability of counts around the trend, the two most recent counts are not contradictory to that trend.

There are only five counts available for East Scotland SMU (Fig 7a) as a whole (and no additional counts since Thompson *et al.* 2019). However, given the decreased suitability of the Firth of Tay and Eden SAC as a proxy for the SMU (14% of the count in 2016 compared to > 90% in the early 2000s) a trend was fitted to both the SMU and the SAC. Although there is evidence of a declining trend, it is clear that there has been a redistribution within the SMU with the catastrophic declines (95% since 2002) restricted to the SAC. A GAM was preferred for the SAC (compared to stable until 2002 and a decline thereafter in Thompson *et al.* 2019). Indeed, there is evidence that the decline may be slowing.

The eastern England SMUs represent the only SMUs which have shown sustained increases in abundance (punctuated by PDV-mediated declines in 1988 and 2002; Thompson *et al.* 2019). Northeast England (Fig 8a) hosts a small number of harbour seals (max count < 100 seals) and the last two counts (2018 and 2019) are c. 14% lower than the three previous counts (highest of the time series). Counts in harbour seals in The Wash and North Norfolk SAC (c. 75% of harbour seals in the SMU) are around 1,000 seals (c. 25%) lower in recent years (five counts; 2019 – 2021) compared to the mean in the previous five years. It is unclear whether this drop represents a step change or the beginning of a rapid decline. See SCOS-BP 21/06 for more detailed examination of the data and associated Discussion.

Grey seal August counts

Independent Estimates (Table 2, Fig 10)

The updated scalar resulted in slightly reduced mean population estimates for 2008 (96,028 compared to 101,196) and 2014 (138,437 compared to 145,889; Russell *et al.* 2016; Table 2). The

total count and population estimate for 2017 was 40,347 and 169,060, respectively, representing a 16% increase compared to 2014.

These three independent estimates are input into the population model (Thomas 2021) after downscaling to make them comparable with the pup production estimates for the appropriate year; only production estimates from the regularly monitored colonies in SMUs 1-9 are incorporated in the population model (c. 92.33, 93.37 and 93.33% for the three independent estimate blocks respectively).

Grey seal August trends

Grey seal trends in August were estimated to be stable in three SMUs (Western Isles, Shetland and East Scotland) and increasing in the other six considered here (Southwest Scotland, West Scotland, North Coast & Orkney, Moray Firth, Northeast England and Southeast England).

In two of the SMUs for which a stable trend was selected, Western Isles (Fig 3b) and East Scotland (Fig 7b), the most recent count is the highest of the time series. There were limited data to fit a robust trend in East Scotland ($n=5$), and for the Western Isles the counts are variable with two periods of increasing counts. Thus Shetland (Fig 5b) is the only SMU for which there is a real possibility of recent declines; an exceptionally low count at the start of the time series precludes the fitting of a robust trend to current data.

Slight increasing trends (with considerable uncertainty) were estimated for West Scotland and its component areas (Fig 2b; only the subareas were included in the combine across SMU trend) as well as North Coast & Orkney (Fig 4b). There was considerable uncertainty around the trend for Northeast England (Fig 8b), indeed it is not clear whether or not the last three counts represent a step increase in abundance or a continuing trend. For Southeast England SMU, the trend was fitted to the three of the five largest haulouts (Donna Nook, The Wash and Blakeney Point; c. 74% of the grey seal abundance in the SMU; Fig 9b). These three haulouts represent the most comprehensive time-series but there are indications that Donna Nook (Humber Estuary SAC) is now in decline (Fig 9b; Thompson and Russell 2021). The more recent popular haulout sites are likely to show different trajectories; data is lacking for Horsey but numbers at Sroby Sands are rapidly increasing (Thompson and Russell 2021).

Some grey seal SACs are designated on the basis of their breeding colonies and have relatively low numbers in August and thus patterns in the August counts are not examined: Treshnish Isles SAC (West Scotland), North Rona (Western Isles), Isle of May (East Scotland). Counts for Faray & Holm of Faray SAC (North Coast & Orkney) have been variable around an average of 375 with no discernable temporal pattern. The remaining SACs (Monachs SACs, the Farnes component of the Berwickshire & North Northumberland Coast SAC, and the Humber Estuary SAC) have significant numbers during both August and breeding. There is no indication of a pattern in the counts for the Monach Islands SAC (Fig 3b; average around 1500; range 1250 - 1991) but the last count was considerably higher (2701). The Farnes component of the Berwickshire and North Northumberland SAC (Fig 8b) used to be the whole count for the Northeast England SAC, it still accounts for >90% and thus the trends will mirror those of the SMU. As mentioned above the Humber Estuary SAC (Fig 9b; Donna Nook) comprises a decreasing proportion of the Southeast England SMU.

Table 2. The three independent estimates for grey seal population (SMUs 1-9) and associated counts.

Region	SMUs	Count			Population estimate (and 95% CIs)		
		2007- 2009	2013- 2015	2016- 2018	2008	2014	2017
Western Isles	3	3,808	4,065	5,773	15,141 (13,099 - 17,753)	16,163 (13,983 - 18,951)	22,954 (19,859 - 26,914)
Western Scotland	1 & 2	2,773	5,438	4,691	11,026 (9,539 - 12,928)	21,622 (18,707 - 25,352)	18,652 (16,137 - 21,869)
North Coast, Orkney & Shetland	4 & 5	10,061	9,664	10,723	40,004 (34,610 - 46,904)	38,425 (33,244 - 45,054)	42,636 (36,887 - 49,991)
North Sea	6 - 9	7,509	15,650	19,160	29,857 (25,831 - 35,007)	62,227 (53,836 - 72,960)	76,183 (65,910 - 89,324)
Surveyed regions		24,151	34,817	40,347	96,028 (83,079 - 112,592)	138,437 (119,770 - 162,317)	160,425 (138,793 - 188,098)

The combined trend across all SMUs (Fig 10) indicates that although the trend is increasing, the increase may be slowing. In contrast to this trend, the counts underlying the independent estimate comprise counts for all haulouts. It would thus be expected that the points would be above the trend line (Fig 10). These counts do not follow the same trajectory as the estimated trend. This may be because the proxies used to fit the SMU trends (Table 1) were based on the availability of a time series of data, and thus did not include some of the more recently established haulout sites which be increasing at a faster rate than the SMU-wide trends (as in the case in Southeast England). Thus, further work is required to determine to what degree the apparent slowing may be an artifact of the proxies used to fit the SMU-wide trends (particularly in Southeast England). However, following such investigation, if it is possible to upscale from proxies to SMU-wide trends, the combined trend could be used as the basis of a time series of independent estimates of population size for the population model. Such an approach would have a number of advantages over the current method which relies on scaling up raw counts (see above). It would increase the amount of data available to the population model (time-series vs three independent estimates); essentially decrease the influence of the day-to-day variability in counts (see Russell & Carter 2021); negate the need for counts over multiple year (3-year survey blocks) to be assigned to a single focal year (potentially masking changes in these years); and the uncertainty around the trends could be propagated into the estimates of population size.

Grey seals pups

The final model estimating trends in grey seal pup production for aerially surveyed SMUs included an estimated of 27 % jump (95% CI: 16.7 – 37.5) in pup production associated with the change from film to digital (delta AIC of -30 compared to a model within the jump). The plots show the pup production trends (and associated confidence intervals) for each SMU if no jump had occurred; in essence, once the jump has been taken into account, the estimates based on both the film and digital surveys are used to fit the trends. The dashed line through the digital surveys shows the same trend but at the higher level of the estimates associated with the digital surveys. For the SMUs which comprise ground-counted colonies, a GAM was selected for Northeast and Southeast England, and a GLM with a declining trend for Shetland.

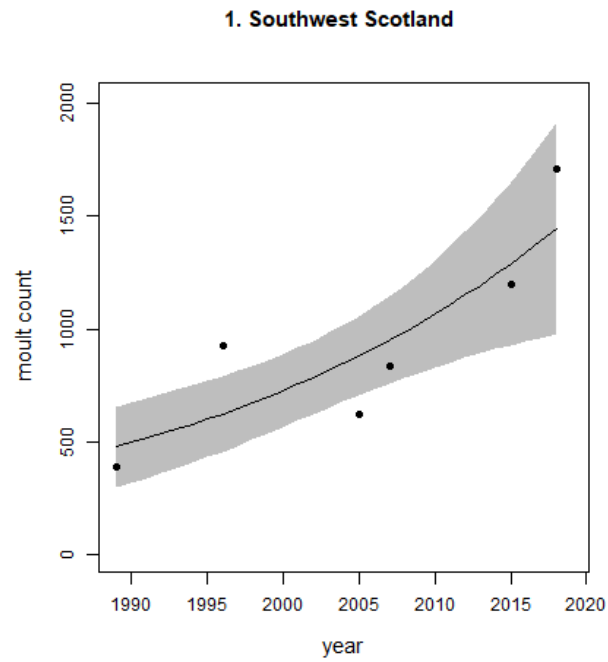
Although pup production had levelled off in West Scotland (early to mid-1990s; Fig 2i c) and Western Isles (mid 1990s; Fig 3c) (Russell *et al.* 2019, the 2016 and 2019 estimates were higher than the first two digital survey estimates (2012 and 2014), which for the Western Isles has resulted in a slight recent increase in the mean predicted trend. This apparent increase is reflected in the Monach Islands SAC which accounts for > 75% of the SMU pup production. In contrast, pup production in North Rona is continuing to decline. In the North Coast & Orkney SMU (Fig 4c), pup production has remained stable since around 2000. The Faray & Holm of Faray SACs indicate that the colony may be in decline. A declining trend was fitted for Shetland (Fig 5c); however, the time-series comprised a subset of colonies and was based on peak counts (which are sensitive to effort, i.e., number and timing of counts) and thus there are doubts as to how robustly these trends represent Shetland as a whole. The Moray Firth SMU (Fig 6c) shows indication that pup production is increasing though it should be noted that there is a limited temporal extent to the data and pup production within this SMU is difficult to accurately estimate. The East Scotland SMU (Fig 7c) is continuing to increase rapidly (mean estimate of c. 28 % between 2014 and 2019), but the two SACs which represent the vast majority of production in the SMU show differing patterns in abundance. The Isle of May SAC, which essentially held the SMUs pup production until the mid 1995s looks to be stable or potentially reduced. In contrast the Fast Castle colony, Berwickshire & North Northumberland Coast SAC, is showing rapidly increasing pup production. Note that although the SAC boundary transects the Fast Castle colony, here all pup production is assigned to the SAC. Pup production in Northeast England, which is entirely encompassed by the Farne Islands component of the Berwickshire & North Northumberland Coast SAC, is also increasing rapidly (mean estimated increase of 53% between

2014 and 2019). Finally, pup production within the Southeast England SMU is continuing to increase exponentially (mean estimate c. 75% between 2014 and 2019) but this is in a large part due to increases in Blakeney Point and Horsey, while the increase at Donna Nook (Humber Estuary SAC) which, up until c. 2000 accounted for the SMUs pup production is now slowing, and thus represents a decreasing proportion of the SMU's pup production.

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(a)



(b)

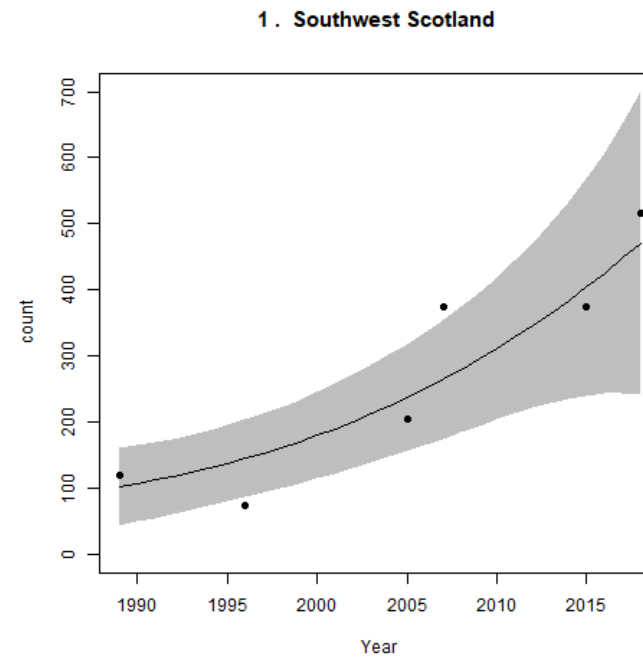


Figure 1. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts in the Southwest Scotland SMU. The filled circles are the values used to fit the trends.

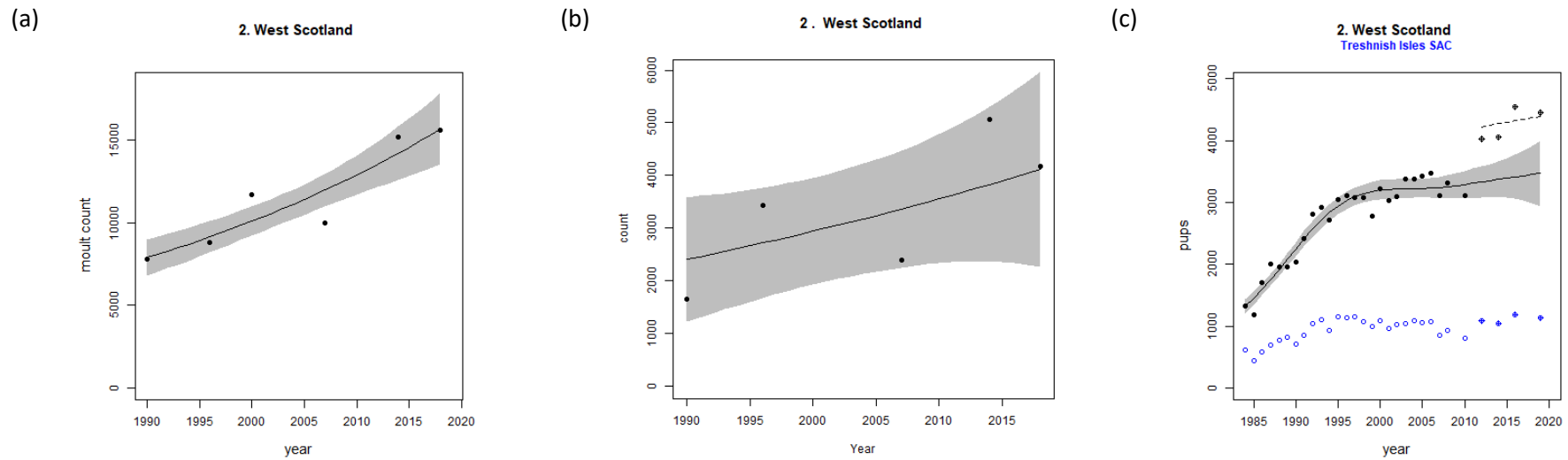
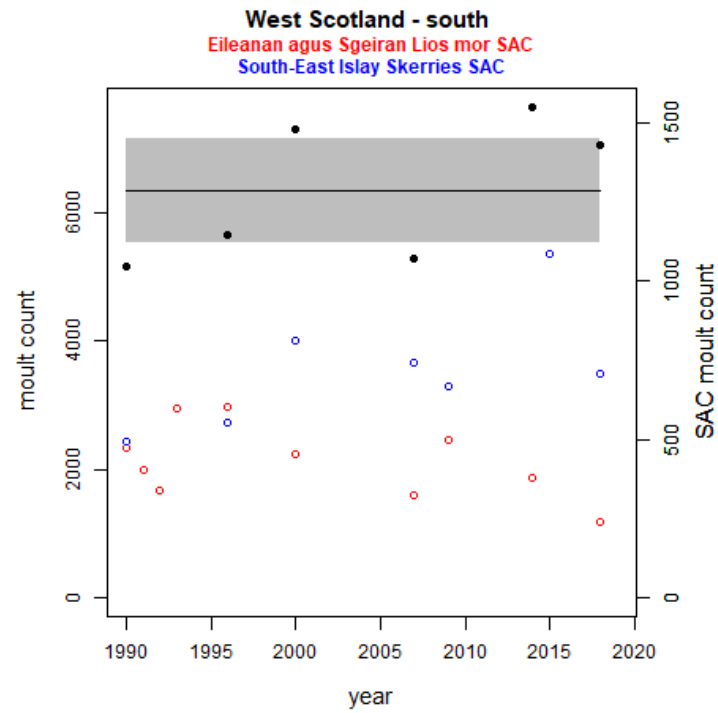


Figure 2i. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts, and grey seal pup production (c) in the West Scotland SMU. The filled (and crossed; c) black circles are the counts used to fit the SMU trends. The dashed line in (c) shows the same trend as the solid line but at the level of pup production predicted for digital survey estimate (crossed circles). The open and crossed coloured circles (c) indicate the SAC estimates for the film and digital surveys, respectively.

(a)



(b)

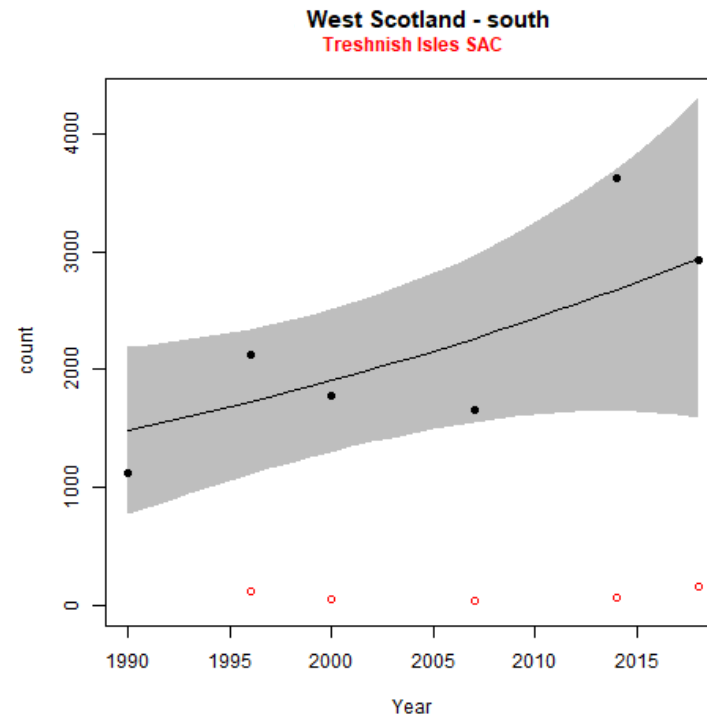
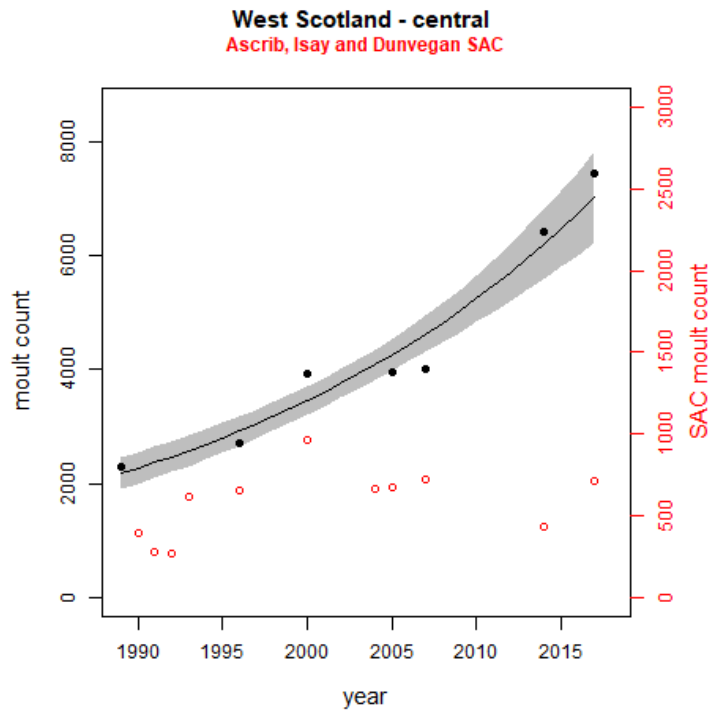


Figure 2ii. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts in the southern area of the West Scotland SMU. The filled black circles are the values used to fit the SMU trends. The open coloured circles indicate the SAC counts.

(a)



(b)

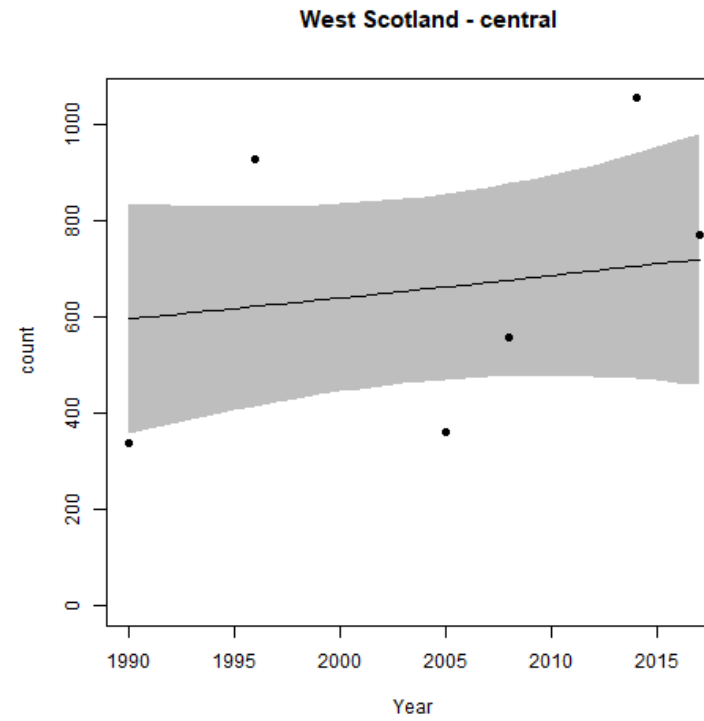
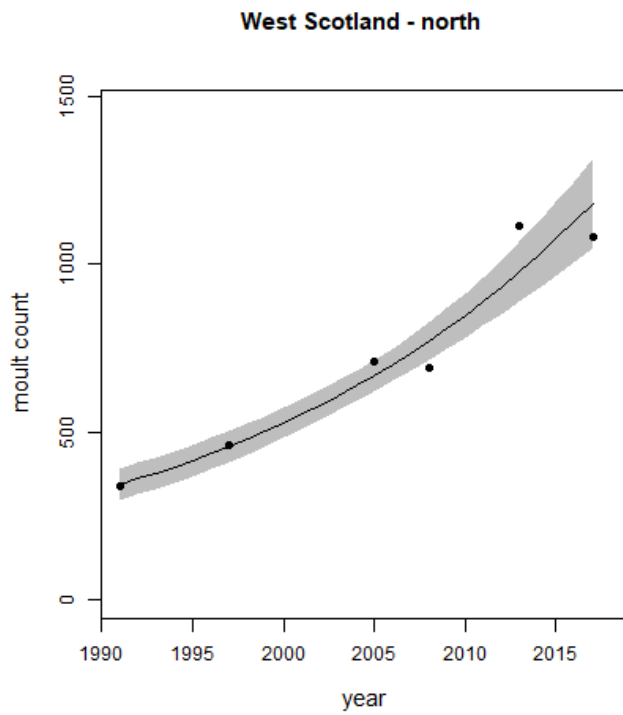


Figure 2iii. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts in the central area of the West Scotland SMU. The filled black circles are the values used to fit the SMU trends. The open circles indicate the SAC counts. The open coloured circles indicate the SAC counts. Note the different axes for the SAC in (a).

(a)



(b)

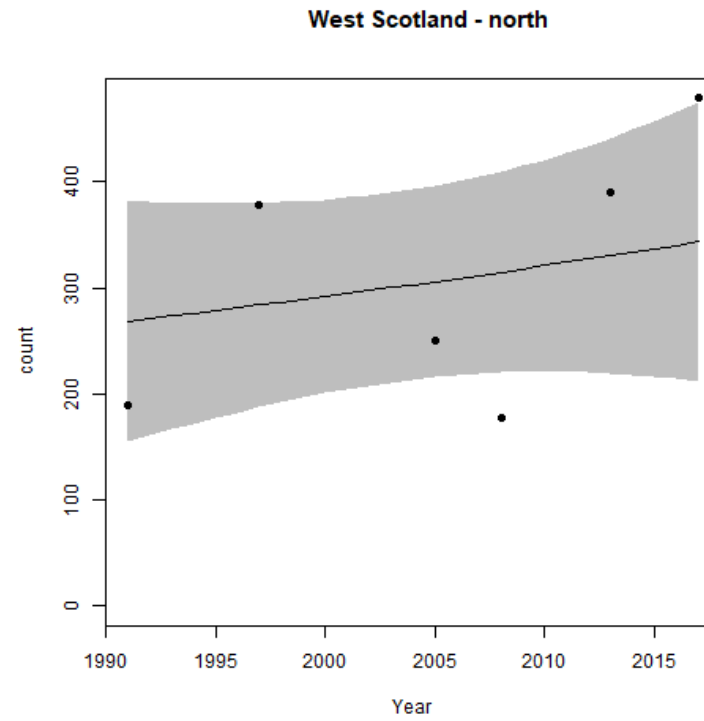


Figure2iv. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts in the southern area of the West Scotland SMU. The filled black circles are the values used to fit the SMU trends.

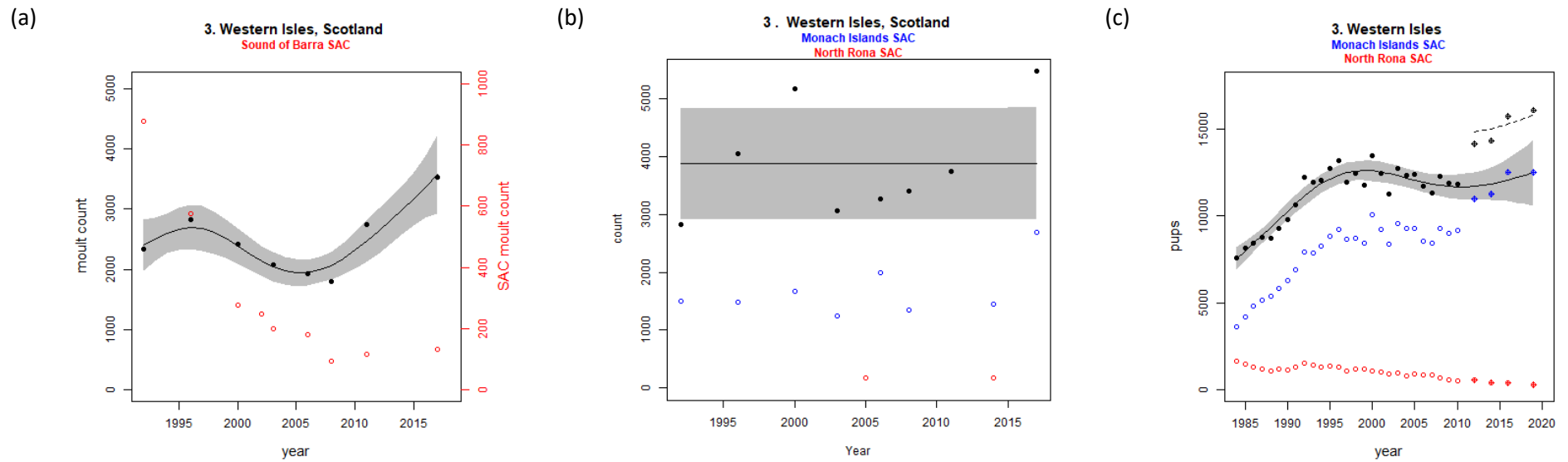


Figure 3. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts, and grey seal pup production (c) in the Western Isles SMU. The closed (and crossed; c) black points are the values used to fit the SMU trends. The dashed line in (c) shows the same trend as the solid line but at the level of pup production predicted for digital survey estimate (crossed circles). The open coloured circles in (a) and (b) indicate the SAC counts. The open and crossed coloured circles (c) indicate the SAC estimates for the film and digital surveys, respectively. Note the different axes for the SAC in (a).

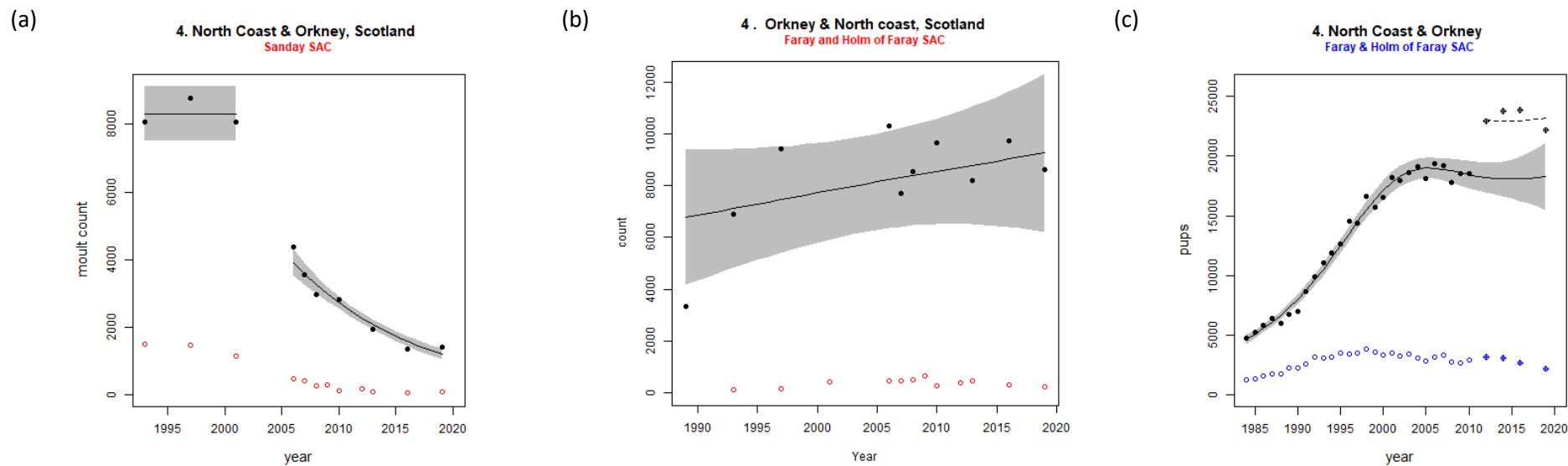


Figure 4. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts, and grey seal pup production (c) in the North Coast & Orkney SMU. The dashed line in (c) shows the same trend as the solid line but at the level of pup production predicted for digital survey estimate (crossed circles). The filled (and crossed; c) black circles are the values used to fit the SMU trends. The open coloured circles in (a) and (b) indicate the SAC counts. The open and crossed coloured circles (c) indicate the SAC estimates for the film and digital surveys, respectively.

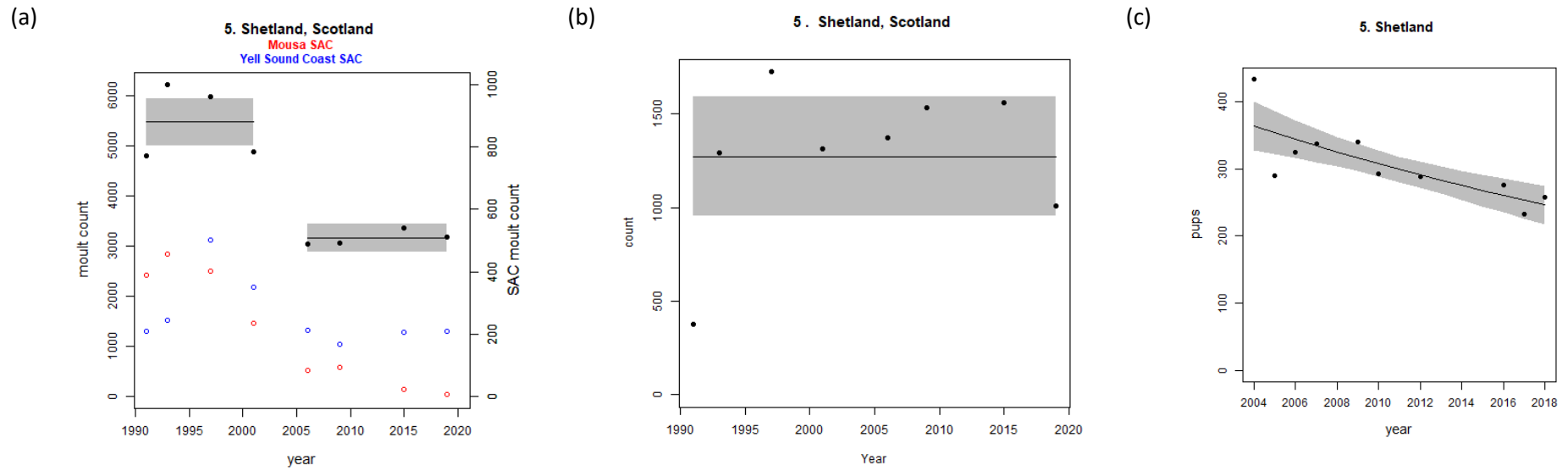


Figure 5. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts, and grey seal peak pup numbers (c) in the Shetland SMU. The filled black circles are the values used to fit the SMU trends. The open coloured circles indicate the SAC counts. Note the different axes for the SACs (a).

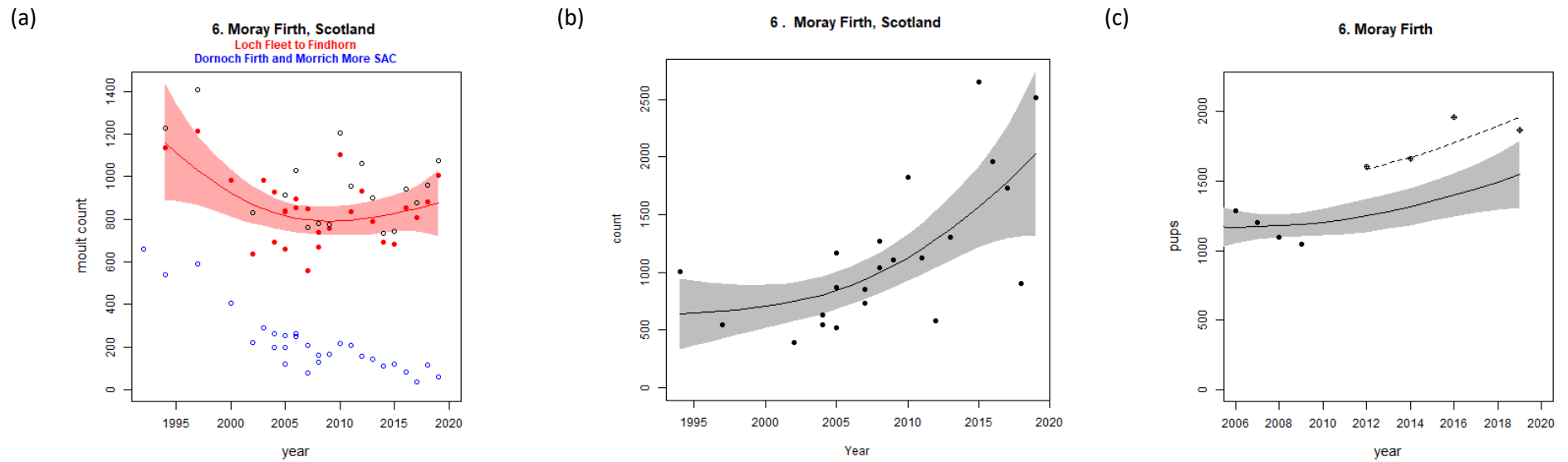


Figure 6. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts, and grey seal pup production (c) in the Moray Firth SMU (subset for a). The open black circles (a) illustrate the SMU-wide counts and were not used for model fitting. The filled (and crossed; c) points are the values used to fit the trends. The dashed line in (c) shows the same trend as the solid line but at the level of pup production predicted for digital survey estimate (crossed circles).

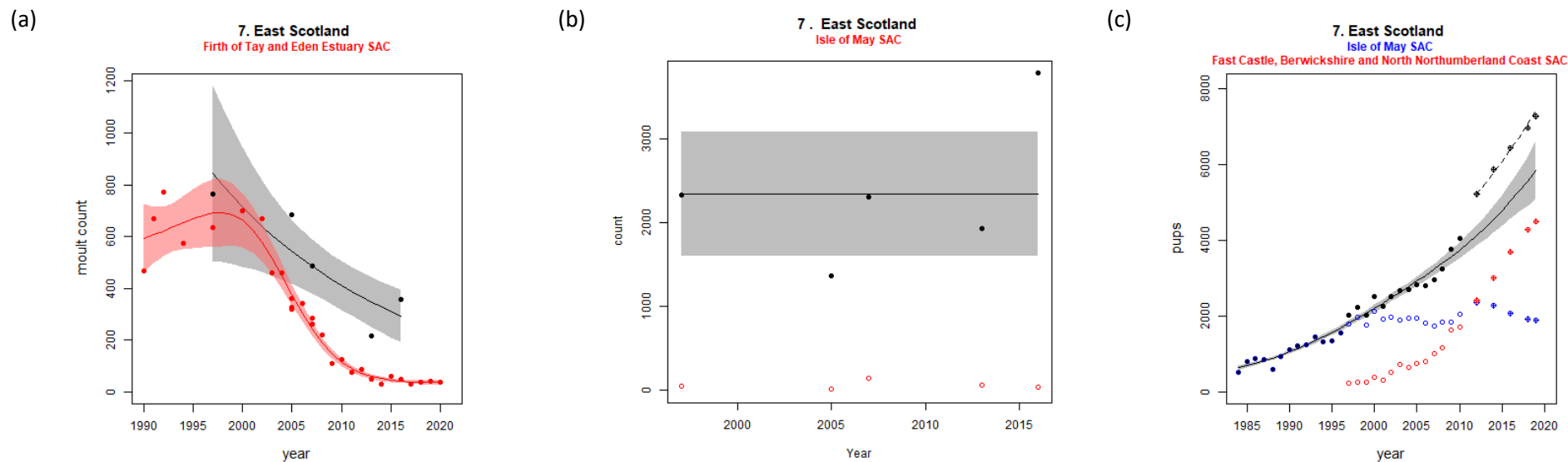


Figure 7. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts, and grey seal pup production (c) in the East Scotland SMU (and the SAC in a). The filled (and crossed; c) black and red points are the values used to fit the trends (b). The dashed line in (c) shows the same trend as the solid line but at the level of pup production predicted for digital survey estimate (crossed circles). The open coloured circles in (b) indicate the SAC counts. The open and crossed coloured circles (c) indicate the SAC estimates for the film and digital surveys, respectively.

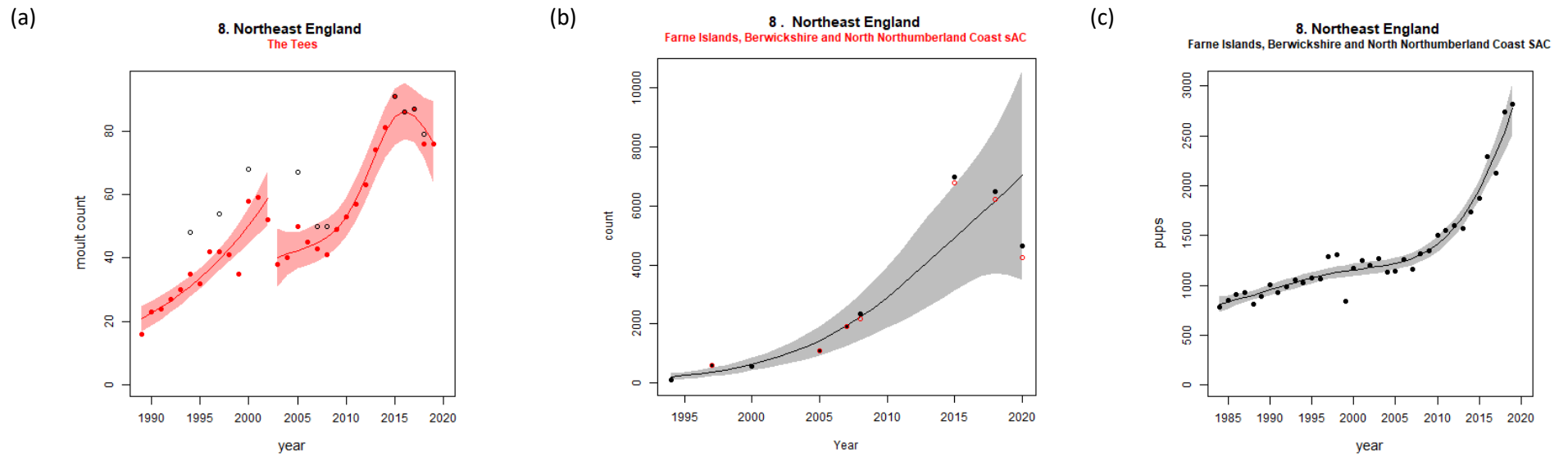


Figure 8. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts, and grey seal pup production (c) in the Northeast England SMU (subset for a). The filled black and red circles are the values used to fit the trends. The open black circles (a) illustrate the SMU-wide counts and were not used for model fitting. The open red circles (b) illustrate the SAC counts.

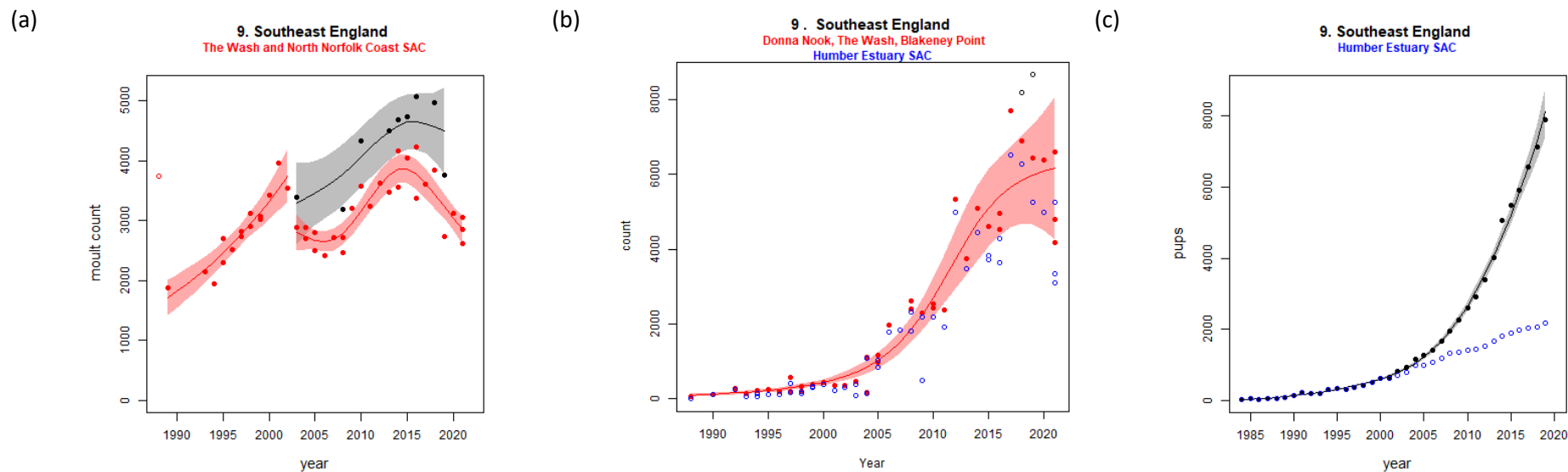


Figure 9. The predicted trend and associated 95% confidence intervals for harbour (a) and grey (b) seal August counts, and grey seal pup production (c) in the Southeast England SMU (and SAC in (a); subset only in (b)). The filled black and red circles are the values used to fit the trends. In (a) the open red circle indicates the single pre-1988 epidemic count (not used for model fitting). The open black circles (b) indicate the SMRU-wide counts and were not used for model fitting. The open blue circles are the counts (b) and production estimates (c) for the grey seal SAC.

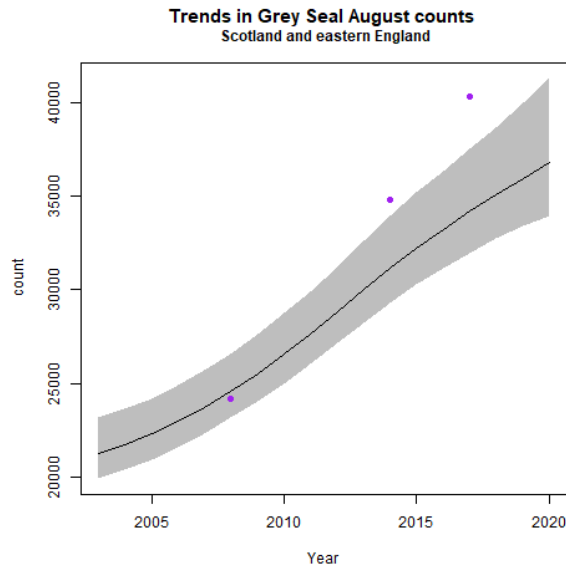


Figure 10. The predicted trend and associated 95% confidence intervals for grey seal August counts for SMUs 1-9. Note that as proxies were used to fit the trend in some SMUs, the predictions do not represent predictions of total counts across the SMUs. In contrast the purple circles, which represent the three counts underlying the independent estimates, are for the entire study area (SMUs 1-9).

Annual review of priors for grey seal population model 2021

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Summary

Prior distributions (Table 1) for the grey seal population model (SCOS-BP 21/05) are required for the following model parameters: adult female survival ϕ_a , maximum pup survival ϕ_{pmax} , fecundity α , shape of density dependence acting on pup survival ρ , region-specific carrying capacity (in terms of pup production) χ_{1-4} , number of adults per female ω , and precision of the pup production estimates ψ . The data used to inform these priors are presented below and in Tables 2 and 3. The resulting prior distributions are shown in Figure 1 and Table 1. These distributions are identical to those used in the previous year's analysis (SCOS-BP 21/05). Further discussion of previous and current prior selection is given in Lonergan (2012; 2014), and Russell (2017). Recent data, and any implications for the current priors, are highlighted. For study sites for which there are multiple estimates for a parameter, only the most comprehensive study is presented. This briefing paper is based on Supporting Information in Thomas et al. (2019).

Table 1. Prior parameter distributions input in Thomas (2021 SCOS-BP 21/05). Be and Ga denote beta and gamma distributions, respectively. Carrying capacity subscripts 1 to 4 refer to North Sea, Inner Hebrides, Outer Hebrides and Orkney regions.

Parameter	Prior distribution	Prior mean (SD)
adult survival ϕ_a	0.8+0.18*Be(1.79,1.53)	0.90 (0.04)
pup survival ϕ_{pmax}	Be(2.87,1.78)	0.62 (0.20)
fecundity α	0.6+0.4*Be(2,1.5)	0.83 (0.09)
dens. dep. shape ρ	Ga(4,2.5)	10 (5)
carrying capacity χ_1	Ga(4,5000)	20000 (10000)
carrying capacity χ_2	Ga(4,1250)	5000 (2500)
carrying capacity χ_3	Ga(4,3750)	15000 (7500)
carrying capacity χ_4	Ga(4,10000)	40000 (20000)
observation precision ψ	Ga(2.1,66.67)	140 (96.61)
sex ratio ω	1.6+Ga(28.08, 3.70E-3)	1.7 (0.02)

Parameters

Adult female survival ϕ_a

Relevant studies are summarized in Table 2. Estimates of annual adult survival in the UK, obtained by aging teeth from shot animals are between 0.935 and 0.96 (Harwood & Prime, 1978; Hewer, 1964; Lonergan, 2012). Capture-mark-recapture (CMR) of adult females on breeding colonies can be used to estimate female survival but may produce underestimates as they are dependent on the assumption that females not returning to the study colony have died. Using capture-mark-recapture (CMR), adult survival was estimated to be between 0.87 and 0.95 (Smout, King & Pomeroy, 2019; see Table 2 for more details). Based on the above data, and the fact that the lower limit on adult survival cannot be lower than 0.8 (Lonergan, 2012), the prior on adult female survival was specified to allow non-zero probability density only between 0.8 and 0.97 (Thomas 2018). However, recent estimates from Sable Island suggest adult female survival may be above this upper bound.

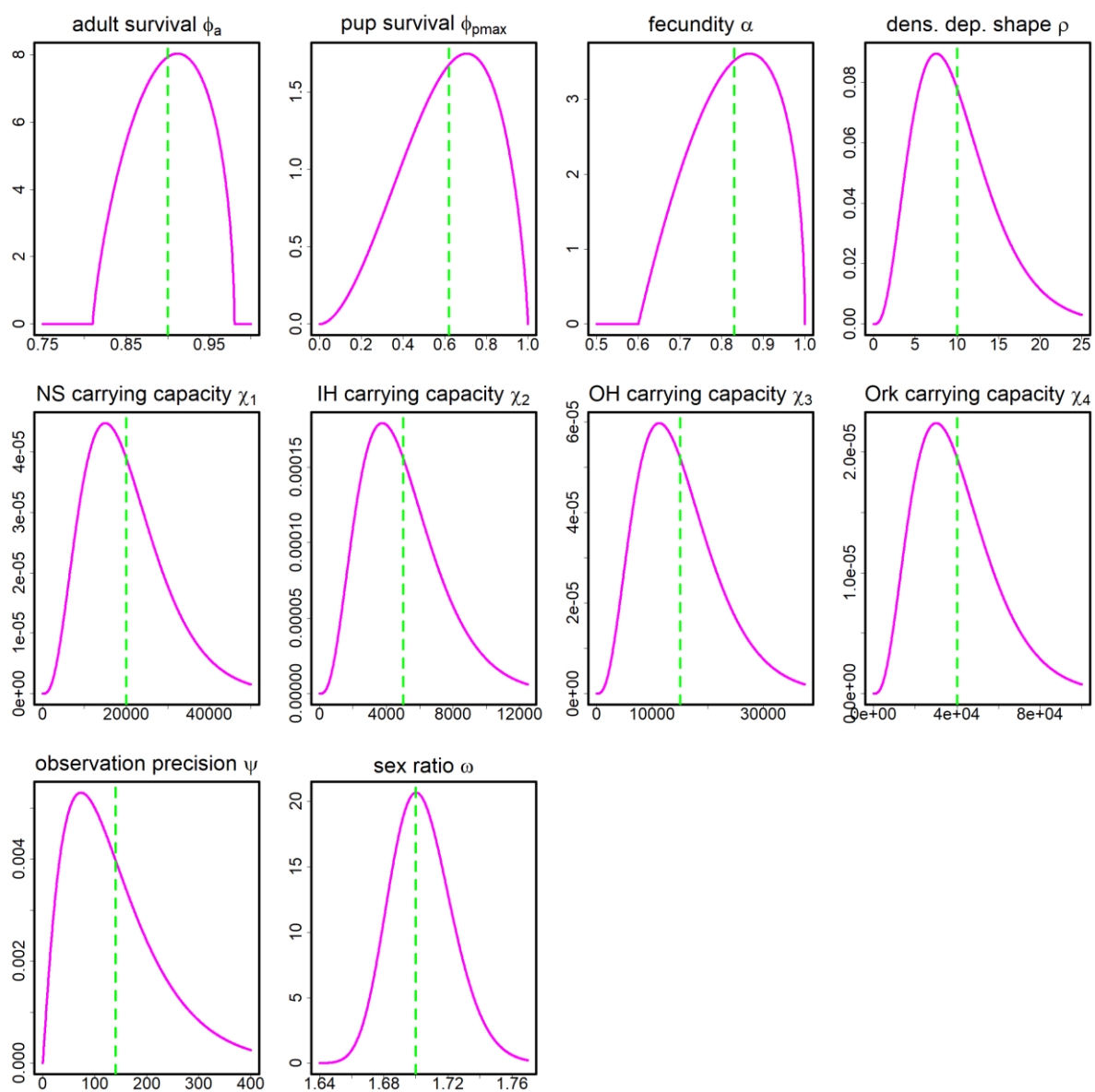


Figure 1. Prior probability density functions for each model parameter input in Thomas (2020), drawn from the distributions specified in Table 1. Carrying capacity subscripts 1 to 4 refer to North Sea, Inner Hebrides, Outer Hebrides and Orkney regions, respectively. Prior means are shown as green dashed vertical lines.

den Heyer & Bowen (2017) used a Cormack-Jolly-Seber model to estimate age- and sex-specific adult survival from a long-term brand re-sighting programme on Sable Island. Average female adult survival was estimated to be 0.976 (SE 0.001), averaged over all animals, but was higher for younger adults (0.989 with SE 0.001 for age classes 4-24) than older adults (0.904 SE 0.004 for age 25+). Rossi et al., (2021) found that females on Sable Island maintained very high annual survival rates (>97%) until age 25, after which survival declines by 8% between ages 25–29 and by another 9% for ages 30+. Males similarly maintained high survival rates (>95%) until age 25, though declines in male survival rates in older age classes were much steeper than in female rates. Thus, as agreed by SCOS in 2018, the upper limit has been increased to 0.98; the resulting distribution is a beta distribution $Be(1.79, 1.53)$ which is scaled (multiplied by 0.18 and added to 0.8) to allow non-zero probability density only between 0.8 and 0.98. The resulting distribution has mean 0.90 and SD 0.04.

Maximum pup survival ϕ_{pmax}

Relevant studies are summarized in Table 2. Data from populations that were growing rapidly and therefore apparently not constrained by density dependence acting on pup survival were required to inform this prior. There are various published estimates of first-year survival during periods of exponential growth (Table 2). Mean estimates of pup survival were between 0.54 – 0.76. On the basis of these estimates, the prior on maximum female pup survival is defined as a diffuse beta distribution $Be(2.87, 1.78)$ which has mean of 0.62 (SD 0.20). Note that Pomeroy, Smout, Moss, Twiss, & King (2010) found high inter-annual variation in pup survival, which is not currently incorporated in the model.

Fecundity α

Relevant studies are summarized in Table 3. For the purposes of this model, fecundity refers to the proportion of breeding-age females (aged 6 and over) that give birth to a pup in a year (natality or birth rate). For the most part, studies have measured pregnancy rather than natality rates. The resulting estimates are thus maxima in terms of fecundity as abortions will cause pregnancy rates to exceed birth rates. Mean estimated adult female pregnancy rates from examination of shot animals were between 0.83 and 0.94 in the UK (Boyd, 1985; Hewer, 1964), and between 0.88 and 1 at Sable Island, Canada (Hammill & Gosselin, 1995). A recent study in Finland (Kauhala *et al.* 2019; Kauhala and Kurkilahti 2020) based on shot animals showed pregnancy rate can fluctuate significantly (between c.0.6 and c.95) in relation to the environment (prey quality). CMR studies report lower estimates, which may be a result of unobserved pupping events (due to mark misidentification, tag loss, or breeding elsewhere), but also because such estimates represent births rather than pregnancy. Such studies, from Sable Island estimate fecundity to be between 0.57 and 0.83 (Bowen, Iverson, McMillan, & Boness, 2006; den Heyer & Bowen, 2017). A recent study from Sable Island demonstrated that fecundity varied as a function of your breeding status in the previous year: non-breeder, first-time breeder, and breeder (in order of lowest to highest). UK estimates of fecundity rates for populations of marked study animals, adjusted for estimates of unobserved pupping events were 0.79 (95% CI 0.77-0.81) and 0.82 (95% CI 0.79-0.84) for a declining (North Rona) and increasing (Isle of May) population, respectively (Smout et al., 2019). Based on the available data, the prior on fecundity (α) is specified as a beta distribution $Be(2, 1.5)$ which is scaled (multiplied by 0.4 and added to 0.6) to only allow probability density between 0.6 and 1. The resulting distribution has mean 0.83 and SD 0.09.

Shape of density dependence acting on pup survival ρ

Pup survival at carrying capacity is not dependent on this parameter, and hence carrying capacity also does not depend on it. Instead, the parameter influences the shape of the population growth trajectory, by determining the shape of the relationship between pup survival and pup production. Fowler (1981) used both theory and empirical data to suggest that most density-dependent change in vital rates happens close to carrying capacity for species with life history strategy typical of large mammals (i.e., long lived and low reproductive rate). Empirical examples (their Figure 4) show relationships consistent with values of ρ in the range 5-10. To avoid being too prescriptive, a diffuse distribution was specified: a Gamma distribution $\text{Ga}(4, 2.5)$, which has a mean of 10 and SD 5.

Region-specific carrying capacity χ_{1-4}

No independent information was available about carrying capacity, and so the priors were specified with a variance wide enough to make their influence on population size estimates negligible. Truly non-informative priors (e.g., improper priors with infinite variance) make the particle filtering algorithm extremely inefficient, since most simulated trajectories are infeasible given the data, hence a trade-off is required between a prior with a large enough variance to be non-informative, but not too large so as to make the algorithm prohibitively inefficient. Having the initial rejection control step in the algorithm helped to some extent in this regard. Gamma distributions with a SD:mean ratio of 1:2, with the mean set subjectively based on expert opinion (Table 1) were found to meet these criteria.

Number of adults per adult female ω

This parameter is also referred to as the sex ratio, although strictly the ratio of males:females is given by $\omega - 1$. Relevant studies (on sex-specific survival rates) are summarized in Table 2. A sex ratio of 0.73:1 was derived from shot samples (Harwood & Prime, 1978). This was based on the following assumptions: that the shot males were a representative sample of the breeding population (≥ 10 years old); that female survival was 0.935; and that survival was the same between the sexes up until age 10. Using telemetry tags and “hat tag” re-sighting data (taking into account detection probability inferred by telemetry data), sex-specific pup survival was estimated (Lonergan 2014; Table 2). Although there were no significant differences in survival between males and females, the mean male survival was lower than females. Combined with data from Hewer (1964), the resulting sex ratio would be between 0.66:1 and 0.68:1 (Lonergan, 2014). Also considered were pup survival estimates derived from shot samples from the Baltic (Kauhala, Ahola, & Kunnasranta, 2012). For Sable Island, Male survival post sexual maturity has been estimated to be 0.98 (SE 0.003) (Brusa et al. 2020 - based on data from Manske et al. 2002). The estimated the sex ratio on Sable was estimated to be 0.69:1 based on estimates of age and sex-specific survival, and assuming a stationary age distribution (Hammill, den Heyer, Bowen, & Lang, 2017). Based on these findings, the prior used was a highly informative scaled Gamma distribution $\text{Ga}(4, 2.5) + 1.6$. This results in a prior mean of 1.7 (SD 0.02); 90% of the prior probability density is between 1.68 and 1.73.

Precision of the pup production estimates ψ

The pup production estimates at colony level from aerial survey data generally have a coefficient of variation of 10% or less. Uncertainty in the ground count estimates is not quantified. The resulting uncertainty in pup production at the region level is hard to predict – if the colony estimates were independent it would be smaller, but they are not independent since they share some parameters. Hence a moderately diffuse prior was specified on ψ ($\text{Ga}(2.1, 66.67)$), implying a prior on CV of pup production (which is $1/\psi$) of 10% with SD 5 (i.e., with 90% of the prior probability density between 5% and 20%).

Table 2. Survival data used to inform the survival and sex ratio priors. CMR refers to Capture-Mark-Recapture studies and can be based on brands (permanent but can be misidentified), passive tagging (can be lost or misidentified), active tagging (can be lost), Photo-ID (can be misidentified). Except for active tagging, estimates of survival depend on the accuracy of re-sighting probabilities and, if appropriate, tag loss. If sex-specific sample sizes are not reported then total *n* is given.

Age classes	females			males			Total n	Time period	Data	Location	Considerations	Source
	mean	uncertainty	n	mean	uncertainty	n						
Pup	0.66		1036	0.66		294		1972, 1975	Aged shot individuals	Farne Islands, UK	Accounted for effect of previous culls on sample structure. Based on life tables.	Harwood & Prime 1978
Pup	0.65	95% CIs: 0.39 - 0.85	180	0.50	95% CIs: 0.25 - 0.75	182		1997 - 1999	CMR (hat tag)	Isle of May and Farne Islands, UK	Tag loss accounted for. Telemetry data used to inform re-sighting probability	Reanalysis of data from Hall, McConnell & Barker 2001; Hall, McConnell & Barker 2002; grey pup seal telemetry data (Carter et al., 2017)
Pup	0.54	95% CIs: 0.18 - 0.86	27	0.43	95% CIs: 0.11 - 0.82	28		2002	CMR (telemetry data)	Isle of May, UK	Tag loss accounted for	Reanalysis of data from Hall, Thomas & McConnell 2009
Pup	0.76			0.38			118	2000 - 2004	Aged shot individuals	Baltic	Samples assumed representative. Based on life tables	Kauhala, Ahola & Kunnasranta 2012
≤ 4	0.73	SE = 0.016	1700				229	2005 - 1985				
	0.33	SE = 0.024	1182				5	1989 - 1998	CMR (brand)	Sable Island, Canada	Includes the data from Schwarz & Stobo (2000)	den Heyer, Bowen & Mcmillan 2014
Adult	0.95		239					1956 - 1966	Aged shot individuals	UK	Samples assumed representative. Based on life tables	Data from Hewer 1974, analysed by Lonergan 2012
≥ 10				0.80		294		1972, 1975	Aged shot individuals	Farne Islands, UK	Accounted for population trajectory. Assumed samples are representative within focal age class.	Harwood & Prime 1978

≥ 7	0.935 (0.90-0.96)		1036				1972, 1975	Aged shot individuals	Farne Islands, UK	As above		Harwood & Prime 1978 (reanalysed by Lonergan 2012)
Adu It	0.94	95% CIs: 0.93 - 0.95	273				1987 - 2014	CMR (brand, flipper tag, photo ID)	Isle of May		Tag loss and differential sighting probability accounted for. Survival confounded with permanent emigration	Smout, King & Pomeroy, 2019
Adu It	0.896	95% CIs: 0.87 - 0.90	584				1993 - 2013	As above	North Rona, UK	As above		As above
≥ 4	0.976	SE = 0.001	3178	1727			1969 - 2002	CMR (brand)	Sable Island, Canada		Tagged as pups. Confounded with permanent emigration (rare)	den Heyer & Bowen 2017
4-24	0.989	SE = 0.001	As above	0.970	SE = 0.002	As above	As above	As above	As above	As above	As above	As above
≥ 25	0.904	SE = 0.004	As above	0.77	SE = 0.01	As above	As above	As above	As above	As above	As above	As above
Adu It	0.976	SE = 0.001	As above	0.943	SE = 0.003	As above	As above	As above)	As above	As above	As above	As above

Table 3. Fecundity data used to inform the fecundity priors. CMR refers to Capture-Mark-Recapture studies and can be based on brands (permanent but can be misidentified), passive tagging (can be lost or misidentified), Photo-ID (can be misidentified). Estimates of fecundity depend on the accuracy of re-sighting probabilities and, if appropriate, tag loss.

Rate	Mean	Uncertainty	n	Time period	Data	Location	Considerations	Source
Pregnancy	0.93		79	1956 - 1963	Shot samples			Hewer 1964
Pregnancy	0.94	95% CIs: 0.89 - 0.97	140	1979 - 1981	Shot samples	Farne Islands, UK		Boyd 1985
Pregnancy	0.83	95% CIs: 0.74 - 0.89	88	1978	Shot samples	Outer Hebrides, UK		Boyd 1985
Pregnancy	0.88-1		526	1968 - 1992	Shot samples	Canada	Aged ≥ 6 years old	Hammill & Gosselin 1995
Birth	0.73	0.015	174	1983 - 2005	CMR (brand)	Sable Island, Canada	Aged 4-15 years. Unobserved pupping not considered (likely rare)	Bowen <i>et al.</i> 2006
Birth	0.83	0.034	32	1983 - 2005	As above	As above	Aged 16-25 year Unobserved pupping not considered (likely rare)	As above
Birth	0.57	0.03	39	1983 - 2005	As above	As above	Aged 26-35 years Unobserved pupping not considered (likely rare)	As above

Birth	0.790	95% CIs: 0.77 - 0.82	584	1993 - 2013	CMR (brand, flipper tag, photo ID)	North Rona, UK	Accounted for unobserved pupping	Smout <i>et al.</i> 2019
Birth	0.82	95% CIs: 0.79 - 0.84	273	1987 - 2014	CMR (brand, flipper tag, photo ID)	Isle of May, UK	As above	As above
Birth	0.79		1727	1992 - 2002	CMR (brand)	Sable Island, Canada	Estimated transitions: unobserved to breeder = 0.41 - 0.64, breeder to breeder = 0.76 - 0.89	den Heyer & Bowen 2017
Birth	0.56		66	2001- 2018	Shot/bycatch samples	Finland	Age 5-6 years old	Kauhala and Kurkilahti 2020
Birth	0.79		460	2001- 2018	Shot/bycatch samples	Finland	Age 7-24 years old	Kauhala and Kurkilahti 2020

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Estimating the size of the UK grey seal population between 1984 and 2020.

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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 1984-2016, 2018 (North Sea region only) and 2019, and (2) independent estimates assumed to be of total population size just before the breeding season in 2008, 2014 and 2017. 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 and prior distributions are identical to those used to provide last year's advice; the data include the new 2019 pup production estimates and 2017 estimate of total population size, as well as slightly revised total population estimates from 2008 and 2014.

Estimated population size in regularly monitored colonies in 2020 was 140,700 (95% CI 129,300-153,500). The population overall is estimated to be increasing at a rate of 1.7% per year.

In a supplementary run, we used an alternative set of pup production estimates derived by making a different assumption about the probability of correctly classifying moulted pups from aerial digital images. The estimate of total population size was almost identical. However, a previous analyses has shown that assumptions made in the pup production model can affect estimates of total population size, so the result obtained here should not be generalized.

Historically one constraint on our ability to investigate and extend the model has been the time taken to fit it using the particle filtering algorithm developed by Thomas and colleagues in 2005. We have recently developed new algorithms that are significantly faster and are undertaking a simulation-based evaluation of the model as well as model extensions. We expect to report our findings at next year's meeting.

Introduction

This paper presents estimates of British grey seal population size and related demographic parameters, obtained using a Bayesian state-space model of population dynamics fitted to pup production estimates (from aerial surveys of breeding colonies) and independent estimates of total population size (from haul-out counts). The model and fitting methods are the same as those employed in recent years and are described in detail in Thomas et al. (2019); the prior distributions on model parameters are the same as those used for the last two years (see Russell et al. (2021) for justification). The data are a time series of regional pup production estimates (1984-2016; 2018 North Sea region only; 2019) of which the 2019 estimates are new for this briefing paper, and independent estimates of total population size (2008, 2014 and 2017) of which the 2017 estimates are new.

We present estimates of population size at the start of the 2020 breeding system (i.e., projected forward one year from the last pup production estimates). Note that all estimates of population size relate to seals associated with the regularly monitored colonies. A multiplier is required to account for the 6-8% of seals that breed outside these colonies.

The pup production estimation method is currently undergoing a revision, and one aspect of estimation that is being examined is the probability of correctly classifying a moulted pup from the film and digital aerial survey images (“PMoult”¹⁸). In the main run, the pup production estimates are based on a PMoult of 0.5 for film and 0.9 for digital images. The change to 0.9 was based on the increased quality of the digital images, compared to the film; this is the value used in previous briefing papers. However, work presented at the SCOS meeting in 2019 suggested that the improvement in correct classification with digital images is substantially less, and so a value less than 0.9 was warranted. To provide a sensitivity analysis, as with the 2020 briefing paper, we present results from a supplementary run of the population model using pup production estimates of 0.5 for both film and digital images.

Methods

Main run

Full details of the population dynamics model, data and fitting methods are given in Thomas et al. (2019). In summary, an age-structured population dynamics model is specified for each of four regions (North Sea, Inner Hebrides, Outer Hebrides and Orkney), with 7 ages included in the model: pups, age 1-5 females (assumed not to reproduce) and age 6+ females (which may breed). The model assumes constant adult (age 1+) survival (indexed by a parameter ϕ_a), constant fecundity (probability that an age 6+ female will birth a pup, α) and density-dependent pup survival with separate carrying capacity in each region (carrying capacity parameters $\chi_1 - \chi_4$ and common parameters for maximum pup survival ϕ_{pmax} and shape of the density dependence function ρ). The modelled pup production is linked to the data by assuming the data follow a normal distribution centred on true pup production and with precision parameter ψ . Adult males are not tracked explicitly in the population model, but instead, the total population size (of males and females) is derived by multiplying estimated adult females by a parameter ω that represents the ratio of total adults to adult females (sometimes called “sex ratio” as shorthand, although sex ratio is actually given by $\omega - 1$). The modelled total population size (age 1+ animals) is linked to the independent estimates using the empirically derived uncertainty on the independent estimates. Informative prior distributions are used on model parameters, as justified in Russell et al. (2021) and summarised in Table 1 (detailed justification for prior distributions is given in Supporting Information of Thomas et al. 2019).

Input data were pup production estimates for 1984-2016, the North Sea region estimate for 2018, and for all regions in 2019 (Russell et al. 2021). The estimates for 1984-2016 are identical to those used in last year’s briefing paper (Thomas 2020); the estimate for the North Sea region in 2018 is almost identical (18,845 vs the previous 16,778). The other source of data is the independent estimates of total population size from 2008, 2014 and, for the first time, 2017 (Russell et al. 2021). The estimates for 2008 and 2014 are approximately 5% lower than those used in previous briefing papers because an updated scaling factor has been used in converting from hauled-out seals counted to population estimate (Russell and Carter 2021). Note that the total population size estimates are assumed independent of one another, when in reality they are based on the same scaling factor. We return to this in the Discussion.

Model fitting, as in previous reports, used a stochastic simulation-based procedure called a particle filter (Thomas et al. 2019). Reliability of reported results depends on the number of simulations. Here, 4.6 billion simulations were used, which gave results accurate to 2-3 significant figures.

¹⁸ To be precise, this parameter is the probability of correctly classifying a light-coated pup as a moulted pup; the pup production model contains an assumption about the proportion of moulted pups that are dark-coated.

Supplementary run

As described earlier, one important parameter in pup production estimation is the probability of correctly classifying moulted pups from the images, PMoult (Russell et al. 2019). This probability has been set at 0.9 for the digital images collected since 2012. As part of an ongoing review of pup production estimation, it was desired to assess the effect of setting PMoult for digital images to 0.5. This results in lower pup production estimates for the digital survey years (post 2010), except in the North Sea region where the majority of pup production estimates are derived from ground counts. A supplementary run of the population model was performed (using 2.2 billion simulations) with these alternative pup production estimates.

Results

Main run

Estimated pup production by region from the model matches the observed values reasonably well although it is clear that the pup production estimates for Inner and Outer Hebrides and Orkney are substantially higher after the advent of digital surveys in 2012 and that this affects the fit: residuals for several years before this are all negative and after are all positive, except for Orkney in 2019 (Figure 1). In the case of Inner and Outer Hebrides, the post-2012 estimates are considerably higher than predicted. A similar tendency is seen in North Sea, but to a much lesser extent. Overall, pup production is estimated to be increasing strongly in North Sea, have stabilized in the decade after 1995 in Inner and Outer Hebrides, and be stabilizing in Orkney (Figure 1).

Total population size estimated using pup production data alone (Figure 2, blue lines) is somewhat larger but considerably less precise than that when the three independent estimates are added (Figure 2, red lines). In both cases, population size is estimated to have grown steadily, although at a slightly decreasing rate. When pup production data and independent estimates are both used (red lines in Figure 2), population size is estimated to have been larger than the independent estimate from 2008 and smaller than that from 2014 and 2017. Posterior mean population size in regularly monitored colonies in 2020 was 140,700 with 95% credible interval (CI) 129,300-153,500. Estimates by region are given in Table 2 and estimates for all years 1984-2020 are given in Appendix 1 (Table A1). The estimated growth in population size between 2019 and 2020 is 1.7%.

Posterior parameter distributions are shown in Figure 3, with numerical summaries in Table 1. The estimates are a little different from those reported by Thomas (2020), likely because of the additional independent estimate. Adult survival is estimated to be slightly higher and pup survival lower (the two are strongly negatively correlated, Thomas et al. 2019); the density dependent shape parameter is somewhat lower and carrying capacity higher. Three regions (Inner Hebrides, Outer Hebrides and Orkney) are estimated to be close to carrying capacity (i.e., posterior mean on carrying capacity parameter close to the pup production), while North Sea is at approximately 60% of carrying capacity (although that estimate is quite imprecise with SE/mean=0.3). Estimated sex ratio is, as previously, unchanged from the prior.

Supplementary run

Despite lower pup production estimates in Inner and Outer Hebrides and Orkney going into the model, the resulting estimates of total population size were very slightly (about 1%) larger (Table 2, last column). The difference is largely caused by a higher population estimate in North Sea, where pup production was least decreased; it is perhaps caused by the slightly lower fecundity estimate (Table 1), although the difference in population estimate is too small to deserve an in-depth examination.

Discussion

Estimated population size in the main run is approximately 3% higher than that reported in last year's briefing paper (Thomas 2020) for comparable years – for example the total population size estimate in 2019 from Thomas (2019) was 133,900 (95% CI 115,300-156,500) while here the estimate for the same year 138,300 (95% CI 127,700-150,500). There have been several updates to the input dataset, but likely the biggest contributor to the change is the introduction of the 2017 independent estimate of total population size, which was larger than the value predicted by the model and hence likely drew the estimates upward. It should be noted that (a) such small changes happen commonly as the data is updated – for example, minor changes to the data used in the 2020 briefing paper produced estimates that were approximately 4% lower than those produced the year before (see Thomas 2020), and (b) all of these changes are well within the estimated credible intervals on total population size.

In this analysis, the three independent estimates of total population size, from 2008, 2014 and 2017, are assumed to be statistically independent of one another. Although they are based on separate aerial surveys of hauled-out seals, in scaling up from counts of seals hauled out to total population size both rely on the same estimate of the proportion of seals hauled out (Russell and Carter 2021). This year, we investigated an approach to deal with this using an observation model that allows each annual haul-out count to follow a binomial distribution with the underlying haul-out probability assumed common across all three counts and following a beta distribution (Appendix 2). However, this model proved to be too restrictive, strongly penalizing population trajectories that do not closely follow the ~6% per-year population growth implied by the values of the three haul-out counts. This growth rate is not supported by the population model fitted to pup production estimates. The new observation model assumes seals haul out independently and that haul-out probability is constant between years – we believe one or more of these assumptions needs to be relaxed before this model will be of use in the population modelling process. Hence, for this briefing paper, we have elected to stick with the assumption used in previous years that the total population estimates come from independent shifted gamma distributions.

Thomas et al. (2019) discuss how sensitive the estimate of total population size may be to the parameter priors, and conclude that fecundity and adult:female ratio are two parameters that strongly affect total population size but for which the prior specification is particularly influential. Hence a renewed focus on priors for these parameters may be appropriate.

In our supplementary analysis, we found very little (1%) change to population size estimates from alternative assumption on pup production estimation. However, we also note that additional analyses undertaken by Thomas (2019) showed that small changes in pup production estimates did influence the total population estimates, so we caution our result here should not be generalized. As noted above, the independent estimates of population size may have been overly dominant in this analysis, and that will change in the future.

One constraint on making inferences from this model has been the time taken to fit it using the particle filtering algorithm used, which was first developed by Thomas et al. (2005) and Newman et al. (2006). The main run presented here was based on runs of 4.6 billion simulations, which took approximately 40 hours computer time, running on 40 processors in parallel. Such run times make it prohibitive to investigate aspects of model performance via simulation and to extend the model to include biologically-relevant factors such as time-varying fecundity. Over the past three years, PhD student Fanny Empacher has been researching alternative more efficient algorithms, and she has been joined in the past year by PhD student Cal Fagard-Jenkin who is working on highly parallel algorithms using Graphics Processing Units (GPUs). Both have made considerable progress and we

anticipate over the next year we will be able to undertake some simulation studies of the model, and also switch estimation to the new algorithms.

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Table 1. Prior parameter distributions and summary of posterior distributions. Be denotes beta distribution, Ga Gamma distribution (with parameters shape and scale, respectively). Analysis uses 1984-2016 and 2018 (North Sea only) pup production estimates, and the 2008 and 2014 total population estimates. Posterior estimates are shown for two runs: a main run, assuming probability of correct classification of moulted pups from digital aerial images is 0.9, and a supplementary run when where this probability is assumed to be 0.5.

Parameter	Prior distribution	Prior mean (SD)	Posterior mean (SD)	
			Main run	Suppl. run
adult survival ϕ_a	0.8+0.17*Be(1.79,1.53)	0.90 (0.04)	0.97 (0.01)	0.96 (0.01)
pup survival ϕ_{pmax}	Be(2.87,1.78)	0.62 (0.20)	0.42 (0.07)	0.49 (0.08)
Fecundity α	0.6+0.4*Be(2,1.5)	0.83 (0.09)	0.91 (0.05)	0.90 (0.06)
dens. dep. ρ	Ga(4,2.5)	10 (5)	3.3 (0.78)	3.81 (1.24)
NS carrying cap. χ_1	Ga(4,5000)	20000 (10000)	33200 (9700)	32100 (10300)
IH carrying cap. χ_2	Ga(4,1250)	5000 (2500)	4110 (457)	3670 (347)
OH carrying cap. χ_3	Ga(4,3750)	15000 (7500)	14000 (1180)	13000 (794)
Ork carrying cap. χ_4	Ga(4,10000)	40000 (20000)	23700 (4290)	20600 (2350)
observation prec. ψ	Ga(2.1,66.67)	140 (96.6)	67.4 (20.7)	74 (20.4)
sex ratio ω	1.6+Ga(28.08, 3.70E-3)	1.7 (0.02)	1.7 (0.02)	1.7 (0.02)

Table 2. Estimated size, in thousands, of the British grey seal population at the start of the 2020 breeding season, derived from a model fit to pup production data from 1984-2016, 2018 (North Sea only) and 2019, and the additional total population estimates from 2008, 2014 and 2017. Estimates from two runs are shown: a main run, assuming probability of correct classification of moulted pups from digital aerial images is 0.9, and a supplementary run when where this probability is assumed to be 0.5. Values in the table are posterior means with 95% credible intervals in brackets.

	Estimated population size in thousands (95% CI)	
	Main run	Supplementary run
North Sea	49.3 (38.1 62.7)	54.0 (41.1 68.9)
Inner Hebrides	9.1 (7.7 11)	8.7 (7.3 10.4)
Outer Hebrides	31 (27.1 35.7)	31 (27 34.7)
Orkney	51.3 (43.9 62.6)	48.7 (41.8 57.3)
Total	140.7 (129.3 153.5)	142.5 (129 156.5)

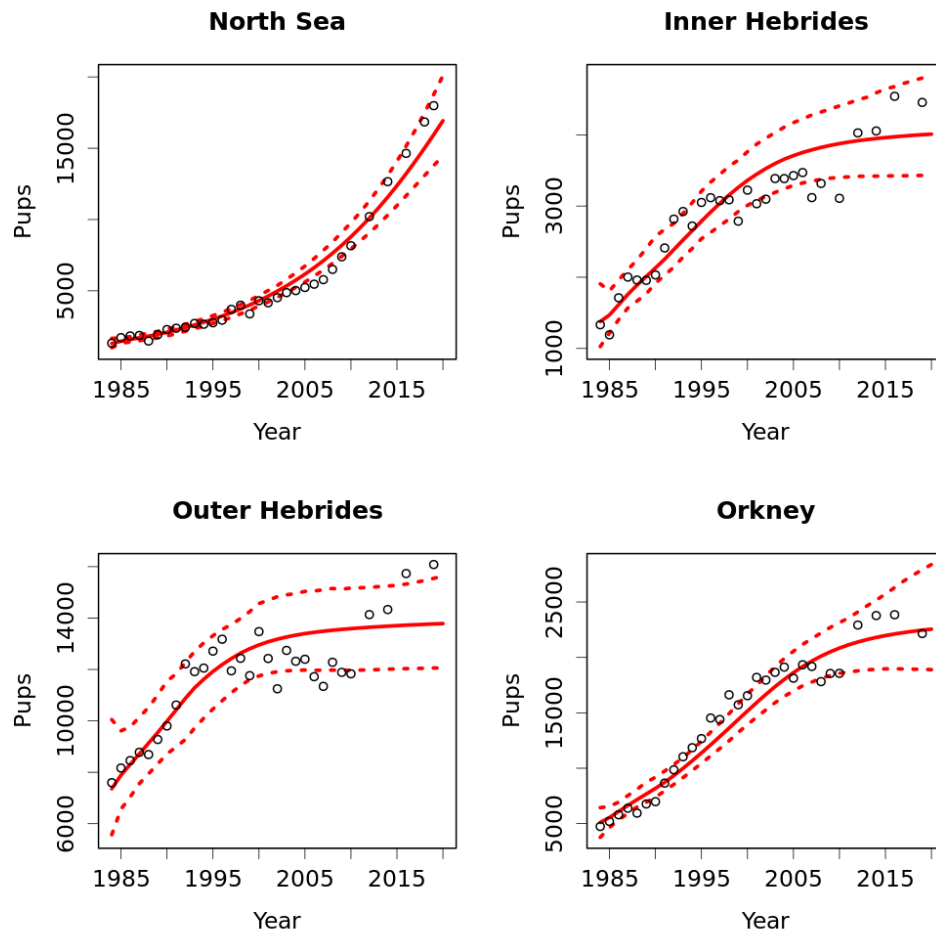


Figure 1. Posterior mean estimates of pup production (solid lines) and 95%CI (dashed lines) from the model of grey seal population dynamics, fitted to pup production estimates from 1984-2016, 2018 (North Sea only) and 2019 (circles) and the total population estimates from 2008, 2014 and 2017.

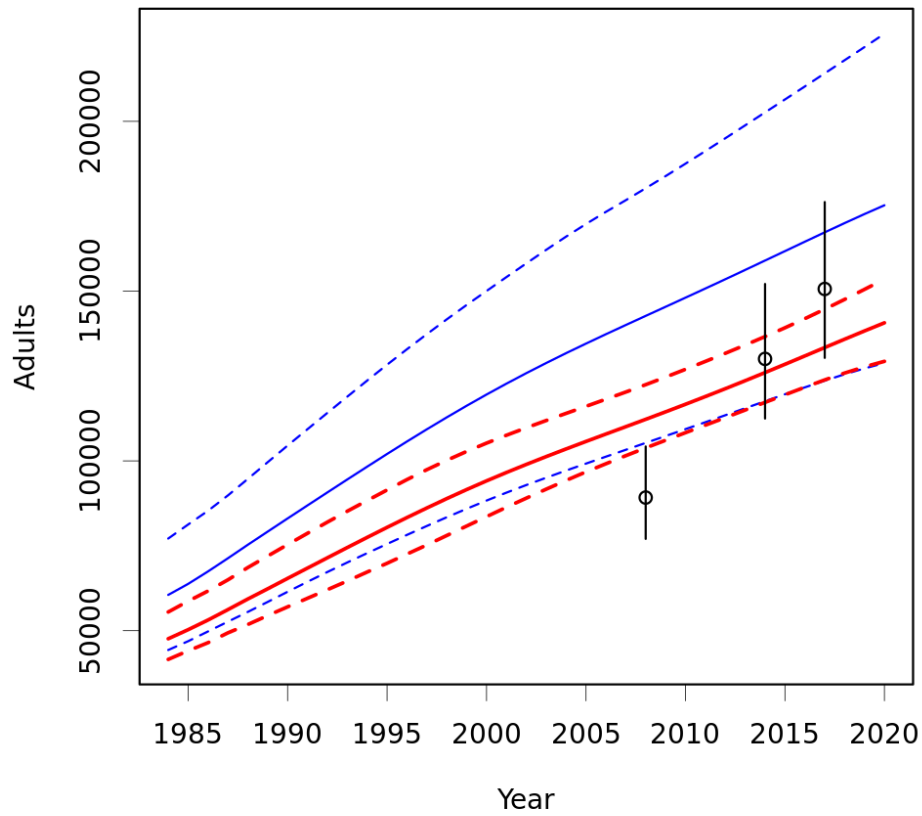


Figure 2. Posterior mean estimates (solid lines) and 95%CI (dashed lines) of total population size in 1984-2019 from the model of grey seal population dynamics, fit to pup production estimates from 1984-2016, 2018 (North Sea only) and 2019, and total population estimates from 2008, 2014 and 2017 (circles, with vertical lines indicating 95% confidence interval on the estimates). Blue lines show fit to pup production data alone, red lines show fit to pup production data and independent estimates.

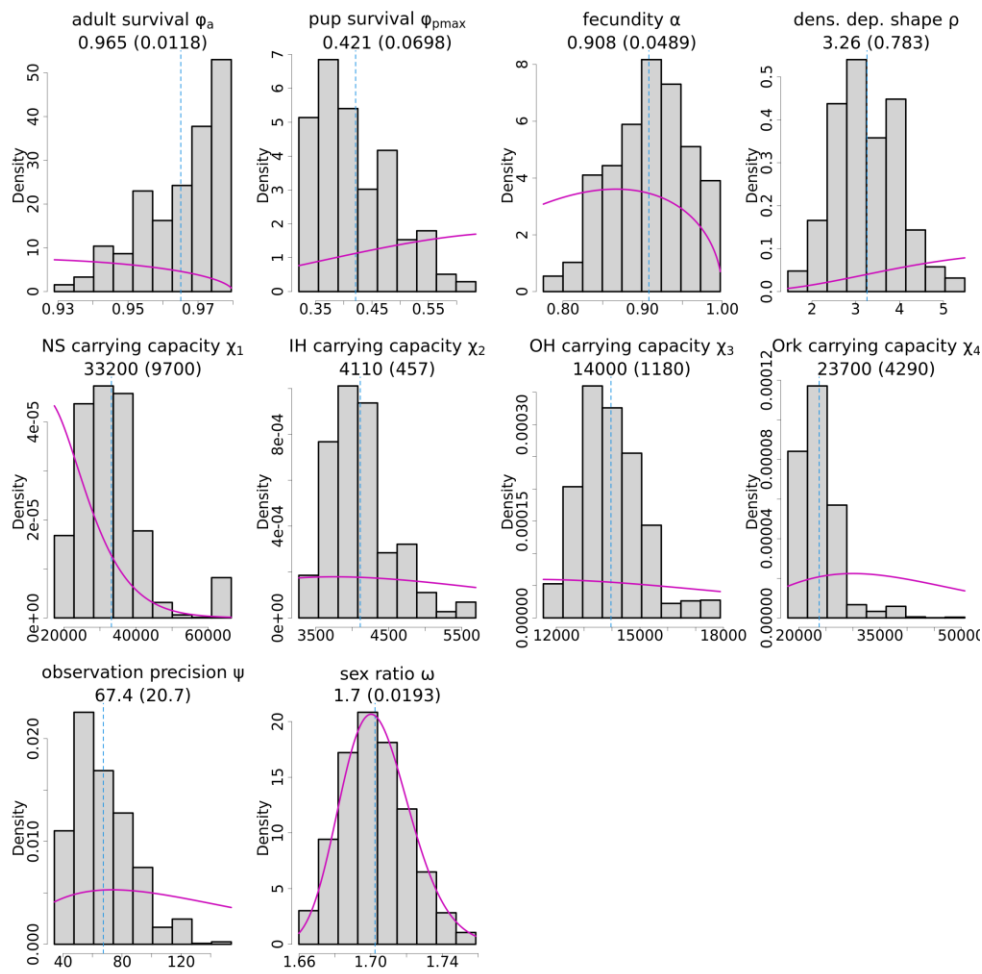


Figure 3. Posterior parameter distributions (histograms) and priors (solid lines) for the model of grey seal population dynamics, fit to pup production estimates from 1984-2016, 2018 (North Sea only) and 2019, and total populations estimate from 2008, 2014 and 2017. The vertical dashed line shows the posterior mean; its value is given in the title of each plot after the parameter name, with the associated standard error in parentheses.

Appendix 1

Table A1. Estimates of total population size, in thousands, at the beginning of each breeding season from 1984-2020, made using the model of British grey seal population dynamics fit to pup production estimates from 1984-2016, 2018 (North Sea only) and 2019, and total population estimates from 2008, 2014 and 2017. Numbers are posterior means followed by 95% credible intervals in brackets.

Year	North Sea	Inner Hebrides	Outer Hebrides	Orkney	Total
1984	4.4 (3.7 5.1)	4.5 (3.9 5.5)	21.5 (17.9 26)	17.2 (14.5 20.4)	47.6 (41.5 55.5)
1985	4.7 (4 5.4)	4.8 (4.1 5.8)	22.4 (18.7 27.3)	18.4 (15.7 21.7)	50.3 (44.1 58.7)
1986	5.1 (4.4 5.8)	5.1 (4.3 6.1)	23.4 (19.5 28.4)	19.6 (16.9 23)	53.1 (46.5 61.7)
1987	5.4 (4.8 6.2)	5.4 (4.6 6.4)	24.4 (20.4 29.4)	21 (18.1 24.5)	56.2 (49.4 65)
1988	5.8 (5.2 6.7)	5.7 (4.9 6.8)	25.4 (21.2 30.5)	22.4 (19.3 26.3)	59.3 (51.8 68.6)
1989	6.3 (5.6 7.2)	6 (5.1 7.1)	26.1 (22.1 31.3)	24 (20.5 28)	62.3 (54.4 72)
1990	6.8 (6 7.7)	6.3 (5.3 7.5)	26.8 (22.8 32)	25.6 (21.8 29.8)	65.4 (57 75.4)
1991	7.3 (6.4 8.3)	6.6 (5.5 7.8)	27.4 (23.4 32.5)	27.2 (23.1 31.7)	68.4 (59.5 78.7)
1992	7.8 (6.9 8.9)	6.8 (5.7 8.1)	27.9 (23.9 32.9)	28.9 (24.5 33.6)	71.5 (62.1 82)
1993	8.4 (7.4 9.6)	7.1 (5.9 8.4)	28.4 (24.4 33.2)	30.6 (26 35.6)	74.5 (64.6 85.2)
1994	9 (8 10.3)	7.3 (6 8.7)	28.8 (24.9 33.5)	32.3 (27.4 37.5)	77.5 (67.3 88.3)
1995	9.7 (8.5 11)	7.6 (6.2 9)	29.1 (25.3 33.7)	34.1 (28.9 39.6)	80.4 (69.8 91.4)
1996	10.4 (9.1 11.8)	7.8 (6.3 9.3)	29.4 (25.6 33.9)	35.8 (30.4 41.6)	83.3 (72.6 94.5)
1997	11.2 (9.8 12.7)	7.9 (6.5 9.5)	29.6 (25.9 34.1)	37.4 (31.9 43.5)	86.1 (75.2 97.4)
1998	12 (10.5 13.7)	8.1 (6.6 9.7)	29.8 (26.1 34.2)	39 (33.5 45.2)	88.9 (78 100.2)
1999	12.9 (11.2 14.7)	8.2 (6.7 9.9)	30 (26.3 34.3)	40.5 (35 46.7)	91.6 (80.8 102.8)
2000	13.9 (12 15.8)	8.3 (6.9 10)	30.1 (26.5 34.4)	41.8 (36.5 48.1)	94.1 (83.6 105.2)
2001	14.9 (12.9 17)	8.4 (7 10.1)	30.2 (26.7 34.5)	43.1 (37.9 49.3)	96.6 (86.3 107.6)
2002	16 (13.8 18.3)	8.5 (7.1 10.2)	30.3 (26.7 34.5)	44.2 (39.1 50.4)	99 (89 109.8)
2003	17.1 (14.8 19.7)	8.6 (7.2 10.3)	30.4 (26.8 34.6)	45.2 (40 51.4)	101.3 (91.7 111.9)
2004	18.4 (15.8 21.2)	8.7 (7.3 10.4)	30.4 (26.8 34.6)	46 (40.8 52.2)	103.5 (94.3 114)
2005	19.7 (16.9 22.7)	8.7 (7.3 10.4)	30.5 (26.9 34.7)	46.8 (41.4 53)	105.7 (96.8 116.1)
2006	21.2 (18 24.4)	8.8 (7.4 10.4)	30.6 (26.9 34.7)	47.4 (41.9 53.7)	107.9 (99.2 118.2)
2007	22.7 (19.2 26.3)	8.8 (7.5 10.5)	30.6 (26.9 34.8)	48 (42.3 54.3)	110.1 (101.5 120.3)
2008	24.3 (20.5 28.2)	8.8 (7.5 10.5)	30.7 (26.9 34.8)	48.5 (42.6 55)	112.2 (103.8 122.4)
2009	26 (21.9 30.3)	8.9 (7.5 10.6)	30.7 (27 34.9)	48.9 (42.9 55.6)	114.4 (106.1 124.7)
2010	27.8 (23.2 32.6)	8.9 (7.6 10.6)	30.7 (27 35)	49.2 (43.2 56.2)	116.7 (108.3 127)
2011	29.7 (24.7 35)	8.9 (7.6 10.6)	30.8 (27 35)	49.6 (43.4 56.7)	118.9 (110.5 129.3)
2012	31.7 (26.1 37.7)	8.9 (7.6 10.7)	30.8 (27.1 35.1)	49.8 (43.5 57.3)	121.3 (112.7 131.7)
2013	33.7 (27.6 40.4)	9 (7.7 10.7)	30.9 (27.1 35.1)	50.1 (43.6 57.9)	123.6 (115 134.1)
2014	35.8 (29.2 43.2)	9 (7.7 10.8)	30.9 (27.1 35.2)	50.3 (43.7 58.7)	126 (117.2 136.6)
2015	38 (30.7 46.3)	9 (7.7 10.8)	30.9 (27.1 35.3)	50.5 (43.8 59.4)	128.5 (119.5 139.2)
2016	40.3 (32.3 49.4)	9 (7.7 10.8)	30.9 (27.1 35.3)	50.7 (43.9 60)	130.9 (121.7 141.9)
2017	42.5 (33.9 52.7)	9 (7.7 10.9)	31 (27.1 35.4)	50.8 (43.9 60.7)	133.4 (123.9 144.7)
2018	44.8 (35.4 56.1)	9 (7.7 10.9)	31 (27.1 35.5)	51 (43.9 61.3)	135.8 (125.9 147.6)
2019	47.1 (36.8 59.4)	9 (7.7 10.9)	31 (27.1 35.6)	51.1 (43.9 61.9)	138.3 (127.7 150.5)
2020	49.3 (38.1 62.7)	9.1 (7.7 11)	31 (27.1 35.7)	51.3 (43.9 62.6)	140.7 (129.3 153.5)

Appendix 2. Alternative observation model for independent estimates.

Let y_t be the count of hauled-out adult (i.e., non-pup) grey seals in year t , where $t = 1, 2, 3$ corresponds to the three years 2008, 2014 and 2016. Let n_t be the total population size of adult grey seals from regularly-monitored colonies in year t . We assume seals haul out independently of one another, and that the probability a seal hauls out, p , is constant between years. Hence, the number hauled out is a binomial random variable

$$y_t \sim \text{Bin}(n_t, p)$$

The haul out probability is not known, and we assume uncertainty in p is described by a beta distribution with parameters a and b . We estimate these parameters by fitting a beta distribution to a non-parametric bootstrap sample of haul-out probabilities derived from the analysis of Russell and Carter (2021). The likelihood for observed haul-out counts $\mathbf{y} = \{y_1, y_2, y_3\}'$ given a, b and $\mathbf{n} = \{n_1, n_2, n_3\}'$ is obtained by integrating over the unknown p :

$$\mathcal{L}(\mathbf{y}|\mathbf{n}; a, b) = \int_{p=0}^1 \left(\prod_{t=1}^3 f_y(y_t | n_t, p) \right) f_p(p | a, b) dp$$

where $f_y(\cdot)$ denotes the binomial probability mass function and $f_p(\cdot)$ the beta probability density function.

Recent changes in status of harbour seals in the Wash and North Norfolk SAC and adjacent sites.

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Abstract

The counts of harbour seals at sites from Donna Nook to Scroby Sands, within the Southeast England Seal Management Unit (SSE SMU), during the August survey in 2019 were approximately 27.5% lower than the five year mean for 2014 to 2018.

The same sites were surveyed in 2020. That count was 8% higher than the 2019 count but was still 21.5% lower than the 2014-2018 mean. Three surveys were carried out in 2021 and the mean harbour seal count was close to the mean of 2019 and 2020 counts and confirms that there has been a decrease.

The total count for the sites between Donna Nook and Scroby Sands has declined by approximately 38% compared to the mean of the previous five years (2019–2021 mean = 3080; 2014-2018 mean = 4296). The count for the Wash and North Norfolk SAC has decreased by approximately 21% (2019 – 2021 mean = 2883; 2014-2018 mean= 3658) over the same time periods while Donna Nook showed a 57% decrease and Scroby Sands showed a 73% decrease.

The harbour seal decline is evident at all sites within the SMU and appears to have affected all sub-sections of the Wash & North Norfolk SAC.

Grey seal numbers have increased within the SMU, but the largest grey seal haulout group at Donna Nook shows a similar levelling off and possible decline, coincident with the harbour seal decline.

Grey seals are expanding their haulout range within the Wash and small groups are now appearing in the sheltered tidal creeks at the southern edge of the estuary where large numbers of harbour seals haulout.

Introduction

This is a preliminary note about recent changes in the aerial survey counts of harbour and grey seals in the Wash and North Norfolk SAC and adjacent sites (within the Southeast England Seal Management Unit (SEE SMU)). Counts of the survey images for 2021 have only recently been completed, so the descriptions of trends in the data should be regarded as preliminary estimates and treated with caution. A full analysis of the trend data will be completed in early 2022 and additional surveys are again planned for August 2022.

Methods

Surveys of the coastline between Donna Nook in Lincolnshire and Scroby Sands in Norfolk were conducted by fixed-wing aircraft using hand-held oblique photography (see Thompson *et al.*, 2019 for detailed methods).

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) and good weather, i.e. good visibility, no rain.

Results

1. 2020 survey

At SCOS 2020 we reported that the count of harbour seals in The Wash and adjacent sites (Donna Nook, Blakeney and Scroby Sands) in 2019 was approximately 27.5% lower than the mean of the previous 5 years (2014-2018). Despite the restrictions due to the Covid 19 pandemic a survey of the coast between Donna Nook, Lincolnshire and Scroby Sands, Norfolk was carried in August 2020. The 2020 count was 8% higher than the 2019 count but was still 21.5% lower than the 2014-2018 mean

Notwithstanding the variability associated with the proportion of the population hauled out and thus available to count, it was thought likely that these lower counts represented a real decrease. The level of decrease and trajectory was unclear, but the data indicated a potential step change decrease of around 25% between 2018 and 2019. Given that the survey area represents the majority of harbour seals in the SEE SMU and encompasses the population in the Wash & North Norfolk SAC, this likely drop in abundance is of immediate and serious concern. This SMU had shown a sustained increase in abundance (punctuated by sudden drops associated with the Phocine Distemper Epizootics) while most SMUs on the eastern and northern coasts had depleted or declining populations (Thompson *et al.*, 2019; SCOS, 2020).

2. 2021 surveys

In response to the perceived decline, funds were provided by Defra and Natural England to supplement the NERC funding and allow additional surveys of the coast between Donna Nook and Scroby Sands. Due to a combination of Covid related travel restrictions and the last-minute collapse of the contracted aerial survey company we were unable to carry out a planned pup census for the area. However, three surveys were carried out during the harbour seal moult, on 12th, 22nd and 23rd August 2021; two covered the entire coastline between Donna Nook and Scroby Sands and one covered the coast between Donna Nook and Blakeney. All three surveys covered the Wash and North Norfolk SAC.

Table 9. counts of harbour seals at Donna Nook, the Wash, Blakeney and Scroby sands during August between 2016 and 2021. n/s = not surveyed

Date	12/08/ 2021	22/08/ 2021	23/08/ 2021	22/08/ 2020	11/08/ 2019	11/08/ 2018	11/08/ 2017	21/08/ 2016	5/8/ 2016
Wash	2724	2439	2837	2866	2415	3632	3210	2992	3762
Donna Nook	153	75	139	157	128	146	290	275	462
Blakeney	135	187	221	258	329	218	271	388	460
Horsey	N/S	9	15	1	16	17	N/S	N/S	N/S
Scroby Sands	N/S	24	25	45	193	210	399	184	211
Total	3012¹	2734	3237	3327	3081	4223	4170	3839	4895

- ¹Total does not include Scroby Sands or Horsey which held ~ 1% of the harbour seals in the other 2021 and the 2020 counts.

3. harbour seals

Counts of harbour seals from surveys between 2016 and 2021 are shown in table 1. The mean harbour seal count for 2021 (2995) was 7% lower than the mean of 2019 and 2020 counts (3206) and confirms that there has been a decrease. The total count for the sites between Donna Nook and Scroby Sands has declined by approximately 38% compared to the mean of the previous five years (2019–2021 mean = 3080; 2014–2018 mean = 4296). The count for the Wash and North Norfolk SAC (i.e. the Wash + Blakeney) has decreased by approximately 21% (2019 – 2021 mean = 2883; 2014–2018 mean = 3658) over the same time periods, while Donna Nook showed a 57% decrease and Scroby Sands showed a 73% decrease. Fitted trends indicate that the Wash & North Norfolk SAC population recovered after the 2002 PDV epidemic, reached a maximum around 2015 at a level close to the pre-epidemic maximum and has declined sharply since then. However, the nature of this decline is still uncertain in terms of whether it represents the beginning of a sustained decline or a step change (similar to those seen in response to the PDV epidemics in the SEE SMU and for unknown reasons in the Shetland SMU). As the Wash and Blakeney counts represent the majority of the SEE SMU population, a similar trajectory is shown by the overall SMU counts.

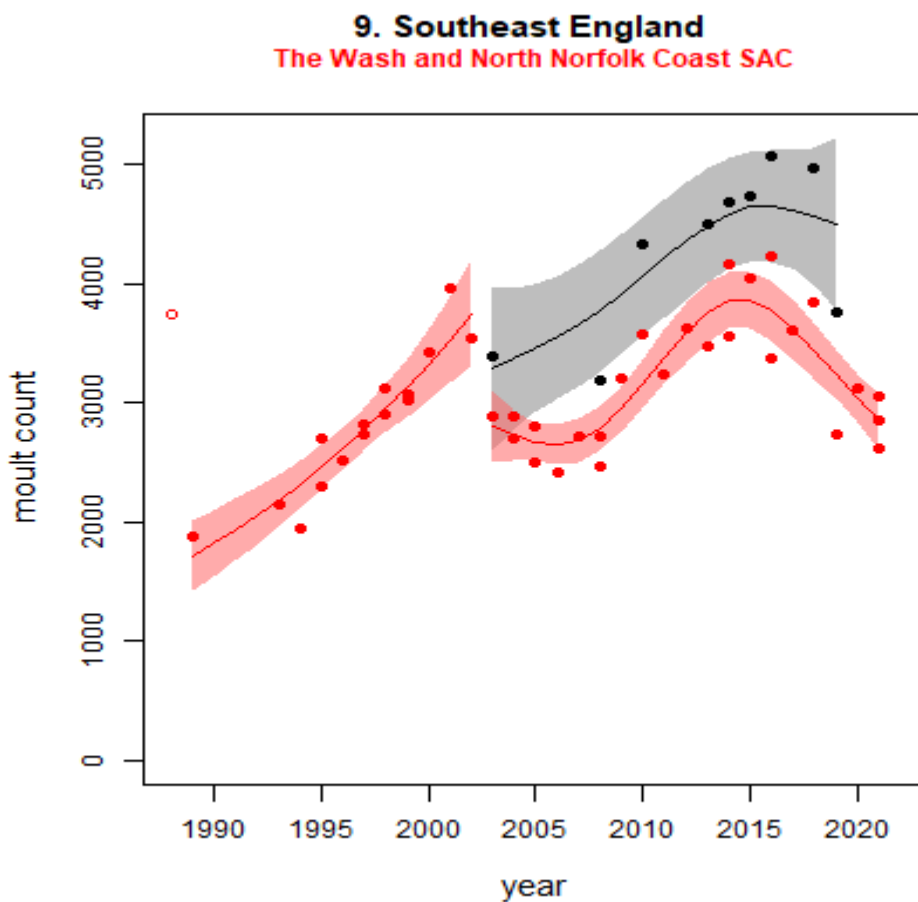


Figure 11. Counts of harbour seals in the Wash and North Norfolk SAC (red) and the total for the Southeast England SMU (grey) during the harbour seal moult in August, between 1988 and 2021, showing the changes in counts after the 1988 and 2002 PDV epidemics. Separate trend lines were fitted were selected (see Russell et al. 2021) to the 1989–2002 counts and post 2002 counts showing recoveries from the two PDV epidemics. Red lines illustrate the mean trend in harbour seal counts (and associated 95 % confidence intervals) for The Wash and North Norfolk SAC and the grey lines show the same for the SMU as a whole (between Donna Nook in Lincolnshire and Goodwin Sands off the Kent coast).

Overall, the harbour seal population in the study area has decreased by approximately 30% since 2018, and the decline appears to be widespread across the area. Comparing counts at the four main haulout areas, The Wash and the adjacent haulout areas at Donna Nook, Blakeney and Scroby Sands, all four areas have declined over the past four years. The patterns differ between sites, with the Wash, and possibly Scroby Sands, showing increases from around 2004 to 2016-18 followed by sharp declines, while at Blakeney there appears to have been a gradual decline over the entire period (2002 – 2021) and at Donna Nook the harbour seal counts were relatively stable until 2018 before declining sharply (figure 2). Counts divided into four subsections of the Wash show that the decrease in harbour seal counts since 2018 has occurred throughout the Wash and does not appear to be localised.

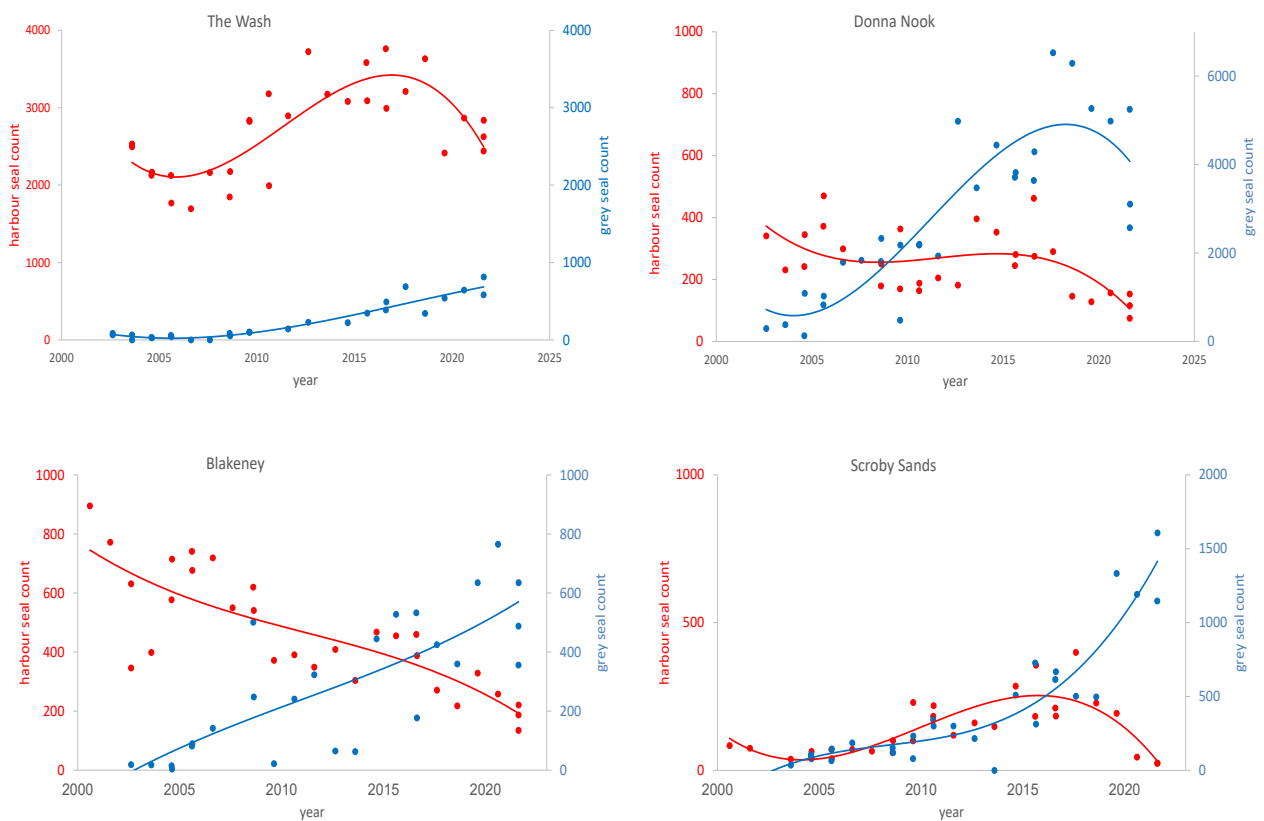


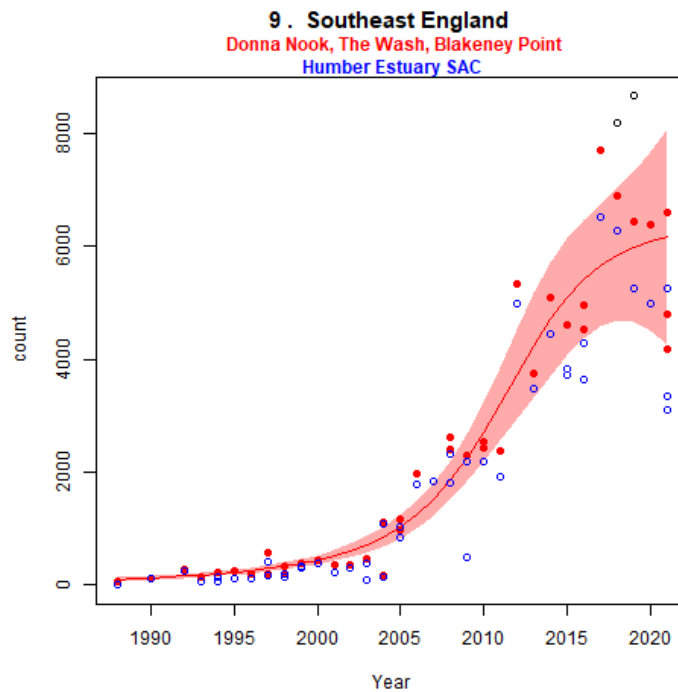
Figure 12. Counts of harbour seals (red) and grey seals (blue) for the period 2002 to 2021, in The Wash, at Donna Nook, Blakeney Point and Scroby Sands. Cubic polynomial lines have been fitted to the count data to illustrate the general patterns. A more formal model fitting procedure will be carried out in due course.

4. Grey seals

Counts of grey seals from surveys between 2017 and 2021 are shown in table 2. Figure 3 shows the trends in grey seal counts in the Humber Estuary SAC (i.e. Donna Nook) and along the coast from Donna Nook to Blakeney point, which are the grey seal haulouts within and adjacent to the Wash and North Norfolk SAC.

Table 10. Counts of grey seals at Donna Nook, the Wash, Blakeney and Scroby sands during August between 2016 and 2021

Date	12/08/ 2021	22/08/ 2021	23/08/ 2021	22/08/ 2020	11/08/ 2019	11/08/ 2018	11/08/ 2017	21/08/ 2016	5/8/ 2016
Wash	1001	583	813	644	540	253	688	491	387
Donna Nook	5248	3105	3339	4982	5265	6288	6526	4288	3640
Blakeney	356	488	635	765	635	360	425	177	533
Horsey	N/S	391	368	504	119	205	N/S	N/S	N/S
Scroby Sands	N/S	1146	1607	1191	1333	497	502	668	615
Total Donna Nook to Blakeney	6605	5713	6762	8086	6440	6901	7639	4956	4560

**Figure 3.** Counts of grey seals on the coast between Donna Nook (blue) and along the coast between Donna Nook and Blakeney (red) during the August surveys between 1988 and 2021. The red trend line (and associated 95% confidence intervals) represent the counts from Donna Nook to Blakeney (see Russell et al. 2021 for more details). The two black open circles indicate the available counts for the SMU as a whole.

The fitted trend (Fig 3) shows that the number of grey seals hauling out in the area has increased dramatically since the 2002 PDV epidemic (note that PDV epidemics are not associated with mortality events in grey seals), but that the rate of increase has clearly slowed and may have stopped over the past three to four years. The counts at Donna Nook, which held around 60% of the SEE SMU grey seal count in 2020 have declined, similar to the pattern seen in the harbour seal population for the Wash & North Norfolk SAC.

The grey seal trends differ between sites (Figs 2 & 3). At Blakeney, Scroby and in The Wash the counts have increased throughout the period.

The distribution of grey seals within the Wash has expanded since the late 2000s (Fig 4) and that expansion has been most pronounced in the last 5 years. During the 2008 and 2011 surveys, grey seals were observed on only five sites within the Wash. During the 2021 surveys grey seals were identified on 21 sites. Importantly, the most recent surveys show that grey seals are now present in small numbers on the sheltered sites in the creeks along the inner (southern) edge of the Wash (Figs 4 & 5).

Although most of the increase in numbers of grey seals has been at the sites on the outer banks at the Northeast corner of the Wash (Fig 5), grey seals are now extending into key harbour seal sites. Indeed, large groups are now found at sites along the edges of the deep channels between the inner banks. Small groups of 1 to 5 individual grey seals are now appearing on sites in the upper reaches of the tidal creeks used by harbour seals. To date, harbour seals still appear to use all the sites now also used by grey seals. Grey seals now outnumber harbour seals on the banks in the Northeast corner of the Wash and on the traditionally large harbour seal sites on Toft and Seal sands in the inner Wash.

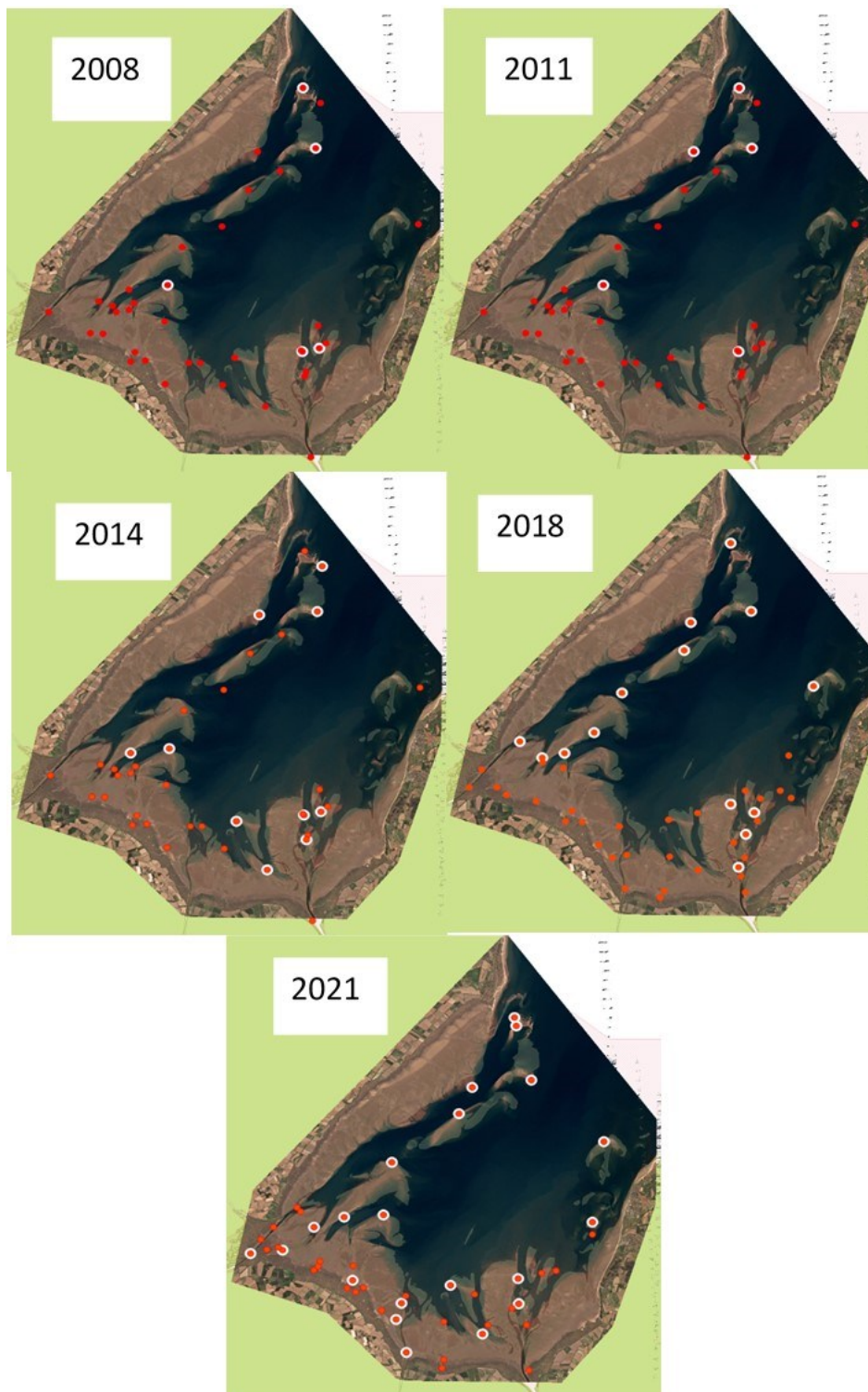


Figure 4. Distribution of harbour (red) and grey (white) seal haulout groups. For clarity the group size has been omitted (see fig 6 below).

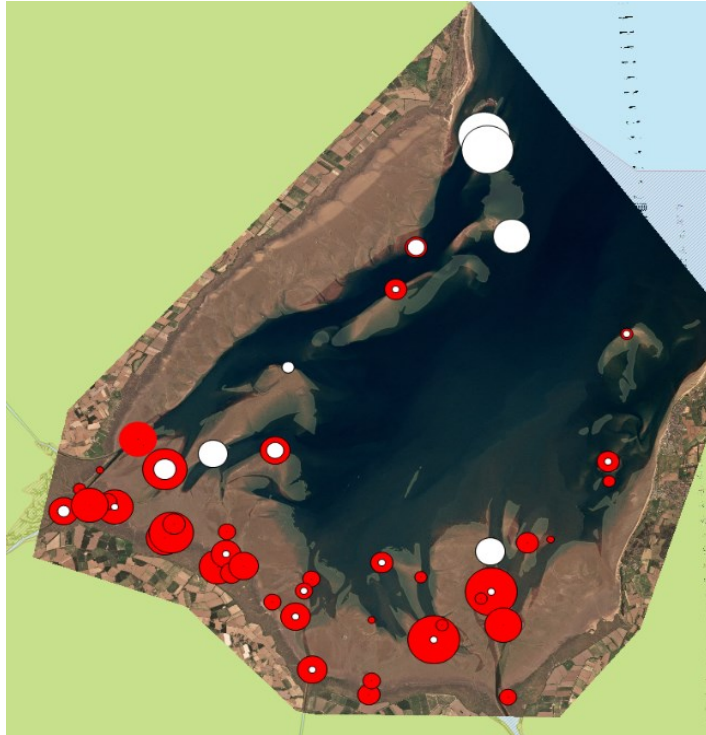


Figure 5. Distribution of harbour seal (red) and grey seal (white) haulout groups in the Wash during the 2021 moult surveys. Group size is indicated by the dot size (max). Grey seal points are superimposed on harbour seal points. All six of the pure white symbols represent sites where grey seal numbers now equal or exceed harbour seal numbers.

On visual inspection, the trends in grey and harbour seal counts by haulout group within the Wash (Fig. 2) does not indicate that the rate of harbour seal decline is closely related to the number of grey seals hauling out in the local area. Further investigation at a finer spatial scale is required, as there are indications that numbers of grey seals may have influenced harbour seal numbers at a limited number of specific sites.

Discussion

The 2020 and 2021 survey results confirm that there has been a significant decline in numbers of harbour seals along the coast between Donna Nook and Scroby Sands. The population appears to have reached a maximum around 2015 and has declined sharply since. The decline is widespread, with counts in all sub-sections of the SMU declining over the same period.

The recent counts suggest a decline of similar magnitude to that caused by the 2002 PDV epidemic. There are no reports of any disease event of sufficient magnitude to explain the drop in numbers, though un-documented/un-observed mortality from disease cannot be ruled out as a possible factor.

The results also indicate that the rapid increase in the numbers of grey seals in the same region has slowed and the numbers may have begun to decrease. Unlike the harbour seals, this change is currently localised to Donna Nook, the largest and most northerly haulout group. Counts of grey seals in the Wash, Blakeney and Scroby Sands have continued to increase.

The grey seal count has grown rapidly since the 2002 PDV epidemic. The magnitude of this change is dramatic; and when scaled up from counts to population it suggests that in 1988 harbour seals outnumbered grey seals ten to one, by 2020, grey seals outnumbered harbour seals by ten to one. Over the same period the total biomass of grey seals associated with these east coast haulout sites increased by at least a factor of 10. Grey and harbour seals generally exploit similar prey resources (Hammond &

Wilson 2016; Wilson & Hammond, 2016,2019), and grey seals are known predators of harbour seals (Brownlow et al. 2016), so it is possible that the increasing grey seal population is significantly affecting harbour seal population dynamics.

The distribution of grey seals in the Wash is expanding. Although most of the increase in numbers is accounted for by growth at sites on banks in the outer part of the Wash, there has also been a continual increase in the number of sites with grey seals. Importantly greys are appearing at sheltered sites in the tidal creeks in the inner Wash. These are important areas for harbour seal pupping. Unfortunately, there are no pup survey data for 2019,2020 or 2021 so no information on the locations of grey seals at the harbour seal breeding sites during the period of decline.

On visual inspection of the August counts, there is no clear indication that the numbers of grey seals hauling out within an area influences the harbour seal trend. Sub-sections of the Wash with widely differing grey seal numbers all show similar declines in harbour seal numbers.

Grey seals could potentially influence harbour seal haulout numbers by depressing the population through direct competition for prey or through direct predation. In addition, the risk of direct predation could directly influence the choice of haulout site or reduce the frequency of hauling out by harbour seals. We do not have any information with which to assess the likelihood of short-term changes in haulout frequency. The widespread nature of the decline discounts the possibility of local re-distribution being the cause of the observed declines. If redistribution were the cause, it would require movement out of the area. Preliminary results from recent surveys in the Thames (SCOS_BP_21/07; Cox *et al.*, 2020) also suggest a decrease in harbour seal counts in 2021. Any redistribution would therefore entail emigration from the SEE SMU probably into the European mainland population. The adjacent European population in the Wadden Sea has also levelled off and has remained apparently stable since 2013 (Wadden Sea 2021). However, because the Wadden Sea population is 6 to 8 times larger it is unlikely that the immigration of 30% of the SEE SMU population would have been detected.

The coincident levelling-off of the summer grey seal counts in Donna Nook may indicate that the overall seal population is approaching or has reached the SMU's carrying capacity. If that is the case, the future trajectory of the harbour seal population will be determined by the intensity of and mechanisms of competition. The extent and severity of such effects are unknown, but the magnitude of and coincident timing of the changes means that grey seals must be considered likely drivers of the observed harbour seal population trends.

Over the same period, i.e., since the 2002 PDV epidemic, there has been a rapid increase in construction of offshore wind farms. Figure 6 shows the trend in installed offshore wind generation capacity in the southern North Sea superimposed on the grey and harbour seal population trajectories. Clearly the trends in grey seal populations and wind farm developments are similar. With current information it is not possible to differentiate between the potential effects of these two stressors, but for conservation and management it is essential that their relative importance can be assessed. It is possible or perhaps likely that more than one natural and/or anthropogenic factor may be implicated in the decline.

Figure 4 highlights another potentially important issue. The 1988 PDV epidemic was unprecedented, but that may be simply a consequence of a lack of historical information. However, the recurrence of PDV in 2002 suggests that the virus may either be in circulation or may be sporadically introduced to the North Sea, e.g., as a result of influxes of Arctic seals. Irrespective of the source, we know that the current European harbour seal population is almost entirely comprised of susceptible animals and another major epidemic is probably imminent (Härkönen & Harding, 2010).

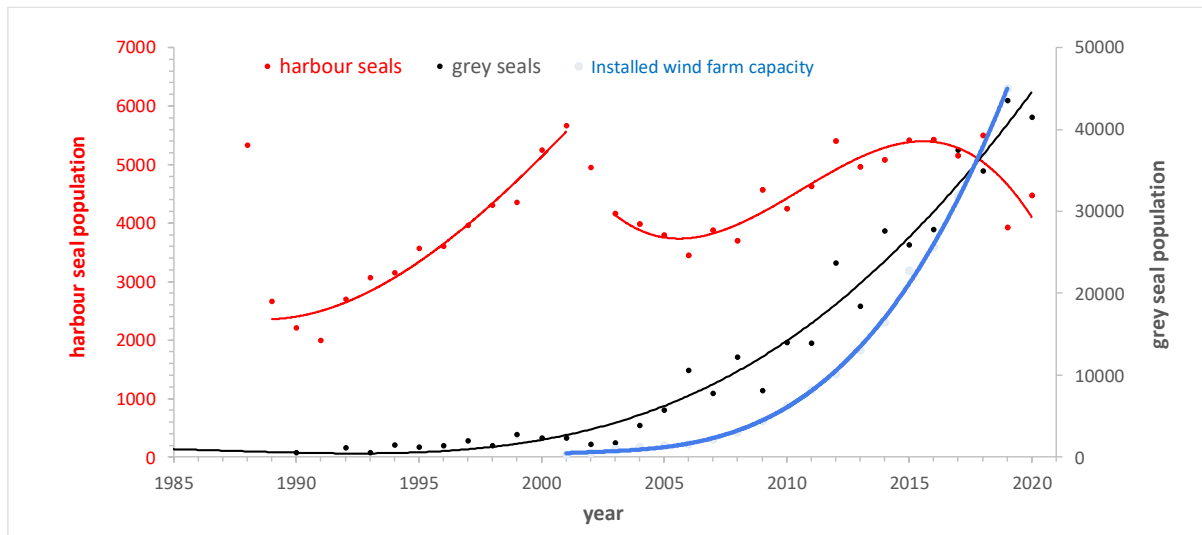


Figure 6. Population estimates of harbour seals (red) and grey seals (black) associated with haulout sites on the coast between Donna Nook and Scroby Sands during the summer between 1988 and 2020 and the trend in installed offshore wind generation capacity (blue). Fitted lines are polynomials for illustration.

The recovery from the 1988 epidemic was a continuous increase until the next epidemic (Figure 1). The recovery from the 2002 epidemic was much slower and the population reached an asymptote prior to the recent decline. The post 2002 recovery coincided with the rapid growth of grey seal numbers and predated the rapid increase in offshore wind farm construction. If a third PDV outbreak occurs soon, the harbour seal population will have to recover in a significantly different environment, with a much larger population of potentially competing grey seals. We do not know what impact the grey seal population will have on the ability of harbour seals to recover.

The variability in the proportion of the population hauled out, and thus variability in counts, means that multiple counts within a year will be required to robustly estimate the scale of the decline and track its trajectory. Therefore in 2022, we plan to carry out a series of three or four complete August surveys between Donna Nook and Scroby Sands and at least one survey during the harbour seal breeding season to estimate pup production.

A report commissioned by Natural England outlined potential future avenues of research and reviewed the current seal telemetry, diet, and health data, which in addition to the survey data, would form the basis for such future work (Russell *et al.* 2021). In brief, there is a clear and pressing need for additional research in the short to medium term to:

- Reliably assess the scale and timing of the decline and monitor its progress
- identify and if possible, rule out as many potential anthropogenic impacts as possible, especially given the rapidly changes anthropogenic landscape
- identify the mechanisms, scale and intensity of competition between grey and harbour seals in the southern North Sea
- establish the likely impact of grey seals on harbour seal populations and to predict the likely consequences of future grey seal population trends
- to investigate the likely impacts of a recurrence of PDV on harbour seal populations in the southern North Sea.

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Report on 2021 Seal Surveys in the Greater Thames Estuary

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Introduction

There are two species of seal that are resident in the UK, both of which are found in the Greater Thames Estuary – the harbour (or common) seal (*Phoca vitulina*) and the grey seal (*Halichoerus grypus*). The southern North Sea grey seal population has been increasing, and until more recently, harbour seal populations on the east coast of England had also generally been increasing (punctuated by major declines due to phocine distemper virus (PDV) outbreaks in 1988 and 2002) (SCOS, 2020). Consistent with other parts of the east coast of England, the Greater Thames Estuary seal populations have seen an increase in numbers, with both harbour and grey seal populations demonstrating high annual growth rates (8.99% pa, bootstrap 95% CI 6.79-11.19 for harbour seals; and 12.62% pa, bootstrap 95% CI 7.71-17.52 for grey seals) (Cox et al., 2020). More recent counts by the Sea Mammal Research Unit (SMRU) in the south-east England Seal Management Unit (SMU), however, may indicate the start of a decline in the harbour seal population of the Wash, North Norfolk Special Area of Conservation (SAC) (SCOS, 2020). The trajectory of other harbour seal populations around the UK coast is variable; generally, populations on the east coast of Scotland and Northern Isles are declining, and those in western Scotland are stable or increasing. With this national context in mind, there is a clear need to continue monitoring the trends in abundance of the Greater Thames Estuary harbour and grey seals.

Surveys of the area were carried out by SMRU and Bramley Associates between 2004 and 2012 (Bramley and Lewis, 2004; Bramley Associates 2005, 2007 and 2010 survey data, unpublished; Bramley Associates, 2012; SCOS, 2020) and ZSL began annual surveys in 2013 (Cox et al., 2020). Harbour seal pup surveys were also carried out in 2011 (SMRU) and 2018 (ZSL). This report presents the latest counts for the Greater Thames Estuary as well as results from the third pup survey of the area (both the population and harbour seal pup surveys had been postponed in 2020 because of the Covid-19 pandemic). In addition, this report presents the findings from an additional aerial survey which was conducted to better understand seal movement and the impact of the multi-day survey methodology used by ZSL.

Methodology

Population and pup surveys were carried out from a light fixed-wing aircraft (Rallye model), based at Southend airport.

The harbour seal pup survey is timed to coincide with when the peak number of pups are expected and the population survey takes place over the harbour seal moult period, which follows whelping. Typically, on the east coast of England, pupping takes place at the end of June-start of July and the

moult occurs over the first two weeks of August. The surveys take place within two hours either side of low tide. To minimise environmental variability, surveys should also ideally happen between 12:00 and 19:00 (SCOS, 2020). However, where Ministry of Defence (MoD) Danger Areas exist, and airspace

restrictions are in place (as in the Greater Thames Estuary) this rule is relaxed. Surveys of coastline and sandbanks that overlap with MoD Danger Areas take place over the weekend, subject to agreement from MoD Range Control.

The pup surveys were scheduled to take place 1st July 2021 – 3rd July 2021. Surveys were successfully completed on 1st and 2nd July, however, weather warnings prevented aircraft flying on 3rd July. Despite efforts, it was not possible to reschedule the 3rd July survey of the Southend area of the Greater Thames Estuary. A total pup estimate for the Greater Thames Estuary is therefore not provided in the results, however, the number and location where pups were recorded in the remainder of the estuary is included.

The population surveys took place on 7th, 8th, and 10th August 2021. Typically, the survey would happen over three consecutive days, however, storms on 9th August prevented this.

A repeat survey of the coastline and sandbanks covered on the 10th was carried out on 11th August. ZSL aim to conduct the seal counts over three consecutive days because the survey team have not found it possible to survey all the coastline and sandbanks of the Thames estuary within a single four-hour window, two hours either side of low tide. The estuary is broadly divided into three 'sections' – Margate, Felixstowe, and Southend – and each section covered on a different day. It has been assumed that the movement of seals between these different sections is limited and therefore the risk of double counting or missing seals that move between survey days is low. A repeat survey was carried out for the first time this year to better understand the impact of multi-day surveys on the counts.

The location of seals was recorded using a Garmin eTrex10 handheld GPS unit. The same unit was used to record the path of the aircraft. Hauled out seals were photographed using a Canon EOS 250D body and Canon EF 100-400mm f/4.5-5.6 L IS USM lens. After the survey, the photos are used to count seal numbers and species at each haul-out site. This is done independently by two people, their counts are compared, any disparities discussed, and a final count agreed.

Results – Pup Survey

On 1st and 2nd July, a total of 1,135 seals were counted, this included 21 harbour seal pups (assumed to be born this year based on size and proximity to an adult harbour seal, presumed to be the mother), 230 harbour seals (all age classes excluding pups) and 99 grey seals (all age classes). There were 785 seals counted that were not identified to species level or age class (pup vs. older). These seals were at two locations in the Estuary – Kentish Knock sandbanks and the Goodwin Sands. A wind farm constructed near the Kentish Knock sandbanks prevented the aircraft flying near enough to the seals to capture photos from which species could be determined with reasonable confidence. Likewise, airspace restrictions over the Goodwin Sands prevented the aircraft flying close enough for the photos needed (except for one location – Goodwin Knoll).

The aerial survey effort that was possible is presented in **Fig. 1**. The distribution and count of harbour seal pups that were seen is presented in **Fig. 2**. Pups were seen across five locations – Margate Sands, Pegwell Bay, Goodwin Knoll, the Blackwater, and Hamford Water. **Fig. 3** shows the distribution and count of harbour and grey seals, for which species identification was possible (all age classes for both species), as well as seals not identified to species level.

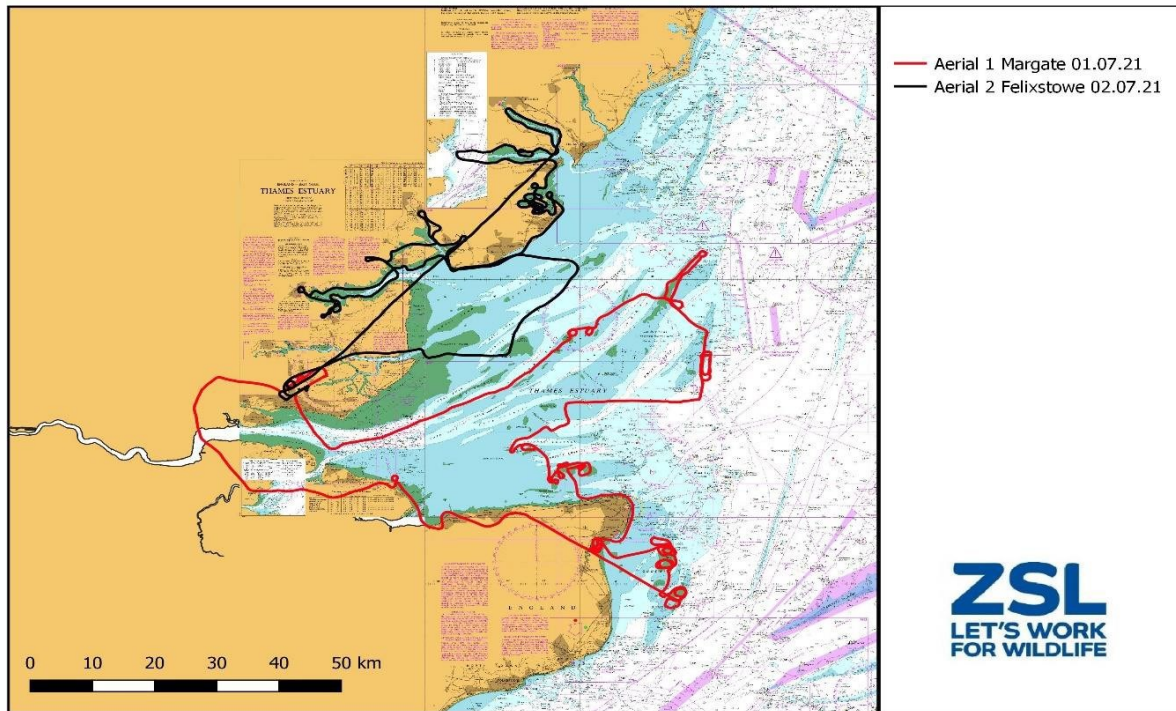


Fig. 1 – Aerial survey effort for pup count, 2021 © Crown Copyright, 2021. All rights reserved. License No. EK00120130801

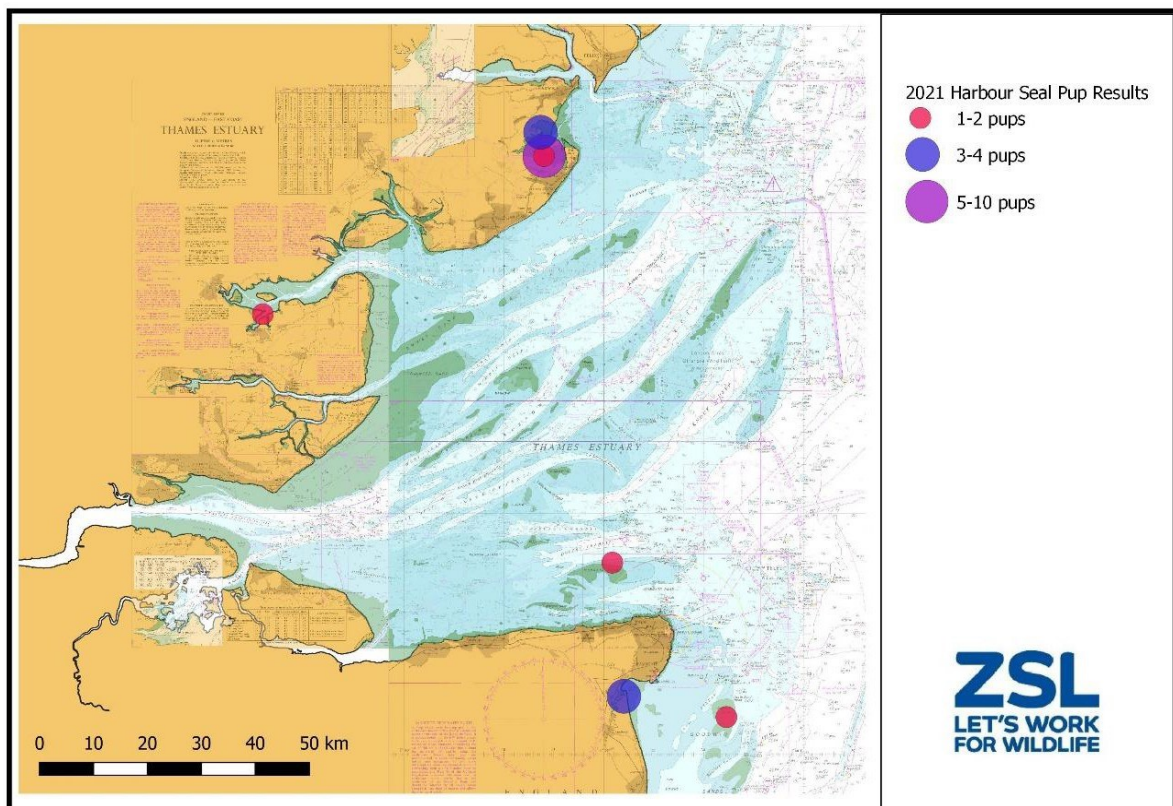


Fig. 2 – Counts of harbour seal pups, 2021 © Crown Copyright, 2021. All rights reserved. License No. EK00120130801

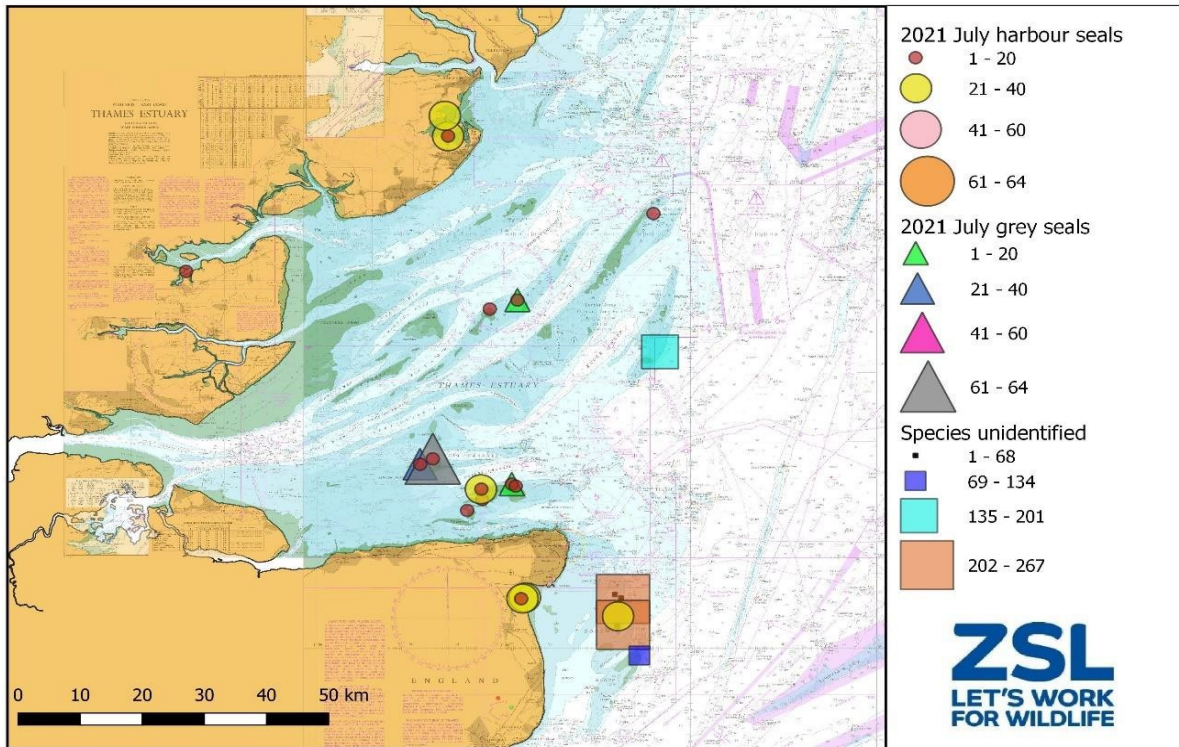


Fig. 3 – Counts of harbour seals, grey seals, and seals (not ID-ed to species) (all age classes), 2021

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Results – Population Survey

A map of the aerial effort for the population survey is presented in Fig. 4. No boat surveys or land-based surveys were conducted in 2021 as it was possible to cover the entire estuary by aircraft.

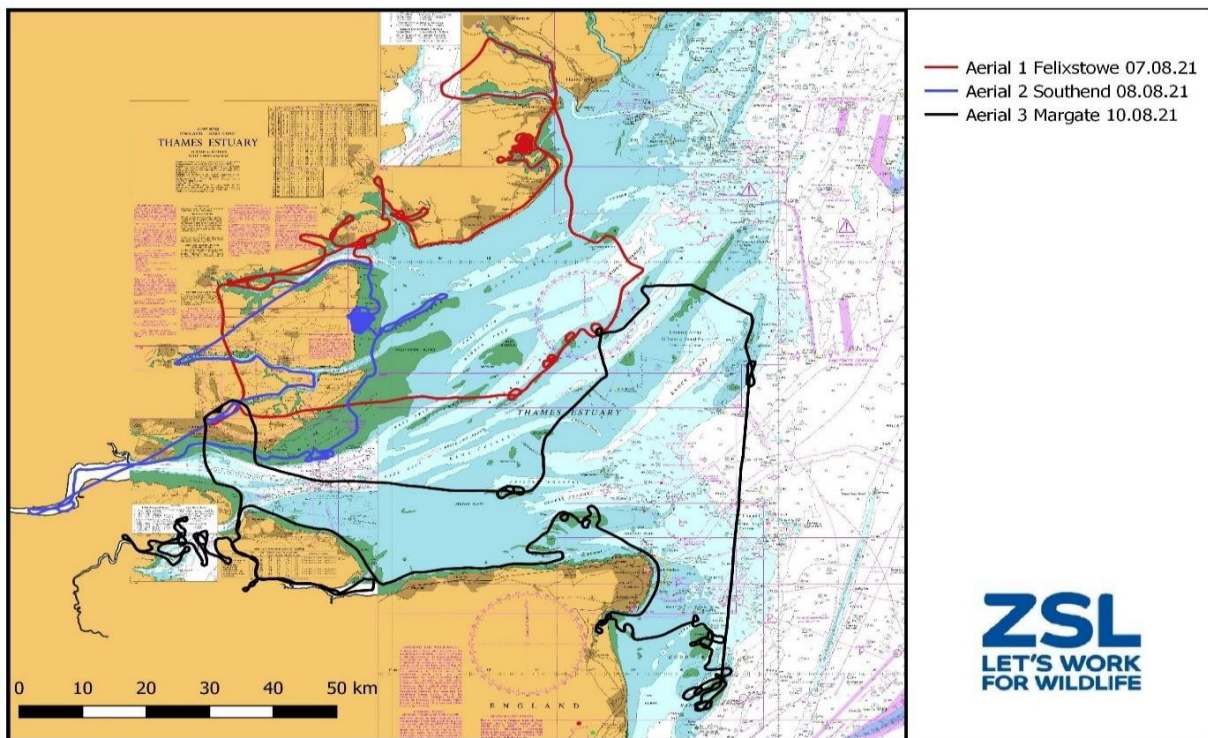


Fig. 4 – Aerial survey effort for population count, 2021 © Crown Copyright, 2021. All rights reserved. License No. EK00120130801

Counts and estimated harbour and grey seal population size for the Thames is presented in **Table 1** below – the latest counts are shown in bold. Populations are estimated by a scaling up of count numbers. This is based on the estimated proportion hauled out during surveys – for harbour seals, this is 72% (0.72, 95% confidence intervals (CI): 0.54-0.88), and for grey seals, this is 23.9% (0.239, 95% CI:

0.192-0.286) (Lonergan et al., 2013; SCOS-BP-16/03).

Table 1: Counts and harbour and grey seal population estimates for the Thames

	2013	2014	2015	2016	2017	2018*	2019	2021**
Harbour seal count	482	489	451	694	795	738	671	498***
Harbour seal population estimate (95% CI)	669 (548-893)	679 (556-906)	626 (513-835)	964 (789-1285)	1104 (903-1472)	1026 (840-1369)	932 (763-1243)	692 (566-922)
Grey seal count	203	449	454	481	575	596	775	749***
Grey seal population estimate (95% CI)	849 (710-1057)	1879 (1570-2339)	1900 (1587-2365)	2013 (1682-2505)	2406 (2010-2995)	2490 (2080-3099)	3243 (2710-4036)	3134 (2619-3901)
Total seal count	685	938	905	1175	1370	1334	1446	1247

*Count completed by SMRU

**Count not completed in 2020 due to Covid-19 restrictions

***Counts updated post press release in September 2021

This year's count excludes seals observed on Kentish Knock sandbanks. As noted above for the pup survey, proximity of the sandbanks to the wind farm meant it was not possible to fly close enough to the seals to take photos from which a total count could be taken, or species identified. Based on observation during the flight, it is estimated that there were ~200 hauled out seals; and based on previous surveys, it is expected to be a mixed species group dominated by grey seals. Airspace restrictions over Goodwin Sands were temporarily lifted to allow those sandbanks to be surveyed for the population count, therefore numbers above include those haul-out sites.

Fig. 5 below shows the distribution and counts of harbour and grey seals in the Thames in 2021. **Fig. 6** below shows the change in harbour and grey seal counts in the Thames since surveys began in 2003 (Cox et al., 2020).

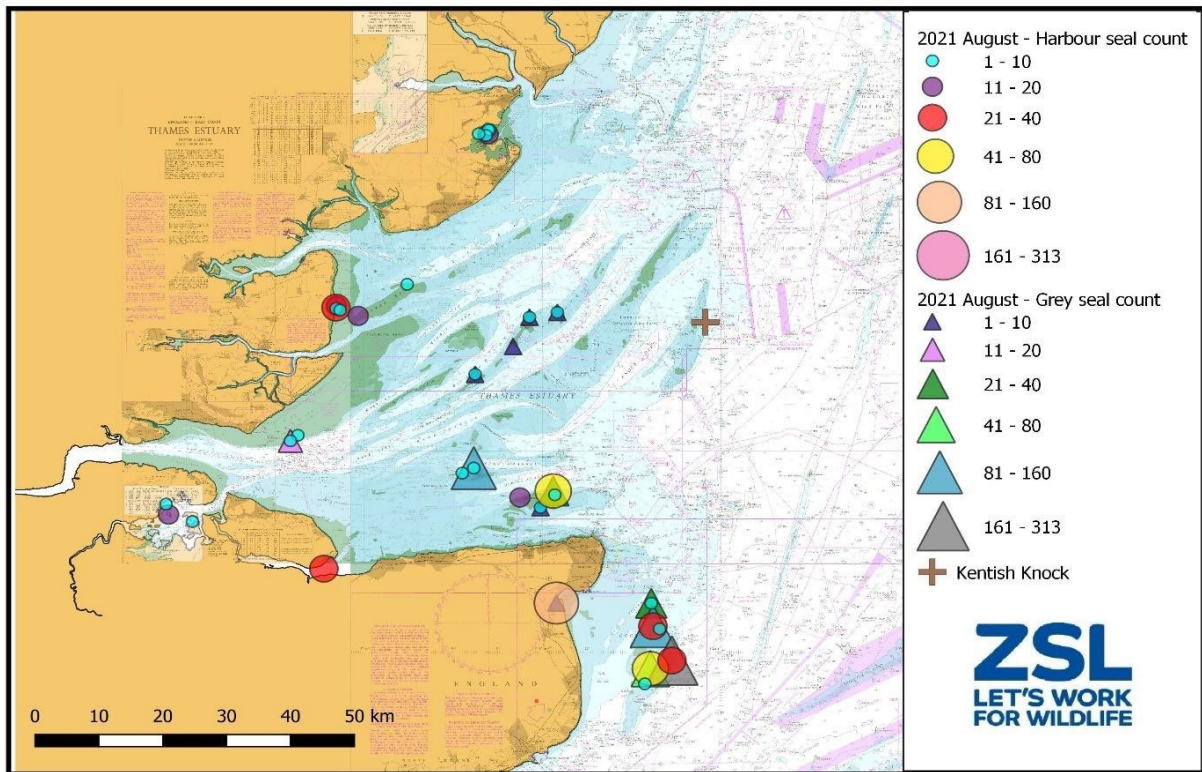


Fig. 5 – Counts of harbour seals and grey seals, 2021 © Crown Copyright, 2021. All rights reserved. License No. EK00120130801

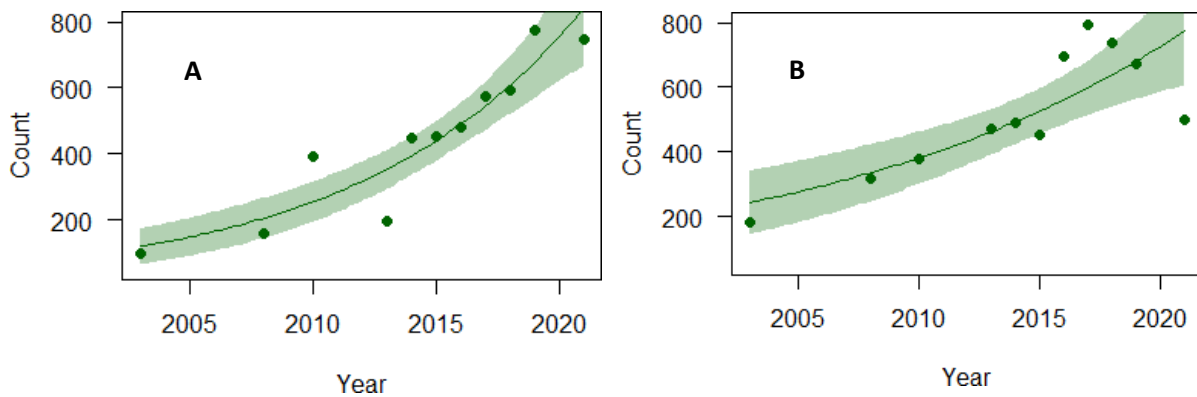


Fig. 6 – Counts and fitted trend for Thames grey seals (A) and harbour seals (B), 2003-2021, 95% CI shown, by SMRU, Bramley Associates and ZSL (see Cox et al., 2020)

Results – Repeat Survey

On 11th August, 398 harbour seals and 635 grey seals were counted (total of 1,033). This count excludes Kentish Knock sandbanks, for the reason explained above. This was a repeat of 10th August flight (same areas surveyed in the same order), during which 352 harbour seals and 714 grey seals were counted, totalling 1,066 seals (similarly excluding Kentish Knock sandbanks). See Fig. 7 for survey route (which can be compared to the ‘Margate route’ in Fig. 4) and Fig. 8 for the distribution and count of seals. Table 2 shows a direct comparison of locations (sometimes combined multiple haul-out sites to represent sandbanks/one location) and seal counts for the two days.

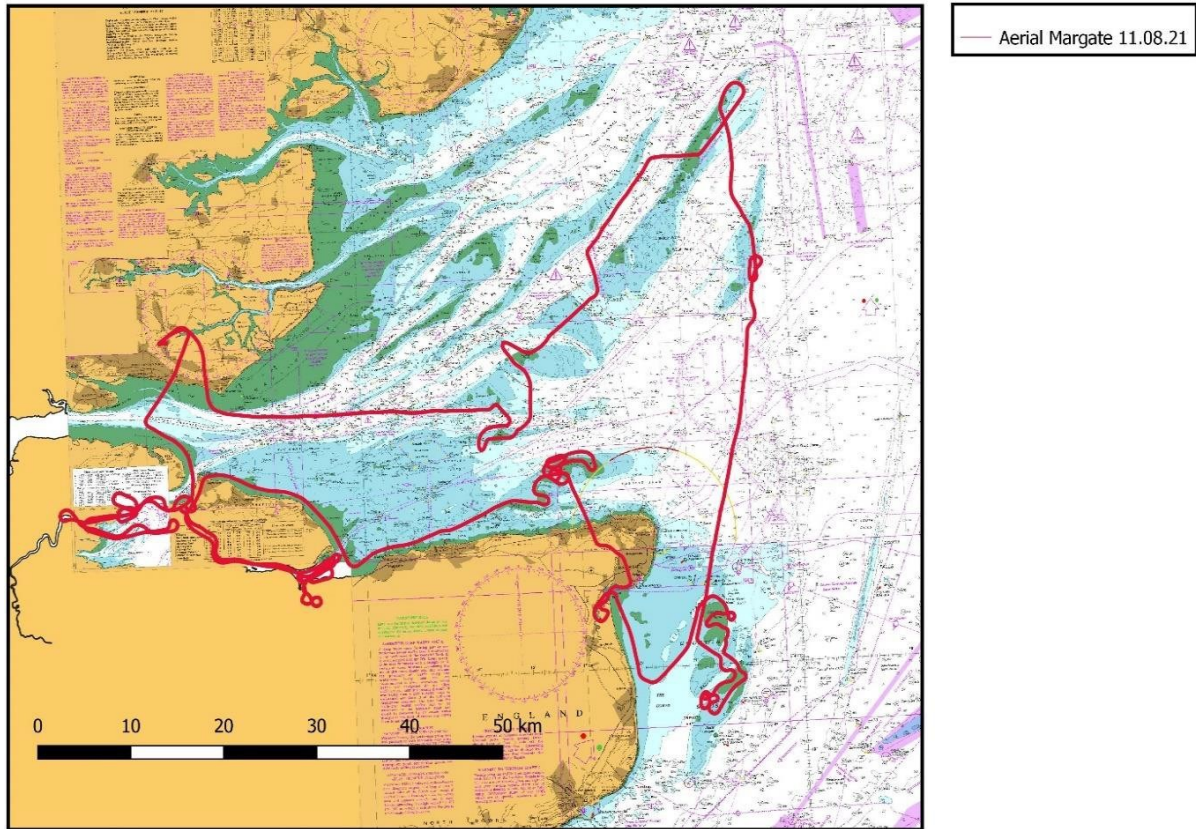


Fig. 7 – Aerial survey effort for repeat count, 11 August 2021 © Crown Copyright, 2021. All rights reserved. License No. EK00120130801

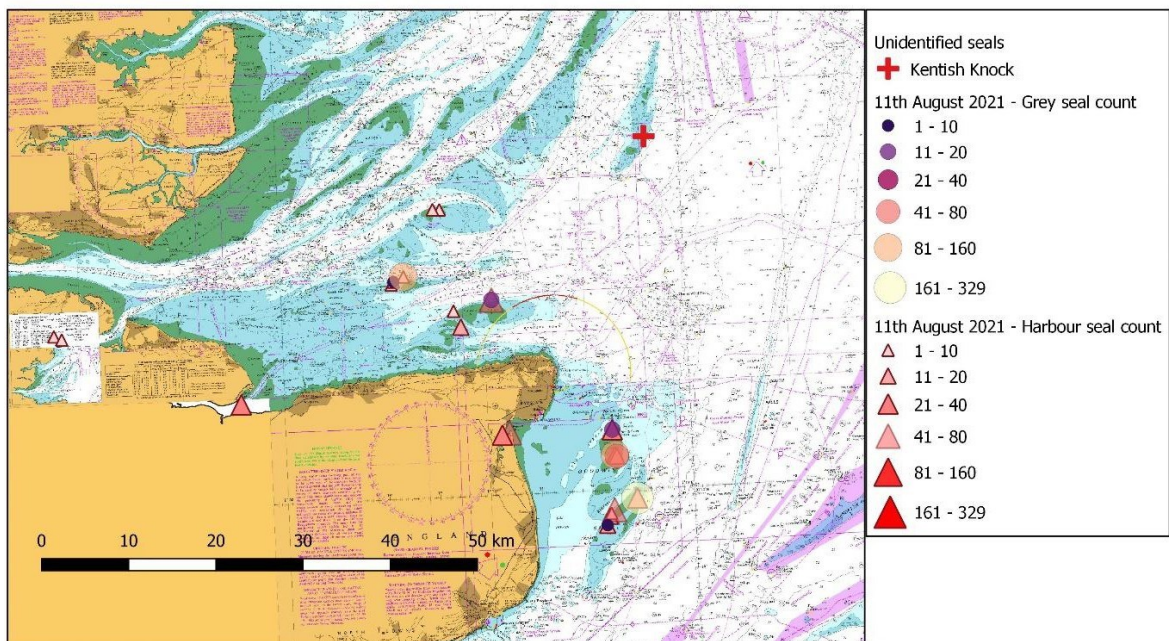


Fig. 8 – Count of harbour seals and grey seals, 11 August 2021 © Crown Copyright, 2021. All rights reserved. License No. EK00120130801

Table 2 – Comparison of counts on 10th and 11th August 2021

Location	No. of harbour seals on 10 th	No. of harbour seals on 11 th	No. of grey seals on 10 th	No. of grey seals on 11 th
Medway	21	17	0	0
Swale	26	31	0	0
Margate Sands	70	71	28	20
Pegwell Bay	97	97	3	0
Goodwin Sands	118	160	574	494
Shingles Patch	0	2	0	0
Pan Sand Ridge	20	20	109	121
Total	352	398	714	635

Discussion

In August 2021, a total of 498 harbour seals were counted, compared with an average of 735 for three surveys in 2017-2019, and an average of 545 for three surveys in 2014-2016. There has been an increase in harbour seal counts since surveys began in 2013 to 2017, and since then there appears to have been a gradual decline. Some variability year-to-year is to be expected, associated with the proportion of the population hauled out and available to count. However, the change in counts since 2017 could also reflect a true decline in harbour seal numbers and requires ongoing monitoring. Considering changes observed in the Wash (the 2019 count was ~27.5% lower than the mean of the previous five years, 2014-2018) and declines elsewhere in the UK, this could be of concern.

In August 2021, a total of 749 grey seals were counted. There has been a sustained increase in grey seal counts in the Thames year-on-year, consistent with the rest of the east coast of England, up until this year. It is suspected that the lower count this year reflects the missed Kentish Knock sandbanks rather than a true decline - a large group of seals were observed at Kentish Knock but could not be photographed and which is typically grey seal dominated. The long-term trend will become clearer with continued monitoring.

Whilst a total pup count for the Thames in 2021 cannot be provided, the survey results show that the Thames estuary is important harbour seal pupping habitat. Further surveys of the entire estuary will be important to build on pup surveys in 2011 and 2018 and monitor trends, especially in determining the cause of any change in population size.

Repeat survey

A comparison of **Figs. 5 and 8** and **Table 2** shows that seals, of both species, were largely seen at the same locations on the Margate route over both days, and in similar numbers, suggesting that there is not a large amount of movement of seals in this short period of time. Overall, there is a +13.1% change in harbour seal counts on 10th and 11th, and -11.1% change in grey seal counts on the same dates. The numbers of each species at each haul-out site do not match exactly though, with the largest difference being at Goodwin Sands (harbour seals +35.6% and grey seals -13.9%), therefore it is possible that there is some level of seal movement within and, most importantly, outside of the Margate 'section'. This therefore means there could be some risk of double counting or missing seals. The difference in numbers between the days could also be due to other reasons though: such as, missed seals/observer bias, errors in photo analysis or environmental factors such as differences in tidal state (although the timing of the surveys is such to minimise this kind of variability). Whilst further, more-resource intensive, research would be needed to fully understand seal movement patterns in the estuary, these results, except for harbour seals at Goodwin Sands, do suggest low level of movement of seals between areas across the 3-day survey period and gives us more confidence that seals are not missed or double counted in large numbers because of multi-day surveys.

Limitations

The surveys conducted in the Thames follow the recognised methodology for harbour seal moult counts in sandy/muddy estuaries, however, there are uncertainties associated with it and limitations of the 2021 survey specifically. These are outlined below:

- When possible, ZSL surveys are timed to coincide with a spring low tide to maximise the time for which coastline and sandbanks are exposed, and therefore available to be surveyed. However, with other constraints on survey dates, such as airspace restrictions, etc., this is not always possible, as was the case this year. As such, certain sandbanks were covered over at the time of surveying – the Barrows, Gunfleet, Long Sands and Knock John.
- Whilst the surveys only took place in fine weather, the conditions in the days around the surveys were unsettled with heavy rain. The third day of the pup survey had to be cancelled and the third day of the population survey postponed by one day because of this. It is possible that the unsettled weather could have affected haul-out behaviour (SCOS, 2020) and therefore the seals available to count.
- Population estimates made from counts do contain considerable uncertainty (SCOS, 2020). During their annual moult, harbour seals spend longer hauled out and the highest proportion of the population is available to count. Some seals will still be at sea though. There is just one UK study that estimates the proportion of harbour seals hauled out during the moult (0.72, 95% CI 0.54-0.88) (Lonergan et al., 2013) and it is this figure that is used to calculate a population estimate. Furthermore, whilst environmental variability is reduced by consistent timing of surveys, the conversion/scaling up factor only represents adult seals – haul-out behaviour could vary with age and sex and the age structure and sex ratio will change over time. The age-sex composition for the Thames population is not known. As such, counts should be considered the minimum number of harbour seals in each area, and population estimates from scaled up counts should be treated with a certain level of caution.
- Counts and especially species ID depend on the quality of the aerial photography and ideally capturing images of the animal's faces/heads. Every effort is made to ensure this but especially where the group of seals is particularly large it is not possible to get a photograph of every animal's face, therefore some assumptions must be made based on other seals in the photographs, position on the shore, position relative to each other, size, etc.

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Provisional Regional PBR values for Scottish seals in 2022

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Abstract

This document estimates PBR values for the grey and harbour seal “populations” that haul out in each of the seven 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:

-]Harbour seals: except for a single count in part of the East Scotland SMU there are no additional count data, so all SMU population estimates, N_{min} values and PBRs are the same as last year.
- Grey seals: a new analysis of the proportion of grey seals hauled out and counted during surveys has reduced the grey seal population estimates by 5.2%. The N_{min} value and the resulting PBRs have been reduced by approximately 3.5%.

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:

$$PBR = N_{min} \cdot (R_{max}/2) \cdot F_R$$

where:

PBR is a number of animals considered safely removable from the population.

N_{\min} is a minimum population estimate (usually the 20th percentile of a distribution).

R_{\max} 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.

F_R 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:

N_{\min} 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 N_{\min} . (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 (Loneragan 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: A revised analysis of GPS/GSM telemetry data from 60 grey seals tagged between 2005 and 2018, allowed more accurate identification of haulout times (SCOS-BP 21/02). The revised estimate of proportion of seals hauled out during the survey window was 25.2% (95% CI: 21.5 – 29.1%), compared with the previous estimate of 23.9% (95% CI: 19.2 - 28.6%) (Russell et al. 2016 SCOS-BP 16/03). The 20th centile of the distribution of scalars from counts to abundances derived from the revised estimate is 3.73, approximately 3.5% lower than the previous scalar.

R_{\max} 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% (Loneragan 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. (Loneragan et al. 2011b). However, the large grey seal population at Sable Island in Canada has grown at nearly 13% p.a. for long periods (Bowen et al. 2003).

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.

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

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 and have not shown any signs of recovery.

2) Western Isles ($F_R = 0.5$) **this may be revised after discussion of the SCA designation for the Western Isles SMU)*

Population was apparently undergoing a protracted but gradual decline during the 2000s, but the 2011 count was close to the pre-decline numbers and a trend analysis suggested no significant change since 1992. 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. The most recent count for the Western Isles was 25% higher than the previous count. On that basis there may be an argument for increasing the recovery factor to bring it in line with the other western Scottish management areas. However, there is an existing conservation order in place for the management unit and it is therefore recommended that the recovery factor is left at 0.5 and reviewed again when a new count is available for the larger, adjacent West Scotland region.

3) West Scotland ($F_R = 1.0$)

The population is largely closed, likely to have limited interchange with much smaller adjacent populations. The most recent count was the highest ever recorded and the population is apparently stable or increasing.

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 or increasing. The intrinsic population growth rate is taken from other similar populations.

5) Moray Firth ($F_R = 0.1$)

Counts for 2019 in the Moray Firth were similar to the previous 5 years, confirming the absence of any overall trend over the past 15 years. The neighbouring Orkney and Tay populations are continuing to undergo unexplained, rapid and catastrophic declines in abundance. Data available from tracking studies suggest there is movement between these three areas. In the absence of a sustained increase in the Moray Firth counts it is recommended that the F_R should be left at its previously recommended 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. All UK populations are either increasing or apparently stable at the maximum levels ever recorded and therefore assumed to be at or close to their carrying capacities (Loneragan et al., 2011b; Thomas et al., 2019; Russell et al., 2019). Available telemetry data and the differences in the regional patterns of pup production and summer haul-out counts (Loneragan et al. 2011a) also suggest substantial long-distance movements of individuals.

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

Seal Management Area	Area Covered
1 Southwest Scotland	English border to Mull of Kintyre
2 West Scotland	Mull of Kintyre to Cape Wrath
3 Western Isles	Western Isles incl. 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 Scotland	Fraserburgh to English border

Results

PBR values for grey and harbour seals for each Seal Management Area for with the full range of F_R values from 0.1 to 1.0 are given in table 1 for harbour seals and table 2 for grey seals. In each table the value corresponding to the recommended F_R is highlighted

Table 1. Potential Biological Removal (PBR) values for harbour seals in Scotland by Seal Management Unit for the year 2021. Recommended F_R values are highlighted in grey cells.

Seal Management Area	2016-2019		PBRs based on recovery factors F_R ranging from 0.1 to 1.0										selected	
	count	N_{min}	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1.0	F_R	PBR
1 Southwest Scotland	1709	1709	10	20	30	41	51	61	71	82	92	102	0.7	71
2 West Scotland	15600	15600	93	187	280	374	468	561	655	748	842	936	1.0	936
3 Western Isles	3532	3532	21	42	63	84	105	127	148	169	190	211	0.5	105
4 North Coast & Orkney	1405	1405	8	16	25	33	42	50	59	67	75	84	0.1	8
5 Shetland	3180	3180	19	38	57	76	95	114	133	152	171	190	0.1	19
6 Moray Firth	1077	1077	6	12	19	25	32	38	45	51	58	64	0.1	6
7 East Scotland	343	343	2	4	6	8	10	12	14	16	18	20	0.1	2
SCOTLAND TOTAL	26846	26846	159	319	480	641	803	963	1125	1285	1446	1607		1147

Table 2. Potential Biological Removal (PBR) values for grey seals in Scotland by Seal Management Unit for the year 2021. Recommended F_R values are highlighted in grey cells.

Seal Management Area	2016-2019		PBRs based on recovery factors F_R ranging from 0.1 to 1.0										selected	
	count	N_{min}	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1.0	F_R	PBR
1 Southwest Scotland	517	1927	12	23	35	46	58	69	81	92	104	116	1.0	116
2 West Scotland	4174	15554	93	187	280	373	467	560	653	747	840	933	1.0	933
3 Western Isles	5773	21512	129	258	387	516	645	774	904	1033	1162	1291	1.0	1291
4 North Coast & Orkney	8599	32043	192	385	577	769	961	1154	1346	1538	1730	1923	1.0	1923
5 Shetland	1009	3760	23	45	68	90	113	135	158	180	203	226	1.0	226
6 Moray Firth	1657	6175	37	74	111	148	185	222	259	296	333	370	1.0	370
7 East Scotland	3683	13724	82	165	247	329	412	494	576	659	741	823	1.0	823
SCOTLAND TOTAL	25412	94695	568	1136	1705	2273	2841	3409	3977	4545	5114	5682		5682

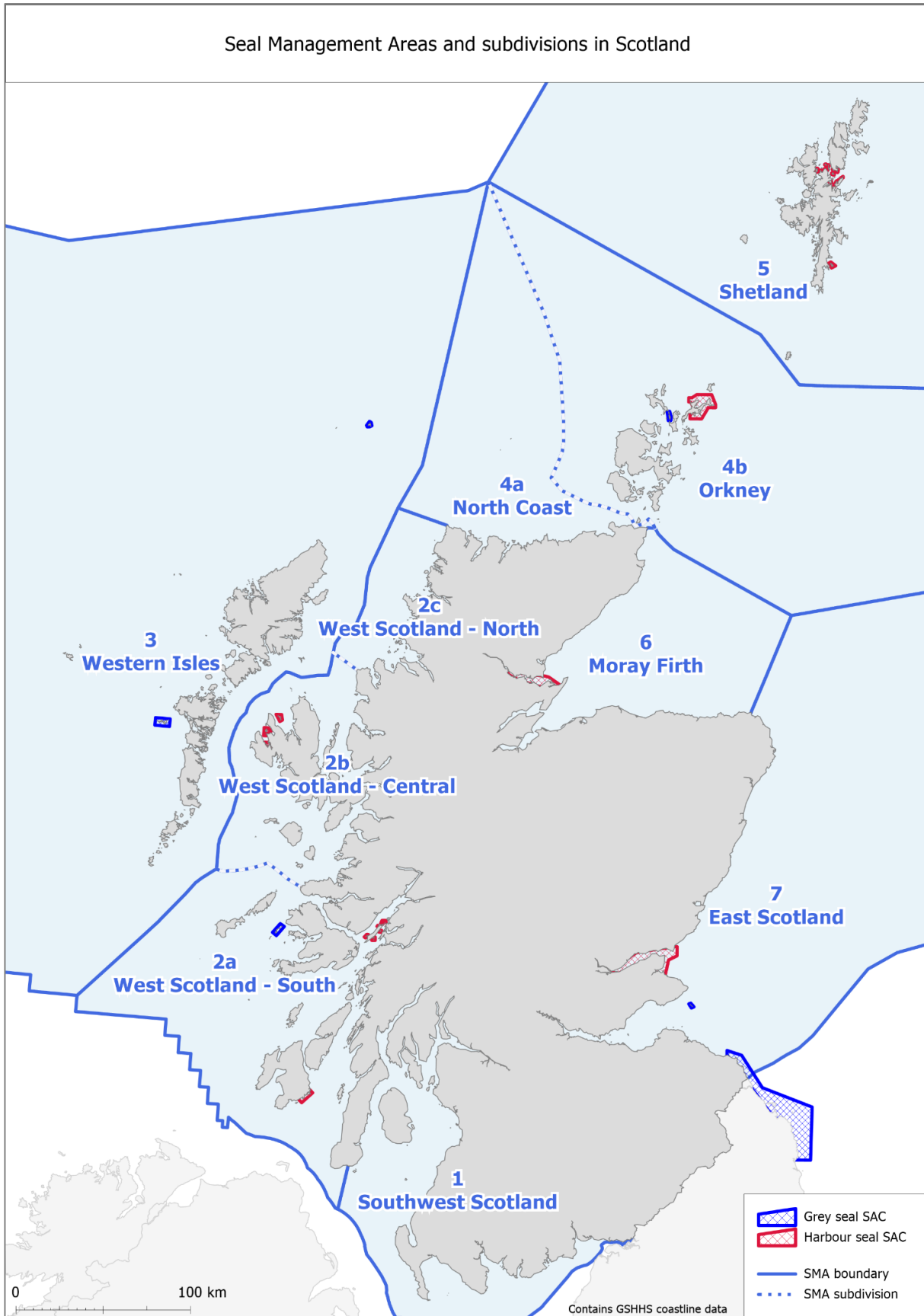


Figure 1. Seal management areas in Scotland. For purposes of PBR calculations West Scotland is treated as a single management unit.

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