Review Of Demographic Parameters And Sensitivity Analysis To Inform Inputs And Outputs Of Population Consequences Of Disturbance Assessments For Marine Mammals

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Review Of Demographic Parameters And Sensitivity Analysis To Inform Inputs And Outputs Of Population Consequences Of Disturbance Assessments For Marine Mammals

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1 Executive summary

SMRU Consulting developed the interim Population Consequences of Disturbance Framework (iPCoD) in 2013 to provide a transparent and robust framework for the prediction and assessment of the effects of marine and offshore renewables projects on the five UK priority marine mammal species.

In 2019, SMRU Consulting were commissioned by Marine Scotland to update the recommended demographic parameters for modelling and to carry out a sensitivity analysis of the outputs to misspecification of population demographic parameters, to ensure that current model outputs were appropriate and that users understand the potential implications of uncertainty in baseline demographic parameters.

The review of demographic parameters involved a desk-based literature review of information that has become available since the previous report was published on the trajectory, size and demographic parameters of the cetacean management units (MUs) and seal management areas (MAs) for the five species of marine mammals most relevant to Scottish waters (grey and harbour seals, harbour porpoise, bottlenose dolphins and minke whales). Several key species experts were consulted in the process of the review.

The evaluation of the sensitivity of outputs to misspecification of demographic parameters focused on the ratio output metric of the counterfactual of population size (the median of the ratio of the impacted to un-impacted population size across all simulated matched replicate pairs). This was felt to be more appropriate than the ratio of the impacted to un-impacted growth rates due to the nature of the impact on marine mammals, with disturbance during discrete periods of construction activity rather than a continuous level of impact over an extended period. Two approaches were used in the sensitivity testing: The effect of changing the value of a single parameter independently of the others (this was done for all species except grey seal) and the effect of adjusting all the demographic parameters systematically to

compare scenarios with the same population trajectory but different combinations of demographic parameters.

Section Four summarises the results of the review of demographic parameters and presents updates where new information has been published since the last review (Harwood and King, 2017). New information has been published on all species to inform our understanding of population trajectories, but not all populations had updated demographic information available. One major difference from the previous review is that recommended values have now been provided for English harbour seal management areas where they were absent before.

Section Five presents the results of the sensitivity analysis. The observed sensitivity to misspecification is complex. It depends both on the proportion of the population that experiences disturbance above the threshold that experts judged is required to affect vital rates, and the magnitude of the effect on the vital rate, as well as on the effect of the parameters on the starting stable age structure of the population.

The analyses demonstrate substantial variation in sensitivity to parameter misspecification among species and demographic variables. The greatest sensitivity was to variation in the value of pup survival for harbour seals and, overall, changes in pup/calf survival had the greatest impact on the counterfactual of population size. This pattern results from the fact that pup/calf survival is the demographic variable that experts considered to be most sensitive to the effects of disturbance. As a result, most of the simulations predicted some effect of the operation being modelled on pup/calf survival, whereas effects of fertility were rare. In addition, the effect of the additional mortality associated with disturbance depends on the absolute value of the demographic rate that is affected. The widest range of pup survival values was used in the analysis of harbour seal population dynamics, which is why these simulations show the greatest effect. Sensitivity to misspecification in other demographic variables is largely a consequence of the effect of changing the value of a particular variable on the stable age structure used to initiate the iPCoD simulations – for example a population with a high adult survival rate will contain a larger proportion of adults and will therefore be more resilient and recover faster after disturbance.

However, despite this complexity and variability, overall, the sensitivity to parameter misspecification is smaller than the variability among simulations, which results from the variability in expert’s opinions on the degree to which vital rates are affected and the various forms of stochasticity (environmental and demographic) that contribute to the simulations. In addition, the sensitivity is only apparent at relatively high (and somewhat unrealistic) levels of impact, and therefore we conclude that the metric of counterfactual of population size is robust to misspecification in demographic rates.

The final section of the report provides a table of recommended parameter values for modelling the impact of disturbance from offshore wind farm construction on UK marine mammal populations using iPCoD. Ultimately, it is the responsibility of the iPCoD user to justify the selection of demographic parameters, and new information may continue to become available from ongoing monitoring and modelling of these populations that may justify a different selection. We recommend that the choices for parameters are discussed and agreed with consultees before assessments are undertaken and we hope that the information presented in this report will guide these discussions. However, in the absence of estimates of demographic parameters, we highlight that, in general, adoption of a higher pup/calf survival rate and a lower adult survival and fertility rate will ensure the most precautionary predictive modelling outcomes.

2 Introduction

SMRU Consulting developed the interim Population Consequences of Disturbance Framework (iPCoD) in 2013 to provide a transparent and robust framework for the prediction and assessment of the effects of marine and offshore renewables projects on the five UK priority marine mammal species. SMRU Consulting have been developing and updating the framework since then with support from a variety of sources, including Marine Scotland Science (MSS), the UK Department of Business, Energy and Industrial Strategy (BEIS) and the Dutch Government.

In 2019, SMRU Consulting were commissioned by Marine Scotland to work on two aspects of the iPCoD framework: 1) to update the recommended demographic parameters for modelling the five UK marine mammal species that are considered in the iPCoD framework, and 2) to carry out a sensitivity analysis of the outputs to variability in baseline demographic parameters to ensure that current model outputs were appropriate and that users understand the potential implications of uncertainty in baseline demographic parameters. Each of these aspects are introduced further below.

2.1 Demographic parameters

There is a need to update the recommended demographic parameters for the UK marine mammal populations for use with the iPCoD model. The current recommendations are based on reviews published in 2014 and 2017, and there have been reported changes in the abundance and demographic parameters of several populations since then. Up to date information is now available on population trends and demographic parameters for several Scottish and UK populations from a variety of sources.

2.2 The sensitivity of output metrics to variations in population dynamics

The adoption of the metrics to provide as standard iPCoD outputs advised by MSS in recent scoping opinions (e.g. MS-LOT 2017b, a) came from a study which examined the sensitivity of output metrics when modelling population consequences of impacts on birds using Population Viability Analyses (PVA) (Jitlal et al. 2017). However, there is need to assess the suitability of these metrics in relation to the iPCoD framework and marine mammals. There are a number of differences between birds and marine mammals in both their population dynamics and in the nature of impacts from offshore renewables and the way in which impacts might manifest at the population level. It is therefore essential that these metrics are shown to be robust to assumptions made about input parameter values and that any sensitivities are understood. These uncertainties and sensitivities should be documented as a matter of routine in reports and assessments that are used to inform decisions and policy development, in order to ensure a precautionary approach to modelling population consequences is implemented.

3 Methods

3.1 Phase 1: Review of demographic information

This phase involved a two stage approach:

1. A desk-based literature review of information that has become available since the previous report was published on the trajectory, size and demographic parameters of the cetacean management units (MUs) and seal management areas (MAs) for the five species of marine mammals most relevant to Scottish waters. The priority marine mammal species considered in the iPCoD framework project are:

    - Grey seal
    - Harbour seal
    - Harbour porpoise
    - Bottlenose dolphin
    - Minke whale

2. The outcomes of this review were then reviewed by and discussed with species experts. The species experts that were involved in this review were:

    - **Bottlenose Dolphins:** As the lead authors of the recent published studies of demographic rates for the Coastal East Scotland bottlenose dolphin MU, Dr Monica Arso Civil (SMRU) and Dr Barbara Cheney (University of Aberdeen) were invited to review the recommendations for the most appropriate demographic parameters for the MU.
    - **Minke Whales:** Dr. Gísli Víkingsson (Head of whale research at the Marine and Freshwater Research Institute, Reykjavík) provided advice on minke whale demographic parameters.
    - **Harbour Porpoise:** Professor Phil Hammond (SMRU) advised on demographic parameters. Professor Hammond has extensive expertise in the statistical and mathematical modelling of marine mammal population parameters and processes. He has led on the strategic monitoring of the harbour porpoise population in the North Sea and adjacent waters since the 1990s (SCANS surveys) and attended the IMR/NAMMCO International Workshop on the Status of Harbour Porpoises in the North Atlantic (December 2018, Tromso, Norway).
    - **Harbour and Grey Seals:** Dr Dave Thompson (SMRU) leads on the delivery of the annual SCOS (Special Committee On Seals) advice to the UK Government which includes the results of the regular UK seal monitoring programme and regular population modelling work. Dr Thompson was consulted to provide advice on the most appropriate demographic parameters for the different harbour and grey seal management areas (SMAs). In addition, Professor Len Thomas (CREEM) and Dr Debbie Russell (SMRU) have also been involved in the consultation, specifically with respect to the grey seal population model that they have both been working on for several years, as reported annually in SCOS reports.

3.2 Phase 2: Evaluate the sensitivity of outputs

In Phase 2, the metrics recommended by MSS in recent Scoping opinions to offshore wind farm developments in Scotland will be evaluated with respect to their sensitivity to differences input parameter values. These metrics are defined below:

Two ratio metrics:

- The median of the ratio of impacted to un-impacted annual growth rate across all simulated matched pairs (or the median counterfactual of population size);
- The median of the ratio of impacted to un-impacted population size across all simulated matched pairs.

A further probabilistic metric was tested in the Jitlal et al. (2017) analysis:

- Centile for un-impacted population which matches the 50th centile for the impacted population.

However, the iPCoD code does not currently provide this probabilistic output as standard so this metric was not assessed, although it is possible to generate probabilistic outputs. It was not possible to assess the sensitivity in other outputs such as the median difference between impacted and un-impacted population as part of the scope for this project.

Two approaches were used to identify the sensitivity of the metrics to the values used for the different demographic parameters:

a) The effect of changing the value of a given demographic parameter independently of the other parameters was assessed. This results in a corresponding change in the population growth rate – this will also allow an implicit assessment of the sensitivity in the model outputs for a given level of impact to variation in the predicted status of the population – whether increasing, decreasing, or stable. This was done for harbour seal, harbour porpoise, bottlenose dolphin and minke whale. In order to generate noticeable population consequences (required in order to carry out a sensitivity analysis), initial simulations for grey seals had to resort to such highly unrealistic extremes of impact that it was deemed that no further simulations would be useful.

b) We also assessed the effect of adjusting all the vital rates simultaneously to compare scenarios with the same population trajectory but different combinations of demographic rates. This examines sensitivity in the model outputs to variation in how a population trajectory is achieved. This analysis was designed drawing from the findings of the Phase 1 review and the systematic sensitivity analyses in Phase 2 (a). The aim was to inform the selection of recommended demographic parameter values for UK MUs and

SMAs. Phase 3 of the project was then used to summarise the recommended values for future simulations and identify any recommended further sensitivity analyses.

3.3 Phase 3: Recommend appropriate demographic parameter values for UK marine mammal MUs and SMAs

This final section of the report presents recommended values for demographic parameters that should be used in future assessment of the effect of underwater noise disturbance on UK marine mammal MUs and SMAs.

4 Phase 1: Review of Demographic Parameters

4.1 Historical values for demographic parameters

Recommended demographic parameter values for each of the five key UK species modelled by iPCoD were initially presented in Harwood and King (2014) (Table 1). Some of these values were obtained from the literature and others were tuned by hand to achieve the observed growth rate for each MU/SMA. For context, this information is provided at the start of each section below for each species.

Table 1

Demographic parameters recommended for each species and MU/SMA in Harwood & King (2014).

Species	MU/SMA	Age calf/pup becomes independent	Age of first birth	Calf/Pup Survival	Juvenile Survival	Adult Survival	Fertility	Growth Rate
		age1	age2	Surv[1]	Surv[7]	Surv[13]		
Harbour Porpoise	North Sea low adult surv.	1	5	0.6	0.85	0.85	0.96	1.000
	North Sea high adult surv.	1	5	0.6	0.85	0.925	0.48	1.000
Grey Seal	All	1	5	0.235	0.94	0.94	0.84	1.010
Harbour Seal	Shetland modified using adult survival	1	4	0.6	0.58	0.89	0.88	0.945
	Shetland modified using fertility	1	4	0.6	0.61	0.94	0.24	0.945
	Orkney & north coast modified using survival	1	4	0.6	0.56	0.86	0.88	0.915
	Moray Firth	1	4	0.6	0.61	0.94	0.88	1.000
	East Coast modified using survival	1	4	0.6	0.5	0.76	0.88	0.820
	South-west Scotland	1	4	0.6	0.61	0.94	0.88	1.000

	West Scotland	1	4	0.6	0.61	0.94	0.88	1.000
	Western Isles	1	4	0.6	0.61	0.94	0.88	1.000
	Northern Ireland	1	4	0.6	0.61	0.94	0.88	1.000
Minke Whale	European waters	1	9	0.70	0.77	0.96	0.91	1.000
Bottlenose Dolphin	All other MUs	2	9	0.8	0.94	0.94	0.25	1.000
	Coastal East Scotland	2	9	0.9	0.947	0.947	0.3	1.018

A number of these values were updated in 2017 (Table 2), however, the corresponding text was not updated.

Table 2

Demographic parameter values recommended for each species and MU/SMA in Harwood & King (2017). Shaded cells indicate parameters that were updated between 2014 and 2017.

Species	**MU/SMA**	**Age calf/pup becomes independent**	**Age of first birth**	**Calf/Pup Survival**	**Juvenile Survival**	**Adult Survival**	**Fertility**	**Growth Rate**
		age1	**age2**	**Surv[1]**	**Surv[7]**	**Surv[13]**		
Harbour Porpoise	North Sea low adult surv.	1	5	0.6	0.85	0.85	0.81	1.000
	North Sea high adult surv.	1	5	0.6	0.85	0.925	0.44	1.000
Grey Seal	All	1	5	0.21	0.94	0.94	0.84	1.010
Harbour Seal	Shetland modified using adult survival	1	4	0.55	0.61	0.867	0.88	0.945
	Shetland modified using fertility	1	4	0.55	0.61	0.94	0.06	0.945
	Orkney & north coast modified using survival	1	4	0.51	0.52	0.867	0.88	0.915
	Moray Firth	1	4	0.55	0.61	0.94	0.88	1.000
	East Coast modified using survival	1	4	0.50	0.5	0.76	0.88	0.820
	South-west Scotland	1	4	0.55	0.61	0.94	0.88	1.000
	West Scotland	1	4	0.55	0.61	0.94	0.88	1.000
	Western Isles	1	4	0.55	0.61	0.94	0.88	1.000
	Northern Ireland	1	4	0.55	0.61	0.94	0.88	1.000
Minke Whale	European waters	1	9	0.70	0.77	0.96	0.90	1.000
Bottlenose Dolphin	All other MUs	2	9	0.835	0.94	0.94	0.25	1.000
	Coastal East Scotland	2	9	0.9	0.94	0.945	0.3	1.018

During subsequent use of the code it was determined that the values provided in Harwood and King (2017) were not always resulting in simulated growth rates that matched observed rates. Therefore, certain demographic parameters were further tuned to achieve the observed growth rates (Table 3) and these updates were provided in the iPCoD version 4 release (August 2018). No change to demographic parameters were made in the version 5 release (March 2019).

Table 3

Demographic parameter values as recommended in the helpfile for the current iPCoD framework (version 5.0). Shaded cells indicate parameters that were updated between Harwood & King (2017) and the current helpfile parameters.

Species	MU/SMA	Age calf/pup becomes independent	Age of first birth	Calf/Pup Survival	Juvenile Survival	Adult Surv	Fertility	Growth Rate
		age1	age2	Surv[1]	Surv[7]	Surv[13]		
Harbour Porpoise	North Sea low adult surv.	1	5	0.6	0.85	0.85	0.958	1.0000
	North Sea high adult surv.	1	5	0.6	0.85	0.925	0.479	1.0000
Grey Seal	All	1	5	0.222	0.94	0.94	0.84	1.0100
Harbour Seal	Shetland modified using adult survival	1	4	0.55	0.61	0.8799	0.88	0.9450
	Shetland modified using fertility	1	4	0.55	0.61	0.94	0.068	0.9450
	Orkney & north coast modified using survival	1	4	0.507	0.52	0.874	0.88	0.9150
	Moray Firth	1	4	0.55	0.61	0.9451	0.88	1.0000
	East Coast modified using survival	1	4	0.5	0.5	0.7701	0.88	0.8200
	South-west Scotland	1	4	0.55	0.61	0.9451	0.88	1.0000
	West Scotland	1	4	0.55	0.61	0.9451	0.88	1.0000
	Western Isles	1	4	0.55	0.61	0.9451	0.88	1.0000
	Northern Ireland	1	4	0.55	0.61	0.9451	0.88	1.0000
Minke Whale	European waters	1	9	0.72	0.77	0.96	0.9	1.0000
Bottlenose Dolphin	All other MUs	2	9	0.86	0.94	0.94	0.25	1.0000
	Coastal East Scotland	2	9	0.9	0.94	0.9497	0.3	1.0180

4.2 Bottlenose dolphin demographic Parameters

4.2.1 Harwood & King 2014

"Lusseau (2013) used the results of capture-recapture analysis of sightings of individually recognisable bottlenose dolphins on the East coast of Scotland to obtain demographic rates that could be used to model the population dynamics of the Coastal East Scotland MU (which has an estimated growth rate of 1.018 - Cheney et al., 2013), and the sub-population within the Moray Firth (which appears to have a growth rate closer to 1.0 - Cheney et al., 2012). We suggest that the latter values are used for the other MUs identified by Anon. (2014)."

4.2.2 Updated demographic parameter values

There have been several recent developments in estimates of the demographic parameters for the coastal East Scotland bottlenose dolphin MU. These are presented in Table 4 and summarised briefly below:

- Quick et al. (2014) estimated fertility rates using photo-ID data between 1989 and 2012 across the entire Coastal East Scotland bottlenose dolphin MU.
- Arso Civil et al. (2017) estimated fertility rates using photo-ID data between 1989 and 2012 across the entire MU (Moray Firth SAC and the St Andrews Bay and Tay Estuary areas).
- Robinson et al. (2017) estimated age at first birth and fertility rate using photo-ID data between 1997 and 2016 for the southern coastline of the outer Moray Firth.
- Arso Civil et al. (2018b) estimated first, second and third year calf survival alongside a combined survival rate for juveniles and adults using photo-ID data collected between 1989 and 2015 across the entire MU (Moray Firth SAC and the St Andrews Bay and Tay Estuary areas).
- Cheney et al. (2019) estimated first and second year calf survival rates and a fertility rate for the Moray Firth SAC portion of the coastal East Scotland bottlenose dolphin MU using photo-ID data collected between 2001 and 2016.

The estimated demographic parameters obtained from each study are shown in Table 4. As noted in the text above, each study used data from different portions of the coastal East Scotland bottlenose dolphin MU. Overall, the calf survival estimates range between 0.55 (for second year SAC calves) and 0.981 (for second year calves across the whole MU), the reported combined juvenile and adult survival rates vary between 0.948 (for both sexes, whole MU) and 0.962 (for females, whole MU) and the fertility rates vary between 0.16 (for southern Moray Firth animals, Robinson et al. 2017) and 0.24 (total MU).

The range in reported fertility estimates is driven by the use of different inter-birth intervals (IBI) and modelling methods between the studies. Robinson et al. (2017)used an IBI of 3.8 years which is much lower than the 4.49 years used in both Quick et al. (2014)and Arso Civil et al. (2017), despite the studies all being conducted on the same population. The fertility estimate in Cheney et al. (2019) accounted for imperfect detection, which the authors state is the reason why it is higher (0.23) than that obtained by Robinson et al. (2017) (0.16), who calculated the proportion of reproductive females with newborn calves. The Cheney et al. (2019) estimate is better aligned with the estimate from Arso Civil et al. (2018b), which took account of individual and temporal heterogeneity in re-sightings. Recent modelling using data up to 2015 resulted in a fertility rate of 0.24, which is considered to be the most up to date estimate for the population (Arso Civil pers. comm.).

The second year calf survival estimate for the SAC presented in Cheney et al. (2019) (0.55) is considerably lower than for first year survival (0.93) and lower than that estimated by Arso Civil et al. (2018b) for the entire MU. The authors suggest that this may be due to a decreased probability of detecting older calves as they spend less time with their mothers; alternatively, the SAC could be acting as a source for the whole population, through the dispersal of second year calves.

Table 4

Demographic rate estimates from various studies of the coastal East Scotland bottlenose dolphin MU.

Parameter	Current	Estimate	Range	Notes	Source
Age at independence	2	3			Arso Civil pers. comm.
Age at first birth	9	9	6+	Median 9 years (SAC)	Cheney et al. (2019)
		8	Range 6-13	Outer Moray Firth	Robinson et al. (2017)
Calf survival	0.9	0.93	95% CI 0.82-0.98	1st year (SAC)	Cheney et al. (2019)
		0.55	95% CI 0.44- 0.65	2nd year (SAC)	Cheney et al. (2019)
		0.865	95% CI 0.785-0.919	1st year (total MU)	Arso Civil et al. (2018a)
		0.981	95% CI 0.797-0.998	2nd year (total MU)	Arso Civil et al. (2018a)
		0.883	95% CI 0.708-0.959	3rd year (total MU)	Arso Civil et al. (2018a)
Juvenile survival	0.94	0.948	95% CI 0.933-0.959	Juvenile + adult both sex (total MU)	Arso Civil et al. (2018a)
		0.962	95% CI 0.941-0.976	Juvenile + adult Female (total MU)	Arso Civil et al. (2018a)
Adult survival	0.9457	0.948	95% CI 0.933-0.959	Juvenile + adult both sex (total MU)	Arso Civil et al. (2018a)
		0.962	95% CI 0.941-0.976	Juvenile + adult Female (total MU)	Arso Civil et al. (2018a)
		0.98		Estimate for F with older calves	Cheney et al. (2019)
Fertility	0.3	0.16	Range 0.08-0.23	IBI 3.8yr (Outer Moray Firth)	Robinson et al. (2017)
		0.22	95% CI 0.22-0.25	IBI 4.49yr (total MU)	Quick et al. (2014)
		0.222	95% CI 0.218–0.253	IBI 4.49yr (3.94-4.93) (total MU)	Arso Civil et al. (2017)
		0.23	SE 0.01	Average rate (SAC)	Cheney et al. (2019)
		0.24	95% CI 0.235-0.273	IBI 4.167 (3.656–4.566) (total MU)	Arso Civil pers. comm.
Growth Rate	1.0180	1.027657		2001-2015 (total MU)	Cheney et al. (2018)
		1.036684		2001-2017 (total MU)	Cheney pers. comm.

In order to check whether the estimated literature values of demographic parameters would result in a population trajectory that met with expectation when modelled in iPCoD, the observed growth rate of the population was determined using monitoring data. The most recent population size using the photo-ID data between 2001 and 2017 was estimated using a Bayesian state space model (Cheney, pers. comm.). This model estimated that the population increased from 125 animals in 2001 (Highest Probability Density Intervals: 104-149) to 222 animals in 2017 (Highest Probability Density Intervals: 190-257) (data provided by Cheney, pers. comm.). Assuming that the population increased at a constant growth rate across that period, the annual growth rate is calculated as 1.0367 (population increases by 3.67% p.a). The data collected prior to 2001 were not included in this analysis as the sampling protocols and the capture probabilities were different.

Table 5 presents the combination of demographic parameter estimates required to achieve a modelled growth rate that matches the observed population growth rate.

Table 5

Demographic parameter estimates to achieve observed growth rate.

Parameter	Estimate	Source
Age calf becomes independent	3	Arso Civil pers. comm.
Age of first birth	9	Cheney et al. (2019)
Calf Survival	0.925	Value tuned by hand within the range of published values
Juvenile Survival	0.962	Arso Civil et al. (2018a)
Adult Survival	0.98	Arso Civil et al. (2018a), Cheney et al. (2019)
Fertility	0.24	Arso Civil pers. comm.
Growth Rate	1.0365	Calculated in iPCoD with ev$val[1]

4.3 Minke whale demographic parameters

4.3.1 Harwood & King 2014

“There is very little empirical information on demographic rates for common minke whales in European waters. However, Hauksson et al. (2011) estimated a fecundity rate and age at first breeding based on samples from the Icelandic whaling industry. Taylor et al. (2007) summarised information on life history parameters for most cetacean species collected worldwide. In Table 5.1, we suggest using their estimate of the adult survival rate for common minke whales, which is based on a longevity estimate of 51 years. They also provided an estimate of calf survival of 0.8 using a ratio of calf survival to adult survival calculated for southern right whales, humpback whales and bottlenose dolphins. If this value is used in population projections with a value of 0.88 (intermediate between the calf and adult survival rates) for juvenile

survival, it implies a population growth rate of 1.049. Until there is a reliable estimate of the growth rate for the European population, we suggest using values for calf and juvenile survival that result in a population growth rate of 1.0."

4.3.2 Updated data

There are still very few data available on appropriate values for European minke whale demographic parameters.

The SCANS III report (Hammond et al 2017) provides an estimate of the North Sea minke whale population size and trend, using data from SCANS I (1994), SCANS II (2005) and SCANS III (2016) as well as five additional estimates from the Norwegian Independent Line Transect Surveys (Schweder 1997, Skaug et al. 2004, Bøthun et al. 2009, Solvang et al. 2015). The estimated size of the North Sea minke whale population in 2016 was 8,900 (CV: 0.24) which is within the range of previous estimates. The results of the trend analysis of estimates in the North Sea show no support for changes in minke whale abundance since 1989. In consultation with Dr Gísli Víkingsson it was advised that no further data on minke whale demographic parameters have been produced since the Hauksson et al. (2011) paper, and that the existing parameters still represent the current state of knowledge.

Therefore, it is recommended that the growth rate for the population remains at 1.0000 and no change in demographic parameter values is suggested.

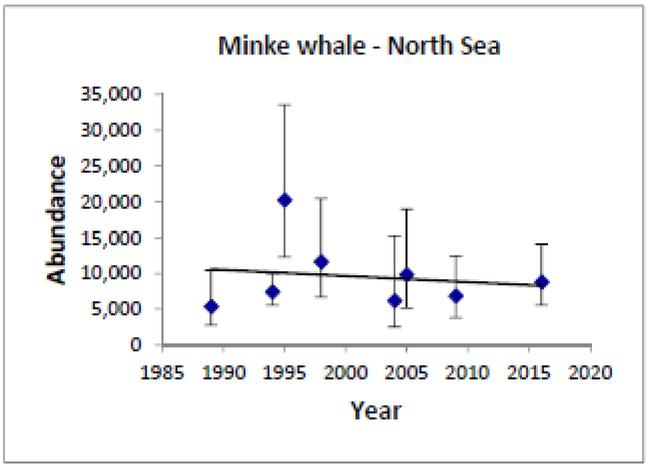


Figure 1: Trend line fitted to time series of abundance estimates for minke whales in the North Sea. Error bars are log-normal 95% confidence intervals. Figure obtained from Hammond et al (2017).

Table 6

Minke whale demographic parameter estimates.

Parameter	Harwood and King (2014)	Hauksson et al. (2011)	Taylor et al. (2007) *Balaenoptera acutorostrata*	Taylor et al. (2007) Range across 58 cetacean species
Surv1 (calf survival)	0.72	-	0.806	0.673 – 0.91
Surv7 (juvenile survival)	0.77	-	0.96	0.914 – 0.99
Surv13 (adult survival)	0.96	-	0.96	0.914 – 0.99
Fertility	0.9	85.0% of the mature females were pregnant. 92.2% mature females were reproductively active.	-	-
age1 (age at independence)	1	-	-	-
age2 (age at first birth)	9	9 years (8 – 10)	8	5 - 20

4.4 Harbour porpoise demographic parameters

4.4.1 Harwood & King 2014

"As part of the development of a model of the potential effects of by-catch on the harbour porpoise population of the North Sea, Winship & Hammond (2008a) estimated survival (which they assumed to be constant across adults and juveniles), age at maturity and maximum birth rate that were compatible with data from by-caught animals and survey data. Moore & Read (2008) used a similar approach for harbour porpoises in the Northwest Atlantic, but they modelled age-specific survival. We used Winship & Hammond's value of 4 for the mean age at maturity to estimate age at first breeding, and their two values for survival (0.85 and 0.925) as alternatives for adult survival. However, we chose a lower value of 0.6 for calf survival, based on Moore & Read's analysis. We then tuned fecundity to achieve the population growth rate of 1.0 suggested by Fig. 7 of Winship & Hammond (2008a) using either of these values."

4.4.2 Updated data

The NAMMCO (2019) report on the International Workshop on the Status of Harbour Porpoises in the North Atlantic (December 2018, Tromso, Norway) was used to inform the review of harbour porpoise demographic parameters. For UK porpoise life history parameters, the NAMMCO (2019) workshop presented data from Murphy et al. (2015), who used data from stranded and by-caught porpoise obtained from UK waters between 1990 and 2012 (n=329, F=127). The study found that the pregnancy rate across all stranded females was 34% and the age at sexual maturity was 4.73 years (50% and 4.92 years for those stranded in “healthy” condition). These pregnancy rate estimates were considerably lower than those estimated in other studies. For example, Read and Hohn (1995) estimated 93% in the Gulf of Maine and Bay of Fundy in the North-west Atlantic and Ólafsdóttir et al. (2003) estimated 98% for waters off Iceland. Given the comparison to other studies of porpoise in the North Atlantic, Murphy et al. (2015) concluded that the UK harbour porpoise population had experienced reproductive failure, likely due to exposure to PCBs (polychlorinated biphenyls). Even the higher pregnancy rate for healthy animals (50%), this is outside the range reported elsewhere.

No data were provided by Murphy et al. (2015) for survival rates of calves, juveniles or adults nor were any updated survival rates presented in the NAMMCO (2019) workshop report. Using the Winship et al. (2008) value for calf and juvenile survival (age 0-4, 0.85) and adult survival (age 5+, 0.925), combined with the Murphy et al. (2015) estimate for fertility (0.34), the calculated population growth rate was slightly greater than one (actually 1.0003). Therefore, calf survival was adjusted downward slightly from 0.85 to 0.8455 to achieve the observed growth rate.

Table 7

Updated recommended harbour porpoise demographic parameters

Scenario	Age calf becomes independent	Age of first birth	Calf Surv	Juv Surv	Adult Surv	Fertility	Growth Rate (GR)	Notes
	age1	age2	Surv[1]	Surv[7]	Surv[13]			
Values from literature	1	5	0.85^1	0.85^1	0.925^1	0.34^2	1.0003	GR too high
Adjust Calf Survival	1	5	0.8455	0.85^1	0.925^1	0.34^2	1.0000	Correct GR achieved

^{1}Winship et al., (2008); ^{2}Murphy, et al., (2015)

4.5 Harbour seal demographic parameters

4.5.1 Harwood & King 2014

"In contrast to the grey seal MUs, there is good evidence that the MUs proposed by Anon. (2014) for harbour seals can be considered as discrete populations for modelling purposes. However, the only MU for which there is good information on demographic rates is the Moray Firth, and we have therefore had to base our suggested demographic rates for the other MUs on the available estimates for this MU. Cordes (2011) provided a fecundity estimate of 0.88 for this population, and an adult survival rate in the range 0.94 - 1.0. Thompson et al. (2013a) used a value of 0.97 for adult survival and a value of 0.7 for pup survival. Recent aerial surveys suggest that the earlier decline in this population "may have been halted" (SCOS, 2012), while modelling studies (Matthiopoulos et al., 2014) suggest that it is now increasing slowly. We therefore suggest using a growth rate of 1.0 for this population. The juvenile survival rate required to give this population growth rate with the values for pup survival, adult survival and fecundity used by Thompson et al. (2013a) is 0.46: substantially less than the survival rate for pups. This seems unlikely. We therefore suggest using an adult survival rate of 0.94 (which is still within the range of Cordes' (2011) estimates) and a pup survival rate of 0.6 for this MU. The juvenile survival rate required to give a population growth rate of 1.0 with these values is 0.61. We suggest that this combination of demographic rates should also be used for all of the MUs whose status is reported to be "unclear' in SCOS (2012).

We estimated the annual rate of decline for the Shetland and Hebrides MUs from the earliest and most recent survey counts in Table 3 of SCOS (2012), and we took the rate of decline for the East Coast of Scotland MU from Lonergan & Thompson (2012). The reasons for the decline in size of these MUs is still unclear. However, there has been a marked increase in the number of harbour seals found dead in UK waters with so-called corkscrew injuries (Bexton et al., 2012), suggesting there may have been a decrease in survival for adult and juvenile animals. In addition, Matthiopoulos et al. (2014) report that "breeding success" (which we refer to as fecundity) was "the most volatile demographic component of the (Moray Firth) population." We therefore considered two scenarios: that fecundity and pup survival had remained at their undisturbed levels in these populations and the decline was entirely the result of a decrease in juvenile and adult survival; that adult and juvenile survival had remained at their undisturbed level and the decline was entirely the result of a decrease in pup survival and fecundity. We were able to replicate the observed decline in the Shetland MU using both scenarios. However, we were *unable to replicate the observed decline in the Orkney and East Coast Scotland MUs by reducing fecundity and pup survival alone. We have therefore only shown the results from the first scenario for these MUs.*

The status of the North-east and South-east England MUs is unclear. Table 4 of SMRU (2012) described the "England" harbour seal population (which is predominantly composed of these two MUs) as "increasing", but subsequent text (p41) indicates that the count for the "English East coast population (Donna Nook to Scroby Sands) in 2011 was 8% lower than the 2010 count. However, this was 26% higher than the mean of counts between 2004 and 2008". The individual counts shown in the accompanying figure (Fig. 7, p42) are highly variable. Hopefully, more recent counts will have clarified this situation but, until these are published, we are unable to offer any suggestions for the best way to model the dynamics of the populations in these MUs."

4.5.2 Updated population trends

It is now considered that there are sufficient data on the trends in abundance for both the Northeast and Southeast England harbour seal SMAs to explore the potential for these to be modelled using iPCoD. Data collected during the harbour seal moult count surveys up until 2017 have been recently modelled using GLMs to estimate population sizes and the population trend within each SMA (Thompson et al. 2019). The most recent data for each UK SMA are summarised below from the information presented in Thompson et al. (2019) and presented in Figure 3:

- **Southeast England:** The selected models for the combined counts for The Wash, Donna Nook and Blakeney Point incorporated two periods of exponential increase; 6.6 % p.a. between 1989 and 2002 and 2.8 % p.a. (95% CIs: 1.34-4.31) between 2003 and 2017. A GAM indicated a levelling off in recent years at The Wash.
- **Northeast England:** Between 2002 and 2017 counts in the Tees have increased at approximately 7.43% p.a. (95% CIs: 6.07-8.80). This SMA is still relatively small, with counts of less than 100 individuals.
- **Combined SE England and NE England:** Adding the two figures together (due to the small size of the NE England SMA) indicates an increase in the combined population between 2007 and 2017 of 3.23% pa.
- **East Scotland**: Trends have been fitted to the Firth of Tay & Eden Estuary SAC as counts within the rest of the SMA are limited. Between 2002 and 2017 the population declined by 18.6% p.a. (95% CIs: 17.1-20.0). However, by 2016 the SAC counts represented only approximately 15% of the SMA total.

When including counts in the Firth of Forth, the total East Scotland SMA counts appear to be relatively stable over recent complete surveys between 200 and 300 animals (Figure 2).

- **Moray Firth:** Trends have been fitted to the area between Loch Fleet and Findhorn as counts within the rest of the SMA are limited. The selected model for this area shows that counts were decreasing at a rate of 5.6% p.a. (95% CIs: 2.5-8.5) between 1994 and 2000, followed by a step change with a drop of ~28% occurring between 2000 and 2003 and no significant trend in counts thereafter.
- **Northern Ireland:** Trends have been fitted to the area between Carlingford Lough and Copeland Islands as counts within the rest of the SMA are limited. Between 2002 and 2011 the population declined by 2.7% p.a. (95% CIs: 1.8-3.5). However, this analysis (Thompson et al. 2019) did not include the most recent surveys in Northern Ireland in 2018 (Duck and Morris 2019). The 2018 count was 7% higher than the 2011 count, and the authors concluded that harbour seal counts in Northern Ireland have not changed markedly since 2003. Therefore, the inclusion of the 2018 data suggests that the population can be considered stable.
- **North Coast & Orkney:** Following a step-change of a 46% decline in counts between 2001 and 2006, between 2006 and 2015 the population size has undergone a continued decline of 10.41% p.a. (95% CIs: 9.29–11.52).
- **Shetland:** The observed population decline in Shetland was modelled as a step-change of a 40% reduction occurring between 2001 and 2005. Between 2006 and 2016 there was no obvious trend in counts with the population size appearing stable.
- **Western Isles:** Between 1992 and 2017 the population size was stable.
- **Southwest Scotland:** Between 1989 and 2015 the population size was stable.
- **West Scotland:** Overall, between 1990 and 2014 the population size was stable. This SMA is very large geographically containing a large proportion of the total UK count (e.g. 49% in 2016). Parts of the range experiencing increases and parts of the range appearing stable. Therefore it is proposed that this could be split into three sub-regions for modelling:
    - **West Scotland North:** In the north sub-unit, the model selected indicates that between 1991 and 2017 the population size has increased by 4.86% p.a. (95% CIs: 4.02-5.70).
    - **West Scotland Central:** In the central sub-unit, the model selected indicates that between 1989 and 2014 the population size has increased by 4.02% p.a. (95% CIs: 3.08-4.97).

- **West Scotland South:** In the south sub-unit, the model selected indicates that between 1990 and 2014 the population size was <u>stable</u>.

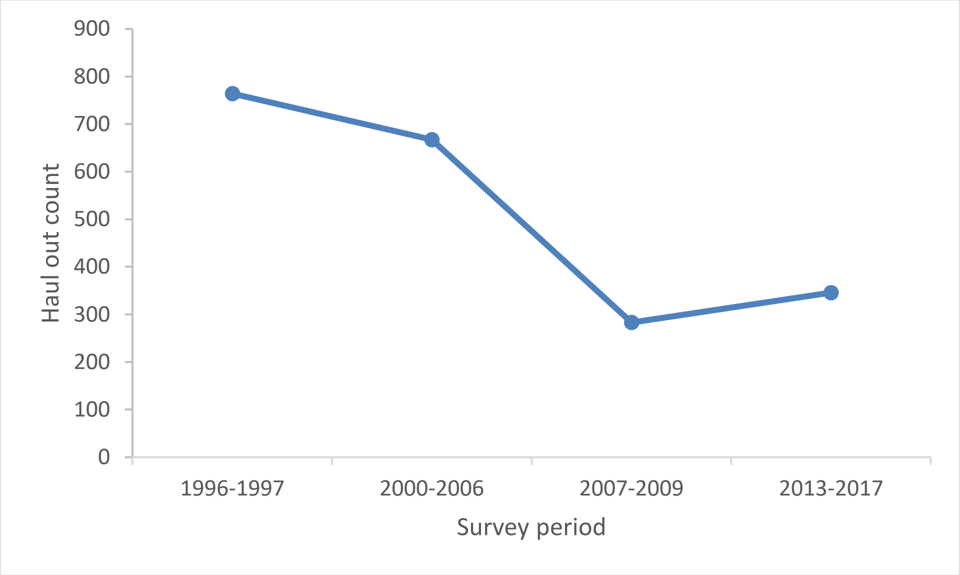


Figure 2: Harbour seal moult survey counts in the whole East Scotland management unit - note that the complete SMASMA is surveyed relatively infrequently.

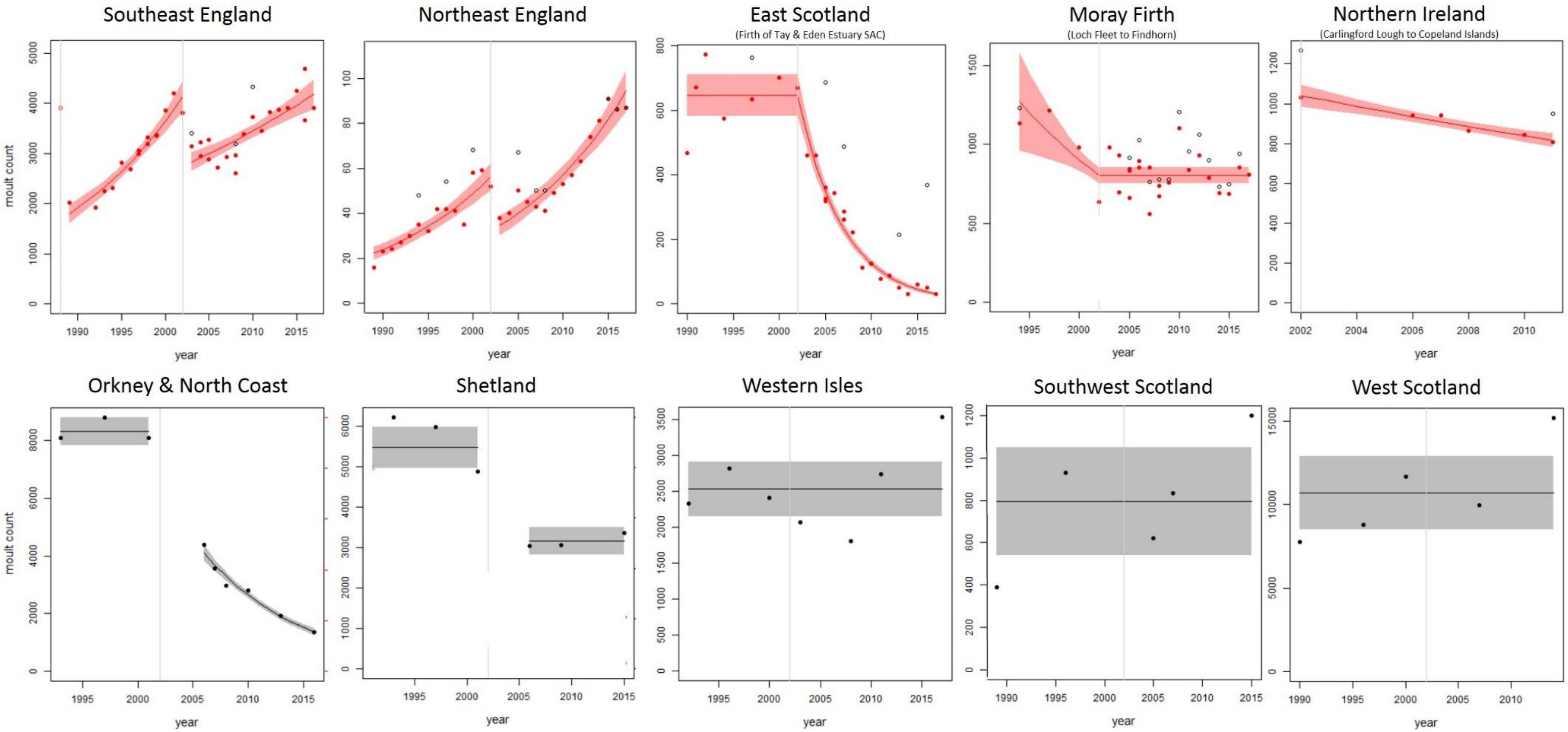


Figure 3: Harbour seal moult survey counts and fitted trends for different SMAs (Thompson et al. 2019). Counts not used in model fits are shown as grey dots. Note, the Northern Ireland estimates do not include the 2018 count data which suggested that the population is stable (Duck and Morris 2019).

4.5.3 Demographic parameters

A review of the available literature on harbour seal population dynamics is provided below. Values for UK populations are explored first but values from other populations are provided for context and to inform the ranges of plausible values for the sensitivity analyses presented in Section 4.

4.5.3.1 Pup survival

Hanson et al. (2013) tagged female pups in 2007, 24 in Orkney (a rapidly declining population) and 24 in west Scotland (stable population). They estimated that female pup survival to age six months was 0.39 (95% CI: 0.297-0.648) and predicted survival to 300 days was 0.12 (95% CI: 0.064-0.42). The pup survival probabilities were the same between the two populations, which led the authors to conclude that low pup survival was not driving the decline in Orkney but it was more likely to be low adult survival. However, the estimated pup survival values are significantly lower than those reported from other populations and from modelling studies.

Hastings et al. (2012) used mark-recapture methods to estimate pup survival at Tugidak Island, Alaska between 2000 and 2007. Both moult and pupping counts for this population had declined drastically between 1976 and 1988. This was followed by a period of stabilisation, then an increase between 1994 to 2000 of 3.4% p.a in the moult counts and 8.63% p.a. in the pupping season. They estimated pre-weaning survival (to ~1 month) between 2000 and 2007 to be 0.74 and post-weaning survival (to ~11 months) to be 0.72-0.82. The survival rate from birth to age one varied by sex and by pup weight. For females, first year survival was 0.549 (95% CI: 0.369-0.717) for small to medium size pups, and 0.82 (95% CI: 0.646-0.919) for large size pups. For males, first year survival was 0.405 (95% CI: 0.248-0.584) for small to medium size pups, and 0.717 (95% CI: 0.520-0.856) for large size pups. The overall first year survival for large pups of both sexes combined was 0.750 (95% CI: 0.570-0.872). The authors note that this is similar to estimates from Härkönen et al. (1999) and Harding et al. (2005) for branded pups from the Skagerrak, where estimated survival to ~nine months was ~0.8-0.85. Assuming a constant survival rate, this gives a first year survival estimate of 0.74-0.81. Harding et al. (2005) estimated a first-year winter survival rate of 0.63 for small pups (95% CI 0.4-0.8) and up to 0.96 for larger pups. This estimate is for post-weaning pups, so first year survival (birth to age 1) will be lower once the effects of pre-weaning mortality are accounted for.

Matthiopoulos et al. (2014) used pup survival data from Thompson et al. (2007) and Härkönen and Heide-Jørgensen (1990) in an age-structured state-space model for the Moray Firth population. Their prior mean for pup survival in 2009 (when the population was stable) was 0.7 (95% CI: 0.62-0.78) and the resulting posterior mean was 0.70 (95% CI: 0.61-0.78).

Svensson (2012) used previously published data from Hansen and Harding (2006), Harding et al. (2007), Jüssi et al. (2008) to parameterise their matrix model of the harbour seal population in the Kattegat–Skagerrak region. The population had shown a high growth rate of ~9% p.a. and the female pup survival parameter of their model was set at 0.86.

Table 8

Range of values in the literature for harbour seal pup survival.

Reference	Data	Estimate	95% CI	Population Trend
Hanson et al. (2013)	Pup survival to 6 months	0.39	0.297-0.648	Both stable and declining
	Pup survival to 300 days	0.12	0.064-0.42	
Hastings et al. (2012)	Pre-weaning survival (to ~1 month)	0.74	-	increasing
	Post-weaning survival (to ~11 months)	0.72-0.82	-	
	Pup survival first year (Female, small-medium)	0.549	0.369–0.717	
	Pup survival first year (Female, large)	0.82	0.646-0.919	
	Pup survival first year (Male, small-medium)	0.405	0.248-0.584	
	Pup survival first year (Male, large)	0.717	0.520-0.856	
	Pup survival first year (Both sexes, large)	0.75	0.570-0.872	
Matthiopoulos et al. (2014)	Pup survival Prior	0.7	0.62-0.78	stable
	Pup survival posterior	0.7	0.61-0.78	
Harding et al. (2005)	Post-weaning (small pup)	0.63	0.4-0.8	
	Post-weaning (large pup)	0.96	-	
Svensson (2012)	Pup survival (no ice)	0.86		increasing

4.5.3.2 Juvenile survival

There are very few published data on harbour seal juvenile survival rates. Hastings et al. (2012) used mark-recapture methods to estimate annual juvenile survival at Tugidak Island, Alaska for males aged 1-3 to be 0.782 (95% CI: 0.706-0.842) and 0.865 for females (95% CI: 0.803-0.910). The apparent survival from birth to age four was 0.38 for females and 0.22 for males. This is lower than the survival to age four of 0.48 estimated by Heide- Jørgensen and Härkönen (1988) for females in the Kattegat–Skagerrak, during a period of rapid population growth. Matthiopoulos et al. (2014) used juvenile survival data from Thompson et al. (2007) and Härkönen and Heide-Jørgensen (1990) to model the Moray Firth population. Their prior mean for annual juvenile survival in 2009 (when the population was stable) was 0.8 (95% CI: 0.72-0.88) and the resulting posterior mean was 0.79 (95% CI: 0.71-0.87).

Svensson (2012) used previously published data from Hansen and Harding (2006), Harding et al. (2007), Jüssi et al. (2008) to parameterise their matrix model of the harbour seal population in the Kattegat–Skagerrak region. The survival for juvenile females was set at 0.86 (the same as for pup survival).

Table 9

Range of values in the literature for harbour seal juvenile survival.

Reference	**Data**	**Estimate**	**95% CI**	**Population Trend**
Hastings et al. (2012)	Male age 1-3	0.782	0.706-0.842	Increasing
	Female age 1-3	0.865	0.803-0.910	
	Female birth to age 4	0.38		
	Male birth to age 4	0.22		
Heide-Jørgensen and Härkönen (1988)	Female birth to age 4	0.48		Increasing rapidly
Matthiopolous et al (2014)	Prior mean in 2009	0.8	0.72-0.88	Stable
	Posterior mean in 2009	0.79	0.71-0.87	
Svensson (2012)	Female age 1-3	0.86		Increasing

4.5.3.3 Adult survival

Cordes and Thompson (2014) used photo-ID data collected at Loch Fleet (Moray Firth SMA) between 2006 and 2011 to estimate an adult female survival rate of 0.95 (95% CI: 0.91-0.37). This analysis has since been updated by Graham et al. (2017) who used re-sightings of 144 individually-recognisable females and 124 males at Loch Fleet collected between 2006 and 2016 to estimate a mean survival probability of 0.97 for females (95% CI: 0.95-0.99) and 0.94 for males (95% CI: 0.90-0.97).

Hastings et al. (2012) used mark-recapture methods to estimate an annual survival between age three and seven of 0.929 for females (95% CI: 0.858-0.966) and 0.879 for males (95% CI: 0.784-0.936) for animals at Tugidak Island, Alaska.

Matthiopoulos et al. (2014) used mark-recapture data collected between 2006 and (Cordes 2011) in an age-structured state-space model of the Moray Firth population. Their prior mean for adult female survival in 2009 (when the population was stable) was 0.97 (95% CI: 0.95-0.99) and the resulting posterior mean was 0.96 (95% CI: 0.94-98).

Svensson (2012) used previously published data from Hansen and Harding (2006), Harding et al. (2007), Jüssi et al. (2008) to parameterise a matrix model of the harbour seal population in the Kattegat–Skagerrak region. The adult female survival parameter of their model was set at 0.97.

Table 10

Range of values in the literature for harbour seal adult survival.

Reference	Data	Estimate	95% CI	Population Trend
Cordes and Thompson (2014)	Female survival (Loch Fleet)	0.95	0.91-0.37	Stable
Graham et al. (2017)	Female survival (Loch Fleet)	0.97	0.95-0.99	Stable
	Male survival (Loch Fleet)	0.94	0.90-0.97	
Hastings et al. (2012)	Female survival (Alaska)	0.929	0.858–0.966	Increasing
	Male survival (Alaska)	0.879	0.784–0.936	
Matthiopolous et al (2014)	Female survival (prior mean)	0.97	0.95-0.99	Stable
	Female survival (posterior mean)	0.96	0.94-98	
Svensson (2012)	Female survival (Sweden)	0.97		Increasing

4.5.3.4 Fertility

Cordes and Thompson (2014) used photo-ID data collected at Loch Fleet (Moray Firth SMA) between 2006 and 2011 to estimate a fertility rate of 0.89 (0.75-0.95) for multiparous females. Graham et al. (2017) used the reproductive histories for 86 multiparous females from the same population between 2007 and 2016 to estimate the mean proportion of breeders was 0.90 (95% CI: 0.87-0.93).

Matthiopoulos et al. (2014) used a prior mean for fertility in 2009 (when the population was stable) was 0.87 (95% CI: 0.84-0.90) and the resulting posterior mean was 0.864 (95% CI: 0.836-0.899).

Svensson (2012) used previously published data from Hansen and Harding (2006), Harding et al. (2007), Jüssi et al. (2008) to parameterise a matrix model of the harbour seal population in the Kattegat–Skagerrak region. The fertility parameter of their model was set at 0.9.

Table 11

Range of values in the literature for harbour seal fertility.

Reference	Data	Estimate	95% CI	Population Trend
Cordes and Thompson (2014)	Fecundity (multiparous females Loch Fleet)	0.89	0.75-0.95	Stable
Graham et al. (2017)	mean proportion of breeders (Loch Fleet)	0.90	0.87-0.93	Stable
Matthiopolous et al (2014)	Prior mean (Moray Firth)	0.87	0.84-0.90	Stable
	Posterior mean (Moray Firth)	0.865	0.836-0.899	
Svensson (2012)	Fertility (Sweden)	0.9		Increasing

4.5.4 Density dependence

The only UK harbour seal population for which there is evidence of density dependence is the Moray Firth SMA where Thompson et al. (2017b) (based on the relationship modelled in Matthiopolous et al. 2014), estimated the carrying capacity to be 2,000 and density dependence was thought to operate on fertility rate. However, the authors highlighted that there was limited understanding of the way in which density dependence operates in this population and that there is a high level of uncertainty over the value for carrying capacity.

Given the apparent decline in the population growth rate in The Wash SAC, it appears that this population may be approaching carrying capacity. However, it is not currently possible to estimate this carrying capacity or to model the mechanism by which density dependence is operating. Therefore, it is recommended that density dependence is not modelled for this population (or any other) until more information is available and the sensitivity of model outputs to density dependent assumptions has been fully explored.

4.6 Grey seal demographic parameters

4.6.1 Harwood & King 2014

“Although Anon. (2014) identified 11 MUs for grey seals in UK waters, telemetry studies (Russell et al. (2013) of UK grey seals have indicated that female seals may breed at colonies that are far removed from the MUs within which they are observed outside the breeding season. We therefore think that it is inappropriate to model the dynamics of each MU separately. We therefore suggest that a single population model is used for all MUs with demographic rates chosen so that the growth rate of the population is 1% per year, the overall growth rate of the British grey seal population in recent years (SCOS, 2012), p8). We based the demographic rates shown in Table 3.1 on the mean estimates in Thomas (2012) for colonies that are monitored annually, and then tuned them to achieve the desired population growth rate.”

4.6.2 Updated data

The recommendation from Harwood and King (2014) was that grey seals are best modelled as a single UK population, rather than modelling individual sub-units as discrete populations as is the case for harbour seals. However, in practice when assessing the impacts of offshore developments, UK Statutory Nature Conservation Advisers generally require assessment at the individual SMA scale because of a concern that assessing local and regional scale impacts against a national population will ‘dilute’ the potential effect and provide an underestimate of the magnitude of impact. Assessment of impacts with reference to individual SMAs is also required for Habitats Regulations Appraisal purposes, where advice in Scoping Opinions is often that the assessment of the potential for a Likely Significant Effect on seals as features of a Special Area of Conservation should be carried out at the SMA scale. It is beyond the scope of this project to redefine appropriate SMAs for grey seals. However, we review information on the relationship between breeding and foraging regions and describe recent modelling of UK grey seal population dynamics based on regional pup production data.

4.6.2.1 Grey seal: foraging and breeding regions

Russell et al. (2013) analysed data from satellite transmitters attached to female grey seals in four regions (Hebrides, Northern Scotland, East Coast and South-East Coast – see their Fig. 1) around the UK. They found that 58% of females that foraged in the East Coast region actually bred in the Northern Scotland region, and

21% of females that foraged in the Northern Scotland region bred in the Hebrides region. It is difficult to determine the most appropriate way to model populations which have such complex spatial dynamics, and it was not possible to resolve this question within the scope of this current project. We, therefore, recommend that future effort is focused on identifying appropriate spatial scales to model the impacts of disturbance on grey seal populations to inform EIAs and HRAs. One option would be to focus iPCoD modelling on the effects of disturbance on animals foraging in each of the regions used by Russell et al. (2013). These individual-level effects could then be translated into population-level consequences by assigning females to their appropriate breeding regions, using the transition probabilities calculated by Russell et al. (2013), or up-dated values for these probabilities (given the small sample size involved in their analysis).

4.6.2.2 Grey seal population modelling

SMRU and CREEM carry out modelling of the UK grey seal population using a Bayesian state-space model that assumes density dependence in pup survival in order to inform NERC’s Special Committee on Seals (SCOS). This model is fitted to annual and, more recently, biennial estimates of pup production and an independent estimate of total population size. Thomas et al. (2019) provide mean estimates of the demographic characteristics of this population. Overall the growth rate for the UK grey seal population has changed from ~6% p.a. in 1984-85 to <1% p.a. since 2002. However, these trends differ among the four breeding regions (North Sea, Inner Hebrides, Outer Hebrides and Orkney) modelled by Thomas et al. (2019). The North Sea region is growing at a near-constant rate. By contrast, growth rates in the other three regions slowed over time, and then exhibited a small decline (Figure 4).

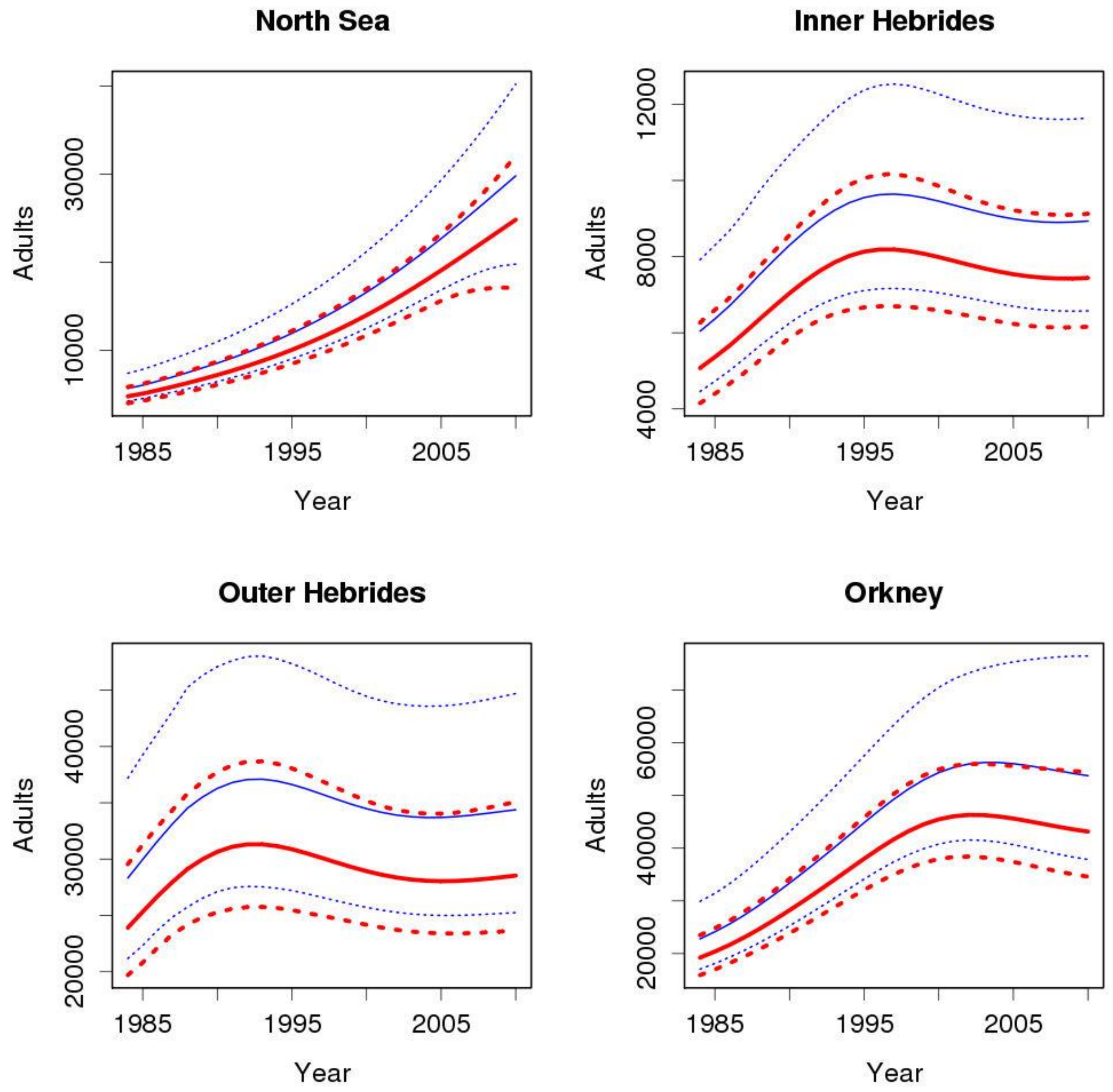


Figure 4: Figure S8 from Thomas et al. (2019) showing posterior mean estimates of regional adult population size (solid lines) and 95% credible interval (dashed lines). Thick red lines show the fit to pup production data plus the total population estimate of 2008; thinner blue lines show the fit to pup production data alone.

The most recent pup production data used to fit this model are already quite old (2010) and so the model may not provide reliable estimates of current population trends and demographic rates. SCOS (2018) presents a model that was fitted to the same time series of pup production estimates, plus additional values from 2011-2014 and a second independent estimate of total population size from 2014. However, survey methods were changed in 2011, and data from the two periods are not comparable. The best way to model changes in the population since 2010 is currently a matter of discussion as part of the ongoing SCOS process.

There are certain notable differences between the values of the demographic parameters used in the Thomas et al. (2019) model, in the SCOS (2018) model and those currently recommended for iPCoD. Both the Thomas et al. (2019) and SCOS (2018) models uses age six as the age of first breeding, whereas iPCoD recommends using age five. However, the key difference is the inclusion of density dependence acting upon pup survival. Thomas et al. (2019) and SCOS (2018) use a density dependent relationship with the same shape for each of the four breeding regions, but different carrying capacities for each region. iPCoD does not currently incorporate density dependence in the grey seal population model.

Table 12

Grey seal demographic parameters obtained from review.

Data Source	**Age independent**	**Age of first birth**	**Pup Survival**	**Juvenile Survival**	**Adult Survival**	**Fertility**	**Growth Rate**	**Carrying Capacity**
	age1	**age2**	**Surv[1]**	**Surv[7]**	**Surv[13]**			
Current iPCoD–whole UK (Harwood and King 2017)	1	5	0.222	0.94	0.94	0.84	1.0100	None
Thomas et al. (2019) prior mean	1	6	0.62	0.90	0.90	0.83	Differs by region	Differs by region
Thomas et al. (2019) posterior mean	1	6	0.48	0.95	0.95	0.90	Differs by region	Differs by region
SCOS (2018) prior mean	1	6	0.62	0.90	0.90	0.83	Differs by region	Differs by region
SCOS (2018) posterior mean	1	6	0.37	0.96	0.96	0.95	Differs by region	Differs by region

4.6.3 Recommended grey seal demographic parameters

We do not provide updated recommendations for grey seal parameter values for modelling disturbance impacts on UK grey seal population using iPCoD. However, we do recommend that future development of iPCoD should include the ability to model impacts on elements of the grey seal population at a finer spatial resolution than the whole UK, as is currently the case. As noted above, one possibility would be to refine the movement model developed by Russell et al. (2013) in order to allocate females in each of the foraging regions to the four breeding regions currently recognised by SCOS (North Sea, Inner Hebrides, Outer Hebrides and Orkney). This would also allow density dependence to be incorporated into the iPCoD grey seal model.

5 Phase 2: Sensitivity of the model outputs to different demographic rates

The adoption of the metrics to provide as standard iPCoD outputs advised by MSS in recent scoping opinions (e.g. MS-LOT 2017b, a) came from a study which examined the sensitivity of output metrics when modelling the population consequences of impacts of offshore renewable developments on seabirds using Population Viability Analyses (PVA) (Jitlal et al. 2017). However, the results of this process may not apply equally well to the output metrics of the iPCoD framework on marine mammals. There are substantial differences in the way offshore renewable energy developments are likely to affect populations of these two taxa. The main impacts on marine mammals are likely to be during the piling installation phase of an offshore wind farm, whereas seabirds are likely to be impacted throughout its entire operational lifespan. It is important that the output metrics reported are robust to misspecification of demographic parameters and that any sensitivities are identified, explained and documented.

Jitlal et al. (2017) recommended a set of metrics that were relatively insensitive to misspecification of the input parameters, and, therefore, would enable a robust assessment of offshore renewable effects on seabirds. The recommended metrics were:

- median of the ratio of impacted to un-impacted annual growth rate
  *If the median of the ratio of impacted to un-impacted annual growth rate equals one, then the median impacted population annual growth rate is no different to the median un-impacted population annual growth rate. If the median of the ratio of impacted to un-impacted annual growth rate <1 then the median impacted annual growth rate is smaller than the median un-impacted annual growth rate. We refer to this as the "**growth rate ratio**".*

- median of the ratio of impacted to un-impacted population size
  *If the median of the ratio of impacted to un-impacted population size equals one, then the median impacted population size is no different to the median un-impacted population size. If the median of the ratio of impacted to un-impacted population size <1 then the median impacted population size is smaller than the median un-impacted population size. We refer to this metric as the "**median counterfactual of population size**" or "**counterfactual**", for short. We also calculate (1 – the counterfactual) x100, which is the percentage difference in size between the impacted and un-impacted populations.*

These metrics have since been recommended by MSS for reporting the population level effects of offshore renewables on marine mammal populations. The following section explores the application of these metrics to marine mammals and provides an assessment of their suitability for marine mammal populations.

A 25-year scenario was simulated for the North Sea harbour porpoise population, with one piling operation involving 150 piling days in year one and a range of estimates for the number of animals disturbed on each day of piling (numDT).

As expected, increasing the number of animals disturbed per day results in a larger difference between the predicted trajectories for the impacted and the un-impacted population at the end of the 25-year simulation. The counterfactual at the end of the 25 year simulation decreased for increasing numDT values, but the median growth rate ratio did not show the same effect (Figure 5).

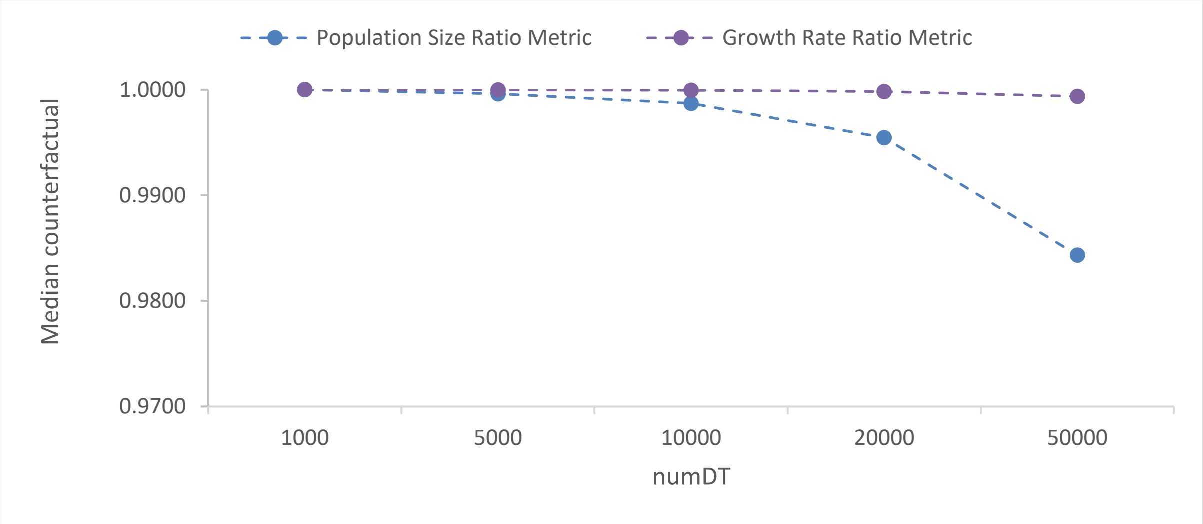


Figure 5: Output ratio metrics for various levels of disturbance after 25 years of simulation for a single operation involving piling for 150 days in year one.

The resulting metrics were then extracted for every year of the 25 year simulation to determine how they changed over time. The median counterfactual of population size declines at the end of year one (immediately after disturbance has ended) as a result of reductions in calf and juvenile survival caused by disturbance. It declines further by the end of year two (one year after disturbance has ended) as a result of the effects of disturbance on fertility. After the end of year two, the ratio increases because the proportion of adult animals in the population has increased. This increase continues until the initial, stable age structure has been established. The ratio then remains stable over the rest of the 25 year simulation (Figure 6). That is, the effect of disturbance on the impacted population remains evident throughout the 25 year simulation.

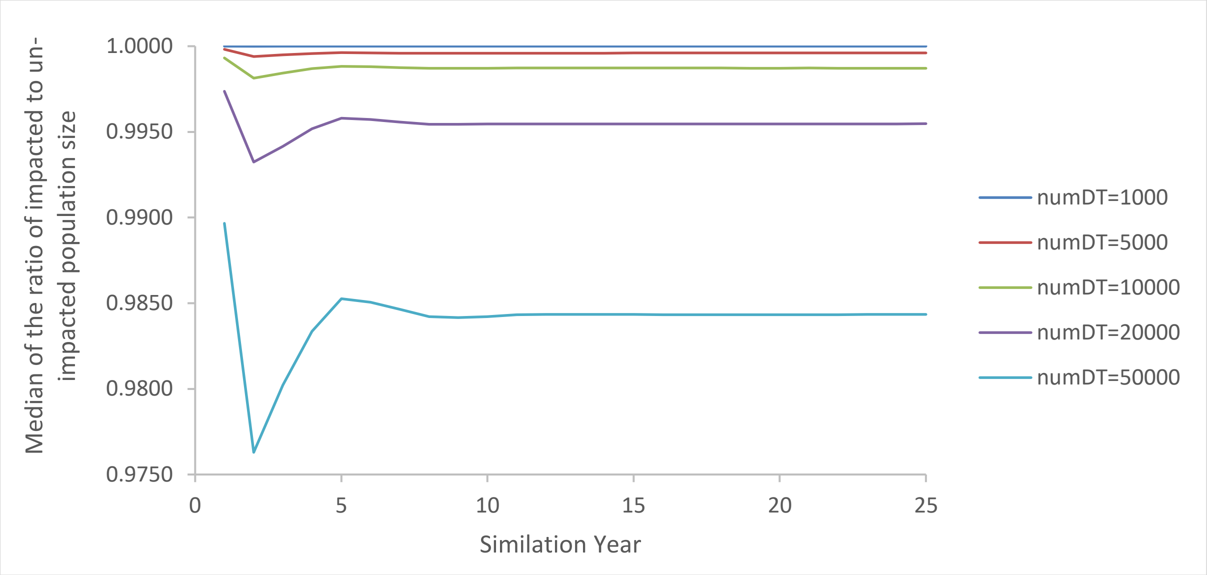


Figure 6 Median of the ratio of impacted to un-impacted population size for various levels of disturbance (numDT) after each year of the 25 year of simulation for a single operation involving piling for 150 days in year one.

A different pattern is evident in the growth rate ratio over the 25 year simulation (Figure 7). The ratio declines at the end of year 1 (immediately after disturbance has ended), remains low at the end of years two, three, four, and five. However, after the end of year five, the ratio increased slowly throughout the remaining years of the simulation, resulting in a median ratio of ~1 by the end of year 25. This is in contrast to the effect simulated by Jitlal et al. (2017), which was the permanent exclusion of birds from the windfarm, resulting in a steady decline in abundance over the whole simulation. Our scenarios result in the following sequence of events:

- A sharp reduction in impacted population size at the end of the first year as a result of increased pup/calf and juvenile mortality, and decreased fertility. The scale of this will depend on the magnitude of the simulated impact and the effects predicted by each of the virtual experts that are sampled.
- A smaller reduction in the following year because of reduced recruitment into the adult age class.
- A gradual increase in the impacted population as a result of the distorted age structure, which will contain relatively more adult animals than the starting age structure. This stops once the stable age structure is re-established.

So, in the first two years the population growth rate is negative, then positive, and finally zero. As a result, the value of the Jitlal et al. (2017) ratio of impacted to un-impacted growth rate changes continuously as the time period over which its calculated increases, rendering it inappropriate as a measure of impact for an effect which does not persist over a long period of time.

The impact of offshore wind farm projects on marine mammals is largely confined to their construction period and, therefore, effects on mean growth rate are gradually diluted as the number of years over which the calculation is made increases. In contrast, the effects of an offshore wind farm on bird populations continues throughout its operation. We, therefore, suggest that the growth rate ratio is only used to assess the short-term effect of disturbance on marine mammals, and that the counterfactual of population size provides a better measure of the impact scenario's long-term population consequences. For cumulative construction scenarios where disturbance from construction may occur every year over a longer period as result of the sequential construction of many offshore wind farms within the same MU/SMA, this metric may be more appropriate. However, the changes to overall population growth rate resulting from multiple operations would still be complex and hard to interpret due to differing amounts of impact from each operation, perhaps affecting different subpopulations if over a wide area, and it would not result in a constant change in growth rate as seen in the scenarios explored in Jitlal et al (2017). In the remainder of the report we therefore focus on the sensitivity of the counterfactual of population size.

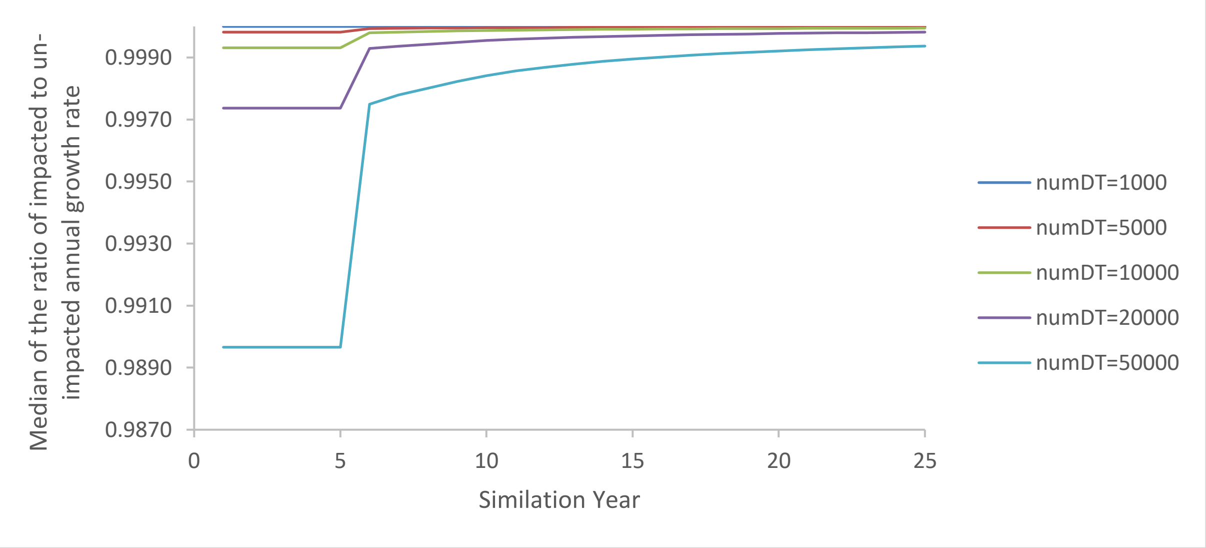


Figure 7: Median of the ratio of impacted to un-impacted annual growth rate for various levels of disturbance (numDT) after each year of the 25 year of simulation for a single operation involving piling for 150 days in year one.

It is important that the demographic parameters values selected for use in iPCoD modelling are as representative of the population being modelled as possible. However, given the limited empirical data for many of the marine mammal populations considered here, there is obvious uncertainty about the appropriate values for these parameters. The aim of the sensitivity analysis was to determine if changes in the value of any particular demographic parameters result in disproportionate changes in population trajectory.

5.1 Systematic changes in a single demographic parameter

Sensitivity to changes in the value of the demographic parameters was first tested by systematically changing the value of individual demographic parameters, within plausible limits, while leaving the other parameter values unchanged. As noted in the introduction, this ignores the effect of these changes on the growth rate of the un-impacted population. However, this method should be effective in identifying which demographic parameters have the most impact on modelled population trajectories.

The plausible limits to the ranges of values selected for the sensitivity analyses were informed by the reported values for each parameter from the wider review for each species presented in Section 3. It is important to note that in some cases parameter values have been included that, when combined with the other fixed values, result in population trajectories not observed in marine mammal populations.

The scenario tested was three consecutive years of pile driving with 118 days of piling in the first year, 138 in the second and 100 in the third (based on indicative piling schedules provided by industry). This was chosen as a compromise between more complex scenarios that would take considerable time to run each individual scenario and more simplistic single year scenarios that would be quick to run but would not represent the potential for cumulative impacts across multiple operations and years.

The levels of effect in terms of the numbers of animals impacted on a given day of pile driving was chosen to ensure a visible population level effect and to allow for some comparability between species (although mindful that there are other differences that complicate such comparisons). It is important to note that these levels of impact are unrealistically high compared to any assessments that we’ve carried out in practice.

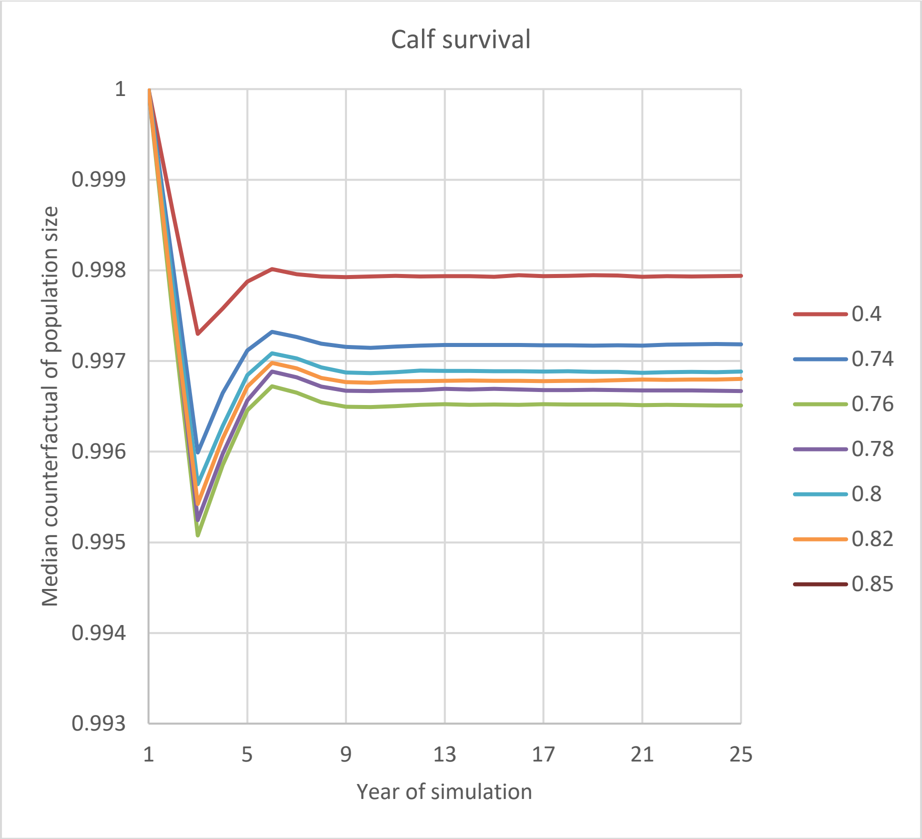
5.1.1 Harbour porpoise

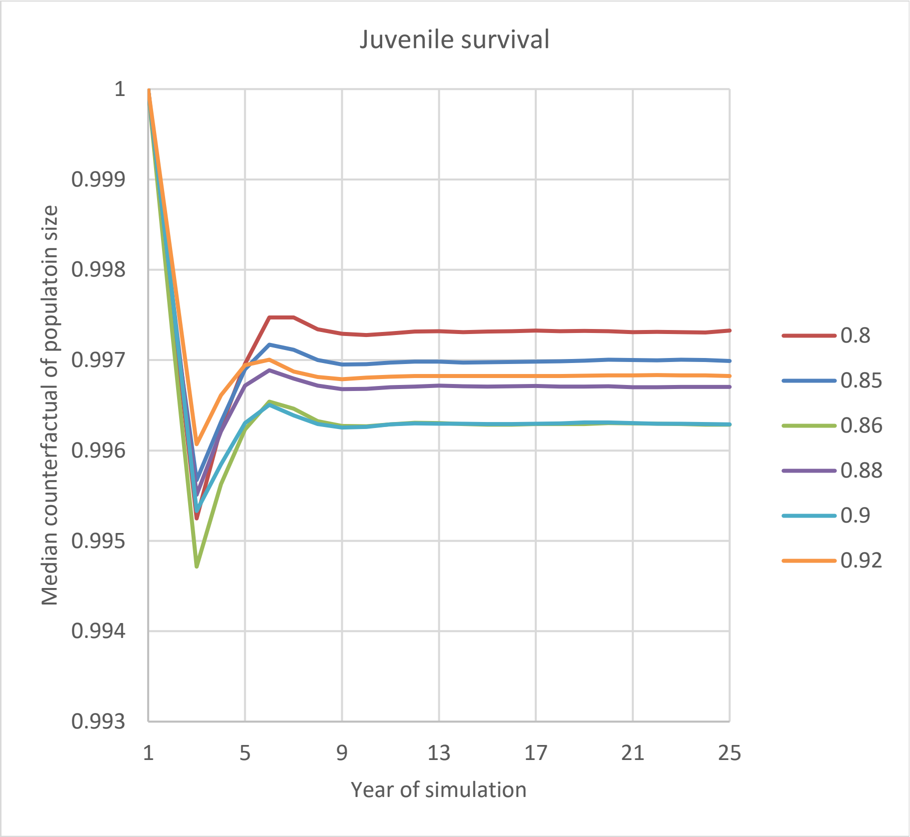
Three levels of impact were tested for harbour porpoises. At the lowest level of impact (numDT=7,000, ~2% of total population size) the largest difference between scenario counterfactuals at the end of the 25 year simulation across all the parameter values tested was only 0.14%.

For numDT=7,000, there was no clear pattern in the direction of the variation in the counterfactual as the parameter value changed for calf survival, juvenile survival or fertility, but it decreased systematically as adult survival was increased (Figure 8). For numDT=7,000, the median predicted difference in size ((1-counterfactual))*100) after 25 years between the impacted and un-impacted populations varied from 0.4% to 0.27% across all scenarios tested.

Overall sensitivity increased with increasing number of individuals predicted to be impacted on a given day at the higher levels of impact (numDT=17,000, Figure 9 and numDT=70,000, Figure 10). For numDT=17,000, ~5% of total population size, the median predicted difference in size ((1-counterfactual))*100) after 25 years between the impacted and un-impacted populations varied from 1.7% to 0.9% across all scenarios tested. There was also a clear pattern of increasing effect size as juvenile survival and fertility were increased, and a decreasing effect size as adult survival was increased (Figure 9).

For numDT=70,000 (~20% total population size), the median predicted difference in size ((1-counterfactual))*100) after 25 years between the impacted and un-impacted populations varied from 7.88% to 4.91% across all scenarios tested. Similar to the numDT=7,000 scenarios, there was also a clear pattern of increasing effect size as juvenile survival and fertility were increased, and a decreasing effect size as adult survival was increased (Figure 10).


Calf survival
Median counterfactual of population size
1
0.999
0.998
0.997
0.996
0.995
0.994
0.993
1
5
9
13
17
21
25
Year of simulation
0.4
0.74
0.76
0.78
0.8
0.82
0.85


Juvenile survival
Median counterfactual of populatoin size
1
0.999
0.998
0.997
0.996
0.995
0.994
0.993
1
5
9
13
17
21
25
Year of simulation
0.8
0.85
0.86
0.88
0.9
0.92

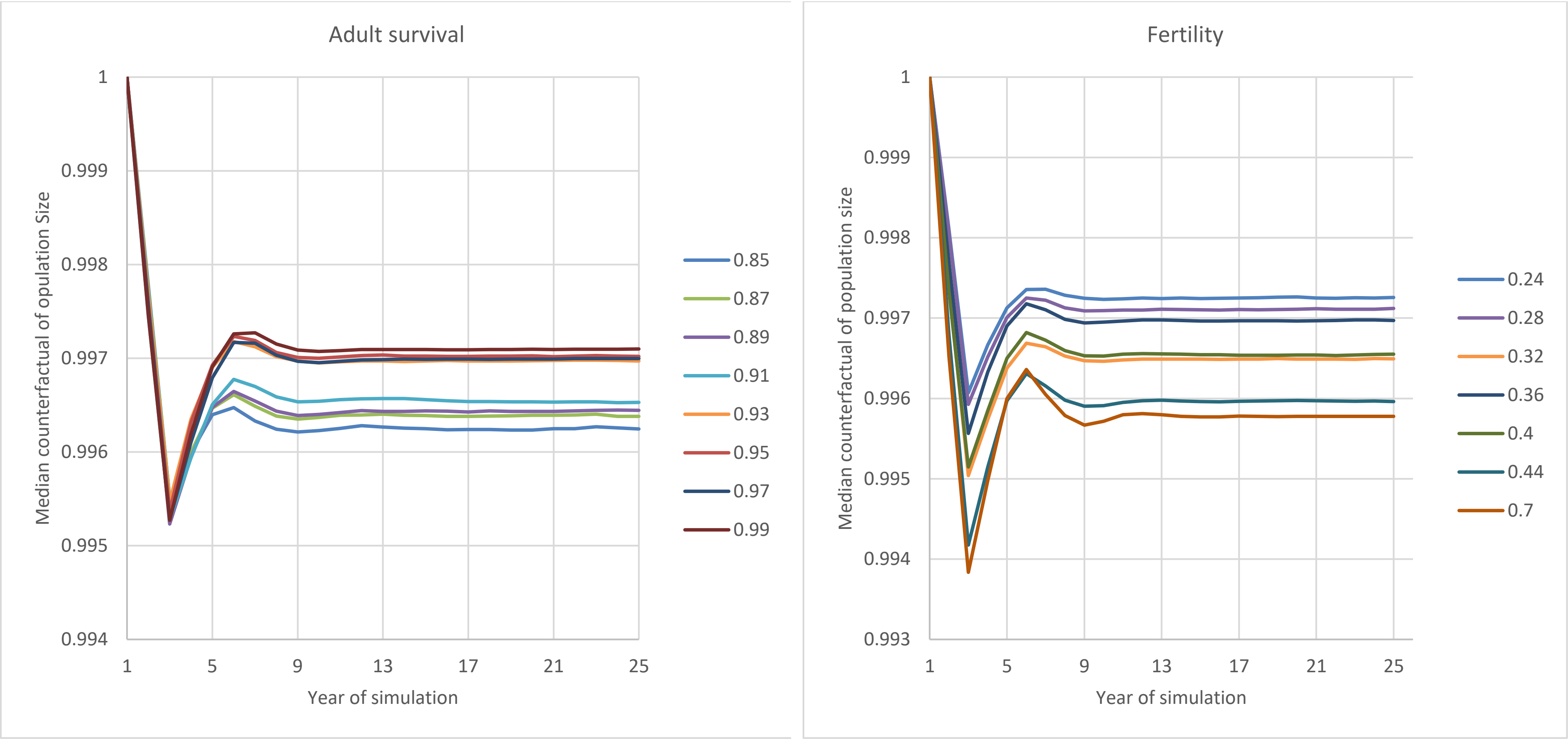
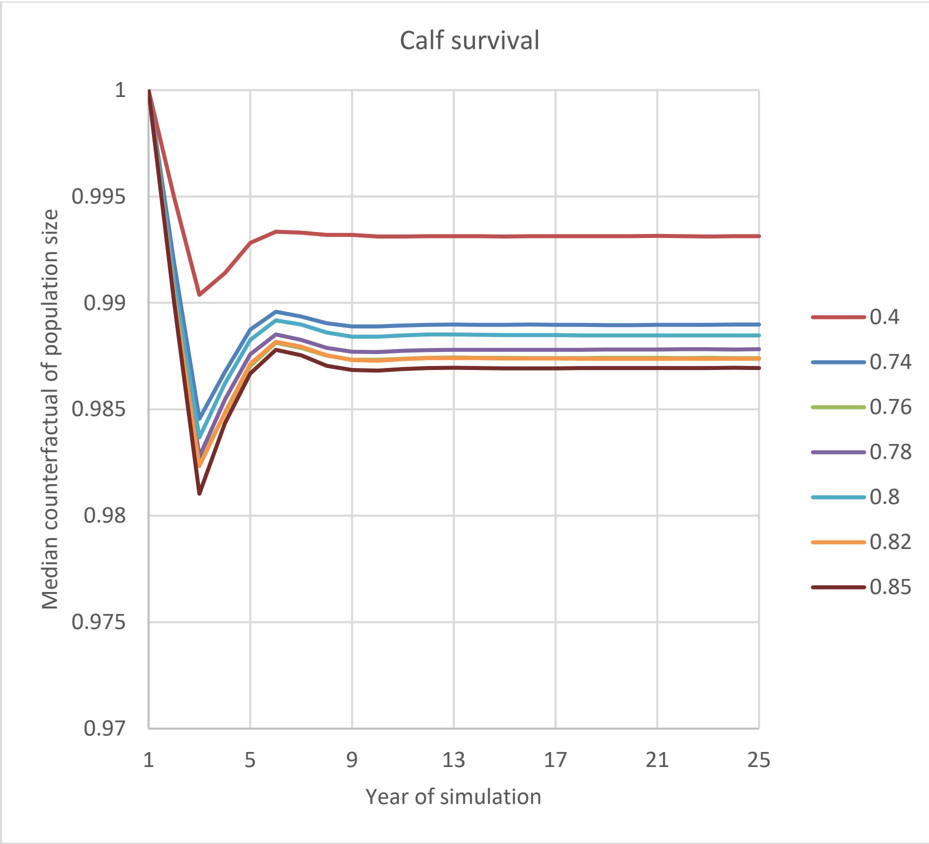
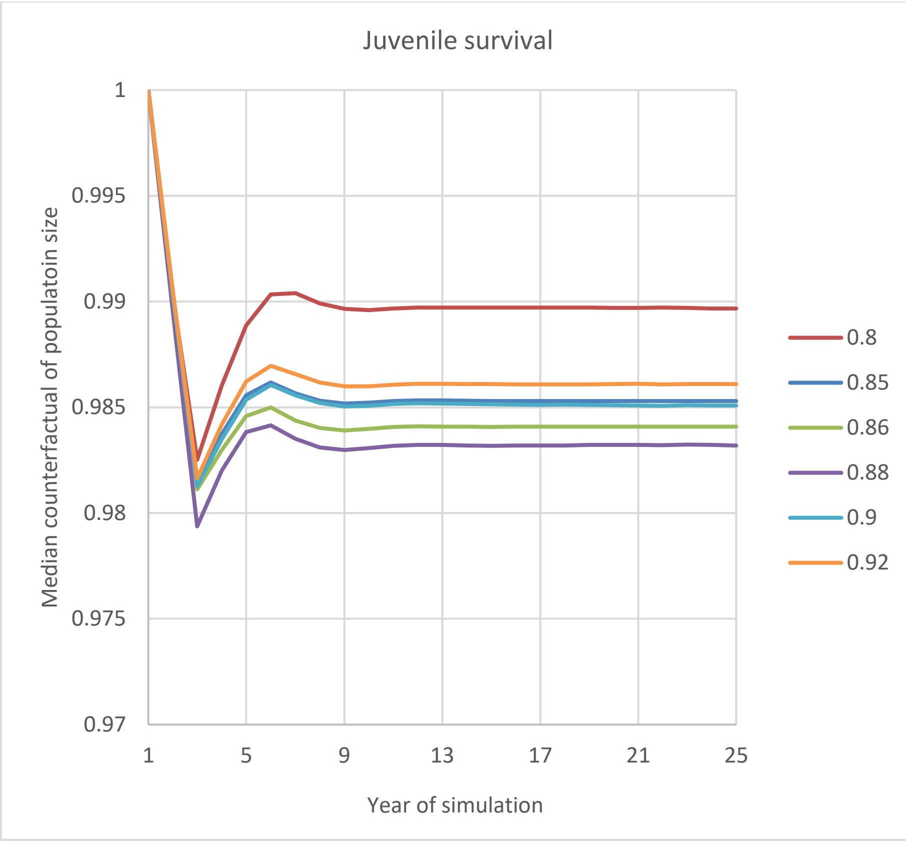


Figure 8: Harbour porpoise sensitivity analysis: exploring the effect of systematically varying demographic parameter values on the counterfactual of population size for a 2% disturbance impact. Total population size was 345,373 and number disturbed per operation per day was 7,000 over a three year piling period. The demographic parameters that were varied were calf survival (top left), juvenile survival (top right), adult survival (bottom left) and fertility (bottom right).


Calf survival
Median counterfactual of population size
1
0.995
0.99
0.985
0.98
0.975
0.97
1
5
9
13
17
21
25
Year of simulation
0.4
0.74
0.76
0.78
0.8
0.82
0.85


Juvenile survival
Median counterfactual of populatoin size
1
0.995
0.99
0.985
0.98
0.975
0.97
1
5
9
13
17
21
25
Year of simulation
0.8
0.85
0.86
0.88
0.9
0.92

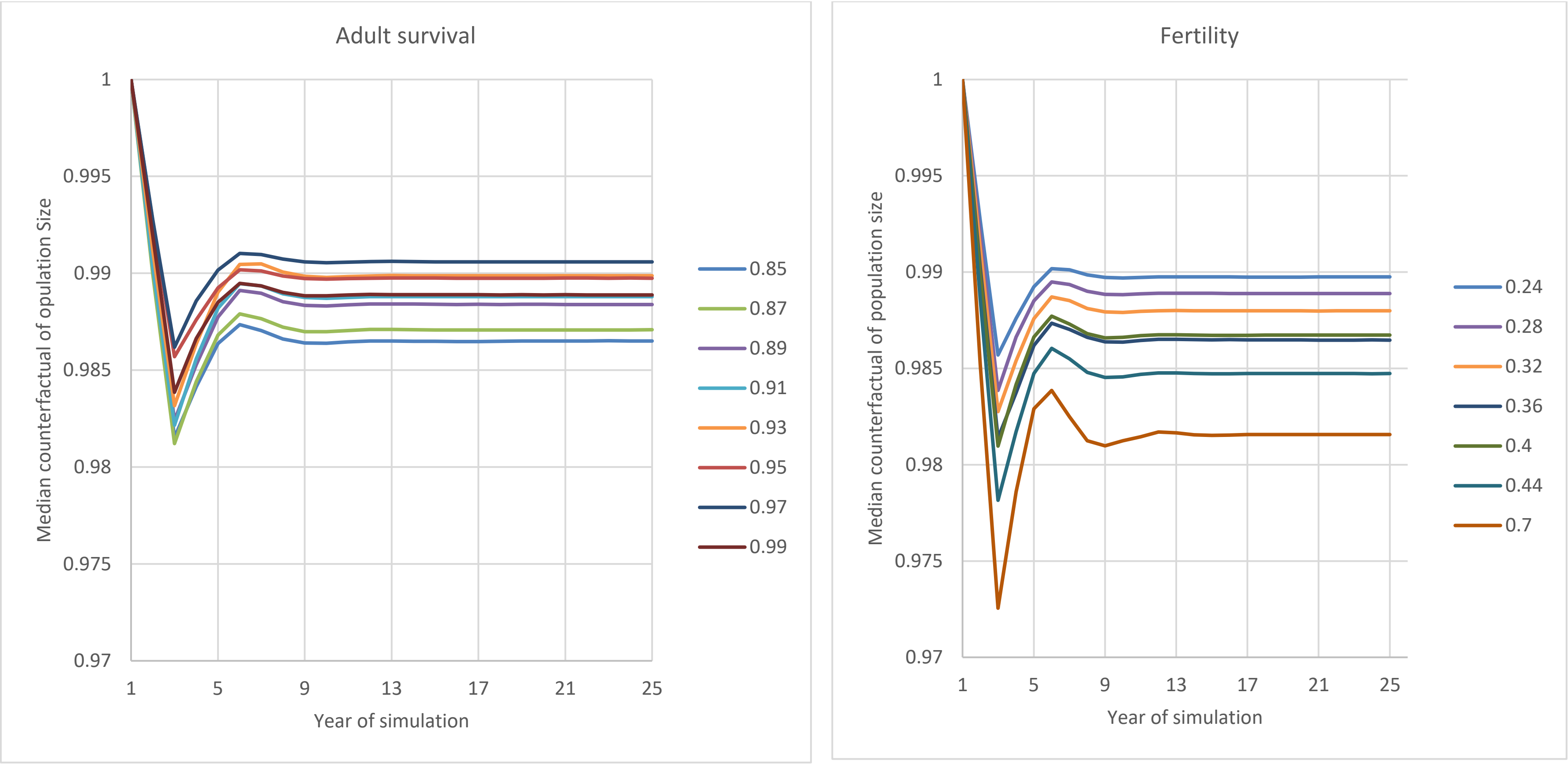
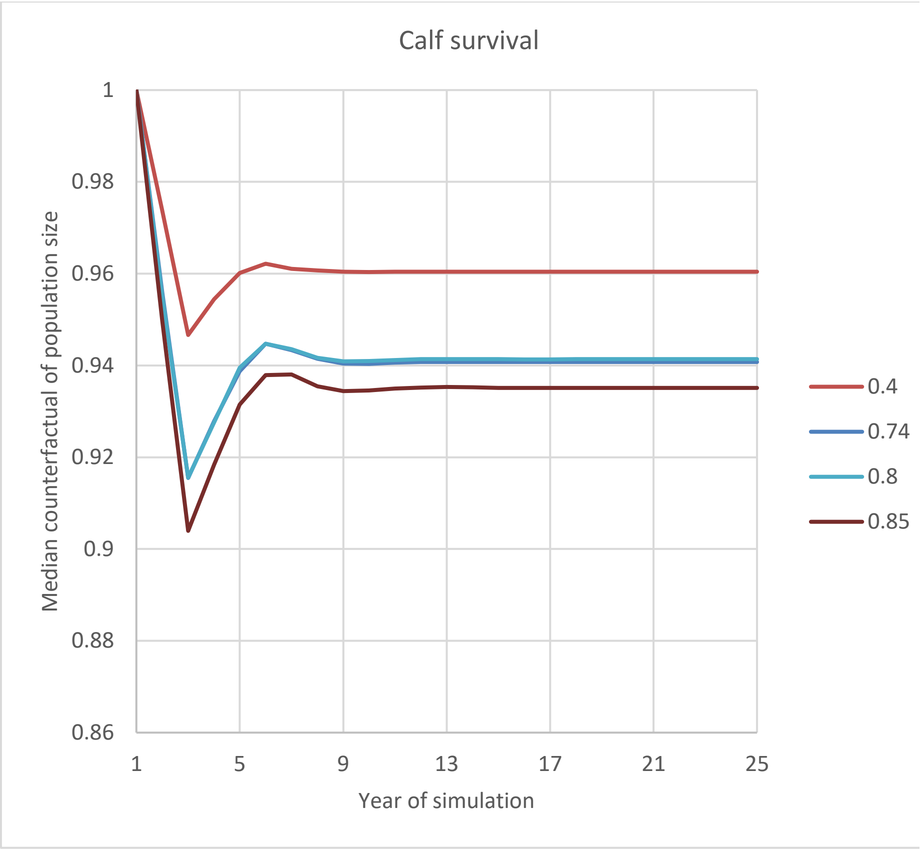
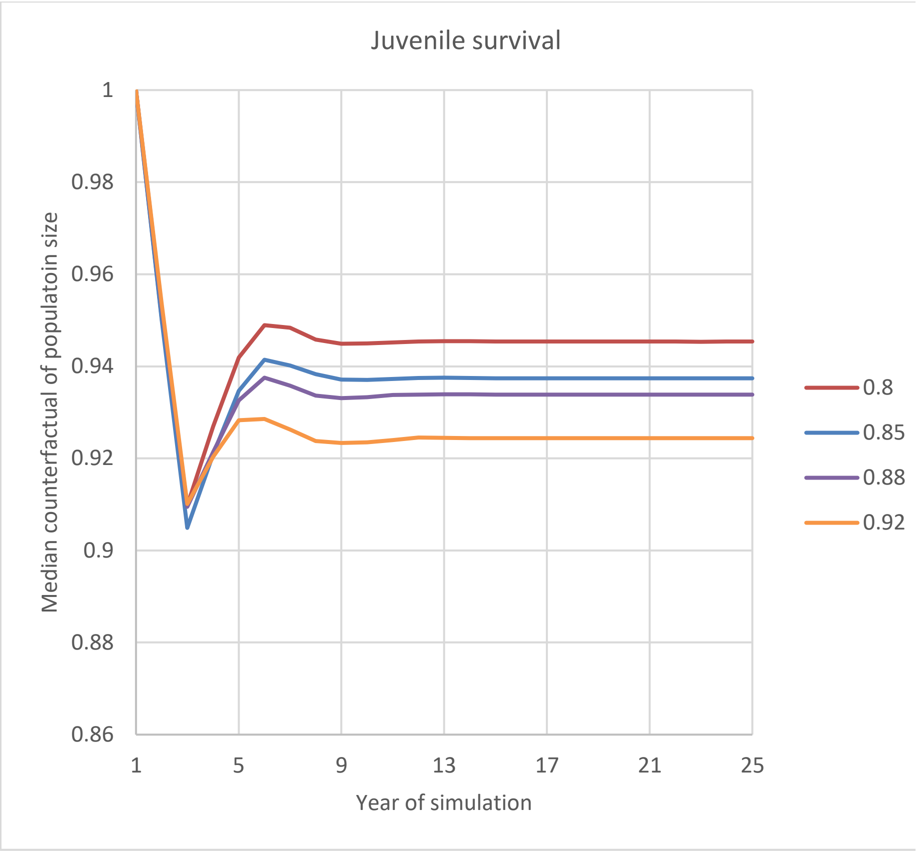


Figure 9: Harbour porpoise sensitivity analysis: exploring the effect of systematically varying demographic parameter values on the counterfactual of population size for a 5% disturbance impact. Total population size was 345,373 and number disturbed per operation per day was 17,000 over a three-year piling period. The demographic parameters that were varied were calf survival (top left), juvenile survival (top right), adult survival (bottom left) and fertility (bottom right).


Calf survival
Median counterfactual of population size
1
0.98
0.96
0.94
0.92
0.9
0.88
0.86
1
5
9
13
17
21
25
Year of simulation
0.4
0.74
0.8
0.85


Juvenile survival
Median counterfactual of populatoin size
1
0.98
0.96
0.94
0.92
0.9
0.88
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Year of simulation
0.8
0.85
0.88
0.92

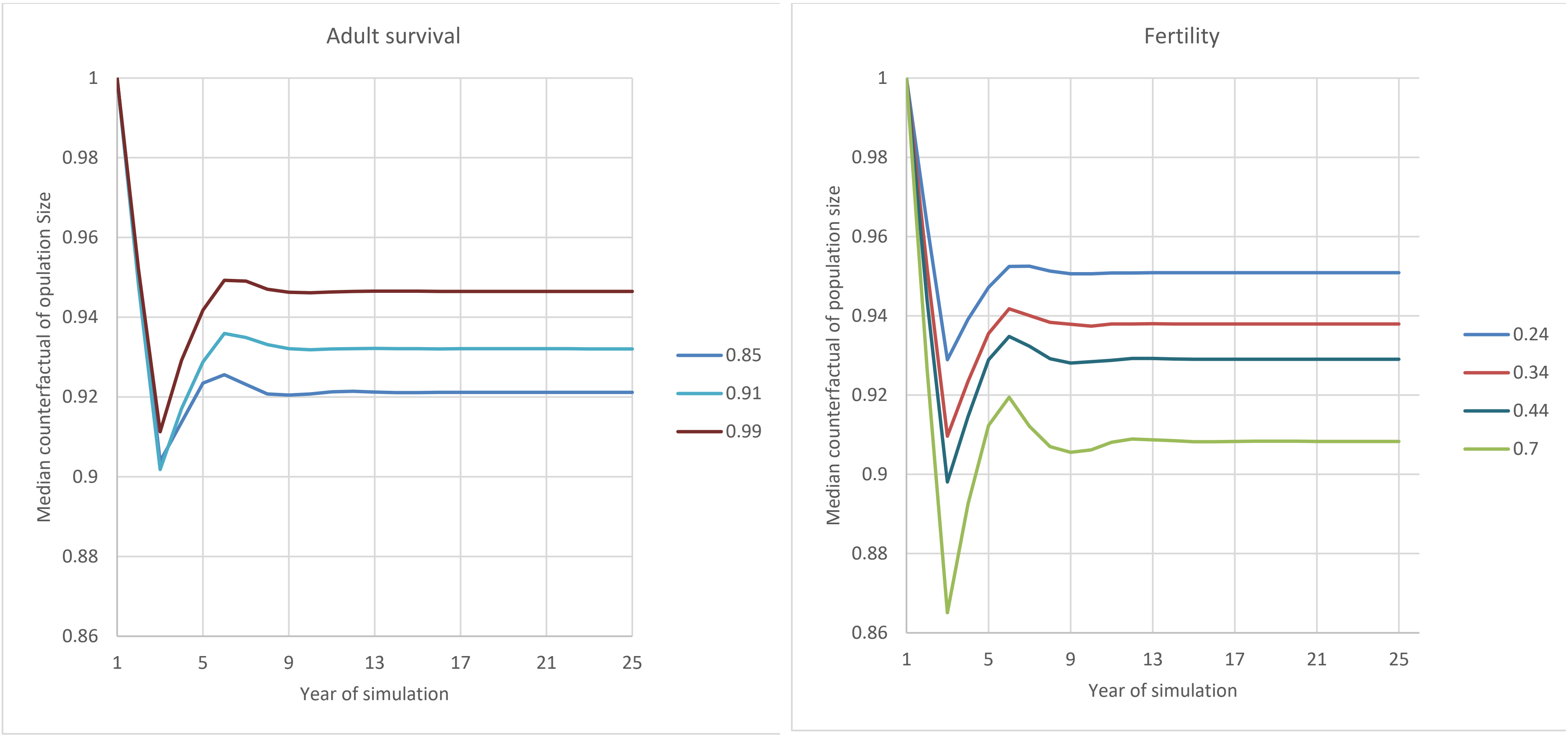


Figure 10. Harbour porpoise sensitivity analysis: exploring the effect of systematically varying demographic parameter values on the counterfactual of population size for a 20% disturbance impact. Total population size was 345373 and number disturbed per operation per day was 70,000 over a three-year piling period. The demographic parameters that were varied were calf survival (top left), juvenile survival (top right), adult survival (bottom left) and fertility (bottom right). Note that a smaller number of scenarios were tested for the highest level of impact (due to time constraints) although the lowest and highest values for each parameter were tested along with a smaller number of intermediate values. Also note the different y-axis scale on all three sets of figures indicating differences in sensitivity.

5.1.2 Bottlenose dolphin

The sensitivity in modelling outcomes to different values of the demographic parameters for bottlenose dolphins is higher than that observed for porpoises. However, a different range of parameter values were tested, therefore, it is difficult to compare sensitivity between species.

For simulations where numDT=50 (~26% of population size), the counterfactual of population size was most sensitive to differences in calf survival rates (0.4-0.95), which resulted in counterfactuals that differed by 6.3% (i.e. a median predicted difference in population size after 25 years of 1.2% with a calf survival of 0.95, and a difference of 7.6% with a calf survival of 0.4). The biggest difference in counterfactuals when juvenile survival rate was varied between 0.7 and 0.97 was 6.5%. The biggest difference when adult survival rates was varied between 0.8 and 0.99 was 5.1%. The biggest difference in counterfactuals when fertility rate was varied between 0.1 and 0.3 was 3.9%. The effect size increased with increasing calf survival, juvenile survival and fertility but decreased as adult survival increased (Figure 12).

This analysis was repeated with a lower level of impact (numDT=25, ~13% of population size). As expected, the overall predicted effect and the resulting sensitivity was lower than when using numDT=50. The biggest difference in counterfactual across all simulations was 3.8%.

Although some of the differences in counterfactuals associated with different parameter values may appear quite large, the variation between replicates within a particular scenario was as great as the variation between scenarios. It would be too complex to display this variability for all simulations, but Figure 11 shows the overlap in the interquartile range for scenarios using the highest and lowest values for adult survival.

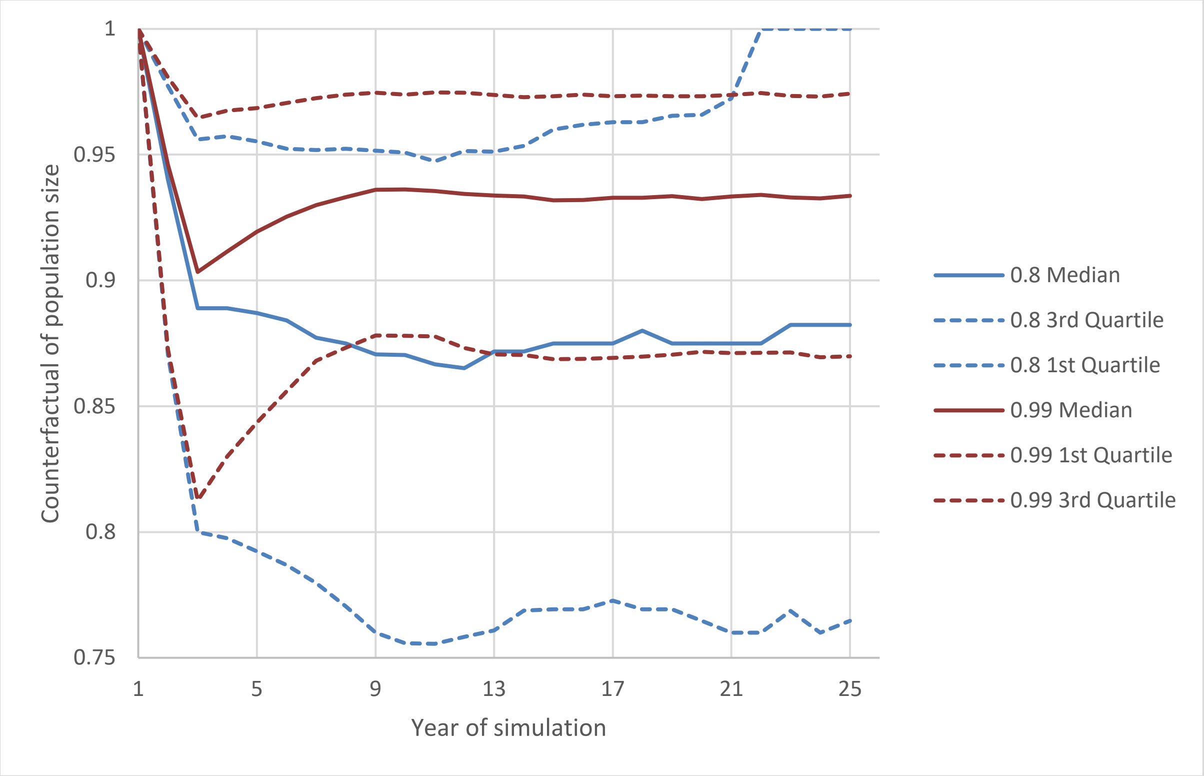
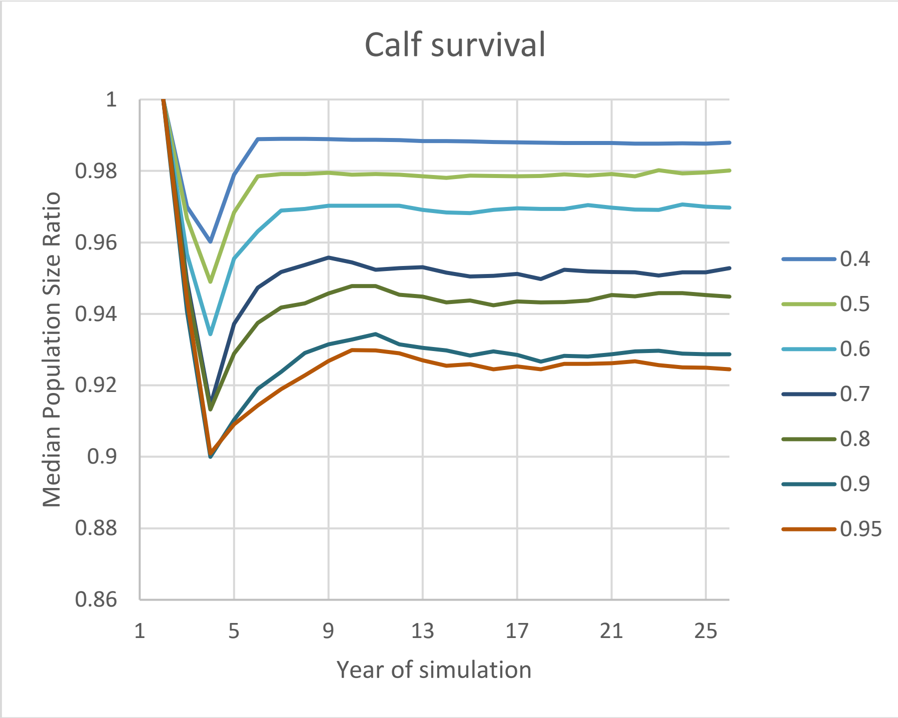
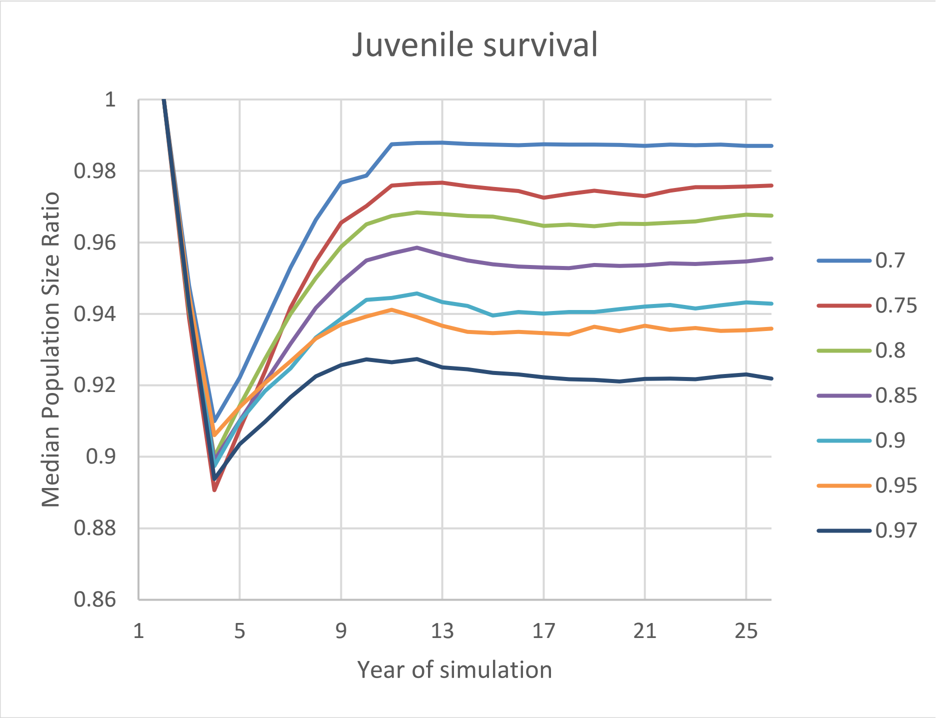
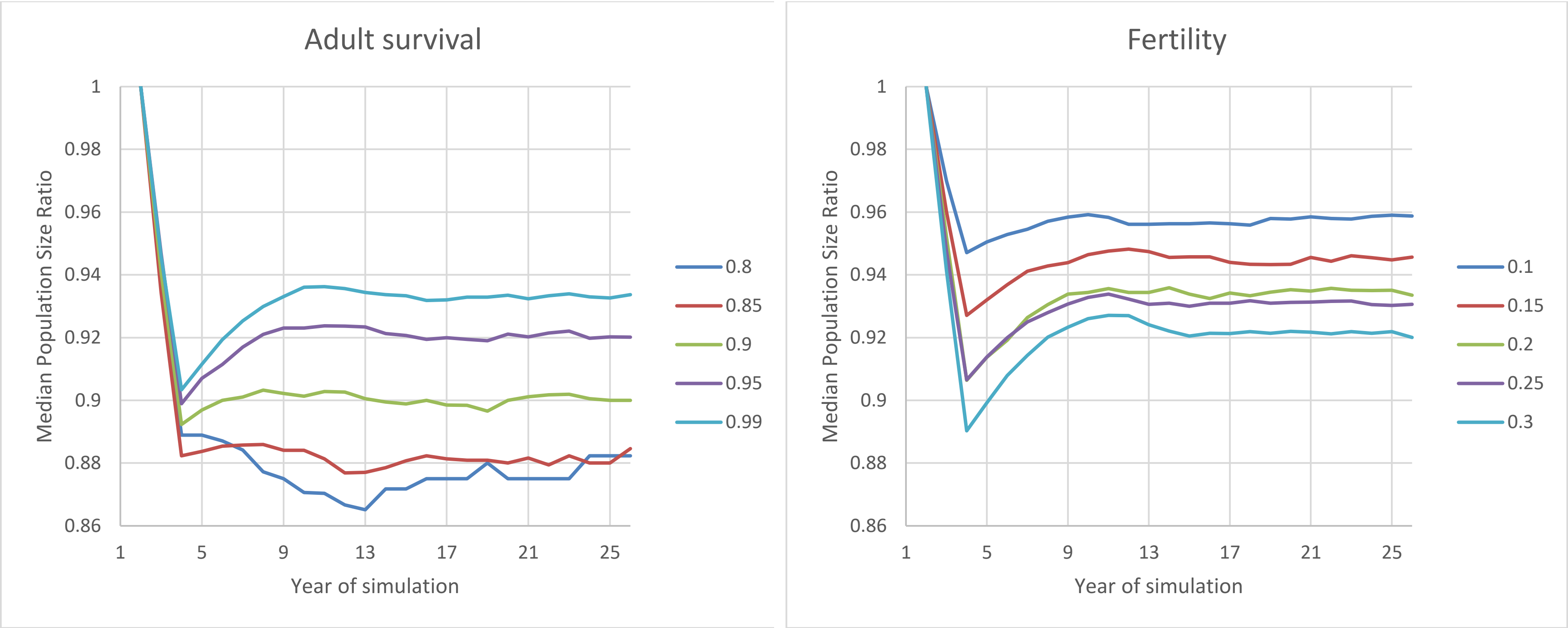


Figure 11: Median counterfactual and interquartile ranges for the outputs of two bottlenose dolphin scenarios with different values for adult survival (0.8 and 0.99) calculated over 500 replicates of the impacted and un-impacted simulated population trajectories.


Calf survival
Median Population Size Ratio
1
0.98
0.96
0.94
0.92
0.9
0.88
0.86
1
5
9
13
17
21
25
Year of simulation
0.4
0.5
0.6
0.7
0.8
0.9
0.95

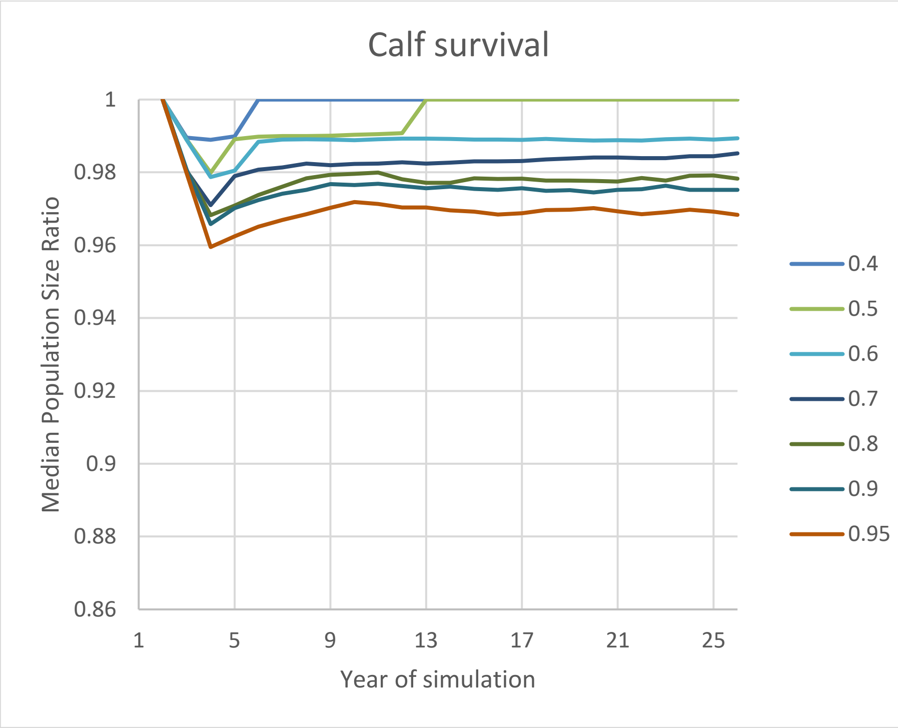

Juvenile survival
Median Population Size Ratio
1
0.98
0.96
0.94
0.92
0.9
0.88
0.86
1
5
9
13
17
21
25
Year of simulation
0.7
0.75
0.8
0.85
0.9
0.95
0.97

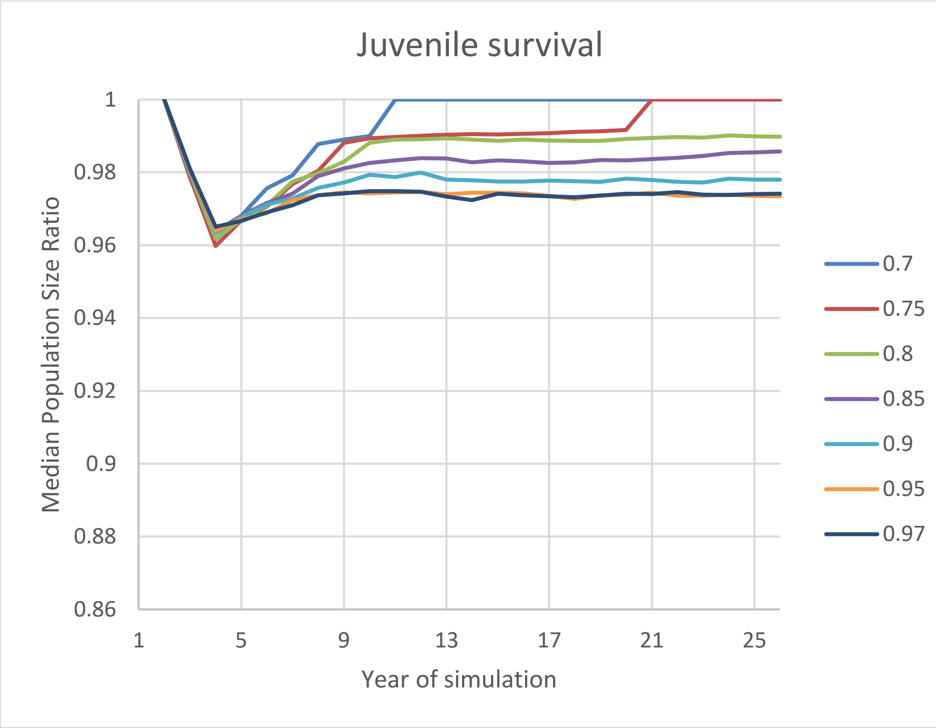


Adult survival

Fertility

Figure 12: Bottlenose dolphin sensitivity analysis: exploring the effect of systematically varying demographic parameter values on the counterfactual of population size for a 25% disturbance impact. Total population size was 195 and number disturbed per operation per day was 50 over a three year piling period. The demographic parameters that were varied were calf survival (top left), juvenile survival (top right), adult survival (bottom left) and fertility (bottom right).


Calf survival


Juvenile survival

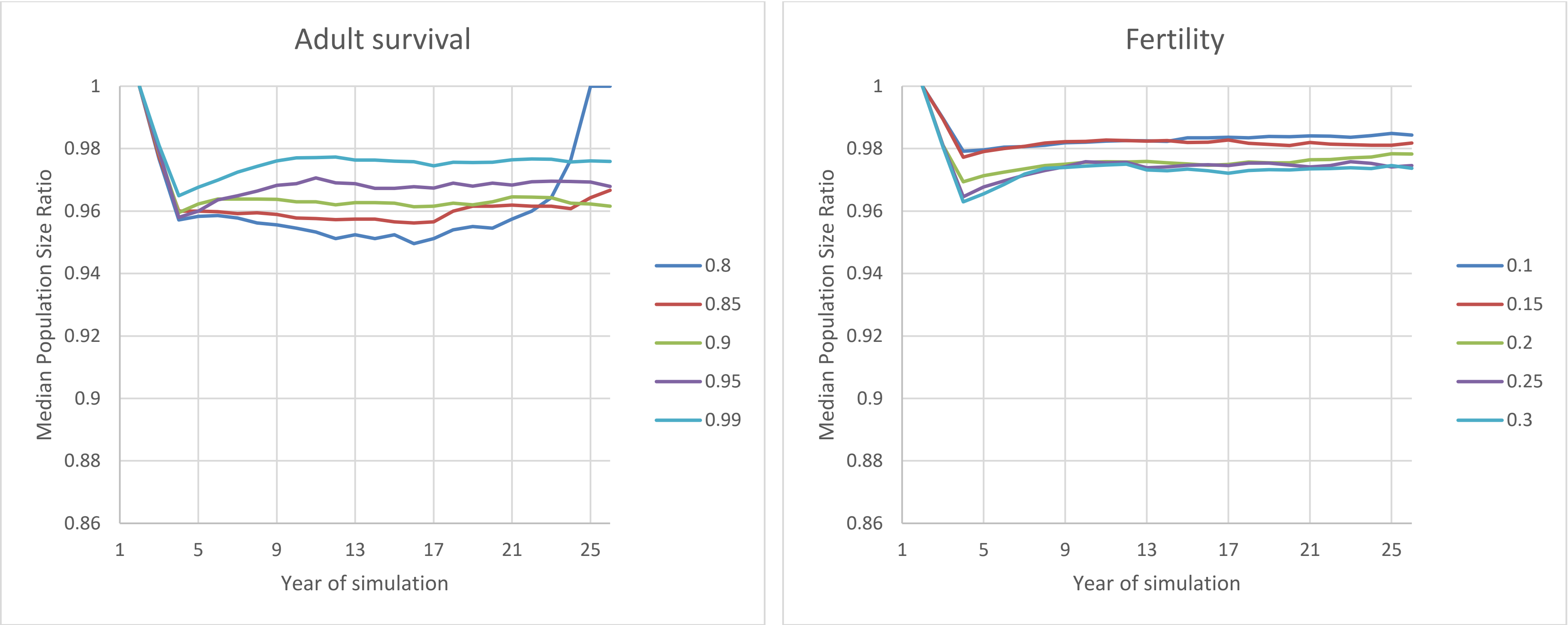
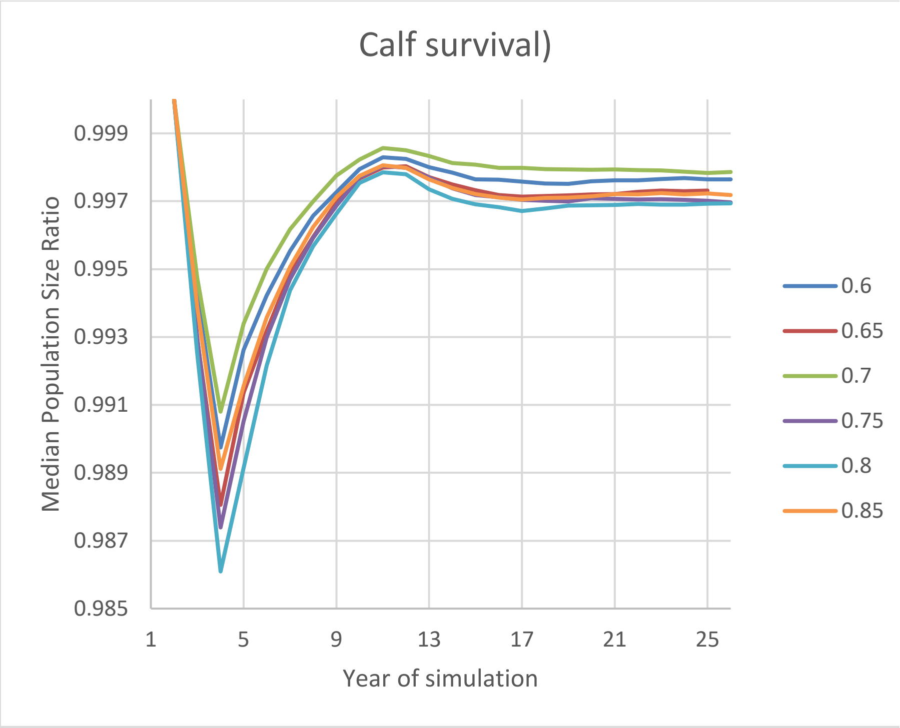


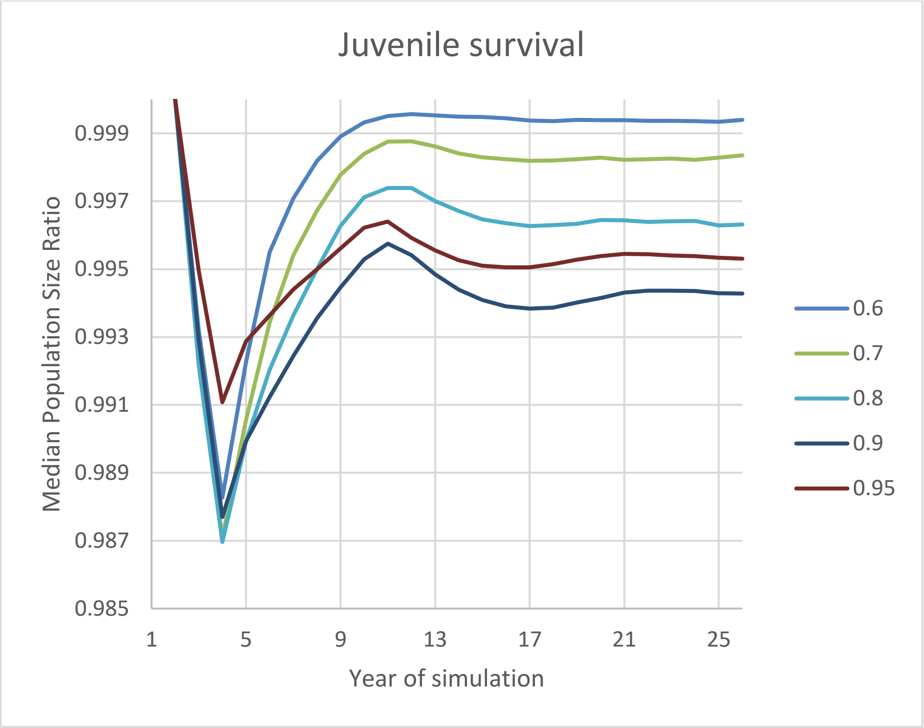
Figure 13: Bottlenose dolphin sensitivity analysis: exploring the effect of systematically varying demographic parameter values on the counterfactual of population size for a 13% disturbance impact. Total population size was 195 and number disturbed per operation per day was 25 over a three year piling period. The demographic parameters that were varied were calf survival (top left), juvenile survival (top right), adult survival (bottom left) and fertility (bottom right).

5.1.3 Minke whale

Model outcomes for minke whales were relatively insensitive to the values used for the demographic parameters (Figure 14), even with an unrealistically high level of impact (20% of total population size).

The biggest difference between counterfactuals after 25 years for calf survival rates between 0.6 and 0.85 was 0.09%; for juvenile survival rates between 0.6 and 0.95 it was 0.51%; for adult survival rates between 0.85 and 0.95 it was 0.13 %; and for fertility rates between 0.75 and 0.99 it was 0.06%. There was no clear pattern in effect size in relation to the direction of change in the demographic parameters (Figure 14).


Calf survival)
Median Population Size Ratio
0.999
0.997
0.995
0.993
0.991
0.989
0.987
0.985
1
5
9
13
17
21
25
Year of simulation
0.6
0.65
0.7
0.75
0.8
0.85


Juvenile survival
Median Population Size Ratio
0.999
0.997
0.995
0.993
0.991
0.989
0.987
0.985
1
5
9
13
17
21
25
Year of simulation
0.6
0.7
0.8
0.9
0.95

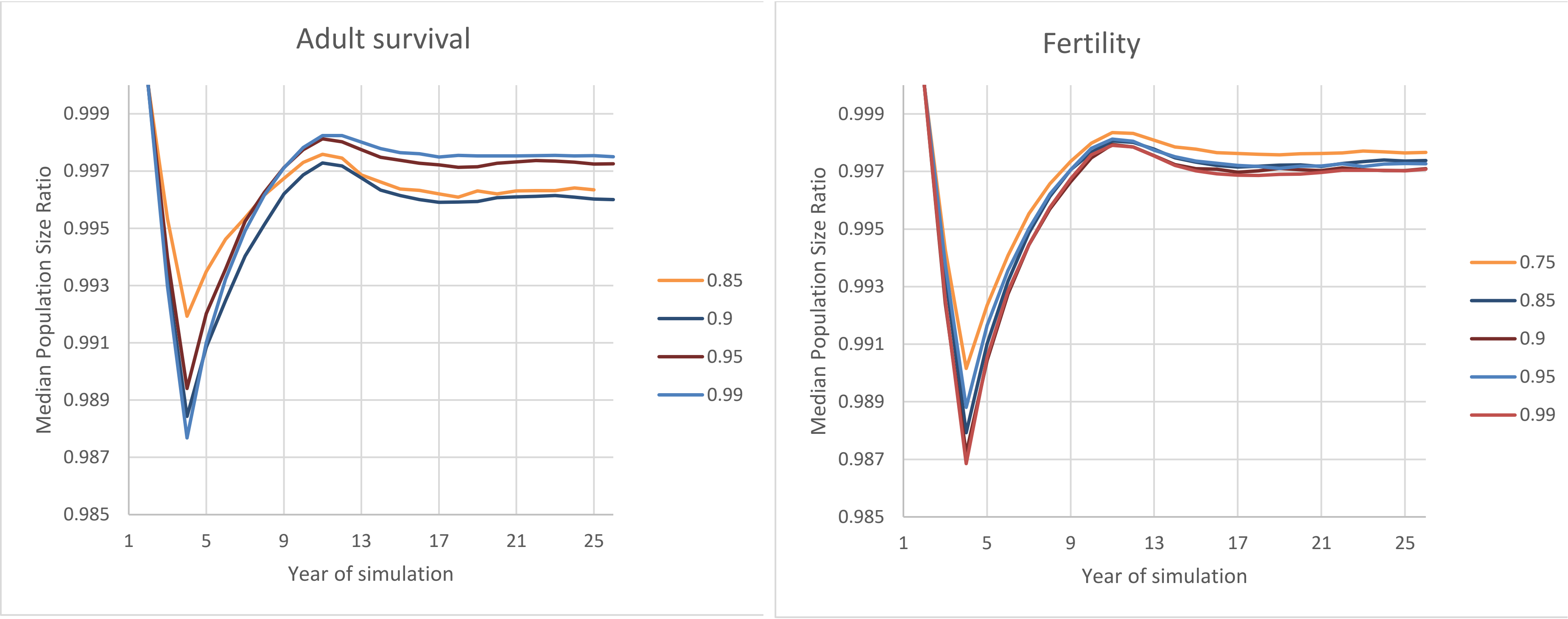


Figure 14. Minke whale sensitivity analysis: exploring the effect of systematically varying demographic parameter values on the counterfactual of population size for a 20% disturbance impact. Total population size was 19,680 and number disturbed per operation per day was 3,936 over a three-year piling period. The demographic parameters that were varied were calf survival (top left), juvenile survival (top right), adult survival (bottom left) and fertility (bottom right).

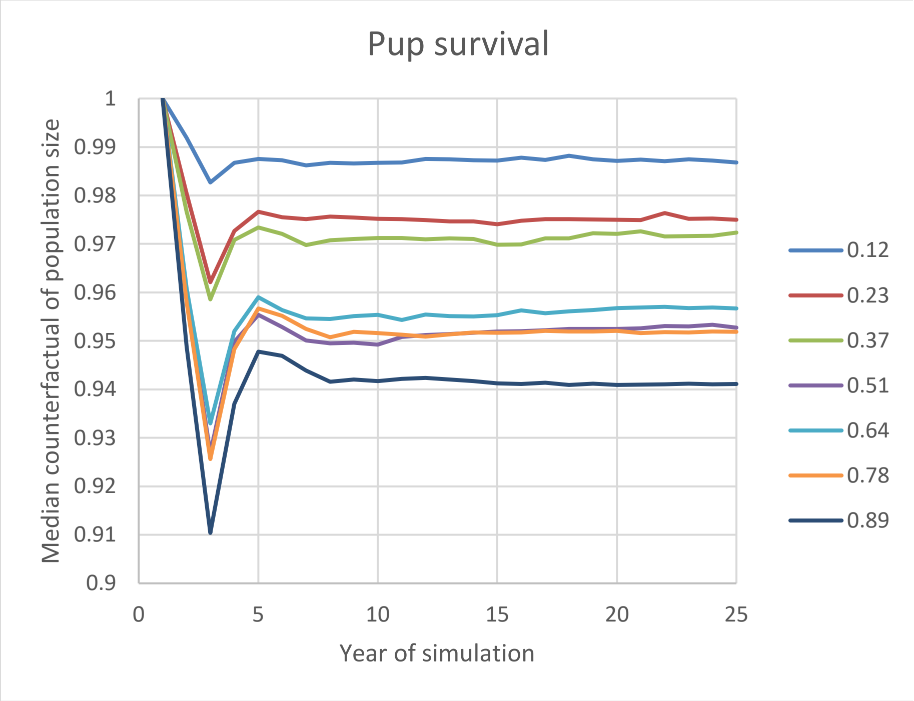
5.1.4 Harbour seal

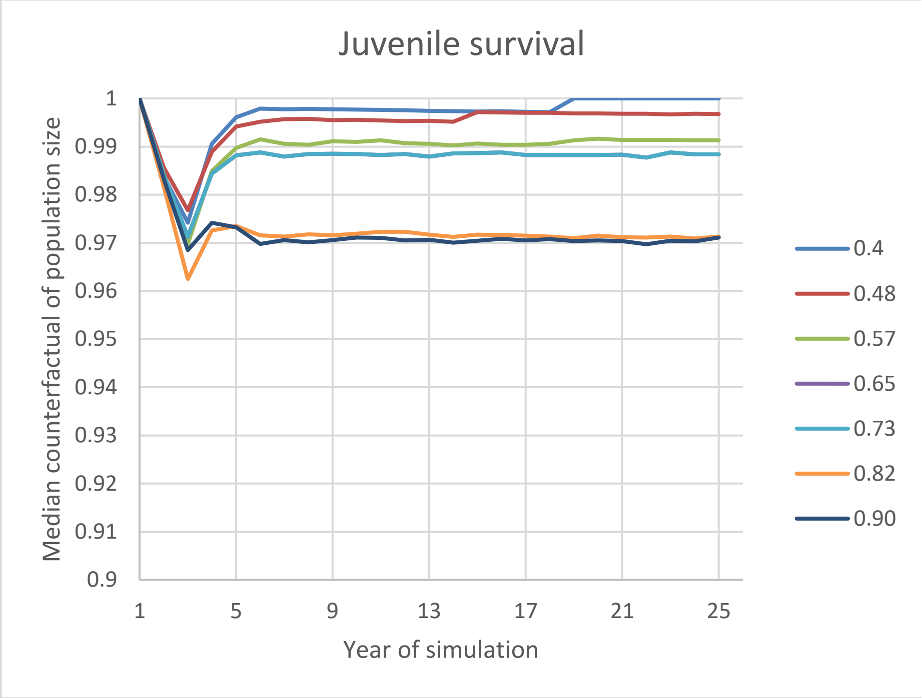
The sensitivity in modelling outcomes to variations in demographic parameter values for harbour seals differ considerably among parameters (Figure 15). A relatively high (and unrealistic) level of impact relative to population size was simulated (numDT = 400, ~33% of total population size). Given the result presented above for harbour porpoises and bottlenose dolphins, where sensitivity increased with effect size, it is likely that the sensitivity will be much lower if a more realistic impact scenario is used.

Predictions were most sensitive to variation in pup survival, although this was not unexpected given the wide range of pup survival rates tested (0.12 - 0.89). Predictions varied from a median counterfactual of 0.987 (equivalent to a 1.3% difference in population size after 25 years) with a pup survival rate of 0.12, to 0.941 (a 5.9% difference) with a pup survival rate of 0.89. Median counterfactual values for simulations with juvenile survival rates between 0.4 and 0.9 ranged from one (no difference between impacted and un-impacted populations), to 0.971 (a 2.89% difference). The effect of adult survival rates between 0.8 and 0.99 on the median counterfactual ranged from a one to 0.975 (a 2.53% difference between populations). Varying the fertility rate resulted in median counterfactuals ranging from 0.984 (a 1.6% difference between impacted and un-impacted populations) to 0.977 (a 2.31% difference).

The effect size increased with increasing pup survival, juvenile survival and fertility but decreased as adult survival increased (Figure 15).

As with the other species, the variability among replicates within a scenario was as great as the variability between scenarios, as demonstrated in Figure 16 where there is broad overlap in the interquartile ranges of the outputs obtained using two extreme values for pup survival.


Pup survival
Median counterfactual of population size
1
0.99
0.98
0.97
0.96
0.95
0.94
0.93
0.92
0.91
0.9
0
5
10
15
20
25
Year of simulation
0.12
0.23
0.37
0.51
0.64
0.78
0.89


Juvenile survival
Median counterfactual of population size
1
0.99
0.98
0.97
0.96
0.95
0.94
0.93
0.92
0.91
0.9
1
5
9
13
17
21
25
Year of simulation
0.4
0.48
0.57
0.65
0.73
0.82
0.90

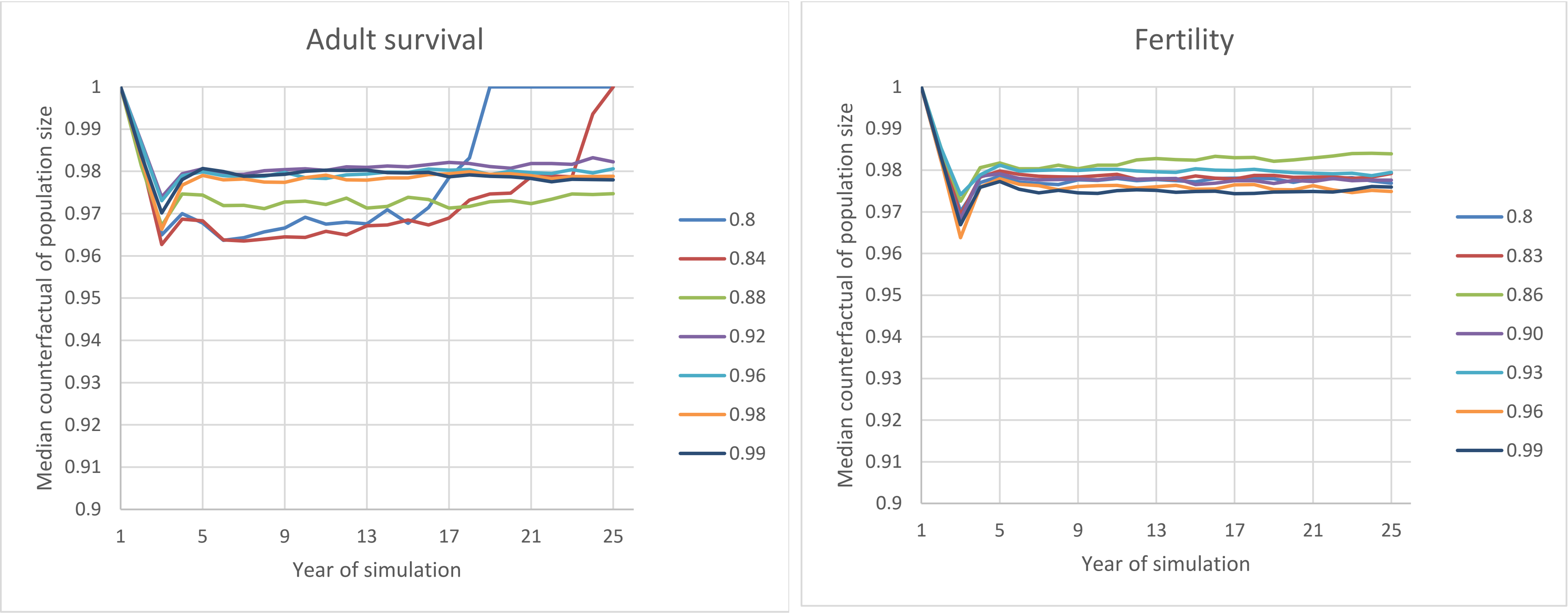


Figure 15: Harbour seal sensitivity analysis: exploring the effect of systematically varying demographic parameter values on the counterfactual of population size for a 30% disturbance impact. Total population size was 1,221 and number disturbed per operation per day was 400 over a three-year piling period. The demographic parameters that were varied were pup survival (top left), juvenile survival (top right), adult survival (bottom left) and fertility (bottom right).

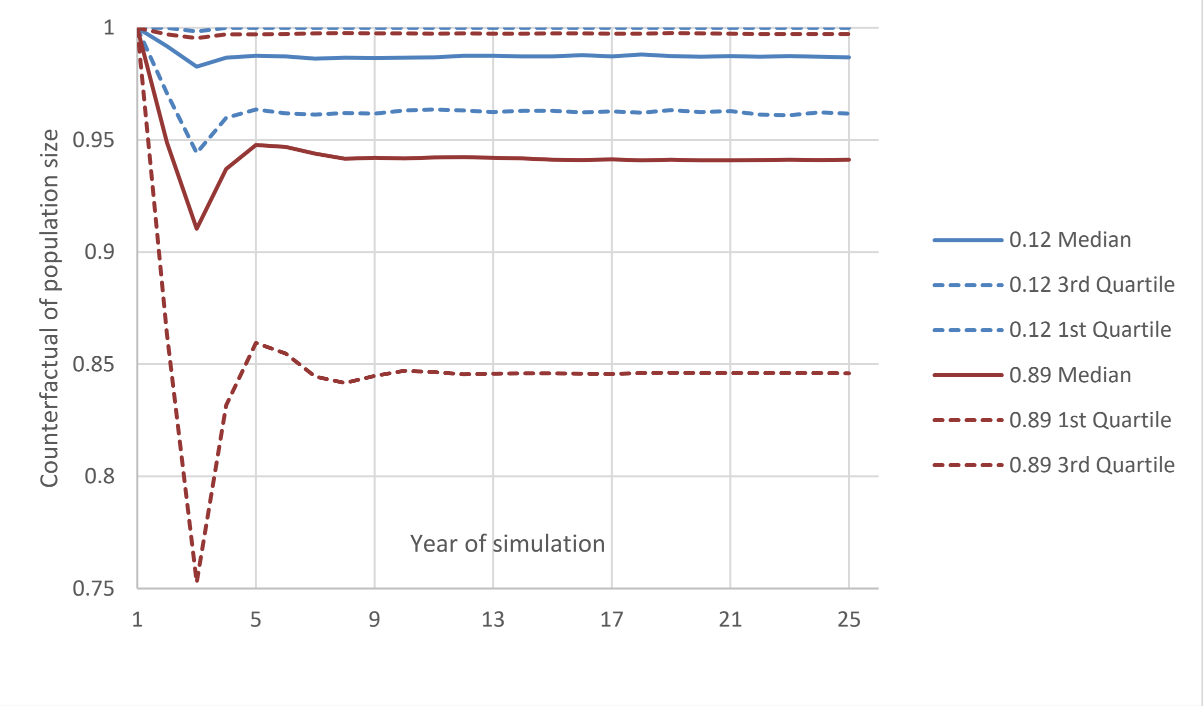


Figure 16: Median counterfactual and interquartile ranges for the outputs of two harbour seal simulations with extreme values (0.12 and 0.89) for pup survival calculated over 500 replicates of the impacted and un-impacted simulated population trajectories.

5.1.5 Summary of sensitivity to systematic changes in single demographic parameters

The analyses described above demonstrate substantial variation in sensitivity to parameter mis-specification among species and demographic variables. The greatest sensitivity was to variation in the value of pup survival for harbour seals and, overall, changes in pup/calf survival had the greatest impact on the counterfactual of population change. There are two main reasons for this. Across all species (with the exception of minke whales, whose calves are assumed not to occur in the sea areas covered by iPCoD) pup/calf survival is the demographic variable that experts considered to be most sensitive to the effects of disturbance. As a result, most of the simulations described in the previous sections predicted some effect of the operation being modelled on pup/calf survival, whereas effects of fertility were rare. In addition, the effect of the additional mortality associated with disturbance depends on the absolute value of the demographic rate that is affected. For example, an additional mortality of 10% in a population where pup/calf survival is 0.89 results in an 81% increase in the number of pups predicted to die in a particular year, whereas the same additional mortality only causes a 1.3% in predicted deaths if pup/survival is 0.12. This is because, in the latter case, most of the pups predicted to die as a result of disturbance would have died from natural causes if there was no disturbance. The widest range of pup survival values was used in the analysis of harbour seal population dynamics, which is why these simulations show the greatest effect. Sensitivity to misspecification in other demographic variables is largely a consequence of the effect of changing the value of a particular variable on the stable age structure used to initiate the iPCoD simulations.

The results described in Section 4.1.1-4.1.4 appear to indicate that iPCOD predictions of the effects of disturbance on bottlenose dolphin and harbour seal population are more sensitive to parameter misspecification than predictions for harbour porpoise and minke whale populations. However, this is partly a consequence of the relatively large size of the harbour porpoise and minke whale populations that are modelled and partly a consequence of the fact that the ranges of calf survival values are greater for harbour seals and bottlenose dolphins compared to those examined for the other species.

The observed sensitivity to misspecification is complex. It depends both on the proportion of the population that experiences some disturbance above the experts' threshold levels and the magnitude of the vital rate that is affected, as well as on the effect of the selected parameter values on the starting stable age structure.

However, despite this complexity, overall the sensitivity to parameter misspecification is smaller than the variability among simulations that results from the variability in expert's opinions on the degree to which vital rates are affected, as well as the effect of environmental stochasticity.

5.2 Adjusting parameter values simultaneously to guide recommendations for future assessments

In this section we describe the results of additional sensitivity analyses for bottlenose dolphin and harbour seals. Both species are of particular interest in Scotland due to their smaller population size, the fact that they are features of SACs and the declining status of some populations. The scenarios used in these simulations were driven by the review of MU/SMA status presented in Phase 1 and the sensitivities identified in the previous section.

5.2.1 Harbour seal

5.2.1.1 Stable populations – Moray Firth, Western Isles, Southwest Scotland, the southern part of Western Scotland, East Scotland[1] and Northern Ireland

As was the case when the last review was undertaken (Harwood and King 2014) the Moray Firth is the only population for which substantial demographic information is available. A sensitivity assessment for this population was carried out using the combinations of parameter values implied by the analyses of Graham et al. (2017) and Matthiopoulos et al. (2014). The relatively high adult survival (0.97) and fertility (0.9) estimates reported in Graham et al. (2017) require relatively low pup and juvenile survival rates (e.g. 0.13 for pup survival and 0.8 for juvenile survival) to replicate the observed population trajectories. The posterior estimates for fertility (0.87), adult survival (0.96) and juvenile survival (0.79) from Matthiopoulos et al. (2014) require a pup survival of 0.1866 to achieve a stable trajectory. Given the sensitivity to variation in pup survival observed in the previous section and the wide range of pup survival rates reported in the literature, we also carried out simulations with a higher pup survival rate, and therefore a lower adult survival (see Table 13). Figure 16 shows the effects of these different combinations of parameter values on the median counterfactual of population size at different effect levels.

There was a concern that the required values for pup survival to achieve the observed growth rate in combination with the other modelled parameters were very low relative to the range of reported values described in Section 4.5.3.1. Given the increased sensitivity in model outcome to low pup survival values we chose to examine an increased pup survival rate and decreased adult survival rate to ensure that any modelling of impact would be conservative given uncertainty in the demographic parameters underlying all the reported stable populations. However, it is important to note that Figure 17 shows that the predictions from the different combinations of parameters only begin to diverge at levels of impact that would be considered unrealistic given the typical magnitude of disturbance impacts reported in EIAs. This is consistent with the results presented in Section 5.1.5. Therefore, the sensitivity to misspecification in demographic parameters is not of concern.

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[1] For the East Scotland harbour seal SMA, there is limited data to inform simulations of impact as the SMA is not regularly monitored in its entirety. We therefore recommend that this population is assumed to be stable.

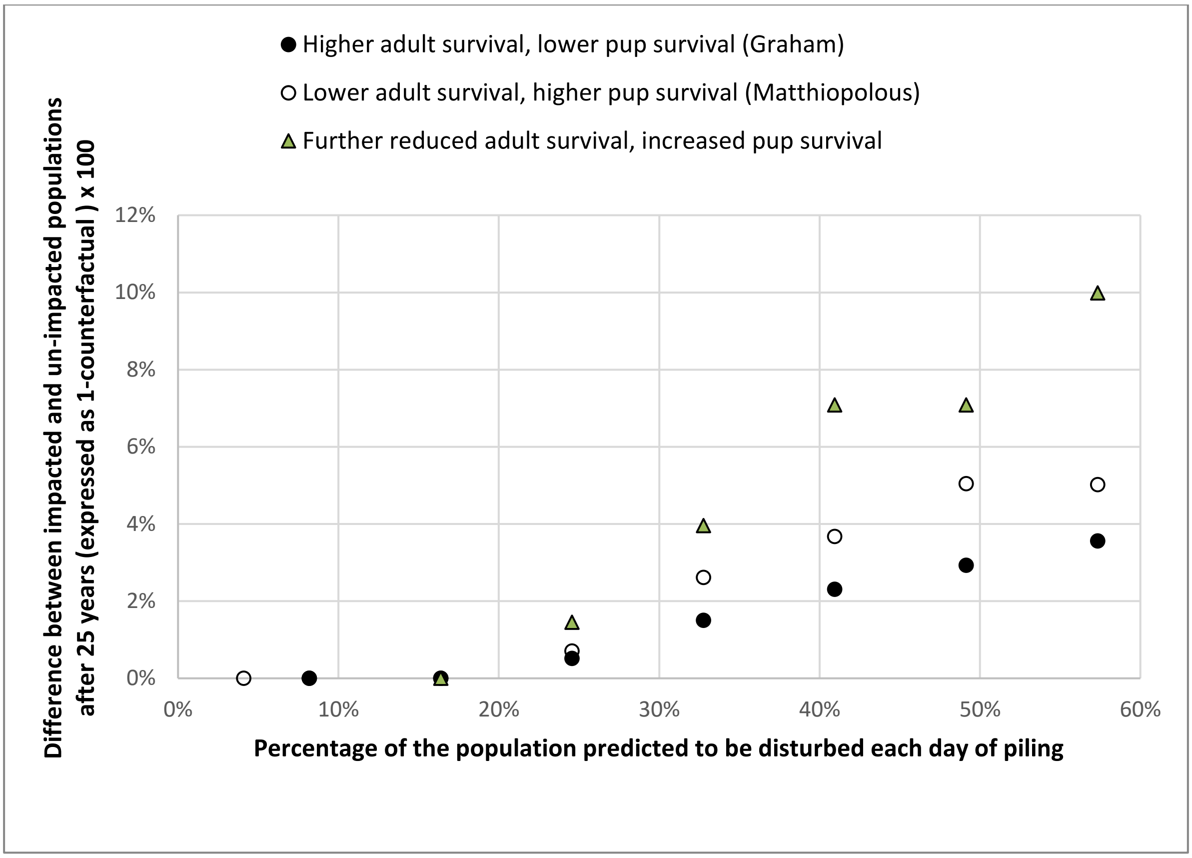

Figure 17: The effect of increasing levels of predicted impact on the predicted effect size (expressed as (one - the median counterfactual of populations size) x100) with different combinations of demographic parameter values.

Table 13

Different combinations of harbour seal parameters explored for modelling stable populations

Scenario	Age calf becomes independent	Age of first birth	Pup Surv	Juv Surv	Adult Surv	Fertility	Growth Rate
	age1	age2	Surv[1]	Surv[7]	Surv[13]		
“Mathiopoulos”	1	4	0.1866	0.79	0.96	0.87	1.00001
“Graham”	1	4	0.13	0.8	0.97	0.9	1.00003
Increased pup survival	1	4	0.4	0.78	0.92	0.85	1.0005

5.2.1.2 Declining populations

The reason for the declines that have been observed in several UK harbour seal SMAs is unclear. Previous studies (Lonergan et al. 2007, Hanson et al. 2015) have concluded that declines in the North Coast & Orkney SMA and the Tay and Eden

Estuary SAC cannot have been driven by reduced fertility alone, because even complete reproductive failure could not account for the rate of decline observed in the moult counts (Thompson et al. 2019). Harwood and King (2014), therefore, assumed that the decline in the Orkney and East Coast regions were a result of a decrease in adult and juvenile survival.

Arso Civil et al. (2018c) reported preliminary results from an integrated population model of the Orkney SMA using simulated datasets, including moult and pup counts, to explore the effect of different demographic rates on population trajectories. Adult mortality has a particularly strong influence on these trajectories; an additional adult mortality of 0.15, beginning after the population began declining was sufficient to account for the observed population trajectory. We suggest that the parameter values calculated by Arso Civil et al. (2018c) (Table 14) are used for modelling the Orkney and North Coast SMA.

Table 14

Updated harbour seal demographic parameters for declining populations

Scenario	**Age calf becomes independent**	**Age of first birth**	**Calf Surv**	**Juv Surv**	**Adult Surv**	**Fertility**	**Growth Rate**	**Notes**
	age1	**age2**	**Surv[1]**	**Surv[7]**	**Surv[13]**			
Declining - Orkney	1	**4**	0.24	**0.86**	**0.80**	0.90	0.8956	Arso Civil et al. (2018)

5.2.1.3 Increasing populations

As described in Section 3.5.2, several UK harbour seal SMAs are currently increasing. These include the English SMAs and parts of the West of Scotland SMA. The factors responsible for these trends have not been identified. In The Wash, which makes up a large proportion of the SE England SMA, the pup count is increasing at a faster rate than the moult count. This could be the result of an increase in the fertility rate. However, given the scale of the increase in counts since 2005, changes in other demographic parameters must be contributing to the increase (Thompson et al. 2017a).

We explored plausible combinations of demographic parameter values that might replicate the observed trajectories for each of the increasing SMAs. Many combinations are consistent with the observed trajectories, and we compared the sensitivity of the iPCoD model outcomes to a given level of impact using the three contrasting combinations shown in Table 15. Combination 1 (with relatively high pup

survival) was most sensitive to the simulated impact, although there is a great deal of overlap across the replicate simulations for each Combination (Table 15).
We assumed that the West of Scottish SMAs would show an equivalent sensitivity, so similar combinations of parameters are suggested for these MUs, with pup survival further increased to achieve the observed level of population growth (Table 16).

Table 15

Alternative scenarios to simulate increasing harbour seal populations.

Scenario	**Growth Rate**	**Pup Surv**	**Juv Surv**	**Adult Surv**	**Fertility**	**Notes**	**Median counterfactual of population size (median % difference)**	**IQR of median difference**
Increasing1	3.23%	0.32	0.86	0.95	0.9	Increased pup survival	0.920 (8.0%)	3-8%
Increasing2	3.24%	0.24	0.86	0.97	0.9	Increased adult survival	0.942 (5.8%)	1-11%
Increasing3	3.26%	0.24	0.86	0.964	0.99	Increased fertility	0.946 (5.7%)	1-11%

Table 16

Conservative parameters for simulating impact on increasing harbour seal populations using iPCoD.

Scenario	**Growth Rate**	**Calf Surv**	**Juv Surv**	**Adult Surv**	**Fertility**
Combined SE & NE England	3.23%	0.354	0.86	0.95	0.9
West Scotland – Central	4.02%	0.32	0.86	0.9588	0.9
West Scotland – North	4.86%	0.397	0.86	0.95	0.9

5.2.2 Bottlenose dolphins

Due to the sensitivities identified in Section 5.1.2 and the range of values presented in the literature for the East Scotland bottlenose dolphin MU, a precautionary approach would be to use a first year calf survival towards the highest end of the reported range, e.g. 0.93 (Cheney et al. 2019), and an adult survival towards the lower end, e.g. 0.948 (Arso Civil et al. 2018a). Combining these values with a juvenile survival of 0.948 requires a fertility rate of 0.405 to achieve the 3.67% annual population growth rate implied by the photo-ID data. However, this value is outwith even the upper confidence interval values for any of the available fertility rate estimates, and thus is considered to be highly unlikely.

Given that we have a number of independent estimates of the individual demographic parameters for this population, derived from empirical studies, we recommend adopting a combination of values from the reported literature that match the observed population trajectory (Table 17).

Table 17

Recommended parameters for simulating impact on the Scottish East coast bottlenose dolphin populations using iPCoD

Growth Rate	Calf Surv	Juv Surv	Adult Surv	Fertility
1.0365	0.925	0.962	0.98	0.24

6 Phase 3 summary and recommendations

Table 18 presents a summary of the recommended demographic parameter values for us with iPCoD in order to model the consequences of disturbance associated with the construction of offshore wind farms for UK marine mammal MUs/SMAs.
There were some sensitivities to variation in input parameters with some species simulations indicating more sensitivity than others – overall harbour seal and bottlenose dolphin simulations were the most sensitive to variation in demographic parameters and minke whales were the least. This is partly driven by differences in the ranges of demographic parameters examined between the species, differing levels of effect and population sizes, as well as differences in the way in which experts thought the vital rates of each species would be affected by disturbance. In general, sensitivity increased with effect size (the number of individuals assumed to be exposed to disturbance on each day of piling). The patterns described below in relation to sensitivity to individual demographic parameters were not evident for the lowest level of impact examined in the harbour porpoise scenarios.

Overall, changes in pup/calf survival had the greatest impact on the counterfactual of population size, with an increasing counterfactual (increasing level of impact) as pup/calf survival increased. This is because experts in the elicitation process considered that calf and pup survival would be most sensitive to disturbance across all species. Sensitivity to misspecification in other demographic variables is a consequence of the effect of parameter values on the starting stable age structure. For example, increasing the value for adult survival increases the proportion of adults in the starting population which enables a quicker recovery from impact because there are more reproducing adults in that population.

Although the median counterfactual of population size varied by as much as 5-8% for the same impact scenario depending on the demographic parameters modelled, the variability within each set of simulations (i.e. the variation between replicates of the same simulation) was always greater than the variability between simulations with different demographic parameters. This variability between replicates within a scenario is due to the uncertainty in the degree to which experts thought that individual vital rates would be affected by disturbance and the amount of disturbance needed to significantly affect fertility or survival. In addition, by comparing the three levels of impact assessed for harbour porpoise, and the different harbour seal combinations in Section 5.2.1. (Figure 17), it is clear that the differences in model outcome between scenarios differing in the specified demographic parameters, only become apparent when very large and unrealistic impact sizes are modelled. As a result, our overarching conclusion is that the metric of counterfactual of population size is robust to misspecification in demographic rates.

Having said this, Table 18 provides a summary of all our recommended parameters, based on the review and the sensitivity analyses presented in this report. Ultimately, it is the responsibility of the iPCoD user to justify the selection of demographic parameters and new information may become available from the monitoring and modelling of these populations that may justify a different selection. We recommend that the choices for parameters are discussed and agreed with consultees before assessments are undertaken and we hope that the information presented in this report will guide these discussions. However, in the absence of estimates of demographic parameters, we highlight that, in general, adoption of a higher pup/calf survival rate and a lower adult survival and fertility rate will ensure the most precautionary predictive modelling outcome.

Table 18

Summary of recommended parameters for all UK marine mammal management units covered in this review

Species	MU/SMA	Age calf/pup becomes independent	Age of first birth	Calf/Pup Survival	Juvenile Survival	Adult Survival	Fertility	Growth Rate
		age1	**age2**	**Surv[1]**	**Surv[7]**	**Surv[13]**		
Harbour Porpoise	North Sea	1	5	0.8455	0.85	0.925	0.34	1.000
Grey Seal	All UK	1	6	0.222	0.94	0.94	0.84	1.01
Harbour Seal	Shetland	1	4	0.4	0.78	0.92	0.85	1.000
	Orkney & north coast	1	4	0.24	0.86	0.80	0.90	0.8956
	Moray Firth	1	4	0.4	0.78	0.92	0.85	1.000
	East Coast	1	4	0.4	0.78	0.92	0.85	1.000
	South-west Scotland	1	4	0.4	0.78	0.92	0.85	1.000
	West Scotland - North	1	4	0.397	0.86	0.95	0.9	0.397
	West Scotland - Central	1	4	0.32	0.86	0.9588	0.9	0.32
	West Scotland - South	1	4	0.4	0.78	0.92	0.85	1.000
	Western Isles	1	4	0.4	0.78	0.92	0.85	1.000
	Northern Ireland	1	4	0.4	0.78	0.92	0.85	1.000
	England NE & SE	1	4	0.354	0.86	0.95	0.9	0.354
Minke Whale	European waters	1	9	0.70	0.77	0.96	0.91	1.000
Bottlenose Dolphin	All other MUs	2	9	0.8	0.94	0.94	0.25	1.000
	Coastal East Scotland	3	9	0.925	0.962	0.98	0.24	1.0365

The sensitivity analysis carried out and reported on here was not exhaustive and there are several parameters that remain to be explored and assumptions that remain to be tested. This includes:

- The sensitivity of outputs to variation in the age at independence and the age at first reproduction.
- That the patterns of sensitivity to misspecification observed here hold under a wider range of impact scenarios, particularly more complex cumulative scenarios over long periods of time.
- The patterns of sensitivity of outputs to misspecification of demographic parameters in relation to variation in the amount of PTS predicted to occur.
- The sensitivity of alternative output metrics to variation in all input parameters (mean, instead of median, population counterfactuals, centile metrics, probabilistic outputs).
- The sensitivity of outputs to the incorporation of density dependence.

There are also several potential future developments in the pipeline for iPCoD (e.g. the development of species specific dynamic energy budget models which could eventually replace the expert elicitation in iPCoD, development of movement models to better estimate aggregate exposure). It will be important to understand how the underlying sensitivities of outputs to variation in inputs may change alongside these developments.

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