Population and Impact Modelling of Marine Mammals in Wales: Recommendations for Regulatory Applications

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Foreword

This report documents work completed in 2023. Publication was delayed to address various issues arising that needed further investigation. Although the report has been released for publication in 2026, it retains the original 2023 publication date to reflect the context in which the analyses were conducted. Key data sources and updates to population parameters, such as abundance and bycatch estimates, are continually being revised and updated. Those used in this report were the best available at the time of the analysis, but users should generally apply the most appropriate or up-to-date data when using the models described. Interpretation of the results presented in this report should, therefore, be made in light of any more recent estimates to ensure relevance and accuracy in current applications.

Dr. T.B. Stringell & Prof. P.G.H. Evans, December 2025

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Adolygwyd yr wybodaeth sydd ar gael ar strwythur y boblogaeth (mewn perthynas ag unedau rheoli), toreithrwydd a thueddiadau, paramedrau hanes bywyd yng nghyd-destun cyfraddau atgenhedlu a goroesi, a'r prif bwysau anthropogenig y maent yn eu hwynebu. Pan oedd hynny’n bosibl, casglwyd gwybodaeth o Gymru, yn rhanbarthol neu o fewn yr uned reoli, ond, fel arall, fe'i cymerwyd o astudiaethau mewn mannau eraill yn y byd. Disgrifir y paramedrau allweddol a ddefnyddir wrth fodelu a'r rhai a awgrymir fel paramedrau perthnasol yn rhanbarthol yn adrannau 2 a 3, ac fe'u crynhoir yn Nhabl 1.

Yn hanfodol i weithredu unrhyw fodel yn llwyddiannus yw bod yr wybodaeth sy'n ymwneud â'r rhywogaeth yn adlewyrchu'r darlun gwirioneddol. O ran mamaliaid morol, mae hyn yn arbennig o heriol, ac mae llawer o ansicrwydd yn bodoli oherwydd yr anawsterau cynhenid wrth gasglu'r data perthnasol.

Mater sylfaenol yw a yw'r ffiniau gofodol a gynigir ar gyfer rhywogaeth benodol yn adlewyrchu'r boblogaeth fiolegol yn gywir o ran ei nodweddion demograffig ei hun. Os nad ydynt, a bod symudiad sylweddol o anifeiliaid i mewn ac allan o'r uned reoli fel y'i diffinnir, yna bydd hynny'n effeithio ar yr amcangyfrif o doreithrwydd y rhywogaeth a thueddiadau ar ei chyfer yn ogystal â'r paramedrau poblogaeth eraill a ddefnyddir mewn unrhyw fodel. Gall ganlyniad hyn fod yn ganlyniadau camarweiniol ar gyfer modelu ymatebion demograffig i effeithiau dynol.

Ar gyfer yr adroddiad hwn, yr unedau rheoli a fabwysiadwyd ar gyfer y pum rhywogaeth o forfilod yw'r rhai a gynigiwyd gan Weithgor Rhyngasiantaethol ar Famaliaid Morol y DU. Nid oes penderfyniad wedi'i wneud ynghylch graddfa ofodol briodol ar gyfer uned reoli ar gyfer morloi llwyd, felly mae pedwar dewis arall wedi'u treialu. Ystyriwyd tri opsiwn ar wahanol raddfeydd gofodol ar gyfer llamhidyddion pan fo ansicrwydd ynghylch yr uned reoli fwyaf priodol ar gyfer anifeiliaid ym Môr Iwerddon. O'r pum model poblogaeth a adolygwyd, canolbwyntiwyd ar PBR, PVA ac iPCoD. Nid ystyriwyd bod CLA yn briodol yng nghyd-destun asesu effeithiau lefel poblogaeth o ganlyniad i dynnu anifeiliaid o boblogaethau mamaliaid morol oherwydd gweithgareddau dynol. Fe'i sefydlwyd gan y Comisiwn Morfila Rhyngwladol i ganiatáu i'r nifer fwyaf o unigolion gael eu dal yn uniongyrchol heb achosi effaith ar y boblogaeth. Model mwy priodol, wedi'i addasu o CLA, yw RLA. Fodd bynnag, nid oeddem yn gallu gweithredu RLA oherwydd problem o ran codio na ellid ei thrwsio o fewn amserlen y prosiect hwn.[1]

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[1] Mae hyn wedi'i drwsio ers hynny, ac wedi'i gynnwys mewn archwiliad ar wahân o RLA gan OSPAR OMMEG fel dangosydd dalfeydd defnyddiol (Genu ac eraill, 2021).

PBR yw'r model mwyaf syml / y model sy’n gofyn am y lleiaf o ddata o'r modelau a adolygwyd a gellid ei archwilio ar gyfer y chwe rhywogaeth, gan mai dim ond amcangyfrif ceidwadol o faint y boblogaeth, y gyfradd atgenhedlu bosibl uchaf, a ffactor adfer sydd eu hangen, gyda'r ddau olaf yn deillio o astudiaethau o effeithiau sy'n ddibynnol ar ddwysedd ar draws ystod o rywogaethau mamaliaid morol (Wade, 1998). Mae PBR yn gwneud sawl rhagdybiaeth, er y gellir archwilio'r rhain ymhellach gan ddefnyddio dulliau sy'n seiliedig ar reolau.

Ar gyfer tair rhywogaeth yn unig (llamhidyddion, dolffiniaid trwyn potel, a morloi llwyd) yr oedd digon o wybodaeth am baramedrau poblogaeth o fewn y rhanbarth ar gyfer mewnbwn ystyrlon i PVA ac iPCoD. Mae gan y dolffin cyffredin a'r morfil pigfain boblogaethau sylweddol y tu allan i'r unedau rheoli a ddynodwyd gan y Gweithgor Rhyngasiantaethol ar Famaliaid Morol, gyda symudiadau ar raddfa fawr posibl i mewn ac allan, a phwysau sy'n gweithredu'n bennaf o'r tu allan i'r uned reoli ddynodedig. Fodd bynnag, er mwyn cyflawnrwydd, rydym wedi rhedeg y rheini hefyd gan ddefnyddio rhai gwerthoedd a argymhellir. Mae diffyg paramedrau poblogaeth allweddol ar gyfer dolffiniaid Risso y gellir rhedeg modelau PVA ac iPCoD arnynt, tra ar gyfer morfilod pigfain dim ond brasamcanu y gellir eu gwneud (o boblogaethau a gynaeafwyd) a heb unrhyw wybodaeth am gyfraddau atgenhedlu na goroesi o fewn y rhanbarth.

Oherwydd ansicrwydd data, bydd modelau megis modelau RLA a modelau gofod cyflwr Bayesaidd o gymhwysedd cyfyngedig ar gyfer poblogaethau mamaliaid morol Cymru nes bod gwybodaeth fanylach ar gael, ac yn benodol tueddiadau poblogaeth ar draws yr uned reoli fwyaf priodol. Mae hyn yn berthnasol i bob rhywogaeth, gan gynnwys dolffiniaid trwyn potel lle nad oes llawer o wybodaeth y tu hwnt i Ardal Cadwraeth Arbennig Bae Ceredigion.

Er bod y PVAau a ddefnyddir yma yn benderfyniaethol, maent yn cynnig offeryn defnyddiol ar gyfer archwilio effaith defnyddio cyfraddau demograffig gwahanol ar dwf poblogaeth, sy'n werthfawr wrth fenthyca gwybodaeth gan boblogaethau eraill. Os yw cyfradd twf y boblogaeth yn agos at un, yna dylid cymhwyso dull rhagofalus. Mae rhedeg nifer o senarios cyfradd ddemograffig yn darparu modd o asesu pa mor sensitif yw'r boblogaeth i newid y paramedrau dan sylw ac ystod y cyfraddau twf poblogaeth a ddisgwylir. Mae PVAau yn cynnig mwy o hyblygrwydd gan eu bod yn caniatáu archwilio effaith stocastigrwydd demograffig a chynnwys dibyniaeth ar ddwysedd. Dim ond i ddolffiniaid trwyn potel oedd stocastigrwydd yn ystyriaeth bwysig mewn gwirionedd oherwydd maint bach eu poblogaeth. Ar ben hynny, mae PVAau yn caniatáu asesu pa mor sensitif yw'r boblogaeth i newid yn y gwahanol gyfraddau demograffig, sy'n ddefnyddiol at ddibenion rheoli gan ei fod yn datgelu ble byddai ymdrech yn fwyaf effeithiol. Yn ein rhediadau o fodelau gwahanol, unrhyw newid yng ngoroesiad anifeiliaid ifanc ac anifeiliaid llawndwf ar gyfer pob rhywogaeth astudio, yn ogystal ag unrhyw newid yng nghyfradd atgenhedlu dolffiniaid trwyn potel, oedd y paramedrau mwyaf dylanwadol. Felly, rheoli tynnu anifeiliaid ifanc a llawndwf yn uniongyrchol, e.e. sgil-ddalfa, fyddai â'r effaith fwyaf ar dwf y boblogaeth. Yn olaf, mae PVAau yn caniatáu asesu a chymharu effaith gwahanol bwysau yn y byd go iawn ar dwf poblogaeth.

Mae iPCoD wedi'i ddatblygu i helpu i werthuso effeithiau nad ydynt mor sylweddol â marwolaeth megis aflonyddwch o sŵn tanddwr, er ei fod yn caniatáu i ddefnyddwyr redeg modelau sy'n archwilio effaith tynnu bodau biolegol yn uniongyrchol yn unig. Fodd bynnag, nid yw'n bosibl nodi a yw’r bodau ifanc sy’n cael eu tynnu yn rhai ifanc neu lawndwf, neu ryw gyfuniad o’r ddau opsiwn. Nid oedd yn glir chwaith a allai iPCoD gael gwared ar nifer ar hap o unigolion bob blwyddyn (h.y. pan fo’n debygol y bydd y sgil-ddalfa yn ystod o unigolion yn hytrach na gwerth penodol). Serch hynny, mae iPCoD yn caniatáu'r un hyblygrwydd i'r defnyddiwr â PVA i fewnbynnu gwahanol baramedrau demograffig a phoblogaeth. Mae bonws pendant yn cynnwys cynhyrchu terfynau hyder o 95% o o ran y duedd boblogaeth a ragwelir. Ond mae'n dal i ymddangos yn gam cymhleth ar gyfer archwilio effeithiau cymharol fach tynnu anifeiliaid o amgylch datblygiadau morol yn unig ac nid yw wedi'i gynllunio'n bwrpasol ar gyfer archwilio effeithiau sgil-ddalfeydd fel y cyfryw. Yng nghyd-destun Môr Iwerddon, ar gyfer poblogaethau mawr (h.y. heb gynnwys dolffiniaid trwyn potel), ni ddangosodd tynnu <5 neu <10 unigolyn y flwyddyn effaith sylweddol ar boblogaethau llamhidyddion na morloi llwyd, yn y drefn honno. Ond datgelodd effeithiau mwy arwyddocaol ar dwf poblogaeth wrth gynnwys sgil-ddalfeydd ar gyfer y rhywogaethau hyn.

Roedd canlyniadau o PVA ac iPCoD yn eithaf tebyg, yn enwedig ar gyfer dolffiniaid trwyn potel. Ar gyfer morloi llwyd, roedd llai o gytundeb, gyda chyfraddau twf cyson is o'r model PVA, o bosibl oherwydd ei fod yn ymgorffori dibyniaeth ar ddwysedd nad yw iPCoD yn ei wneud.

Mae gan bob model ei fanteision a'i anfanteision penodol ei hun y mae angen eu gwerthuso'n ofalus, yn enwedig pan fo'r rhan fwyaf o baramedrau demograffig a phoblogaeth yn ansicr. Yn y cyd-destun hwn, byddem yn argymell cymhwyso PVA oherwydd ei symlrwydd a'i hyblygrwydd cymharol. Credwn fod PVA yn fwy addas at y diben hwn o'i gymharu ag iPCoD, ac yn y rhan fwyaf o achosion roedd cytundeb da rhwng y cyfraddau twf a gynhyrchwyd gan y naill ddull neu'r llall, sy'n cefnogi'r defnydd o'r dull PVA symlach / hawdd ei ddefnyddio ac addasadwy. Er y gellir ychwanegu mwy o gymhlethdod at PVAau, nid oeddem yn teimlo bod hyn yn bwysig nac yn briodol gan y byddai rhywun yn ychwanegu cymhlethdod at ansicrwydd, ac felly fe ddewisom gadw'r modelau'n gymharol syml. Mae PVA yn ymddangos yn addas iawn at y diben penodol hwn, gan gynnwys nodweddion neu bosibiliadau ychwanegol defnyddiol a llawn gwybodaeth, gan gynnwys dadansoddi swyddogaethau trosglwyddo, archwilio stocastigrwydd demograffig, a phrofi gwahanol senarios. Gellir defnyddio PVA ar y cyd â PBR hefyd.

Executive Summary

Five modelling approaches were reviewed to better understand likely population responses to anthropogenic pressures, such as fisheries bycatch and release of contaminants into the marine environment, for six marine mammal species regularly occurring in Welsh waters: harbour porpoise, bottlenose dolphin, common dolphin, Risso's dolphin, minke whale, and grey seal. The models considered were Potential Biological Removal (PBR), Population Viability Analysis (PVA), Catch Limits Algorithm (CLA), Removals Limit Algorithm (RLA), and interim Population Consequences of Disturbance (iPCoD).

Available information was reviewed on population structure (in relation to Management Units), abundance and trends, life history parameters in the context of reproductive and survival rates, and the main anthropogenic pressures they face. Where possible, information was collated from Wales, regionally or within the Management Unit but otherwise it was taken from studies elsewhere in the world. Key parameters used in modelling and those suggested as regionally relevant parameters are described in sections 2 and 3 and summarised in Table 1.

Crucial to the successful implementation of any model is that the information relating to the species should reflect the true picture. For marine mammals, this is particularly challenging with many uncertainties existing due to the intrinsic difficulties in collecting the relevant data.

A fundamental issue is whether the spatial boundaries proposed for a particular species correctly reflect the biological population in terms of its own demographic characteristics. If they do not, and there is significant movement of animals in and out of the Management Unit as defined, then that will affect the estimate of abundance and its trend as well as the other population parameters used in any model. The consequence can be misleading results for modelling demographic responses to human impacts.

For this report, the Management Units adopted for the five cetacean species are those proposed by the UK Inter-Agency Marine Mammal Working Group. No decision has been taken on an appropriate spatial scale for a grey seal Management Unit, so four alternatives have been trialled. Three options at different spatial scales were considered for harbour porpoise where there is uncertainty over the most appropriate Management Unit for animals within the Irish Sea. Of the five population models reviewed, focus was placed upon PBR, PVA and iPCoD. CLA was not considered appropriate in the context of assessing population level effects of removals from marine mammal populations due to human activities. It was established by the International Whaling Commission to allow direct takes to be maximised without causing a population effect. A more appropriate model adapted from CLA, is RLA. However, we were unable to implement RLA due to a coding issue that could not be fixed within the time frame of this project[2].

PBR is the most simplistic/least data demanding of the models reviewed and could be examined for all six species, since it only requires a conservative estimate of population size, the maximum potential reproductive rate, and a recovery factor, the last two derived

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[2] This has since been fixed, and included in a separate examination of RLA by OSPAR OMMEG as a useful bycatch indicator (Genu et al. 2021).

from studies of density dependent effects across a range of marine mammal species (Wade 1998). PBR makes several assumptions, although these may be explored further using rule based methods.

For only three species (harbour porpoise, bottlenose dolphin, and grey seal) was there sufficient information on population parameters from within the region for meaningful input to PVA and iPCoD. Both common dolphin and minke whale have sizeable populations outside the Management Units designated by the Interagency Marine Mammal Working Group, with potential large-scale movements in and out, and pressures operating largely from outside the designated Management Unit. However, for completeness, we have run those as well using some recommended values. Key population parameters for Risso’s dolphin upon which to run PVA and iPCoD models are lacking, whilst for minke whale they can only be approximated (from harvested populations) and with no information on reproductive or survival rates from within the region.

Because of data uncertainties, models such as RLA and Bayesian state space models will be of limited applicability for Welsh marine mammal populations until more detailed information is available, and in particular population trends across the most appropriate Management Unit. This applies to all species, including bottlenose dolphin where there is little information beyond Cardigan Bay Special Area of Conservation.

Although PVAs used here are deterministic, they offer a useful tool for exploring the impact of using different demographic rates on population growth, which is valuable when borrowing information from other populations. If the population growth rate is close to one, then a precautionary approach should be applied. Running multiple demographic rate scenarios provide a means to assess how sensitive the population is to changing the parameters in question and the range of expected population growth rates. PVAs offer more flexibility in that they allow the exploration of the impact of demographic stochasticity and inclusion of density dependence. Stochasticity was really only an important consideration for bottlenose dolphins due to their small population size. Furthermore, PVAs allow the assessment of how sensitive the population is to change in the different demographic rates, which is useful for management purposes as it reveals where effort would be most effective. In our model runs, any change in juvenile and adult survival for all study species as well as any change in reproductive rate of bottlenose dolphins were the most influential parameters. Therefore, managing direct removals, e.g. bycatch, of juveniles and adults would have the largest impact on population growth. Finally, PVAs allow assessment and comparison of the impact of different real-world pressures on population growth.

iPCoD has been developed to help evaluate sublethal effects such as disturbance from underwater noise, although it does allow users to run models only exploring the impact of direct removals. However, it is not possible to specify whether these removals should be juvenile, adult or some combination. It was also not clear whether iPCoD could remove a random number of individuals each year (i.e. where bycatch is likely to be a range of individuals rather than a set value). Nevertheless, iPCoD allows the user the same flexibility as PVA to input different demographic and population parameters. A definite bonus includes the production of 95% confidence limits around the predicted population trend. But it still appears to be a complex step for purely exploring relatively small impacts of removals around marine developments and is not purpose-built for exploring impacts of bycatch as such. In the context of the Irish Sea, for large populations (i.e. excluding bottlenose dolphin), removing <5 or <10 individuals per year did not show a significant impact on the populations of harbour porpoise or grey seal, respectively. But it revealed more significant impacts on population growth when including bycatch for these species.

Results from PVA and iPCoD were rather similar, particularly for bottlenose dolphin. For grey seal, there was less agreement with consistently lower growth rates from the PVA model, possibly because it incorporates density dependence which iPCoD does not.

All models come with their own specific pros and cons which need to be carefully evaluated particularly when most demographic and population parameters are uncertain. In this context, we would recommend application of PVA due to its relative simplicity and flexibility. We believe PVA is more suitable for this purpose compared to iPCoD, and in most cases there was good agreement between growth rates generated by either method, which supports the use of the simpler/user friendly and adaptable PVA approach. Although more complexity can be added to PVAs, we did not feel this was important or appropriate as one would be adding complexity to uncertainty, and therefore we chose to keep the models relatively simple. PVA seems well suited for this particular purpose, including useful and informative additional features or possibilities including transfer function analysis, exploring demographic stochasticity, and testing various scenarios. PVA may also be used in conjunction with PBR.

1. Introduction

Thirty species of marine mammal are found in UK waters and all are highly protected. Yet several species are subject to a variety of impacts from human activity. One major pressure on populations is anthropogenic mortality or removals, of which bycatch from fisheries is a major concern. There is also a potential for mortality to occur from interactions with Marine industry, for example, tidal energy devices and vessel collisions. There is therefore a need to better understand the potential consequences of such removals on the population making it a critical aspect of environmental assessments to determine the degree of impact from marine industrial development.

The purpose of this report is to review, analyse and compare several different existing models available for assessing population level effects of removals from marine mammal populations. We explore the use of three of these: PBR (Potential Biological Removals), PVA (Population Viability Analysis) and iPCoD (interim Population Consequences of Disturbance), briefly reviewing two others - RLA (Removals Limit Algorithm), CLA (Catch Limits Algorithm). Six species that occur regularly in Welsh waters (Baines & Evans, 2012) are considered: harbour porpoise (*Phocoena phocoena*), bottlenose dolphin (*Tursiops truncatus*), common dolphin (*Delphinus delphis*), Risso's dolphin (*Grampus grypus*), minke whale (*Balaenoptera acutorostrata*), and grey seal (*Halichoerus grypus*).

The extent of our knowledge on population parameters used in the models varies among species, so some models may be appropriate for one species but not another. Each of the models is evaluated, and the appropriate ones run under different parameterisation scenarios or parameterisations, with a comparison of model outputs and analysis of which parameters are most influential. Descriptions are provided for how uncertainty in input parameters is dealt with by the modelling frameworks, and assessments made on the levels of confidence that can be placed in model outputs given the data available for the relevant populations. Recommendations for the most appropriate model to be used under different regulatory scenarios are provided.

1.1. Legislation

For marine developments, a Habitats Regulations Assessment (HRA) is typically required. The Appropriate Assessment (AA) stage of the HRA establishes whether or not the development could cause Adverse Effect on Site Integrity (AEOSI) to sites of European importance (Special Areas of Conservation). If AEOSI cannot be ruled out, the project cannot be consented without a derogation under Article 6(4). Mortality impacts from marine developments could include collision with construction vessels, kills from unexploded ordnance, etc, although currently a key focus in Welsh waters is from potential collision with tidal energy devices, particularly the rotating blades of the turbines and their tethers. Mortality associated with fisheries such as bycatch is not formally assessed under HRA (article 6(3)) but should be considered in population modelling to inform the decision making process.

Two other assessment procedures are relevant: the Strategic Environmental Assessment (SEA) Directive, European Directive 2001/42/EC "on the assessment of effects of certain plans and programmes on the environment" which has been transposed into UK law through the SEA Regulations; and the Environmental Impact Assessment (EIA) Directive, Council Directive 85/337/EEC (as amended and consolidated) “on the assessment of the effects of certain public and private projects on the environment”; transposed into UK law through the EIA Regulations.

All cetacean species have also been designated within Annex IV as European Protected Species (EPS) under the EU Habitats Directive, whilst bottlenose dolphin, harbour porpoise, grey seal and harbour seal are listed under Annex II requiring Special Areas of Conservation as part of the Natura 2000 network. EPS are protected under Article 12 (and transposing regulations) from deliberate killing (or injury), capture and disturbance throughout its range. Within the UK, these regulations became enshrined in law in England and Wales under the Conservation of Habitats and Species Regulations 2017 (as amended). From 1st January 2021, with the UK no longer part of the European Union, some amendments come into force (Conservation of Habitats and Species Regulations 2017 (regulation 9(1), as amended by the 2019 Regulations). These currently apply to inshore waters up to 12 nautical miles and explain how the amendments to the legislation work; they do not cover offshore waters beyond 12nm (which are covered by the Offshore Regulations) but similar processes are expected to be applied. Most of the changes described above involved transferring functions from the European Commission to the appropriate authorities in England and Wales. All other processes or terms in the 2017 Regulations have remained unchanged and existing guidance is still relevant. The obligations of a competent authority in the 2017 Regulations for the protection of sites or species do not change.

The assessment of whether a marine development (plan or project) could cause an adverse effect to Special Areas of Conservation requires reference to the site’s conservation objectives. A common conservation objective theme for all marine mammal Annex II species is ‘population viability’ and for potential anthropogenic removals (mortality), this is the principal objective on which to base an assessment of whether an effect is adverse or not. Unacceptable levels of removals can be defined as those having an impact on the wider populations of the species in their natural range, with the reference population being levied at the Management Unit (MU) scale.

The conservation objectives for harbour porpoise SACs are that the species is a viable component of the site; the intent of this objective is to minimise the risk of injury and killing or other factors that could restrict the survivability and reproductive potential of harbour porpoise using the site. Specifically, this objective is primarily concerned with operations that would result in unacceptable levels of those impacts on harbour porpoises using the site. Unacceptable levels can be defined as those having an impact on the Favourable Conservation Status (FCS) of the populations of the species in their natural range. The reference population for assessments against this objective is the MU population in which the SAC is situated (IAMMWG 2015). Site based measures should therefore be aligned with the existing strict protection measures in place throughout UK waters.

The conservation objectives for bottlenose dolphin and grey seal SACs are that the population is maintaining itself on a long-term basis as a viable component of its natural habitat, and, for grey seal, that populations should not be reduced as a consequence of human activity whilst for bottlenose dolphin, populations should be increasing as stated in the Restoration and Recovery conservation objective.

Even where there is focus upon Special Areas of Conservation, the wider population and the pressures upon it need to be considered. Site based measures should be aligned with the existing strict EPS protection measures in place throughout UK waters (and beyond). Thus, although several relevant developments in Welsh waters concern offshore renewables, if a population affected by those developments is exposed to, for example, significant levels of bycatch, most likely outside of Welsh waters, that should be accounted for in models used to inform on acceptable levels of removal.

2. Population Parameters

Critical to the successful application of any population model is selection of the most appropriate values for population parameters based upon best available evidence, including alternatives to cover the range of uncertainty that exists. The parameters required in the models include some of the following: abundance estimate at the appropriate spatial scale, trend in population size, reproductive rates, and survival rates.

Each species has been considered in turn, together with a review of the spatial scale at which to run the models, and the evidence for the different parameters required by the various models.

Pressures facing each species in the region have also been described, and in this context Figure 1 depicts the Divisions used by ICES for aggregating information on bycatch prior to determining bycatch rates.

All the models depend upon an appropriately defined population management unit. If the population boundaries assigned do not align with the true biological population, then this may give misleading results for modelling demographic responses to human impacts. This is a fundamental issue, whose influence is often underrated. For marine mammals where there are generally no obvious physical barriers, population boundaries are frequently unclear, and indeed may be gradual with isolation being a function of geographic distance, modified in some cases by habitat or oceanographic variables (Evans & Teilmann 2009). This introduces a measure of uncertainty which is compounded by the practical difficulties in sampling animals for indicators of population structure.

In the past, population structure has been largely determined from molecular genetic evidence. However, genetic differences observed may reflect evolutionary aspects of population separation involving tens, hundreds or thousands of generations rather than contemporary population structure (Taylor & Dizon 1999, Fontaine et al. 2010). The use of a suite of approaches has increasingly been advocated, incorporating genetic, morphometric and ecological information (e.g. variation in cranial and other body measurements, life history parameters, evidence of movements from telemetry and photo-ID). From that, the concept of the Management Unit has developed (see Moritz 1994, Taylor & Dizon 1999, Palsbøll et al. 2007, Evans & Teilmann 2009, Sveegaard et al. 2015). This focuses more upon defining populations that are demographically independent of one another, where population dynamics depend largely upon local birth and death rates rather than immigration. In this way, the emphasis is upon the contemporary dispersal rate of individuals rather than the historical amount of gene flow.

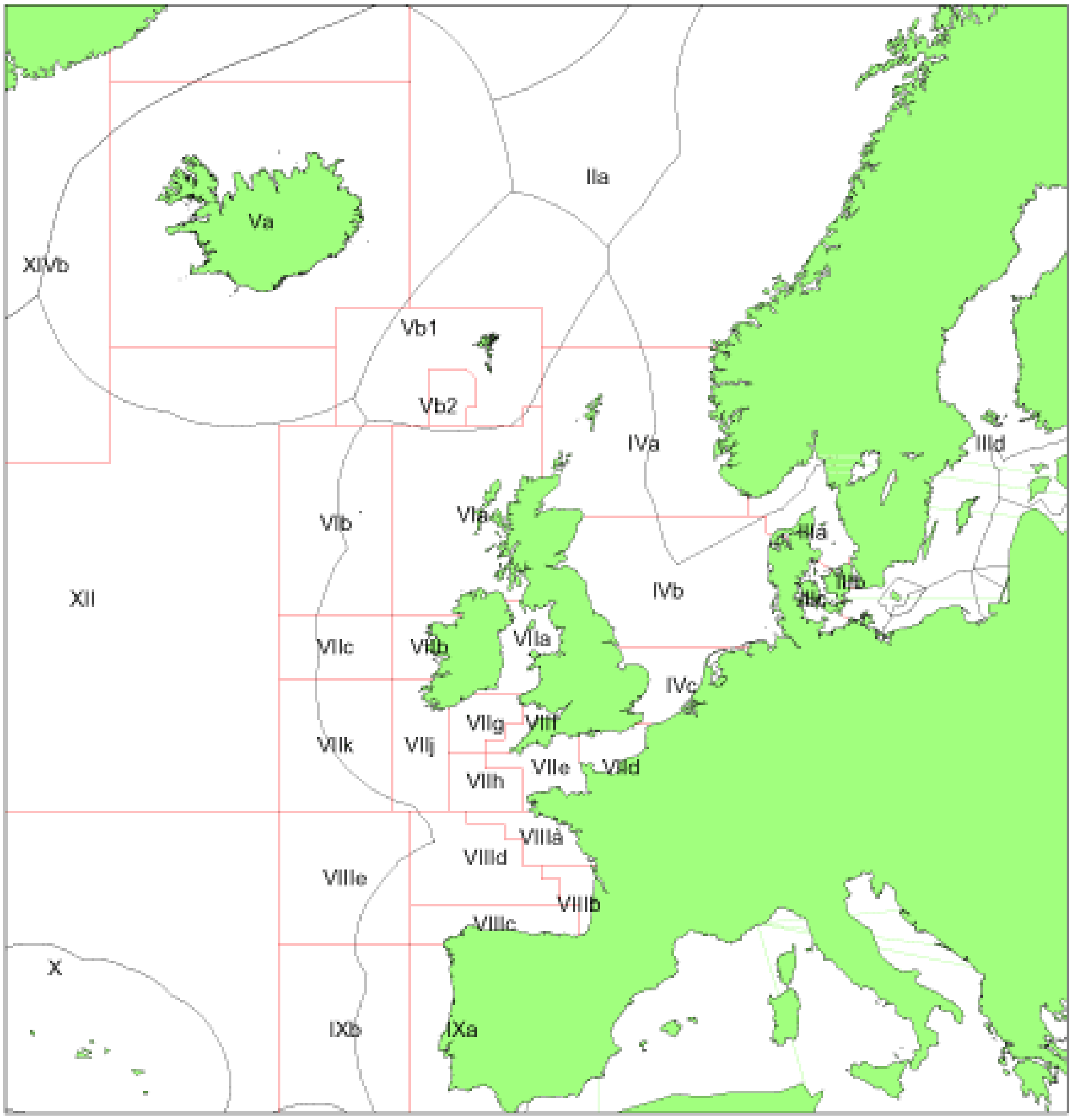

Figure 1: ICES Subareas and Divisions in the European North Atlantic (from IAMMWG 2020).

2.1. Harbour Porpoise

2.1.1. Management Unit

In the case of harbour porpoises in the waters west of the UK, there remains uncertainty where population divisions are best made. There are a few options to consider.

The ASCOBANS Population Structure workshop recommended the Celtic & Irish Seas (excluding West Scotland & NW Ireland) as a separate Management Unit (Evans & Teilmann 2009). Broadly the same boundaries have been followed by ICES WGMME (2014) and IAMMWG (2015, 2021). On the other hand, Fontaine et al. (2017), largely on genetic grounds, proposed a wider area that extends south into the Bay of Biscay, whilst recognising that the Irish Sea may represent a genetic transition zone between porpoises located in the Celtic Sea, Bay of Biscay and Western Channel (some of which have affinities with animals from the Iberian Peninsula) and the ''pure'' porpoises of northern waters (including West Scotland), and assessment units along these lines were proposed at the IMR/NAMMCO workshop, with the Irish Sea tentatively included with the Celtic Sea, Bay of Biscay and Western Channel (NAMMCO & IMR 2019). The other difference with the earlier assessments was that all of western and SW Ireland were included with West Scotland, the boundary for the Celtic and Irish Sea MU being placed instead in southern Ireland (at the western border of ICES divisions 7.g,h – see Figure 1).

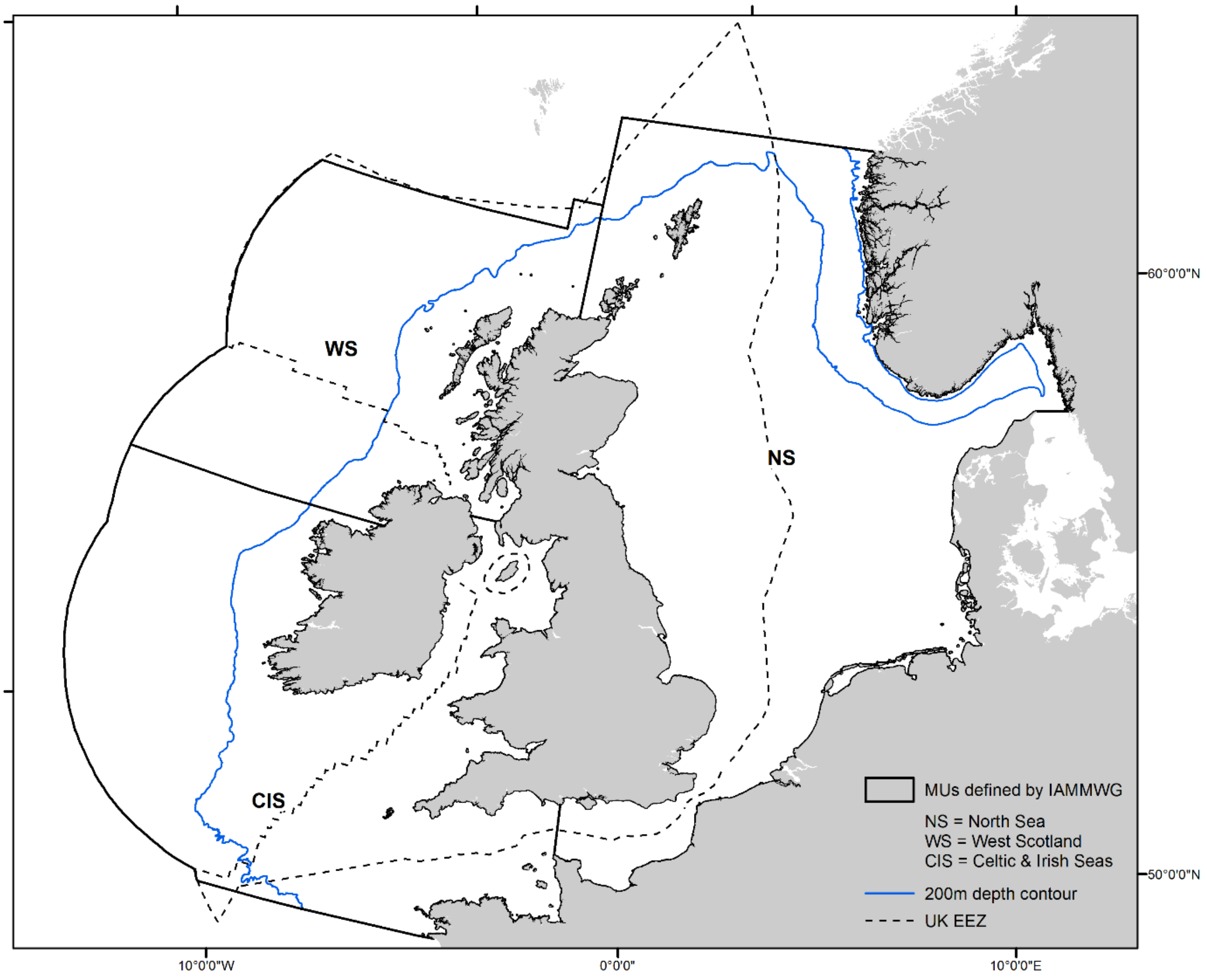

Figure 2: Harbour Porpoise Management Units (MUs), noting that this species is largely confined to the continental shelf (i.e. waters < 200m depth). The UK portion of the MUs is delimited by the UK EEZ (from IAMMWG 2021).

Genetic sampling effort remains patchy (e.g. very few samples are available from the Republic of Ireland or from the Bay of Biscay) and is reliant upon strandings, whose origins are usually unknown and potentially influenced by prevailing currents. Most evidence for the Irish Sea Assessment Unit (NAMMCO & IMR 2019) is based upon microsatellite genotyping, along with the fact that animals around the Iberian Peninsula north to SW Britain are generally of larger body size than those further north and within the North Sea (Fontaine et al. 2017). The question remains open where boundaries are best drawn to identify demographically distinct populations, and in particular where the Irish Sea should be placed. For the time being, the area defined by IAMMWG (2015, 2021) as the Celtic and Irish Seas (CIS) (Figure 2) has been adopted. Additionally, we model two further areas described by IMR & NAMMCO (2019) (see also ICES WKMOMA 2021, ICES 2021): the Irish Sea AU (ICES Division 7.a) and the Celtic Seas AU (southern Ireland, western English Channel, and shelf seas of the Bay of Biscay: ICES Divisions 7.e,f,g,h, 8.a,b). However, in the draft and forthcoming OSPAR indicator assessments for QSR 2023, the OSPAR Marine Mammal Expert Group (OMMEG) is opting to combine the Irish Sea with the Celtic Seas to form the Irish and Celtic Seas AU. Western Ireland and West Scotland are combined into an AU called the West Scotland & Ireland AU, which we do not model here, although the IAMMWG (2015) CIS MU covers much of these AUs. It should be noted that these delineations are based largely upon microsatellite DNA analysis rather than multiple lines of evidence, and that genetic sampling is based upon strandings from around the British Isles with relatively few from Ireland, and with variable sample sizes and periods of sample collection between areas.

Parameters adopted: We model three units: Celtic and Irish Seas Management Unit (CIS MU) (IAMMWG 2015, 2021), Irish Sea Assessment Unit (IS AU), and Celtic Seas Assessment Unit (CS AU) (IMR & NAMMCO 2019), and for the time being recommend the first (Figure 2).

2.1.2. Abundance and Trend

Wide-scale line-transect surveys (SCANS-II in July 2005 and SCANS-III + ObSERVE surveys in summer 2016) have provided two sets of abundance estimates for harbour porpoise for the Celtic and Irish Seas Management Unit (referred hereafter as CIS MU) (Hammond et al. 2013, 2021, Rogan et al. 2018). IAMMWG (2021) calculated abundance estimates for the CIS MU, which indicate a decline in numbers from 98,807 (CV: 0.30; 95% CI: 57,315-170,336) in 2005 to 62,517 (CV: 0.13; 95% CI: 48,324-80,877) in 2016, amounting to an average annual decline of 4%.

The summer 2016 abundance estimate for the Irish Sea, corresponding to the IS AU (blocks E & F from SCANS-III; Hammond et al. 2021) was 9,376 (CV: 0.33; 95% CI: 4,985-16,364), and 25,261 (CV: 0.52; 95% CI: 12,125-46,113) in the Celtic Sea (including southern & SW Ireland) corresponding to the CS AU (strata 4 & 8 from ObSERVE survey and blocks B & D from SCANS-III; Rogan et al. 2017, Hammond et al. 2021).

Two point estimates alone provide weak evidence for a trend, and therefore it is prudent to examine any other independent evidence available. During the 1990s, concerns were expressed by ASCOBANS that high levels of bycatch (estimated at an average of 2,200 porpoises per year) in the Celtic Sea, amounted to annual mortality of c. 6% of the population (Tregenza et al. 1997, Hammond et al. 2002), which was deemed unsustainable, i.e. exceeding both the 1.7% maximum limit and 1.0% precautionary limit for all anthropogenic removals adopted by Parties to ASCOBANS (ASCOBANS 2000, Anon 2000). This led to conservation action by the European Commission with the introduction of EC Regulation 812/2004 (European Union 2004), which dictated the mandatory deployment of pingers. Since then, in combination with reduced fishing effort, bycatch rates in the region have declined to c. 1.1-2.4% of the population by 2016 (ICES 2018). A comparative analysis of stranded porpoises from the CIS MU (confined to samples from the UK) found marked differences in life history parameters for animals between the periods 1990-99 and 2000-13, with slower growth and maturation (Murphy et al. 2020). This could reflect density dependence if the population was increasing towards carrying capacity after 2000, or, for example, it may simply be the result of food shortage due to extrinsic factors.

Parameters adopted: 62,517 (CV: 0.13; 95% CI: 48,324-80,877) with a 4% p.a. declining trend in CIS MU (IAMMWG 2021); 9,376 (CV: 0.33; 95% CI: 4,985-16,364) in the Irish Sea, corresponding to the IS AU (blocks E & F from SCANS-III; Hammond et al. 2021); and 25,261 (CV: 0.52; 95% CI: 12,125-46,113) in the Celtic Sea (including southern & SW Ireland) corresponding to the CS AU (strata 4 & 8 from ObSERVE survey and blocks B & D from SCANS-III; Rogan et al. 2017, Hammond et al. 2021).

2.1.3. Reproductive Rates

Harbour porpoise reproductive rates have been calculated in several regions of the North Atlantic. They are typically much higher in the North-west Atlantic compared to the North-east (Murphy et al. 2015, 2020). Age at sexual maturity is later in European populations (between 4 and 5 years) than in North American & Icelandic populations (between 3 and 4 years) (Murphy et al. 2015, NAMMCO/IMR 2019). Differences in age at sexual maturity may exist between samples according to cause of death (e.g. infectious disease vs physical trauma) (Murphy et al. 2015). On the other hand, Murphy et al. (2020) found that there was no significant difference in female age at sexual maturity for animals believed to be healthy and which died of physical trauma (e.g. bottlenose dolphin attack) (average 4.67 years) compared with those dying of infectious disease (average 4.39 years).

Pregnancy rates from stranded or bycaught animals in Iceland and eastern North America range between 0.72 and 0.98, suggesting an annual reproductive cycle, whereas in Europe, they vary between 0.34 (North Sea) and 0.68 (Celtic & Irish Sea) which is suggestive of a biennial cycle (Murphy et al. 2015, Kesselring et al. 2017, NAMMCO / IMR 2019). Murphy et al. (2020) recently analysed stranded porpoises from the CIS MU (UK coasts only) and found that whereas 68% (17 out of 25 mature females) were pregnant in the sample from the period 1990-99, only 54% (19 out of 35 mature females) were pregnant in the sample from the period 2000-2013. The pregnancy rate was calculated using data that was obtained outside the conception period (May to September) and included all causes of death groups. Biases in terms of estimated pregnancy rates may exist for particular causes of death, but they were broadly comparable across the time periods (56% trauma and 24% infectious disease for period 1, and 63% trauma and 29% infectious disease for period 2).

Parameters adopted: We have used the estimate of age at first reproduction of 5 years and an inter-birth interval of 2 years (Murphy et al. 2015), with a reproductive rate of 0.54, based upon the 2000-2013 strandings pregnancy rate, as the preferred value, providing 0.34 and 0.68 as alternatives from the North Sea & Celtic & Irish Sea strandings data respectively (following Murphy et al. 2020).

2.1.4. Survival Rates

Information on age structure and longevity, where it exists, is informative when considering observed trends in abundance. Within the Celtic and Irish Seas, the average age of 127 female porpoise and 129 male porpoise from strandings collected over the period 1990-99 was 2.77 and 3.08 years respectively, and for 269 female and 271 male porpoises from the period 2000-13 was 3.33 and 3.22 years respectively (Murphy et al. 2020). Maximum age increased from 15 to 21 years for females and decreased from 18 to 15 years for males between the two time-periods (Murphy et al. 2020). Although a maximum age of 22 years was reported, approximately 80% of necropsied porpoises were 5 years old or less, and only 5% were aged 12 years or older (Murphy et al. 2020).

Survival rates have not been estimated for the CIS MU. Lockyer (1995) obtained an annual adult survival rate of 0.80 for males and 0.82 for females from stranded animals collected around the UK. Winship et al. (2008) used a value for calf and juvenile survival (age 0-4 years, assuming a constant rate) of 0.85, and adult survival (age 5+ years) of 0.925. Hammond et al. (2019) estimated annual natural survival rates of 0.85 for age 0-1 years, 0.87 for age 1-2 years, and 0.91 for age 2+ years, based on Winship (2009). These applied to North Sea porpoises (where the population is believed to be stable) and assumed the age at independence was at 1 year and the age of first birth was at 5 years (following a gestation period of 10-11 months). Taking account of a pregnancy rate for this population of 0.34 (Murphy et al. 2015), Sinclair et al. (2020) adjusted age-related survival rates to simulate no growth in the population, as follows: 0.8455 (0-1 year), 0.85 (1-4 years) and 0.925 (age 5+ years). Different values may be applicable to the Celtic and Irish Sea MU.

For their iPCoD models, Sinclair et al. (2020) then followed Moore & Read (2008) and Harwood & King (2014) in estimating age-specific survival, age at maturity and maximum birth rate that were compatible with data from bycaught animals and survey data for the region in question (in this case, the North Sea). They used 4 years for the mean age at sexual maturity (i.e. 5 years for age of first birth), and the two values for survival (0.85 and 0.925) as alternatives for adult survival, but chose a lower value of 0.60 for calf survival, based on Moore & Read's analysis. They then tuned fecundity to achieve the population growth rate of 1.0 using either of these values.

Parameters adopted: Since survival rates have not been estimated previously for the CIS MU, we have trialled age-specific estimates encompassing the ranges outlined above, with 0.60 as the preferred value for calf survival and 0.85 as an alternative; 0.85 for juvenile survival; and 0.90 as the preferred value for adult survival with 0.85 and 0.925 as alternatives (Winship et al. 2008, Sinclair et al. 2020), given that some evidence suggests a decline in the CIS population (Hammond et al. 2017, Murphy et al. 2020).

2.1.5. Pressures

ICES WGMME (2019) has developed threat matrices by ecoregion for all marine mammal species regularly occurring in NW Europe. Threat levels were classified as high, medium or low, using the following criteria:

High = evidence or strong likelihood of negative population effects, mediated through effects on individual mortality, health and/or reproduction;

Medium = evidence or strong likelihood of impact at individual level on survival, health or reproduction but effect at population level is not clear;

Low = possible negative impact on individuals but evidence is weak and/or occurrences are infrequent.

These are used here in the pressure assessments for the six species under consideration.

Harbour porpoises from the CIS MU face a range of pressures (ICES WGMME 2019) which also vary spatially within the Management Unit. Bycatch and contaminants are both listed as high threats, as defined above (ICES WGMME 2019).

A bycatch risk assessment was made for harbour porpoise within the region by ICES in 2018 (ICES WGBYC 2018). Most bycatch of porpoises occurs in static gillnets (métiers GNS - set gillnets, GND - drift nets, GTR - trammel nets). Gillnetting effort in West Scotland and North-west Ireland (ICES Divisions 4a, b) is very low, with most effort off South-west England and southern Ireland in ICES Divisions 7.f, g, h (see Figure 1 for division boundaries). Most of the fishing effort in SubArea 7 is undertaken by the UK but also by France and Ireland, with limited effort by Germany and Denmark (Evans et al. 2021). Using data submitted by the member states pooled across 2015 and 2016 (the latest years available at the time), bycatch rates were highest in Division 7.g in set gillnets although in 2016 they were highest in 7.f. In SubArea 7, this was estimated to be between 620 and 1,391 (95% CI) porpoises, with lower and upper 95% bycatch mortality estimates of 1.08-2.42% of the abundance estimate of 57,491 porpoises for SubArea 7 (ICES WGBYC 2018).

ICES WGBYC (2018) noted that historically, harbour porpoises have been recorded as bycatch in Division 7.h, but none was reported in 2015 or 2016 despite having the highest observer coverage (~3%) of all observed netting in divisions within Subarea 7. If we assume that the patterns of fishing have not changed in recent years, then one might expect bycatch still to be occurring in this division. Therefore, the bycatch rate for 7.h calculated from pooled data over 2008–2013 was applied to the current 2015–2016 assessment. This resulted in an upward bycatch estimate for 2016 of between 706–1,514 (95% CI) porpoises.

An assessment of bycatch as a percentage of the best population estimate was also reported for the Celtic and Irish Seas (CIS) Assessment Unit in the OSPAR Intermediate Assessment of 2017; it was concluded that there was 1.06-1.37% annual mortality due to bycatch in this Assessment Unit based upon a bycatch estimate of 1,137-1,472 (OSPAR IA, 2017). However, the OSPAR assessment used best abundance estimates from SCANS-II in 2005 of 107,300 (CV: 0.3) and bycatch data pooled across years 2006–2013; ICES WGBYC (2018) in their bycatch risk assessment has used the new estimates from SCANS-III and ObSERVE in 2016 which report lower densities of harbour porpoise in the CIS MU area compared to previous surveys, resulting in an average annual bycatch estimate of 1.08-2.42% (620-1,391) of the abundance (estimated at 57,491 for Subarea 7).

Note that there is very little static gillnetting within Division 7.a (Irish Sea) and no bycatch from there recorded within the ICES database.

From the above figures, one can see that, by all accounts, bycatch estimates have been above the 1% precautionary environmental limit recommended by ASCOBANS. There are many uncertainties to these estimates, but given that vessels of <12m length are for the most part not included in these data despite making up the majority of vessels in the region and that not all EU member state fleets fishing in ICES Area 7 are monitored or bycatch data recorded, bycatch rates are at best minima which suggests the level of take may be unsustainable within the CIS Management Unit, although bycatch rates do appear to have declined since 2015 (ICES WGBYC 2019).

For the Irish Sea alone, we use an annual bycatch estimate of 12 (range 6-27); and for the Celtic Sea, we use an annual bycatch estimate of 738 (range 284-2,340) (ICES WKMOMA 2021).

Contaminants (particularly persistent organic pollutants such as PCBs) are also considered a high threat to the species in the region (NAMMCO & IMR 2019, ICES WGMME 2019). There is widespread evidence for its effects on reproduction for females breeding for the first time (Murphy et al. 2015) but also for males (Williams et al. 2021), and in compromising immune responses to disease (Jepson *et al*. 2005, 2016). Adults are not the only age group that may be affected, given recent findings on juveniles (Williams et al. 2020a). PCB levels remain high, with UK hotspots in the Irish Sea and around Southwest England (Jepson et al. 2016, Williams et al. 2020b).

Underwater noise for porpoises in the Celtic and Irish Seas was identified by ICES WGMME as a medium threat, i.e. where there is evidence or a strong likelihood of impact at an individual level on survival, health or reproduction but an effect at a population level is not clear (ICES WGMME 2019). Prey depletion was also identified as a medium threat (ICES WGMME 2019), occurring through overfishing or mediated through climate change impacts, either of which might also increase fatal consequences of interaction between bottlenose dolphins and harbour porpoise (see, for example, Boys 2015).

Parameters adopted: For the area of the CIS MU, an annual average removal (mortality) of 1,000 porpoises from bycatch was used in the models; this value was chosen as a little below the mid-point of the range 706-1,514 annual porpoise bycatch on the basis that, since 2015, fishing effort in SubArea 7 has declined (ICES WGBYC 2018, 2019). Given the various sampling uncertainties and difficulties in matching up areas for an assessment of the CIS MU, this approximation was deemed more appropriate than a precise value. For the Irish Sea alone, we use an annual bycatch estimate of 12, and for the Celtic Sea of 738, following ICES WKMOMA (2021). Removals were apportioned in the ratio 60% juveniles and 40% adults to take account of the fact that juveniles amongst small cetaceans are believed to be more susceptible to bycatch (Mannocci et al. 2012). For the impact of contaminants, we have proposed a 10% reduction in reproductive rates as a precautionary estimate, and for marine developments, we adopted NRW's AEOSI recommendation of the removal of 4 harbour porpoises per year (NRW 2020).

2.2. Bottlenose Dolphin

2.2.1. Management Unit

Management Units for coastal bottlenose dolphins have been drawn up largely on the basis of photo-ID studies and re-sightings of known individuals. Comparisons with photo-ID catalogues of individuals in western Ireland, West Scotland, the North Sea, and the

Channel have produced no confirmed matches with those photographed from within the Irish Sea (Pesante et al. 2008a, Feingold & Evans 2012, Lohrengel et al. 2017).

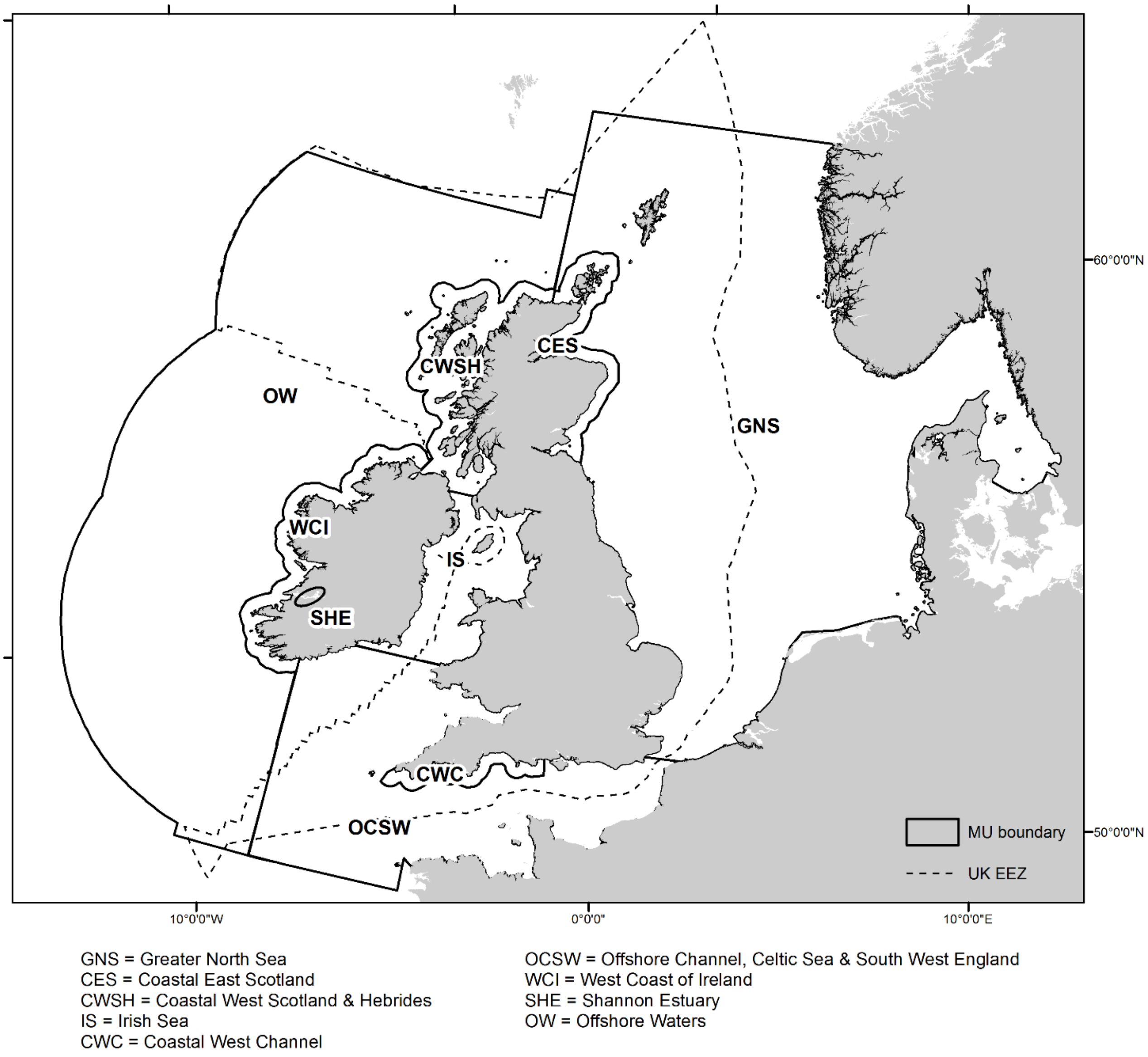

Figure 3: Bottlenose Dolphin Management Units (MU) (from IAMMWG 2021)

Although photo-ID studies have been concentrated within Cardigan Bay, West Wales, there have been surveys also in North Wales and the Isle of Man, which confirm that individuals either seasonally or for more protracted periods occupy coastal regions in other parts of the Irish Sea (Pesante et al. 2008b, Feingold & Evans 2012, 2014, Lohrengel et al. 2017).

Parameter adopted: The Irish Sea (IS) is recommended as the appropriate Management Unit (Evans & Teilmann 2009, Evans 2012, ICES 2014, IAMMWG 2015, 2020; Figure 3).

2.2.2. Abundance and Trend

For coastal populations of bottlenose dolphins, capture-mark-recapture (CMR) estimates from photo-ID, where they exist, are favoured over line-transect surveys because the line-transect design does not adequately sample predominantly inshore distributions (Buckland et al. 2004, Evans & Hammond 2004, Cheney et al. 2018). Combined SCANS-III and ObSERVE survey results in summer 2016 yielded an abundance estimate in IS MU of 293 but with wide confidence limits (CV: 0.54; 95% CI: 108-793) (Hammond et al. 2017, Rogan et al. 2018, IAMMWG 2020). This was based upon just five sightings in all of the Irish Sea with only one of these within Cardigan Bay (see Figure 4b in Hammond et al. 2017).

Between 2005 and 2018, when the wider area of Cardigan Bay was monitored, CMR closed population models yielded estimates of 195 (CV: 0.36; 95% CI: 162-263) in 2005 rising to a peak of 318 (CV: 0.39; 95% CI: 251-440) in 2009, then declining to 141 (CV: 0.57; 95% CI: 117-262) in 2014 (Lohrengel et al. 2017), with the most recent estimate being 186 (CV: 0.38; 95% CI: 152-263) in 2018 (Sea Watch Foundation unpublished data; see also ICES WGMME, 2019). Cardigan Bay clearly does not form a closed population, with movement in and out of the region (Feingold & Evans 2012, 2014, Lohrengel et al. 2017). Some of this can be accounted for within the closed population models but the population for the entire Irish Sea MU is almost certainly larger than the Cardigan Bay estimates.

The population in Cardigan Bay increased between 2001 and 2008 but has since declined again to a similar value to twenty years ago (Lohrengel et al. 2017).

Parameters adopted: 318 (CV: 0.39; 95% CI: 251-440) (Lohrengel et al. 2017) was used as the highest value obtained from the CMR monitoring time series. As an alternative, we also used the 2016 SCANS-III/ObSERVE estimate of 293 (CV: 0.54; 95% CI: 108-793) from IAMMWG (2021). No long-term trend has been identified for this population although within Cardigan Bay there has been a decline over the last 10 years (Lohrengel et al. 2017). Since trends in other parts of the Irish Sea are not available, it is difficult to determine an overall trend for the IS MU.

2.2.3. Reproductive Rates

Sample sizes to estimate mean age at first birth for the Cardigan Bay population are considered too small for a reliable value, particularly given the potential bias introduced with movement into and out of Cardigan Bay. For the population in the Outer Moray Firth, age at first birth is estimated at 8 (range 6-13) years (Robinson et al. 2017), whilst within the Moray Firth SAC, the median is 9 years (Cheney et al. 2019).

The mean inter-birth interval of known adult female bottlenose dolphins within the Cardigan Bay population is 3.4 (range 2-7) years (Pesante et al. 2008, Feingold & Evans 2014). Elsewhere in the UK, the population in the Outer Moray Firth has been estimated to have an average inter-birth interval of 3.8 years (Robinson et al. 2017) whilst across the coastal East Scotland MU, it has been estimated (by different methods) at somewhere between 3.66 and 4.93 years (Quick et al. 2014, Arso Civil et al. 2017, Sinclair et al. 2020).

Fertility rates for bottlenose dolphins in the Outer Moray Firth have been estimated to average 0.16 (range 0.08-0.23) (Robinson et al. 2017) but for the coastal East Scotland MU overall at somewhere between 0.22 and 0.24 (95% CI: 0.218-0.273, combining different analyses to indicate lower and upper 95% limits) (Quick et al. 2014, Arso Civil et al. 2017, Sinclair et al. 2020). The coastal East Scotland population is believed to be increasing at an estimated rate of 3.65% per annum, and Sinclair et al. (2020) opted for a fertility rate of 0.30 for their iPCoD population model. Given that fertility rates for the East Scotland population may be slightly higher than for the one in the Irish Sea since the population is growing whereas there is no evidence for that in the Irish Sea population, we have used 0.22 with alternatives of 0.16 and 0.30 for reproductive rates in the models for this species.

Parameters adopted: We have used 9 years as the age at first birth (Cheney et al. 2019) and an inter-birth interval of 3 years (Feingold & Evans 2014), with a fertility rate of 0.22 and alternatives of 0.16 and 0.30 (Robinson et al. 2017, Arso Civil et al. 2017, Cheney et al. 2019, Sinclair et al. 2020).

2.2.4. Survival Rates

Average age of independence for bottlenose dolphin calves in Cardigan Bay is estimated to be three years (Pesante et al. 2008, Feingold & Evans 2014) which is the same age as has been determined for calves in the Moray Firth (Arso Civil et al. 2018, Sinclair et al. 2020). Calf survival is therefore considered as applying to years 0-3. In the Cardigan Bay population this has been estimated at 0.85 (0-1 year), 0.83 (1-2 years), and 0.93 (2-3 years), although the latter is based on a small sample size (Pesante et al. 2008, Feingold & Evans 2014). In the coastal East Scotland population, calf survival is estimated at 0.865 (0-1 year), 0.981 (1-2 years), and 0.883 (2-3 years) (Arso Civil et al. 2018). For the Moray Firth SAC, Cheney et al. (2019) estimated calf survival at 0.93 (0-1 year) and 0.55 (1-2 years). The authors suggest that the low survival estimate for calves of 1-2 years 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.

Once calves become independent, it is rarely possible to determine their age because they are usually unidentifiable, having not acquired markings whilst they were with their parent. Thus, for the most part, juvenile and adult survival cannot be distinguished. Arso Civil et al. (2018) therefore estimated juvenile and adult survival combined for the East Scotland MU at 0.948 (for both sexes), and 0.968 (for females only). Sinclair et al. (2020) suggested a calf (0-3y) survival rate of 0.925, juvenile survival rate of 0.962 and adult survival rate of 0.98 in their iPCoD model, to align with a population growth rate of 3.65% p.a., and on the assumption that juvenile survival will likely be slightly lower than adult survival. For other MUs in the UK, a calf survival of 0.86 (0-2 years), and juvenile and adult mortality of 0.94 (Sinclair et al 2020).

We do not have survival rates for juveniles/adults from the Irish Sea MU, because of potential movement out of Cardigan Bay, given that adult females are significantly more likely to move into the bay during years 0-2 of their calf's life, and then leave again (Lohrengel et al. 2017, Duckett 2018). If the population overall is stable, then survival rates are likely to be slightly lower than the ones estimated for the East Scotland MU.

Longevity in UK bottlenose dolphins is unknown but the long-term study in Florida found the life span to be 40-45 years in males and 50 years in females (Wells & Scott 1999).

Parameters adopted: For the models, we have proposed survival rates of 0.87 for calves (assessed from Cardigan Bay data), 0.93 (with 0.96 as an alternative) for juveniles, and 0.94 (with 0.96 as an alternative) for adults, slightly reduced on parameters for the East Scotland MU given that the latter population appears to be increasing at a rate of nearly 4% per year while the Cardigan Bay population appears to be stable over the long term (20 years) or decreasing in the last decade.

2.2.5. Pressures

Among the pressures on coastal bottlenose dolphins in the Irish Sea, contaminants is identified as a high threat (ICES WGMME 2019), largely down to very high PCB levels that are of conservation concern and the fact that one of the UK PCB hotspots in terms of bottlenose dolphin contaminant levels is around Northwest Wales (Jepson et al. 2016). Animals from Cardigan Bay have a history of high Persistent Organic Pollutant levels (Morris et al. 1989, Law & Allchin 1994, Law et al. 1995), and photo-ID studies have now shown that individuals from Cardigan Bay (particularly in the northern part) regularly range across the north-east Irish Sea including Liverpool Bay (Feingold & Evans 2014, Lohrengel et al. 2017).

Underwater noise, recreational disturbance, vessel strike, and prey depletion are all identified as moderate threats (ICES WGMME 2019). There is no evidence that bycatch is a serious threat in the Irish Sea (ICES WGBYC 2018, 2019), and there are very few strandings diagnosed as bycatch reported (one individual possibly bycaught, between 2011 and 2017, Deaville 2018, which is consistent with the nature of the fisheries (primarily potting and bottom trawling) in the region.

There are three cases from Wales between 2009 and 2019 of bottlenose dolphin deaths diagnosed as physical trauma, one of which was a calf with acute trauma that could clearly be attributed to a vessel strike (Penrose 2020). There has been increasing concern expressed over interactions between recreational speedcraft and bottlenose dolphins in Cardigan Bay over the last ten years, with physical evidence also of non-lethal strikes on individuals (Feingold & Evans 2014, Lohrengel et al. 2017, Vergara-Peña 2019). We therefore believe this pressure should be included in any modelling of removals.

Parameters adopted: For the models, we have proposed zero bycatch, 10% reduction in reproductive rates from contaminants, removal of 0.3 individuals per year from vessel strike, and 0.67 individuals per year from marine developments (as in NRW's AEOSI position statement, NRW 2020).

2.3. Common Dolphin

2.3.1. Management Unit

Currently, there is little evidence for sub-structuring of the common dolphin population within the NE Atlantic (Evans & Teilmann 2009, ICES WGMME 2012). Indeed, it almost certainly extends beyond areas for which abundance estimates have been obtained (Murphy et al. 2019). IAMMWG (2015, 2020) has proposed for management purposes using a combination of the Celtic & Irish Sea (CIS) and the Greater North Sea (GNS) ecoregions as the Management Unit (Figure 4). Few common dolphins occur in the GNS ecoregion, whilst the greatest abundance occurs along the continental shelf edge in the Bay of Biscay and west of the British Isles and Ireland in the CIS ecoregion (Hammond et al. 2017, Rogan et al. 2018, Waggitt et al. 2020). ICES undertook an assessment of the impact of estimated bycatch rates on the common dolphin population, conducted as part of the EU Emergency Measures, and agreed that the Management Unit should be taken as the entire eastern North Atlantic. Since abundance estimates did not exist for areas beyond the ASCOBANS Agreement Area, those from SCANS-III and Irish ObSERVE surveys were adopted (ICES, 2020, ICES WGBYC 2020, ICES WKEMBYC 2020).

Parameter adopted: Celtic and Greater North Sea (CGNS) Management Unit (IAMMWG 2015, 2020) (Figure 4).

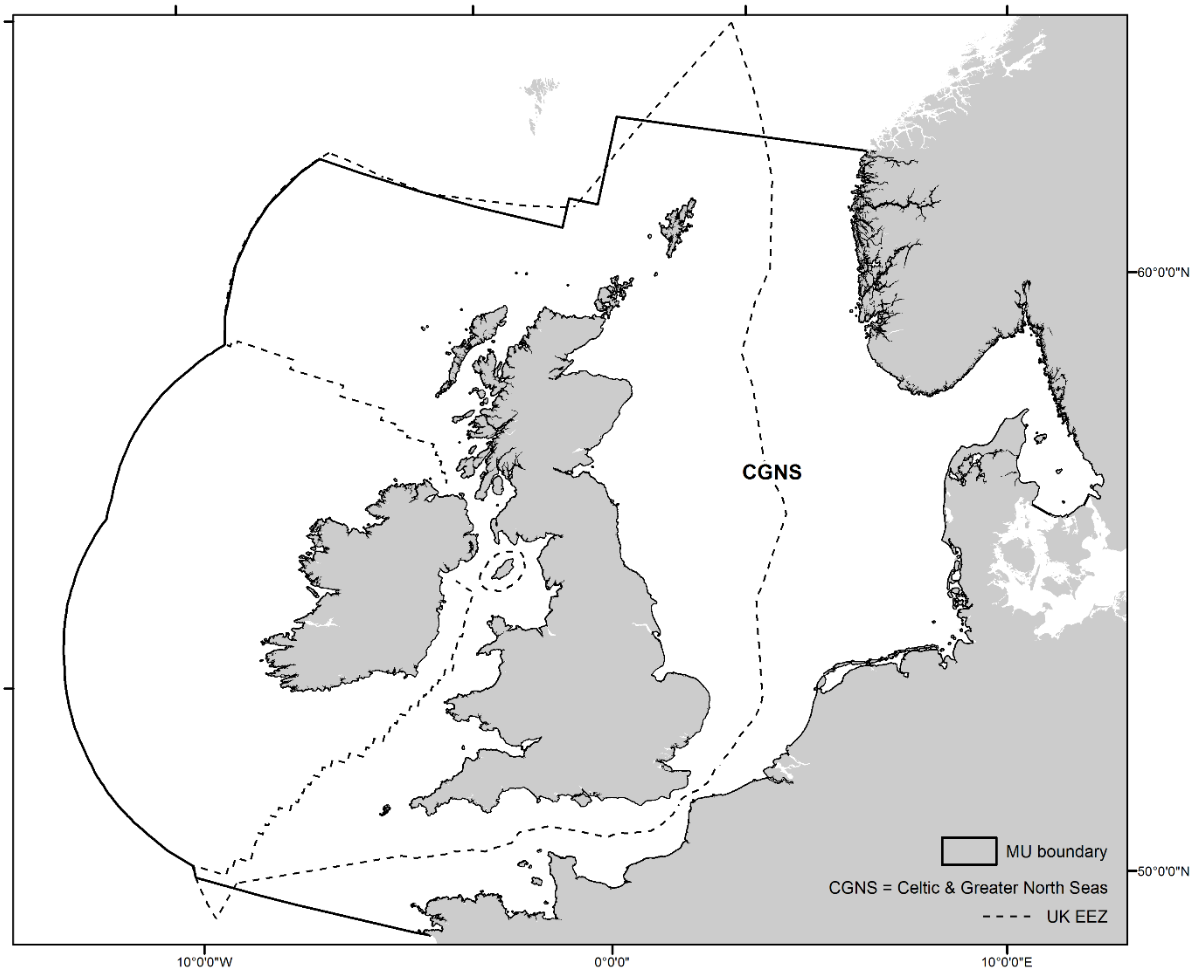

Figure 4: Management Units for common dolphin, Risso's dolphin, and minke whale (from IAMMWG 2021).

2.3.2. Abundance and its Trend

Two abundance estimates exist from wide-scale surveys in the region under consideration (Celtic and Greater North Sea MU). 62,364 (CV: 0.14; 95% CI: 47,098-82,579) common dolphins were estimated in summer 2005 (from SCANS II survey) and 102,656 (CV: 0.29; 95% CI: 58,932-178,822) common dolphins in summer 2016 (from SCANS III/ObSERVE surveys) (IAMMWG 2020, based upon Hammond et al. 2013, 2017, Rogan et al. 2018).

For the wider area considered as closer to the management unit, ICES WKEMBYC (2020) used an estimate for common dolphin abundance of 634,286 (CV: 0.307). Since during the surveys, common and striped dolphin could not always be distinguished (particularly on aerial surveys), this was calculated by taking estimates of abundance for positively identified common dolphins and then correcting these to include a proportion of the abundance of common or striped dolphins that were unidentified to species. This was done separately for SCANS-III ship, SCANS-III aerial and ObSERVE aerial surveys, by multiplying the estimate of unidentified common or striped dolphins by the proportion of identified sightings that were common dolphins.

In 2005, the total summer abundance for Northeast Atlantic shelf waters surveyed during SCANS-II was 54,995 (CV 0.21, 95% CI: 36,500–82,800) (revised from Hammond et al. 2013). This was supplemented by the CODA survey conducted in July 2007 in offshore waters of the ASCOBANS Agreement Area, which estimated a total abundance of 118,264 (CV 0.38, 95% CI: 56,900–247,000) (Hammond et al. 2009). These estimates are corrected for animals missed along the track-line and also for responsive movement. An overall revised estimate combining SCANS-II and CODA surveys (2005/07) is 173,219 (CV 0.27, 95% CI: 103,000-290,000) common dolphins (Evans 2020).

The significant increase in common dolphin numbers between the surveys in 2005 and 2016 both around the British Isles and over the wider area, almost certainly reflects movement into the region from unsurveyed areas further offshore and south of the ASCOBANS Agreement Area, although there may also be a genuine population increase. Annual regional surveys show great variation between years and between seasons (see, for example, Rogan et al. 2018 for Irish waters), suggesting variable movement of segments of the population between years. We therefore do not have a useful time series. We can assume the population is not decreasing, and may be increasing, but any rate of increase value would be arbitrary unless informed by life history analysis of dead animals from the region.

Parameter adopted: 102,656 (CV: 0.29; 95% CI: 58,932-178,822) (IAMMWG 2020).

2.3.3. Reproductive Rates

Although maximum life span for common dolphins in the NE Atlantic is 30 years, 98% of females sampled from the UK, Ireland, France and Spain were less than 20 years of age (Murphy et al. 2007, 2009). With an average age at sexual maturity for females of 8.2 years and an inter-birth interval of 3.79 years, together, these figures suggest a low lifetime reproductive output of possibly four to five calves per female, assuming an older age was attained (Murphy et al. 2007, 2009). Mean generation length was estimated at 12.94 years (Murphy et al. 2007).

In the same study, no significant differences were observed in the proportion of pregnant females, proportion of mature females simultaneously pregnant and lactating, average age attained at sexual maturity, or in nutritional condition of females between two different time periods (1991–1999, and 2000–2006). An annual reproductive rate of 33% was estimated for bycaught individuals using data from 46 mature females, and 26% from 248 mature females that had stranded (Murphy et al. 2009). Reproductive rates are believed to be lower in stranded animals that are more likely to be unhealthy than bycaught ones, and for this reason we have adopted a fertility rate of 0.33, given also the fact that despite bycatch, the population appears to be increasing.

A later study by Mannocci et al. (2012) based upon analysis of 406 females of known age stranded between 1972 and 2006 (but with two-thirds since 2000) in France, largely from the Biscay coast, obtained ages of individuals ranging from 0-28 years. Of these, at least 151 (37%) died as a result of fishery interactions. The age-at-death distribution was multimodal with peaks in the age groups of juveniles (i.e. 2–5 years) and younger sexually mature adults (i.e. 9–12 years). This is quite different from what is expected under a stable age distribution where the greatest frequency is expected for yearlings, followed by juveniles and then adults and demonstrates that bycatch affected age classes unequally.

For example, between 2 and 5 years, about 50% of the individuals surveyed died as a result of bycatch. Average age at sexual maturity (age at which 50% of females are mature) was estimated at 8.24 years with mean age at first reproduction of 9.23 years. Inter-birth interval varied with age: 3.33 (8-11 years), 2.13 (12-15 years), 3.03 (16-19 years), and 7.14 (20-23 years). Mean population growth rate was estimated as 0.945 (i.e. declining at mean rate of 5.5% per year). Maximum population growth rate was calculated to be 4.5% per year. However, age at first reproduction may be underestimated and pregnancy rates overestimated leading to a slight overestimation of population growth rate (Mannocci et al. 2012). The reason for this is that they are based on post-mortem data, so it is not possible to determine whether females were primiparous or multiparous, leading to the assumption that females all conceive immediately after attainment of sexual maturity. For calculating pregnancy rates, intra utero mortality is not considered, leading to the production of calves being potentially overestimated.

In another study in Galicia, NW Spain, between 1990 and 2009, females reached up to 24 years of age, and males to 29 years (Read 2016). Females in the region attained sexual maturity at an average age of 8.4 years, and males at 10.5 years (Read 2016). Using a sample size of 80 mature females, estimates of the annual pregnancy rate varied between 31% and 38% (the higher estimate did not exclude females that were sampled during the mating period), equivalent to an inter-birth interval of 2.5–3 years (Read 2016).

Using a larger sample (n= 334 males, 223 females) from NW Spain, covering the period 1990 to 2013, first year of reproduction was calculated to be 9 years (i.e. sexual maturity was reached around 8 years) (Saavedra 2017). Annual pregnancy rate averaged 0.333, and inter-birth interval was 2.5 years (Saavedra 2017). The generation length was calculated to be 15.05 years (Saavedra 2017). Annual population growth rate was 2.01% (Saavedra 2017), and we have used the same value.

Parameters adopted: We have used the estimate of age at first reproduction of 9 years (Mannocci et al. 2012, Read 2016, Saavedra 2017), inter-birth interval of 3.8 years (Murphy et al. 2007, 2009) and fertility rate of 0.33 (Murphy et al. 2009, Saavedra 2017).

2.3.4. Survival Rates

From stranding samples on the French Biscay coast, Mannocci et al. (2012) noted that survival rates decreased at a constant rate between 0 and 20 years: 0.90 (0-2 years), 0.60 (0-5 years), <0.30 (0-12 years). They estimated juvenile survival rate: at 0.92 and adult survival rate at 0.84. These low survival rates are almost certainly influenced by high bycatch mortality in the region.

In a study in Galicia, NW Spain, between 1990 and 2009 annual adult survival rate was estimated at 0.872, with no significant differences observed between males and females Read 2016). This low rate again probably reflects mortality from bycatch which is greatest in the Bay of Biscay (ICES WGBYC 2019, 2020).

With a larger sample from NW Spain, covering the period 1990 to 2013, calf survival was estimated at 0.877, and non-calf survival at 0.932 under natural conditions whereas non-calf survival dropped to 0.844 when bycatch was included (Saavedra 2017).

Parameters adopted: In the absence of survival estimates from the CIS population and bearing in mind the only existing estimates are from the Bay of Biscay which have long been subject to a high bycatch rate, we have used a calf survival rate estimate of 0.80 (0-1 year), and both juvenile and adult survival rates of 0.95 (>1 year).

2.3.5. Pressures

Bycatch is the most important pressure facing common dolphins in the Celtic and Irish Seas (the area where by far the greatest portion of the IAMMWG Management Unit under consideration is inhabited by the species – Hammond et al. 2017, Evans 2020) is identified as bycatch, with contaminants, underwater noise, and prey depletion considered as moderate threats (ICES WGMME 2019).

In the ICES Celtic Seas ecoregion, highest numbers of dolphins caught were estimated to be in bottom otter trawl (OTB) and gillnet (GNS) fisheries targeting demersal fish mainly in ICES divisions 7.e,f,g), capturing 276 dolphins (95% CI: 151–427) and 192 dolphins (95% CI: 85–299) respectively. The total amount of annual bycatch reported by member states in recent years (2016–2018) across all métiers amounted to 720 dolphins (95% CI: 278–1,345) (ICES WGBYC 2020).

Further south, in the Bay of Biscay and Iberian Peninsula ecoregion, the highest numbers of dolphins caught annually were estimated to be in the trammel net fisheries for demersal fish (GTR_DEF) in ICES division 8.a, amounting to 2,061 dolphins (95% CI: 1,202–3,092). The mean annual bycatch in recent years (2016–2018) across all métiers amounted to 3,973 dolphins (95% CI: 1,998–6,599). In 2017 and 2018, the mortality inferred from French strandings in the Bay of Biscay and the Western Channel were respectively estimated at 9,300 (5,800–17,900) and 5,400 (3,400–10,500) common dolphins (ICES WGBYC 2020).

During the review of emergency measures for the common dolphin, whereas the estimate of annual bycatch used for the entire region (Bay of Biscay and Iberian Peninsula, Celtic and Irish Seas, Greater North Sea) from at-sea bycatch monitoring was 3,973 (95% CI: 1,998-6,598), the equivalent one from strandings and drift modelling was revised downwards to 6,620 (95% CI: 4,411-10,827) (ICES 2020). Using those figures, an assessment (utilising Potential Biological Removal, PBR) indicated an annual anthropogenic mortality limit of 4,927 common dolphins for the Northeast Atlantic Management Unit, based on the estimated population size of 634,286 (ICES 2020).

Population impacts on common dolphins from contaminants, underwater noise disturbance and prey depletion are not possible to assess quantitatively. Evidence of reproductive impairment from high PCB levels has been demonstrated by Murphy et al. (2010, 2019). Climate change may also be having an impact, and a significant positive relationship was found between common dolphin abundance and sea surface temperature (Evans & Waggitt 2020). In response to climate warming, stocks of some typical prey species of common dolphin such as anchovy and sardine have extended their range northwards becoming regular even in the North Sea (Beare et al. 2004).

Parameters adopted: The most obvious anthropogenic source of mortality is bycatch and for the CIS MU area we have used a removal rate of 720 per year (ICES WGBYC 2020). Note that this would need to be increased substantially if the Bay of Biscay was included with common dolphins moving between there and the Celtic and Irish seas, with a different abundance estimate and number bycaught needing to be applied. Removals from other pressures cannot be assessed but are believed to be low, although contaminants may affect reproductive rates.

2.4. Risso's Dolphin

2.4.1. Management Unit

Population parameters for Risso's dolphin are poorly known globally. Although a few genetic studies have been undertaken, with DNA evidence for individuals from the eastern North Atlantic differing from those in the Mediterranean, there is almost no information upon which to base identification of Management Units. IAMMWG (2020) proposes that the same management unit used for common dolphin (CGNS) is also used for this species (Figure 4). As with common dolphin, the true biological population is probably wider than this (Evans 2020), but this presents a more practical approach than using the entire NE Atlantic which likely represents the extent of the population. Risso's dolphins photo-identified off the coast of Anglesey have been re-sighted around the Isle of Man, Bardsey Island, off the Pembrokeshire coast and in the Scottish Hebrides, indicating movements around western UK at a minimum (Stevens 2014). Nevertheless, there may be some sub-structuring, since at Bardsey Island and north Anglesey, c. 12% of identifiable individuals repeatedly return to the same locations (De Boer et al. 2013, Stevens 2014, Sea Watch Foundation unpublished data).

Parameter adopted: Celtic and Greater North Sea (CGNS) Management Unit (IAMMWG 2015, 2020) (Figure 4).

2.4.2. Abundance and Trend

The only abundance estimate available for the CGNS MU, derived from the SCANS-III and Irish ObSERVE surveys in summer 2016 (Hammond et al. 2017, Rogan et al. 2018), is 12,262 (CV: 0.46; 95% CI: 5,227-28,764) (IAMMWG 2020). There is no information on population trends for Risso's dolphins in this region.

Parameter adopted: 12,262 (CV: 0.46; 95% CI: 5,227-28,764) (IAMMWG 2020).

2.4.3. Reproductive Rates

No life history parameters exist for the species from within the population unit area proposed by IAMMWG. In an adjacent region, the Faroe Islands, local drives in 2009 and 2010 of two pods of Risso's dolphin were analysed and provided an estimate of average age of sexual maturity of 8 years in 16 females (Bloch et al. 2012). Life history parameters are available for populations elsewhere in the world; a recent study in South Africa estimated average age at sexual maturity of 7.1 years for males and 7.8 years for females (Plön et al. 2020). These are lower than in a study of Risso's dolphins from Japan which determined average age of sexual maturity of 10-12 years in males and 8-10 years in females, and an inter-birth interval of 2.4 years (Amano & Miyazaki 2004).

Parameters adopted: In the absence of any information on life history parameters from the UK, we recommend values from elsewhere: age at first reproduction of 9 years (Bloch et al. 2012), and an Inter-Birth Interval of 2 years (Amano & Miyazaki 2004).

2.4.4. Survival Rates

No survival estimates are available anywhere in the world. Maximum lifespan recorded is 34 years, but most individuals were less than 27 years (Amano & Miyazaki 2004).

Taylor et al. (2007) used default values for survival rates of 0.798 for juveniles and 0.95 for adults, which will be used here. These were applied by Taylor et al. (2007) using the following procedure: for adult survival, where the oldest individual of the species was less than 50 years, adult survival rate was equated to 0.95 (based upon bottlenose dolphin population parameters from the Florida population which is the best studied of all small cetaceans). They then averaged the ratio of calf to non-calf survival across three well-studied species (southern right whale, humpback whale and bottlenose dolphin), and multiplied that value (0.84) by the non-calf survival rate for all cases without empirical estimates of calf survival.

Parameters adopted: Juvenile survival rate of 0.80 and adult survival rate of 0.95 (Taylor et al. 2007). However, since these are generalised default values (Taylor et al. 2007), they were not modelled in PVA.

2.4.5. Pressures

ICES WGMME (2019) considered three pressures as moderate for this species: contaminants, underwater noise, and bycatch, although evidence in support of specific effects is limited (Evans 2013, 2020).

From post-mortem examinations of 35 Risso’s dolphins stranded in the UK between 1991 and 2010, 17% were diagnosed as by-catch, 14% live-stranded, 14% died from starvation, and 11% died as a result of infectious disease (Deaville & Jepson 2011). Between 2011 and 2017, 15 stranded Risso’s dolphins were investigated at post-mortem (ten in Scotland, three in England, and two in Wales). A cause of death was established in 14 examined individuals, of which four died as a result of meningoencephalitis, two as a consequence of live stranding, two from infections of the gastro-intestinal tract, two were neonatal deaths, one died as a result of impact by vessels (vessel strike), one from physical trauma of unknown origin, one from a peritonitis, and one from a spinal deformity (Deaville 2018).

Parameters adopted: Too little information exists to be able to recommend levels of anthropogenic removals for Risso’s dolphin. However, it is worth noting that in Welsh seas, the species frequently occurs in areas where marine developments in the form of tidal turbines are proposed (Baines & Evans 2012, Evans & Waggitt 2020), so there is potential for physical strikes.

2.5. Minke Whale

2.5.1. Management Unit

There is some genetic evidence from minke whales sampled on the summer feeding grounds for the existence of two sympatric stocks in the North Atlantic, but otherwise there is no evidence of population structure (Anderwald et al. 2011). The implication is that minke whales range extensively across the North Atlantic in summer, but segregate to some extent in winter on at least two breeding grounds somewhere (as yet unidentified) offshore. For purposes of management within the UK, IAMMWG (2015, 2020) recommends the use of the Celtic and Greater North Sea (CGNS) Management Unit for this species (Figure 4).

Parameters adopted: Celtic and Greater North Sea (CGNS) MU (IAMMWG 2020).

2.5.2. Abundance and Trend

Two abundance estimates are available from widescale surveys covering the Celtic & Irish Sea and Greater North Sea regions (CGNS MU). In July 2005, SCANS-II survey estimated 20,136 (CV: 0.20; 95% CI: 14,061-28,786) minke whales (IAMMWG 2020), derived from Hammond (2013) with a revised estimate by Hammond et al. (2017). And in summer 2016, the SCANS-III and Irish ObSERVE surveys (IAMMWG 2020, derived from Hammond et al. 2017, Rogan et al. 2018) yielded an abundance estimate of 20,118 (CV: 0.18; 95% CI: 14,061-28,786) minke whales. These show no trend between the two years of survey.

Parameters adopted: 20,118 (CV: 0.18; 95% CI: 14,061-28,786) (IAMMWG 2020). We have assumed that the population is stable (Hammond et al. 2017, IAMMWG 2020).

2.5.3. Reproductive Rates

Life history parameters have long been difficult to obtain for minke whales because of uncertainties in age determination. New methods (using aspartic acid analysis) in more recent years have been used on hunted minke whales from the Barents Sea, Norwegian and northern North Seas and indicate age at sexual maturity of 5.8 and 7.8 years (using two different age determination methods) (Olsen 2002, Olsen & Sunde 2002). They indicated age at first birth of 9 years and maximum longevity at 32+ years (Olsen & Sunde 2002). In Greenland, during the 1970s to early 1980s, maximum longevity was 33+ years (Christensen 1981). Here, age at sexual maturity was 6 years in males and 7 years in females estimated using traditional ear plug methods (Christensen 1981). Apparent pregnancy rates averaged 0.90, indicating an inter-birth interval of 1 year (Christensen 1975).

During the 1970s, a pregnancy rate from Norwegian animals of 0.97 also indicated annual breeding. This was at a time when the population was probably well below carrying capacity following years of exploitation. A recent study (also using the aspartic acid method) from Iceland estimated average age at sexual maturity to be 6 years, an ovulation rate of 0.877, and pregnancy rate of 0.745, with 92.2% of mature females reproducing (Hauksson et al. 2011). For the eastern North Atlantic population, Sinclair et al. (2020) recommended an average age of first birth at 9 years and a fertility rate of 0.90.

Parameters adopted: In the absence of other information, we have used 9 years as the age of first birth and a fertility rate of 0.90, with an inter-birth interval of 1 year (Sinclair et al. 2020).

2.5.4. Survival Rates

Sinclair et al. (2020) used the above data to estimate minke whale population parameters in European waters assuming a stable population, and proposed a calf survival rate of 0.70 (0-1 year), juvenile survival of 0.77 (1-8 years), and adult survival of 0.98. In the absence of other information, we have adopted the same values for calf and juvenile survival, but have favoured a rather lower adult survival rate estimate (0.96), as suggested by Taylor et al. (2007), due to the fact that minke whales on the west coast of the UK are subject to bycatch (Northridge et al. 2010, Leaper et al. 2022). Maximum age is estimated to be 40-50 years (Stewart & Leatherwood 1985).

Parameters adopted: We have used 0.70 for calf survival rate (0-1 year), 0.77 for juvenile survival rate (1-8 years) (Sinclair et al. 2020), and 0.96 for adult survival rate (Taylor et al. 2007).

2.5.5. Pressures

The most obvious cause of human-induced mortality in the eastern North Atlantic is hunting, with Norway, since 1998, taking on average, between 600 and 800 minke whales each year since 1998 (Evans 2020, based on IWC data). ICES WGMME (2019) has identified underwater noise from seismic surveys as a high level threat, with bycatch and vessel strike as medium threats. Several studies have reported negative behavioural responses to seismic sound at tens of kilometre range, mainly in baleen whales including minke whales (whose communication signals and presumably hearing sensitivity are in the same low frequency range), with masking at mid-range and the risk of auditory injury at close range (Richardson et al. 1995, Nowacek et al. 2007, Southall et al. 2007).

Minke whales are known to suffer entanglement in creel lines, particularly in Scottish waters (Northridge et al. 2013), whilst individuals have also been found with lost/discarded netting around them (including one in Cardigan Bay near New Quay). In fact, between 2011-17, about 36% (n=25) of all examined cases involved gear entanglement, with the majority recorded in cetaceans found stranded in Scotland (Deaville 2018). About 5% of 64 minke whales examined over the same period had died of vessel strike (Deaville 2018).

Parameters adopted: At this point, too little information exists to be able to recommend levels of anthropogenic removals for minke whale.

2.6. Grey Seal

2.6.1. Management Unit

It is well known that there is extensive grey seal movement around the British Isles. Russell et al. (2013) analysed data from satellite transmitters attached to females in four regions (Hebrides, Northern Scotland, East Coast, and South-East Coast) and found extensive movement between regions, with females breeding at colonies far removed from the seal MUs within which they are observed. Within the UK, 14 Seal Management / Monitoring Units (SMUs) are recognised for monitoring, reporting and assessment (SCOS 2020; SCOS 2022). Harwood & King (2014) concluded that it was inappropriate to model the dynamics of each SMU separately, and recommended that a single population model should be used including all UK SMUs, 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, Harwood & King 2014). On the other hand, it would be more appropriate to set MUs internationally to recognise movements across national boundaries. Since 2012, the overall UK population growth rate is still estimated at c. 1% per annum (SCOS 2020) but demographic differences clearly exist between regions, for example, between Ireland and the Hebrides, the former appearing to be continuing to increase whereas the latter stabilised several years ago, and at some sites, e.g. North Rona, are actually declining. (Smout et al. 2019, Thomas et al. 2019, SCOS 2020). UK Statutory Nature Conservation Advisers generally require assessment at the scale of individual Seal MUs or combinations of those. However, there is some disagreement about the appropriateness of the boundaries of the grey seal SMUs – which only extend to UK waters - especially in SW Britain where photo-ID data and telemetry studies demonstrate movements of seals not only around the Irish Sea but also encompassing Southwest England, Northwest France and Ireland (Russell et al. 2019, Carter et al. 2020, Langley et al. 2020; Luck et al. 2020, C. Luck pers. comm.). NRW presently utilise the large OSPAR Region III area (west coast of UK + Ireland) as an interim MU for the species – this MU was used in recent marine development applications and is the basis for reporting under OSPAR and MSFD. However, such a large MU is somewhat un-pragmatic. We therefore have proposed running population models at a large scale – but smaller than the OSPAR Region III area - that encompasses ICES divisions 7a,e,f,g,h (see Figure 1). There is some uncertainty whether to include Southwest Ireland (7j). Here, tracks of individuals from the Blasket Islands (Co. Kerry) show that they have entered the Irish Sea, although most travel was up the west coast of Ireland (in ICES 7b area) and to the Hebrides off west Scotland (C. Luck pers. comm.). We have therefore also considered as an option the aforementioned area but with the inclusion of ICES divisions 7j and 7b.

Given the various uncertainties and different management needs, we have run population models at four increasing spatial scales: Wales (SMU12), Celtic & Irish Seas (ICES divisions 7a,e,f,g,h), Celtic & Irish Seas including all of Ireland (ICES divisions 7a,b,e,f,g,h,j), and OSPAR Region III.

Parameters adopted: Option 1: Wales ($\text{Wales}_{\text{SMU}}$); Option 2: Irish Sea, SW England, southern Ireland, NW France (Celtic and Irish Seas ICES areas: CIS_{ICES}); Option 3: Irish Sea, SW England, all-Ireland, NW France (Celtic and Irish Seas and Ireland ICES areas: $\text{CIS+Ire}_{\text{ICES}}$); and Option 4: western Britain, all-Ireland (OSPAR Region III: OSPAR 3).

2.6.2. Abundance and Trend

Grey seal abundance in the UK is largely derived from pup production estimates. A major challenge in estimating abundance and trends for the region of interest is that pup surveys have tended not have been undertaken systematically but at different times in different regions (see Baines et al. 1995, Westcott 2002, Westcott & Stringell 2003, 2004, McMath & Stringell 2006, Strong et al. 2006, Stringell et al. 2014, Bull et al. 2017, Morgan et al.

2018, Robinson et al. 2020 for Wales; Duck & Morris 2012, Culloch et al. 2018 for Northern Ireland; Kiely et al. 2000, O'Cadhla et al. 2008, 2013, Morris & Duck 2019 for Republic of Ireland; Vincent et al. 2005, 2017 for France).

The Sea Mammal Research Unit (SMRU) has developed a population model to estimate all-age population size (Thomas et al. 2019, SCOS 2020). The population model is based on two sets of data: 1) a region-specific time series of pup production; and 2) overall (i.e. not region-specific) estimates of grey seal population size from August harbour seal surveys.

SCOS (2020) has estimated pup production for Wales (2,250), SW England (450), the Irish Sea portion of Northern Ireland (250, using the latest 2019 estimate for Strangford Lough; and for France (c. 70, Vincent et al. (2017)). In the Irish Republic, pup production along the south and east coasts is estimated at c. 230 (O'Cadhla et al. 2013, C. Luck pers. comm.), making a total of 2,250 for Wales SMU option 1, and 3,250 for CIS MU option 2.

On the west coast of the Irish Republic, pup production is estimated at c. 1,875, giving an estimate of pup production for all of the Irish Republic at c. 2,100 (SCOS 2020). Thus, for MU area option 3, this provides a total pup production of 5,020 (3,245 + 1,875).

Population estimates used in the models are:

Option 1 (Wales (Wales_{SMU})): N_{min} of 4,972 and N of 5,300, based upon a pup production estimate of 2,250, a scalar of 2.21 for N_{min}, (minimum population size) and multiplier of 2.36 for average population size (N) (SCOS 2020, 2021; D. Russell, *pers. comm*).

Option 2 (Irish Sea, SW England, southern Ireland, NW France (CIS_{ICES})): N_{min} of 7,475 and N of 8,125, based upon a pup production of 3,250, a scalar of 2.3 for N_{min} and a multiplier of 2.5 for N (O'Cadhla et al. 2013, Stringell et al. 2014, NRW 2020).

Option 3 (all of the above but also including all of Ireland (CIS+Ire_{ICES})): N_{min} of 11,546 and N of 12,550, based upon a pup production of 5,020, a scalar of 2.3 for N_{min} and a multiplier of 2.5 for N (Stringell et al. 2014, NRW 2020).

Option 4 (OSPAR Region III: western Britain, all of Ireland (OSPAR 3)): N_{min} of 60,780 and N of 64,854, based upon an August count of 16,295 and a multiplier of 3.73 for N_{min} and 3.98 for N (SCOS 2020, 2021; D. Russell, *pers. comm*).

Rates of increase are available for rather few sites within the Irish Sea, and show some variability across time periods and between sites: 1) for Skomer Island (SW Wales), it was 10.2% p.a. between 2011 and 2015 (cf. 8.9% p.a. from 1983-93, but relatively stable in between) (Bull et al. 2017); 2) for the three regularly monitored colonies in Pembrokeshire-Ramsey Island, Skomer, and Marloes Peninsula, the recent rate of increase has been c. 6% p.a. (Bull et al. 2017a, b, Lock et al. 2017, Morgan et al. 2018, SCOS 2021); 3) for Northern Ireland as a whole, the average rate of increase was 4.9% p.a. between 1995 and 2014 (Culloch et al. 2018); whereas 4) for Strangford Lough, the rate of increase averaged 5.2% p.a. over the same period (Culloch et al. 2018). For the Republic of Ireland, an average rate of increase of 3.8% pa has been calculated for grey seals of all ages from aerial thermal imaging surveys, when comparing 2017/2018 with 2011/2012, but with regional differences and the greatest increase (11.5% p.a. occurring in the east, southeast and southwest regions (Duck & Morris 2019).

For an interim trend, we have used a 5% p.a. increase, recognising that in the south and southwest, the increase may be larger (potentially up to c. 10% p.a.), but it is lower elsewhere. Relatively high rates of increase are likely to be a combination of high reproductive rates and movements into breeding colonies from elsewhere.

Parameters adopted: Option 1 (Wales ($\text{Wales}_{\text{SMU}}$)): N_{min} of 4,972 and N of 5,300; Option 2 (Irish Sea, SW England, southern Ireland, NW France (CIS_{ICES})): N_{min} of 7,475 and N of 8,125; Option 3 (all of the above but also including all of Ireland ($\text{CIS+Ire}_{\text{ICES}}$)): N_{min} of 11,546 and N of 12,550; Option 4 (OSPAR III Region: western Britain, all of Ireland (OSPAR3)): N_{min} of 60,780 and N of 64,854. For all, a population increase of 5% p.a. was used.

2.6.3. Reproductive Rates

Age at sexual maturity has been estimated in a number of North Atlantic populations. In Iceland, it has been estimated at 4.9 years for males and 4.0 years for females, with females having their first pup on average at 5.3 years (Hauksson 2007). Similar values have been obtained in Norway, with females reaching sexual maturity (primiparity) on average at 4.7 years (Øigard et al. 2012) and having their first pup at 5.35 years (Wiig 1991). In the UK, female age at first reproduction is believed to be on average 6 years (Harwood & King 2014), although as populations are increasing, an age at first reproduction of 5 years has been proposed for those parts of the UK experiencing an increase, bearing in mind that density-dependent effects seem to apply mainly to pup survival and so may decline as seals at particular sites reach carrying capacity (Sinclair et al. 2020).

Fertility rate is estimated at 0.84 for a population growing at 1% per annum (Sinclair et al. 2020), but is probably higher than this in the Celtic and Irish Seas region that we are considering, given that the population appears to be increasing at a significantly faster rate. Thomas et al. (2019) used a prior mean of 0.83 and a posterior mean of 0.90, and SCOS (2018) a prior mean of 0.83 and posterior mean of 0.95. We therefore propose a fertility rate of 0.90 for this population.

In Norway, grey seals have been estimated to have a pregnancy rate of 0.81 (Øigard et al. 2012), but note that actual birth rates may be lower if abortions occur. In Iceland, Hauksson (2007) found that by 7 years of age, 90% of females were pregnant.

Parameters adopted: For age at first reproduction, we use 5 years (Hauksson 2007, Sinclair et al. 2020) with an inter-birth interval of 1 year and fertility rate of 0.90 (Thomas et al. 2019).

2.6.4. Survival Rates

For their interim PCoD model, Sinclair et al. (2020) recommend using a low pup survival rate of 0.22 rising to a constant (0.94) after year 1. Thomas et al. (2019), on the other hand, when modelling population trends by region, used a posterior mean for pup survival of 0.48 and a prior mean of 0.62, whereas SCOS (2018) used a posterior mean of 0.37 and a prior mean of 0.62. For both juvenile and adult survival, Thomas et al. (2019) used prior and posterior means of 0.90 and 0.95 respectively, whilst SCOS (2018) used prior and posterior means of 0.90 and 0.96 respectively. Both the Thomas et al. (2019) and

SCOS (2018) models use age six as the age of first breeding, whereas the iPCoD model used by Sinclair et al. (2020) 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 (Sinclair et al. 2020).

Male grey seals are estimated to live for 30 years and females for up to 46 years (females) (Bonner 1981).

Parameters adopted: We have used a pup survival rate of 0.40 (0-1 year), following SCOS (2018) and Thomas et al. (2019), a juvenile survival rate of 0.93 and adult survival rate of 0.94, as they more closely mirrored demographic rates. These have been applied to all four area options.

2.6.5. Pressures

Grey seals face a variety of pressures in the region, but possibly the most important one is bycatch (ICES WGBYC 2019, 2020, Luck et al. 2020).

The UK is the only European country with a long-running, dedicated observer programme for bycatch of protected, endangered or threatened species (ICES 2020), and considerable data gaps exist regarding bycatch levels among non-UK fishing fleets and those operating in neighbouring countries. As a result, most countries have scarcely reported any bycatch of seals under dedicated observer schemes although, more recently, Ireland has placed dedicated observers on fishing vessels specifically to assess seal bycatch (Cosgrove et al. 2013, Luck et al. 2020). Within the UK fleet, seal bycatch within ICES divisions 7.a,e,f,g,h in 2016 was an estimated 379 (7.a: 8; 7.e: 181; 7.f: 163; 7.g: 16; 7.h: 11); 361 (7.a: 8; 7.e: 179; 7.f: 153; 7.g: 10; 7.h: 11) in 2017; and 297 (7.a: 3; 7.e: 159; 7.f: 122; 7.g: 4; 7.h: 9) in 2018 (Northridge et al. 2017, 2018, 2019). These were all in static gillnets, particularly tangle or trammel nets. It is important to note, however, that UK bycatch is not determined directly each year but estimated according to effort from sample bycatch rates.

In Irish waters, annual estimates of seal bycatch in static gillnets, from 2011 to 2016, ranged between 202 (90% CI: 2-433) and 349 (90% CI: 6-833) seals per annum (Luck et al. 2020). Those values were calculated by constructing a model of bycatch rate as a function of known predictors of seal bycatch, and using this to predict bycatch rates throughout the Irish Exclusive Economic Zone based upon fishing effort; measures of effort incorporated estimates of gillnet length and soak time. Although grey and harbour seals have not been distinguished, it is thought that almost all bycatch involves grey seals.

Combining estimates of bycatch for 2018 within the Irish EEZ (6.a,b, 7.a,b,c,g,h,j) with those given above from the UK EEZ portion of divisions 7.a,e,f,g.h for the same year, total seal bycatch is 494 (197 within the Irish EEZ + 297 from the UK fleet, based on data provided by A. Kingston & C. Luck). If the smaller area is considered (i.e., excluding 7.b,c,j,k), the total would be 339. In Irish waters, bycatch appears to be highest in divisions 7.j (105), followed by 7.g (31) and 7.b (25). For comparison, estimates in other Irish divisions in 2018 were 12 (7.a), 6 (7.c), 0 (7.h), and 20 (7.k) (C. Luck pers. comm.).

It should be noted that within UK waters these do not include bycatch from other fleets (mainly France) active in the region besides British vessels. However, no seal bycatch was reported by other countries within this region in 2018 (but note that uncertainty remains since for most vessels there were no dedicated observers on board). Bottom trawls occasionally (e.g. by one unnamed country in 2016 and 2017) have also reported high bycatch, although this may be unrepresentative of the fleet as a whole (ICES WGBYC 2019).

Other pressures identified as moderate threats by ICES WGMME (2019) include contaminants, underwater noise, and prey depletion. Although all three have been shown to have negative effects upon grey seals generally (see, for example, Bergman & Olsson 1986, Russell et al. 2016, Jones et al. 2017, Aarts et al. 2018, Robinson et al. 2018), there is very little information on their impact on the species in this region beyond the fact that they co-occur. There has also been concern expressed over potential collision risk from tidal turbines if and when they are introduced (Benjamins et al. 2015, Sparling et al. 2015, 2018, Onoufriou et al. 2019), but ICES WGMME (2019) felt there was insufficient information at this stage to assess their impact.

Parameters adopted: For removals from bycatch, we have used the values 3 per year for Wales (option 1: Wales$_{\text{SMU}}$), 339 per year for Irish Sea, SW England, southern Ireland, NW France (option 2: CIS$_{\text{ICES}}$), 394 per year for Irish Sea, SW England, all-Ireland, NW France (option 3: CIS+Ire$_{\text{ICES}}$), and 494 per year for OSPAR Region III area (ICES Subarea 7) (option 4: OSPAR3). Bycatch was apportioned in the ratio 60% juveniles and 40% adults to take account of the fact that juveniles are believed to be more susceptible to bycatch (ICES WKMOMA 2021). For the impact of contaminants, we have arbitrarily proposed a 10% reduction in reproductive rates as a precautionary estimate, and for marine developments, adopted NRW's AEOSI position statement of the removal of 9 grey seals per year.

3. Population Models

Two main approaches have been taken to assess population level effects of anthropogenic pressures. The first is a rule-based method which aims to determine a threshold for the number of deaths that should not be exceeded if the population is to be sustained. The second method, often adopted in impact assessments, is the use of a Population Viability Analysis (PVA) or predictive modelling approach. The latter may involve matrix models as used in iPCoD (Harwood & King 2014, King et al. 2015, Harwood et al. 2017) or individual (or agent-based) models such as DEPONS (Nabe-Nielsen et al. 2014), both of which were developed to address sub-lethal impacts such as underwater noise disturbance from offshore renewable energy construction activities.

The most commonly used rule-based method is the Potential Biological Removal (PBR), developed in the United States by the National Marine Fisheries Service (Wade 1998, NMFS 2005) to meet Marine Mammal Protection Act requirements to limit anthropogenic removals and applied particularly to bycatch. It is the least data demanding of all the models requiring only an estimate of abundance within the designated area. It can therefore be applied to all the species under consideration in this report.

Another rule-based method that has been frequently used but within a specific context is the Catch Limit Algorithm (CLA) within the International Whaling Commission's Revised Management Procedure (RMP) (IWC 1999). Under the CLA procedure, the limit of removals is calculated as a function of population parameter estimates that are derived by fitting a relatively simple, deterministic population model to time-series of estimates of absolute abundance. It was first applied to fisheries (Cooke 1999), but then incorporated into the RMP for sustainable harvesting of whales. The RMP does not contain the notion of a target level; instead, it aims at achieving the maximum continuing yield consistent with controlling risk (Cooke et al. 2012). For catches set according to the RMP, the expected mean population size, as a fraction of carrying capacity, is scenario dependent.

Removals Limit Algorithm (RLA) (Hammond et al. 2019) is based upon the International Whaling Commission's CLA but there are some important differences in management objectives. The aim for commercial whaling is to maximise catches in a sustainable manner whereas the management objective for an unintentional removal such as bycatch should be to reduce it towards zero. Thus, the CLA target of 72% of carrying capacity was not a conservation objective but merely a tuning level to achieve a balance between conservation and utilisation of the population. For this reason, we do not consider CLA appropriate for management of anthropogenic pressures in a UK context and recommend RLA as its replacement. RLA fits a population dynamics model to a time series of abundance and removals data (Hammond et al. 2019). It assumes a population with density-dependent growth and subject to anthropogenic removals. Model simulations are used to tune the parameters of the RLA until the conservation objectives are met, at which point the limit to removals can be determined. It is a precautionary procedure in the face of uncertainty and utilises extra information where it exists but nevertheless is data demanding as it requires regular estimates of population size, total bycatch and other sources of mortality. Only for bottlenose dolphin, do we have a time series of abundance estimates, but this is focused only on part of the Management Unit area. For other species we either have just one (Risso's dolphin) or two abundance estimates (harbour porpoise, common dolphin, minke whale) whilst for grey seal, abundance estimates exist for portions of the region at different times. Time series of bycatch data (albeit with great uncertainties attached) exist for harbour porpoise, common dolphin and grey seal, but information on removals from bycatch in other species and for all species from other anthropogenic pressures is very limited so that we usually have to make best guesses at these from post-mortem examinations of strandings. We were planning to run RLA on bottlenose dolphin but unfortunately encountered coding issues that prevented us from proceeding. However, we would like to attempt this in future for a direct comparison once the coding issue has been solved.

Population Viability Analysis (PVA) is a species-specific method of risk assessment initially established to determine the probability that a species will go extinct within a given number of years (Beissinger & McCullough 2002, Morris & Doak 2002, Reed et al. 2002). In the past this commonly involved the use of age-structured or Leslie matrix models (Boyce 1992, Beissinger & McCullough 2002, Caswell 2002). However, for long-lived species the later developed stage-structured or Lefkovitch matrix models are typically more practical. It is useful for target setting rather than setting thresholds (such as bycatch limits), but it provides a good modelling framework for better understanding how a population is likely to respond to different management scenarios. It has been more widely used for birds (Maclean et al. 2007, Freeman et al. 2014, Cooke & Robinson 2017) and terrestrial mammals (e.g. Tufto et al. 1999, Wood et al. 2007), although it has been used on some relatively data-rich marine mammal species, such as seals, bottlenose dolphin and killer whale (e.g. Harding et al. 2007, Thompson et al. 2000, Olsen et al. 2014, Lacy et al. 2017). PVA requires information on life history and demographic parameters including age at sexual maturity, reproductive rates, and age- or stage-related survival rates, and it incorporates density-dependent regulation. Where demographic rates are not known expert opinion can be used or estimates can be borrowed from other areas, preferably where the population is behaving similarly. Only three of the six marine mammal species have much of that information from within the designated Management Unit areas: namely harbour porpoise, bottlenose dolphin, and grey seal. However, by borrowing information from other areas, it is possible at least to explore some of the implications of using those particular parameters on common dolphin and minke whale.

Finally, SMRU Consulting (St Andrews, Scotland), has developed an interim Population Consequences of Disturbance (iPCoD) model, in response to the need to assess the impact of disturbance from construction of offshore renewal energy devices (Harwood et al. 2013, Harwood & King 2014). Although the focus was initially on how to address sub-lethal effects such as disturbance which might disrupt feeding, reduce energy intake, and thus have population consequences on vital rates, iPCoD can also incorporate removals in terms of direct mortality. Currently, however, it cannot incorporate density-dependence. For a direct comparison with Population Viability Analysis, iPCoD was run on the same three marine mammal species: harbour porpoise, bottlenose dolphin, and grey seal. For other species there is insufficient data available from within the region.

Table 1 summarises the population parameters we have identified for potential use in the models in the light of the information presented in section 2. Table 2 summarises the input parameters needed for each model.

In summary, three population models are compared: PBR (Potential Biological Removal) for all six species, PVA (Population Viability Analysis) for five of the six species (excluding Risso’s dolphin), and iPCoD (interim Population Consequences of Disturbance) for three species harbour porpoise, bottlenose dolphin, and grey seal.

Table 1: Population Parameters identified for the six Marine Mammal study species to inform the modelling process ([1] abundance estimate for CIS MU; [2] abundance estimate for Irish Sea AU; [3] abundance estimate for Celtic Sea AU (excludes Bay of Biscay); [4] abundance estimate from photo-ID capture-mark-recapture analysis; [5] abundance estimate from SCANS-III survey, 2016 (IAMMWG 2021); [6] abundance estimate for Wales SMU; [7] abundance estimate for CIS MU but excluding W Ireland; [8] abundance estimate for CIS incorporating all of Ireland; [9] abundance estimate for OSPAR III Region (western Britain and all of Ireland), [10] Recorded as decreasing although trends not well known in every area; [11] Increase followed by a decrease; note that values in parentheses are particularly uncertain).

Species	**Harbour Porpoise**	**Bottlenose Dolphin**	**Common Dolphin**	**Risso's Dolphin**	**Minke Whale**	**Grey Seal**
Population Unit	Various	Irish Sea	Celtic & Greater North Sea	Celtic & Greater North Sea	Celtic & Greater North Sea	Various
Population Abundance	62,517[1] 9,376[2] 25,261[3]	318[4] 293[5]	102,656	12,262	20,118	5,300[6] 8,125[7] 12,550[8] 64,854[9]
Population Trend	decreasing[10]	no trend[11]	unknown	unknown	unknown	increasing
Growth Rate	0.96	1.00	unknown	unknown	unknown	1.05
Age at First Reproduction (years)	5	9	9	9	9	5
Fertility / Reproductive Rate	0.54	0.22	[0.33]	unknown	[0.90]	0.90
Inter-Birth Interval (years)	2	3	4	[2]	[1]	1
Calf/Pup Survival Rate	0.60	0.87	0.80	unknown	[0.70]	0.40
Juvenile Survival Rate	0.85	0.93	[0.95]	[0.80]	[0.77]	0.93
Adult Survival Rate	0.90	0.94	[0.95]	[0.95]	[0.96]	0.94

Table 2: Comparison of Population Models, their requirements and scope.

Requirements:	**PBR**	**PVA**	**RLA**	**CLA**	**iPCoD**
Initial Population Estimate			√	√	
Best Population Estimate (N)	√	√			√
Minimum Population Size (N_{min})	√				
Population Time Series			√	√	
Sex Ratio		√	√	√	√
Age at First Reproduction		√	√	√	√
Reproductive Rate		√	√	√	√
Age-Specific Survival Rate		√	√	√	√
Maximum Reproductive Rate (R_{max})	√		√	√	
Recovery Factor (f)	√				
Carrying Capacity (K)	√	√	√	√	
Includes Density Dependence		√	√	√	
No. days of disturbance					√
No. animals affected daily					√
Link between disturbance & demographic rates					√
Useful for Anthropogenic Removals	√	√	√	√ Directed takes only	√
Useful for Sub-lethal Impacts		√			√

3.1. Potential Biological Removal

The Potential Biological Removal (PBR) level is the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal population while allowing it to reach or maintain its Optimum Sustainable Population (OSP) at or above the level that will result in maximum productivity (Maximum Net Productivity Level – MNPL) (Wade 1998), keeping in mind the carrying capacity of the habitat and the health of the ecosystem of which they form a constituent part. For marine mammals this level is thought to be between 50% and 85% of carrying capacity and is more likely to be at the lower end of the range (Taylor & DeMaster 1993). A population is referred to as a stock in the Marine Mammal Protection Act, and defined as a management unit that in the best case delineates a demographically isolated biological population.

The minimum population estimate (N_{min}) is defined as the 20th percentile of a log-normal distribution based on an estimate of the number of animals in the population. This is equivalent to the lower limit of a 60% 2-tailed confidence interval (Barlow et al. 1995). Population simulations have demonstrated that this goal can be achieved by setting it at this value (Wade 1998, NMFS 2005). The default maximum theoretical productivity rate (R_{max}) is 0.04 for cetaceans and 0.12 for seals. These values are used as a default in the absence of species specific information but when such data are available they should be used. The recovery factor, expressed as F_R or simply *f*, is intended to compensate for uncertainty and possible unknown estimation errors. A recovery factor of 0.1 often is the default used for endangered or threatened populations of marine mammals, 0.3 for near-threatened species, and 0.5 used for those populations depleted, threatened or of unknown status (Wade 1998, NMFS 2005). Since the status of most species is not clearly known, a recovery factor of 0.5 tends to be used.

The main strength of the PBR model is its simplicity and the fact that it does not require a lot of biological information which is often not readily available for marine mammal species. Parameters can be set low to be precautionary. As noted above, it is routinely used by management bodies in the United States to determine limits of anthropogenic removals, typically bycatch (NMFS 2005). In the US the conservation objective for cetacean PBR limits is 50% of carrying capacity within a 100-year period (NMFS 2005). Although conservation objectives have not been finalised in Europe, provisionally they have been set at maintaining the population at 80% carrying capacity with an 80% probability within a 100-year period, as applied for the OSPAR bycatch indicator. This reduces the default recovery factor (f) slightly below 0.5, and has thus been referred to as a modified PBR (Genu et al. 2021). As with all models, there are some limitations: it makes assumptions on population growth rates, the influence of density dependence, estimates of carrying capacity which is rarely known, and a recovery factor which can be set low if uncertainty is high, or the population status is unfavourable (NMFS 2005), or it can be set using a rule-based decision matrix.

The PBR equation is as follows:

$$PBR = 0.5 * R_{max} * N_{min} * f$$

where R_{max} is the maximum annual recruitment rate, N_{min} is a conservative estimate of population size (20th percentile), and *f* is a recovery factor between 0.1 and 1 (Wade 1998;

Taylor et al. 2000; Hunter & Caswell 2005; Niel & Lebreton 2005; Dillingham & Fletcher 2008).

We used the recommended default values for R_{max} of 0.04 and 0.12, for cetaceans and pinnipeds, respectively (Wade 1998, Taylor et al. 2007). However, we acknowledge that even among cetaceans, species can have very different life histories which will ultimately influence R_{max}. Take for example harbour porpoise which have a relatively fast life history as they can produce their first offspring around 4 years of age and are capable of breeding every year or every other year, or a minke whale which is capable of producing a calf every year. Compare these to bottlenose dolphins which have a slower life history, as they do not mature until 9 years of age and have an average inter-birth interval of 3-4 years. On the other hand, we know that harbour porpoise (Learmonth et al. 2014, Read 2016, Murphy et al. 2020) do not live nearly as long as bottlenose dolphins (Wells & Scott 1999). These factors will affect the potential population growth rate. For this reason, we also calculated PBR using an alternative R_{max} value to explore what the consequences would be of lowering or raising R_{max}. The alternative R_{max} value was chosen by adjusting the default value according population growth and given what we know about their life history while also considering potential vulnerability as a result of population size. For bottlenose dolphins, we tested a lower alternative R_{max} of 0.03 due to their small population size and slow life history. For harbour porpoise and minke whale, we tested a higher R_{max} of 0.05 due to their faster life history (in terms of their high reproductive rate and for harbour porpoise also their relatively young age at maturity). For common and Risso's dolphin, we explored a lower alternative R_{max} of 0.03 to be precautionary as a result of higher uncertainty in their population parameters and due to their relatively slow life history. For grey seals, the default R_{max} is already quite high, and as the population may be nearing carrying capacity and to be precautionary, we tested the consequence of an R_{max} of 0.10. N_{min} (NMFS 2005) was calculated based on the best estimates of abundance (N) (Table 3; also see Tables 12, 14, and 16-18 in the PVA section for estimates of N used). For bottlenose dolphins and grey seals, this involved two different estimates of N allowing the assessment of two different population paradigms.

$$N_{min} = \frac{N}{e^{0.842*(\ln 1+CV^2)^{0.5}}}$$

We calculated PBR based on all recovery factors from 0.1 to 1 by increments of 0.1 resulting in ten values for both the default R_{max} and the alternative R_{max} (Figure 5). For our recommended values of PBR, we always used the default R_{max} and chose a recovery factor of 0.2 for bottlenose dolphins due to the small population size, 0.8 for grey seals due to its larger population size and growth, but 0.5 in Wales in order to retain some precaution because colonies around Wales are largely within SACs that require careful management, and 0.5 for all other species because of uncertainty in population estimates.

Table 3: Values for maximum reproductive rate (R_{max}), minimum population size (Nmin) and recovery factor for the six marine mammal study species ([1] from abundance estimate for CIS MU; [2] from abundance estimate for IS AU; [3] from abundance estimate for CS AU (excludes Bay of Biscay); [4] from abundance estimate from photo-ID capture-mark-recapture analysis; [5] from abundance estimate from SCANS-III survey, 2016 (IAMMWG 2021); [6] from abundance estimate for Wales SMU; [7] from abundance estimate for CIS MU but excluding W Ireland; [8] from abundance estimate for CIS incorporating all of Ireland; [9] from abundance estimate for OSPAR III Region (western Britain and all of Ireland)

Species	**Default R_{max}**	**Alternative R_{max}**	**N_{min}**	**Recovery factors (f)**	**Recommended Recovery factor (f)**
Harbour porpoise	0.04	0.05	56,061[1] 7,152[2] 16,732[3]	0.1-1.0	0.5 0.5 0.5
Bottlenose dolphin	0.04	0.03	232[4] 191[5]	0.1-1.0	0.2 0.2
Common dolphin	0.04	0.03	80,811	0.1-1.0	0.5
Risso's dolphin	0.04	0.03	8,479	0.1-1.0	0.5
Minke whale	0.04	0.05	17,309	0.1-1.0	0.5
Grey seal	0.12	0.10	4,972[6] 7,475[7] 11,546[8] 60,780[9]	0.1-1.0	0.5 / 0.8 0.8 0.8 0.8

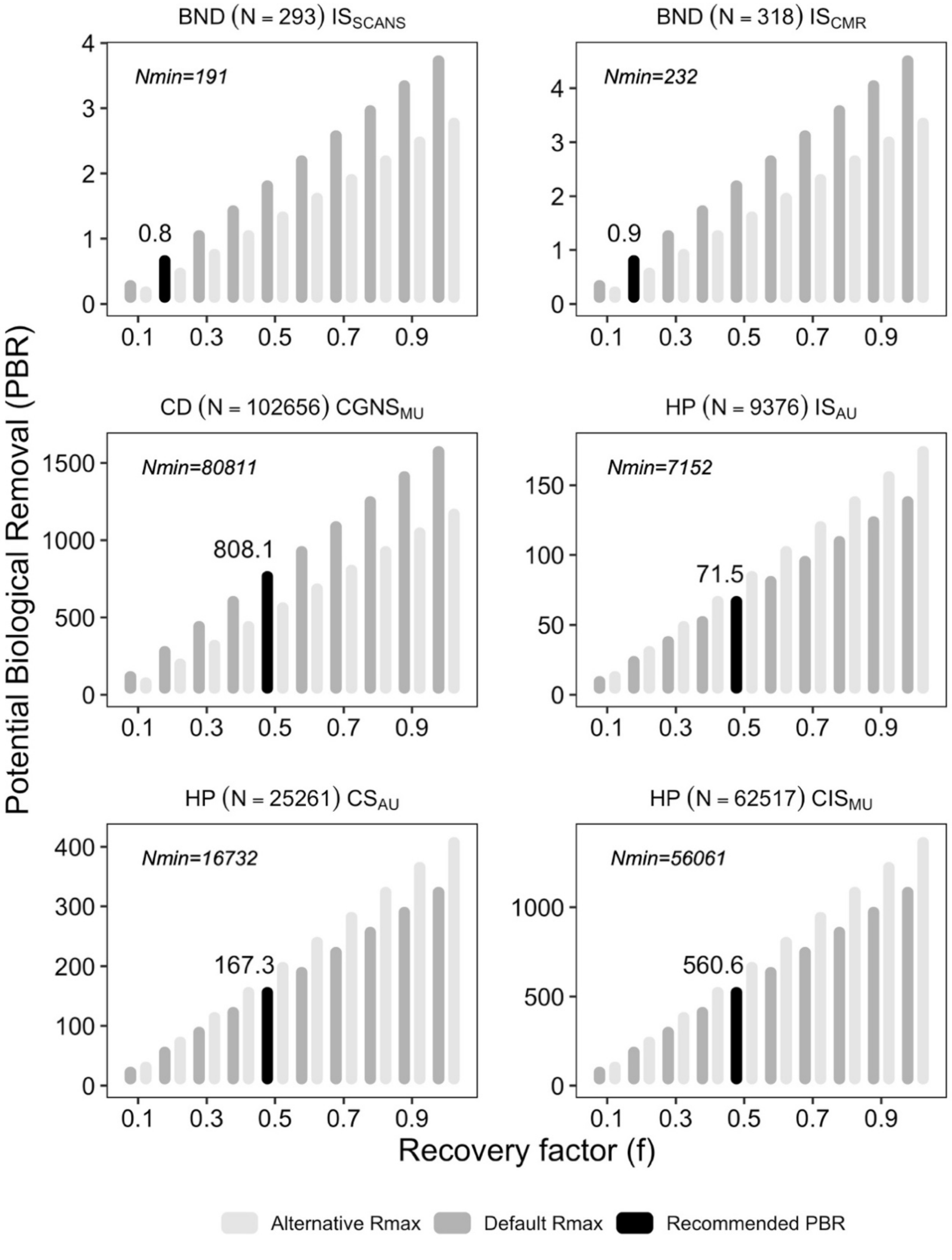

Figure 5: Potential Biological Removal (PBR) limits for bottlenose dolphin, common dolphin and harbour porpoise (see next page for Minke whale, Risso's dolphin, grey seal) across the full range of Recovery Factors (*f*). PBR was calculated both for the default (0.04 for cetaceans; dark grey bars) and the alternative R_{max} (light grey bars; see Tables 4-11). The single black bar indicates the recommended recovery factor (f) and PBR value for the species (always based on the default R_{max}).

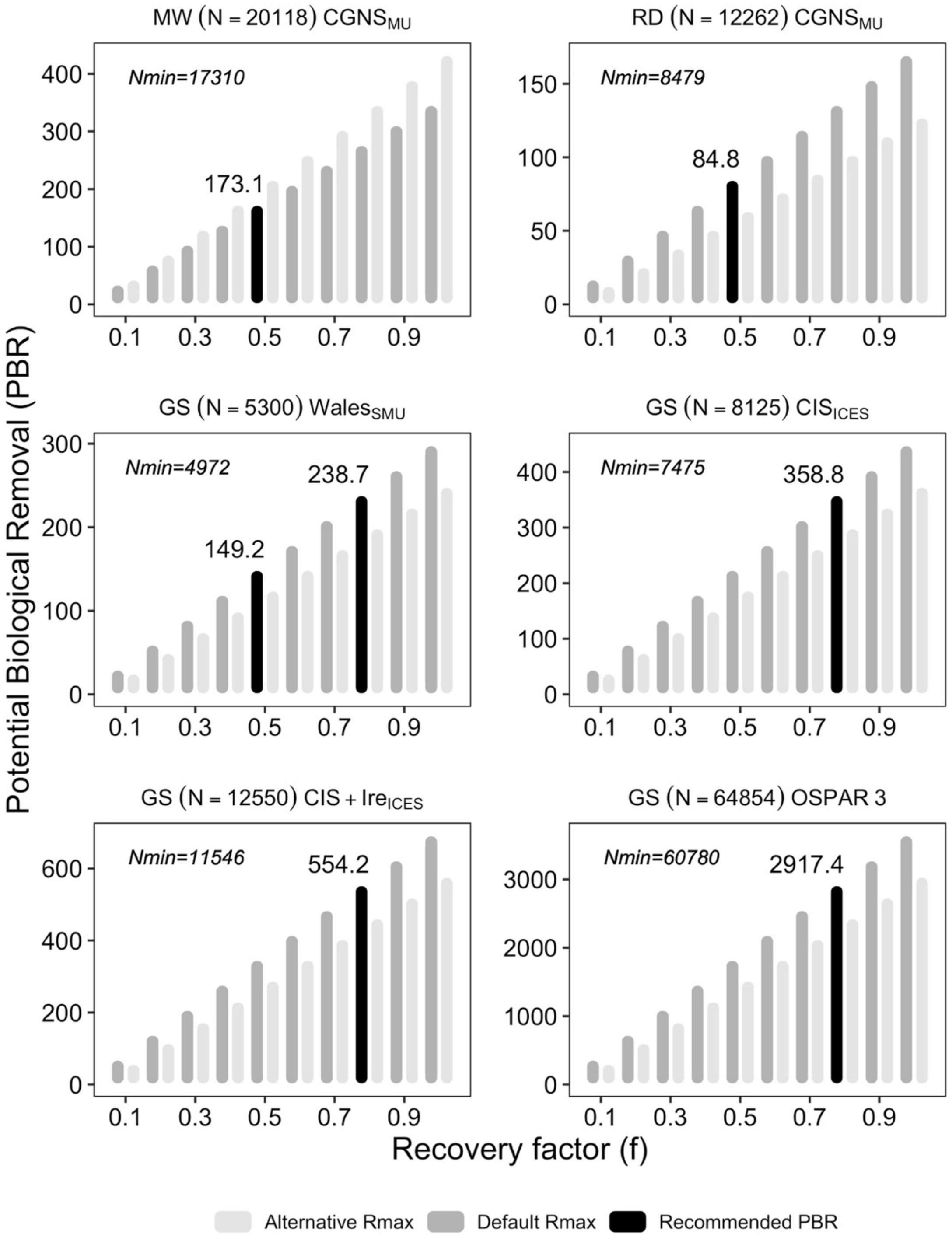

Figure 5 cont.: Potential Biological Removal (PBR) limits for minke whale, Risso's dolphin and grey seal across the full range of Recovery Factors (*f*). PBR was calculated both for the default (0.04 for cetaceans and 0.12 for pinnipeds; dark grey bars) and the alternative R_{max} (light grey bars; see also Tables 4-11). The black bars indicate the recommended recovery factor (f) and PBR value for the species (always based on the default R_{max}). A lower f value is included for grey seals in Wales as a precautionary measure since most are within SACs that require careful management.

Table 4: Estimates of PBR for harbour porpoise at three spatial scales (Celtic & Irish Seas Management Unit (CIS_{MU}), Irish Sea Assessment Unit (IS_{AU}), and Celtic Seas Assessment Unit (CS_{AU})) for both the default and alternative R_{max} values, as well as the range of recovery factors (f). Bold values indicate the recommended recovery factor (f) and PBR value for the species.

Harbour porpoise (N=62,517) CIS_{MU}

R_{max} default	**f**	**PBR**	**R_{max} alternative**	**f**	**PBR**
0.04	0.1	112	0.05	0.1	140
0.04	0.2	224	0.05	0.2	280
0.04	0.3	336	0.05	0.3	420
0.04	0.4	448	0.05	0.4	561
0.04	**0.5**	**561**	0.05	0.5	701
0.04	0.6	673	0.05	0.6	841
0.04	0.7	785	0.05	0.7	981
0.04	0.8	897	0.05	0.8	1121
0.04	0.9	1009	0.05	0.9	1261
0.04	1	1121	0.05	1	1402

Table 4 cont.

Harbour porpoise (N=9,376) IS_{AU}

R_{max} default	f	PBR	R_{max} alternative	f	PBR
0.04	0.1	14	0.05	0.1	18
0.04	0.2	29	0.05	0.2	36
0.04	0.3	43	0.05	0.3	54
0.04	0.4	57	0.05	0.4	72
0.04	**0.5**	**72**	0.05	0.5	89
0.04	0.6	86	0.05	0.6	107
0.04	0.7	100	0.05	0.7	125
0.04	0.8	114	0.05	0.8	143
0.04	0.9	129	0.05	0.9	161
0.04	1	143	0.05	1	179

Table 4 cont.

Harbour porpoise (N=25,261) CS_{AU}

R_{max} default	f	PBR	R_{max} alternative	f	PBR
0.04	0.1	33	0.05	0.1	42
0.04	0.2	67	0.05	0.2	84
0.04	0.3	100	0.05	0.3	125
0.04	0.4	134	0.05	0.4	167
0.04	**0.5**	**167**	0.05	0.5	209
0.04	0.6	201	0.05	0.6	251
0.04	0.7	234	0.05	0.7	293
0.04	0.8	268	0.05	0.8	335
0.04	0.9	301	0.05	0.9	376
0.04	1	335	0.05	1	418

Table 5: Estimates of PBR for bottlenose dolphins using SCANS III/ObSERVE survey estimate (N=293) for both the default and alternative R_{max} values, as well as the range of recovery factors (f). Bold values indicate the recommended recovery factor (f) and PBR value for the species. The spatial scale adopted is the Irish Sea Management Unit (IS).

Bottlenose dolphin (N=293) IS_{SCANS}

R_{max} alternative	f	PBR	R_{max} default	f	PBR
0.03	0.1	0.3	0.04	0.1	0.4
0.03	0.2	0.6	0.04	**0.2**	**0.8**
0.03	0.3	0.9	0.04	0.3	1.1
0.03	0.4	1.1	0.04	0.4	1.5
0.03	0.5	1.4	0.04	0.5	1.9
0.03	0.6	1.7	0.04	0.6	2.3
0.03	0.7	2.0	0.04	0.7	2.7
0.03	0.8	2.3	0.04	0.8	3.1
0.03	0.9	2.6	0.04	0.9	3.4
0.03	1	2.9	0.04	1	3.8

Table 6: Estimates of PBR for bottlenose dolphins (N=318) from CMR analysis for both the default and alternative R_{max} values, as well as the range of recovery factors (f). Bold values indicate the recommended recovery factor (f) and PBR value for the species. The spatial scale adopted is the Irish Sea Management Unit (IS).

Bottlenose dolphin (N=318) IS_{CMR}

R_{max} alternative	**f**	**PBR**	**R_{max} default**	**f**	**PBR**
0.03	0.1	0.3	0.04	0.1	0.5
0.03	0.2	0.7	0.04	**0.2**	**0.9**
0.03	0.3	1.0	0.04	0.3	1.4
0.03	0.4	1.4	0.04	0.4	1.9
0.03	0.5	1.7	0.04	0.5	2.3
0.03	0.6	2.1	0.04	0.6	2.8
0.03	0.7	2.4	0.04	0.7	3.2
0.03	0.8	2.8	0.04	0.8	3.7
0.03	0.9	3.1	0.04	0.9	4.2
0.03	1	3.5	0.04	1	4.6

Table 7: Estimates of PBR for common dolphins for both the default and alternative R_{max} values, as well as the range of recovery factors (f). Bold values indicate the recommended recovery factor (f) and PBR value for the species. The spatial scale adopted is the Celtic and Greater North Sea Management Unit ($CGNS_{MU}$).

Common dolphin (N=102,656) $CGNS_{MU}$

R_{max} alternative	f	PBR	R_{max} default	f	PBR
0.03	0.1	121	0.04	0.1	162
0.03	0.2	242	0.04	0.2	323
0.03	0.3	364	0.04	0.3	485
0.03	0.4	485	0.04	0.4	646
0.03	0.5	606	0.04	**0.5**	**808**
0.03	0.6	727	0.04	0.6	970
0.03	0.7	849	0.04	0.7	1131
0.03	0.8	970	0.04	0.8	1293
0.03	0.9	1091	0.04	0.9	1455
0.03	1	1212	0.04	1	1616

Table 8: Estimates of PBR for Risso's dolphins for both the default and alternative R_{max} values, as well as the range of recovery factors (f). Bold values indicate the recommended recovery factor (f) and PBR value for the species. The spatial scale adopted is the Celtic and Greater North Sea Management Unit ($CGNS_{MU}$).

Risso's dolphin (N=12,262) $CGNS_{MU}$

R_{max} alternative	f	PBR	R_{max} default	f	PBR
0.03	0.1	13	0.04	0.1	17
0.03	0.2	25	0.04	0.2	34
0.03	0.3	38	0.04	0.3	51
0.03	0.4	51	0.04	0.4	68
0.03	0.5	64	0.04	**0.5**	**85**
0.03	0.6	76	0.04	0.6	102
0.03	0.7	89	0.04	0.7	119
0.03	0.8	102	0.04	0.8	136
0.03	0.9	114	0.04	0.9	153
0.03	1	127	0.04	1	170

Table 9: Estimates of PBR for minke whale for both the default and alternative R_{max} values, as well as the range of recovery factors (f). Bold values indicate the recommended recovery factor (f) and PBR value for the species. The spatial scale adopted is the Celtic and Greater North Sea Management Unit ($CGNS_{MU}$).

Minke whale (N=20,118) $CGNS_{MU}$

R_{max} default	f	PBR	R_{max} alternative	f	PBR
0.04	0.1	35	0.05	0.1	43
0.04	0.2	69	0.05	0.2	87
0.04	0.3	104	0.05	0.3	130
0.04	0.4	138	0.05	0.4	173
0.04	**0.5**	**173**	0.05	0.5	216
0.04	0.6	208	0.05	0.6	260
0.04	0.7	242	0.05	0.7	303
0.04	0.8	277	0.05	0.8	346
0.04	0.9	312	0.05	0.9	389
0.04	1	346	0.05	1	433

Table 10: Estimates of PBR for grey seals for both the default and alternative R_{max} values, as well as the range of recovery factors (f). Bold values indicate the recommended recovery factor (f) and PBR value for the species. The spatial scales adopted are Option 1: Wales (Wales_{SMU}); Option 2: Irish Sea, SW England, southern Ireland, NW France (Celtic and Irish Seas ICES areas: CIS_{ICES}); Option 3: Irish Sea, SW England, all-Ireland, NW France (Celtic and Irish Seas and Ireland ICES areas: CIS+Ire_{ICES}); and Option 4: western Britain, all-Ireland (OSPAR Region III: OSPAR 3).

Grey seal (N=5,300) Wales_{SMU}

R_{max} alternative	**f**	**PBR**	**R_{max} default**	**f**	**PBR**
0.10	0.1	25	0.12	0.1	30
0.10	0.2	50	0.12	0.2	60
0.10	0.3	75	0.12	0.3	89
0.10	0.4	99	0.12	0.4	119
0.10	0.5	124	0.12	**0.5**	**149**
0.10	0.6	149	0.12	0.6	179
0.10	0.7	174	0.12	0.7	209
0.10	0.8	199	0.12	**0.8**	**239**
0.10	0.9	224	0.12	0.9	268
0.10	1	249	0.12	1	298

Table 10 cont.

Grey seal (N=8,125) CIS_{ICES}

R_{max} alternative	f	PBR	R_{max} default	f	PBR
0.10	0.1	37	0.12	0.1	45
0.10	0.2	75	0.12	0.2	90
0.10	0.3	112	0.12	0.3	135
0.10	0.4	150	0.12	0.4	179
0.10	0.5	187	0.12	0.5	224
0.10	0.6	224	0.12	0.6	269
0.10	0.7	262	0.12	0.7	314
0.10	0.8	299	0.12	**0.8**	**359**
0.10	0.9	336	0.12	0.9	404
0.10	1	374	0.12	1	449

Table 10 cont.

Grey seal (N=12,550) CIS+Ire$_{ICES}$

R_{max} alternative	f	PBR	R_{max} default	f	PBR
0.10	0.1	58	0.12	0.1	69
0.10	0.2	115	0.12	0.2	139
0.10	0.3	173	0.12	0.3	208
0.10	0.4	231	0.12	0.4	277
0.10	0.5	289	0.12	0.5	346
0.10	0.6	346	0.12	0.6	416
0.10	0.7	404	0.12	0.7	485
0.10	0.8	462	0.12	**0.8**	**554**
0.10	0.9	520	0.12	0.9	623
0.10	1	577	0.12	1	693

Table 10 cont.

Grey seal (N=64,854) OSPAR III

R_{max} alternative	f	PBR	R_{max} default	f	PBR
0.10	0.1	304	0.12	0.1	365
0.10	0.2	608	0.12	0.2	729
0.10	0.3	912	0.12	0.3	1094
0.10	0.4	1216	0.12	0.4	1459
0.10	0.5	1520	0.12	0.5	1823
0.10	0.6	1823	0.12	0.6	2188
0.10	0.7	2127	0.12	0.7	2553
0.10	0.8	2431	0.12	**0.8**	**2917**
0.10	0.9	2735	0.12	0.9	3282
0.10	1	3039	0.12	1	3647

3.2. Population Viability Analysis

As noted in the introduction to this section, Population Viability Analysis (PVA) is a species-specific method of risk assessment established to determine the probability that a population will go extinct within a given number of years (Beissinger & McCullough 2002, Reed et al. 2002). Typically, Leslie matrix models have been used in the past (Boyce 1992), but are not very suited for application to long-lived species (Kendall et al. 2019) since information on the survival and reproductive rates for each year of the animal's lifetime is needed. Instead stage-structured or Lefkovitch matrix models can be applied however these still require several types of demographic data (for example, annual survival rates of different stage classes, age at first reproduction, density-dependent regulation, etc), although the exploratory process can borrow information from other populations. One of the strengths of PVA is that it allows one to explore a range of scenarios to identify which population parameters (e.g. survival at different life-stages, reproductive rates) may be most critical in being impacted by an anthropogenic pressure by comparing results of models with and without the population-level consequences of a particular human activity either separately or in combination (Akçakaya & Sjögren-Gulve, 2000). Although its value is enhanced with more data on population parameters, incomplete information does not necessarily preclude meaningful results. PVA can incorporate uncertainties in the data, and establish the relative effects of these uncertainties through transfer function analysis. It can also identify the parameter(s) which deserve highest priority in terms of obtaining more precise estimates as well as time series of estimates where possible. RLA and iPCoD can do the same although the latter as yet cannot incorporate density dependence which may well be relevant (Akçakaya & Sjögren-Gulve, 2000).

3.2.1. Lefkovitch (stage-structured) population models

We used a stage-structured or Lefkovitch matrix model to project population trends for species where sufficient information existed on demographic parameters, namely bottlenose dolphin, harbour porpoise, grey seal, common dolphin and minke whale. Risso’s dolphins were excluded due to significant uncertainty in demographic and population parameters. The PVA models were constructed manually in R, although we used some functionality with the R packages `popdemo` (Stott et al. 2018) and `popbio` (Stubben & Milligan 2007). Given the fact that sex-specific demographic rates were typically not available and that marine mammal populations rely heavily on the number of females, we ran female-only models. Therefore, the total population abundance for the MU was adjusted according to the species sex ratio. Additionally, the reproductive rate was halved allowing only female offspring to enter the population each year, assuming a 50:50 sex ratio at birth. The life cycle model for each of the three species included three life history stages, namely calf/pup (c), juvenile (j), and adult (a) within which demographic rates are constant (Figure 6). Only adults were categorised as reproductive and contributed to annual recruitment. The stage-structured model includes three different probabilities, namely the probability of reproducing each year (F), the probability of surviving any given year (year *t*) and transitioning to the next stage in the following year (year *t+1*) (G), and the probability of surviving year *t* and remaining in the same stage in year *t+1* (P) (Figure 6).

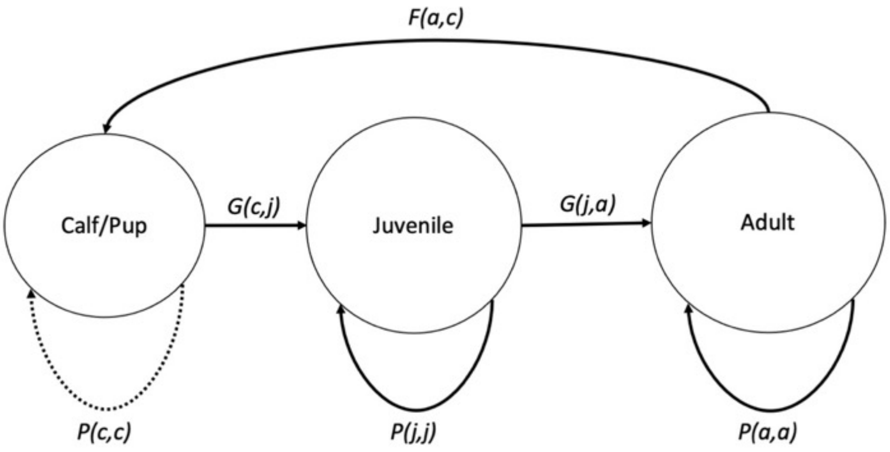

Figure 6: Life cycle model with the stage calf/pup (c), juvenile (j) and adult (a) showcasing the probability of reproduction F(a,c), the probability of surviving and remaining in the same stage (P) and the probability of surviving and transitioning to the next stage.

Below is the corresponding stage-structured matrix (A) based on the illustrated life cycle model (Figure 6).

$$A = \begin{bmatrix} P(c,c) & 0 & F(a,c) \\ G(c,j) & P(j,j) & 0 \\ 0 & G(j,a) & P(a,a) \end{bmatrix}$$

Grey seal, harbour porpoise, and minke whale offspring only spend one year as a pup or calf, which means that if the individual survived the year (year *t*) it would have to transition to a juvenile the following year (year *t+1*). Therefore, the probability that a calf/pup survives year *t* and remains a calf/pup is zero (P(c,c)=0).

In contrast, bottlenose dolphins and common dolphins can spend up to 3 years as calves (i.e. dependent upon their mother) and therefore in any given year, there was both a probability of calves surviving year *t* and remaining a calf in year *t+1* and surviving year *t* and transitioning to a juvenile in year *t+1*. A resulting population growth rate (λ) which is <1 indicates a decline, a λ=1 indicates a stable population, and finally a λ>1 indicates the population is increasing.

A density-dependent reproductive rate was applied using the equation below (Taylor and Demaster 1993):

$$Density\ dependence = F_K + (F_0 - F_K) * \left(1 - \left(\frac{N}{K}\right)^z\right)$$

where F_K is the reproductive rate at carrying capacity (F_K=0.1), F_0 is reproductive rate at lower population abundance, N is population abundance in year t, K is the carrying capacity, and z is a parameter that shapes the form of the relationship between maximum and minimum reproductive rates (see Taylor and Demaster 1993 for further details).

3.2.2. Demographic rates used in PVA:

Below are the demographic rates chosen for the stage-structured population model based on the information in Section 2 for five of the six species considered (Risso's dolphin had to be excluded due to lack of values for necessary population parameters), as well as the demographic rate scenarios tested for bottlenose dolphin, harbour porpoise and grey seal.

Table 11: Harbour porpoise population data and demographic rates used in PVA.

Parameter	**Selected Value**	**Justification/references**
MU abundance	62,517 (CIS_{MU}) 9,376 (IS_{AU}) 25,261 (CS_{AU})	2016 SCANS 3 & ObSERVE survey estimates (Rogan et al. 2018, Hammond et al. 2021, IAMMWG 2021)
Carrying capacity	170,000 (CIS) 16,400 (IS) 46,100 (CS)	Upper 95% confidence limit from highest population estimate (Hammond et al. 2013, 2017)
Sex ratio (M:F)	1:1	Default assumption
Female abundance	31,259 (CIS) 4,688 (IS) 12,630 (CS)	50% of population size
Calf survival	0.60 0.85	Moore & Read 2008, Sinclair et al. 2020 Sinclair et al. 2020
Calf stage (years)	1	Murphy et al. 2015
Juvenile survival	0.85	Sinclair et al. 2020
Juvenile stage (years)	4	Murphy et al. 2015, 2020
Adult survival	0.85 0.90 0.925	Based on range of values from Winship et al. 2008, Sinclair et al. 2020
Age at first reproduction	5	Murphy et al. 2015, 2020
Reproductive rate	0.34 0.54 0.68	Murphy et al. 2015, 2020 Murphy et al. 2015, 2020 Murphy et al. 2015, 2020

Table 12: Harbour porpoise demographic rate scenarios included in PVA, and their resulting mean population growth rates (λ).

Demographic Scenario	**Calf survival**	**Adult survival**	**Reproductive rate**	**λ (62,517) (CIS MU)**	**λ (9,376) (IS MU)**	**λ (25,261) (CS MU)**
1	0.60	0.85	0.34	0.933	0.934	0.934
2	**0.60**	**0.85**	**0.54**	**0.965**	**0.967**	**0.967**
3	0.60	0.85	0.68	0.984	0.987	0.987
4	0.85	0.925	0.34	1.009	1.013	1.013
5	0.85	0.925	0.54	1.031	1.045	1.042
6	0.85	0.925	0.68	1.038	1.059	1.054

No single combination in Table 12 is necessarily the true one. The population appears to be decreasing and, therefore, scenarios 1 or 2 come closest to the observed trend. If reproductive rates are 0.54 as suggested from previous analyses, then scenario 2 may be the most appropriate one to use.

Table 13: Details of bottlenose dolphin population data and demographic rates necessary for use in the PVA. Two sets of analyses were run based on two different estimates of abundance (N) for the Irish Sea management unit (MU), one based on capture-mark-recapture data (CMR) and another an estimate from line transect data from the SCANS-III & ObSERVE surveys (IAMMWG 2021)

Parameter	**Selected Value(s)**	**Justification/reference sources**
MU abundance	N_{CMR} = 318 N_{SCANS} = 293	Lohrengel et al. 2017 Hammond et al. 2021, Rogan et al. 2018, IAMMWG 2021
Carrying capacity	450	Upper 95% confidence limit from highest annual population estimate (Lohrengel et al. 2017)
Sex ratio (M:F)	1:1	Default assumption
Female abundance	N_{CMR} = 159 N_{SCANS} = 147	50% of population size estimated from CMR 50% of population size estimated from SCANS-III
Calf survival	0.87	Average value for 0-3 year olds from Cardigan Bay population (Pesante et al. 2008; Feingold & Evans 2014)
Calf stage (years)	3	Pesante et al. 2008, Feingold & Evans 2014
Juvenile survival	0.93 0.96	Pesante et al. 2008, Feingold & Evans 2014 Arso Civil et al. 2018
Juvenile stage (years)	6	Cheney et al. 2019
Adult survival	0.94 0.96	0.96 option midway between estimates of 0.94 (Arso Civil et al. 2018) and 0.98 (Sinclair et al. 2020) from East Scottish population
Age at first reproduction	9	Cheney et al. 2019
Reproductive rate	0.16 0.22 0.30	Robinson et al. 2017 Arso Civil et al. 2017 Sinclair et al. 2020

Table 14: Bottlenose dolphin demographic rate scenarios included in Population Viability Analysis (PVA), and their resulting mean population growth rates (λ).

Demographic Scenario	**Juvenile survival**	**Adult survival**	**Reproductive rate**	**λ (N=293)**	**λ (N=318)**
1	0.93	0.94	0.16	0.982	0.982
2	**0.93**	**0.94**	**0.22**	**0.992**	**0.991**
3	0.93	0.94	0.30	1.004	1.002
4	0.96	0.96	0.16	1.002	1.001
5	0.96	0.96	0.22	1.011	1.009
6	0.96	0.96	0.30	1.021	1.018

The bottlenose dolphin population in Cardigan Bay shows no trend, since over the 20 years of monitoring the population increased and then decreased, returning to a similar level to the start of the monitoring programme. The picture is complicated by the fact that only a portion of the population is being sampled and animals may migrate in and out of the study area from year to year. Based upon the parameters we have been able to measure, scenario 2, or possibly 3, may be closest to the true situation.

Table 15: Details of common dolphin population data and demographic rates

Parameter	**Selected Value**	**Justification/references**
MU abundance	102,656	2016 SCANS 3 & ObSERVE survey estimates (Rogan et al. 2018, Hammond et al. 2021, IAMMWG 2021)
Carrying capacity (K)	150,000	Unknown, so arbitrarily set at 1.5x the current estimate.
Sex ratio (M:F)	1:1	Default assumption
Female abundance	51,328	50% of population size
Calf survival	0.80	Modified for CIS MU from Saavedra 2017, given lower bycatch rates
Calf stage (years)	3	Mannocci et al. 2012
Juvenile survival	0.95	Modified for CIS MU from Saavedra 2017, given lower bycatch rates
Juvenile stage (years)	5	Saavedra 2017
Adult survival	0.95	Modified for CIS MU from Saavedra 2017, given lower bycatch rates
Age at first reproduction	9	Mannocci et al. 2012, Read 2016, Saavedra 2017
Reproductive rate	0.33	Murphy et al. 2019, Saavedra 2017

Table 16: Details of minke whale population data and demographic rates

Parameter	**Selected Value**	**Justification/references**
MU abundance	20,118	2016 SCANS 3 & ObSERVE survey estimates (Rogan et al. 2018, Hammond et al. 2021, IAMMWG 2021)
Carrying capacity (K)	30,000	Unknown, so arbitrarily set at 1.5x the current estimate.
Sex ratio (M:F)	1:1	Default assumption
Female abundance	10,059	50% of population size
Calf survival	0.70	Sinclair et al. 2020
Calf stage (years)	1	Sinclair et al. 2020
Juvenile survival	0.77	Sinclair et al. 2020
Juvenile stage (years)	8	Sinclair et al. 2020
Adult survival	0.96	Taylor et al. 2007
Age at first reproduction	9	Sinclair et al. 2020
Reproductive rate	0.90	Sinclair et al. 2020

Table 17: Details of grey seal population data and demographic rates necessary for use in the PVA. Four sets of analyses were run based on different estimates of abundance (N) from four different spatial scales of the management unit (MU).

Parameter	**Selected Value**	**Justification/references**
MU abundance	N_{Wales}= 5,300 N_{CIS} = 8,125 $N_{CIS+IRE}$ = 12,550 $N_{OSPAR\ 3}$= 64,854	O'Cadhla et al. 2013, Stringell et al. 2014, Morris & Duck 2019, Vincent et al. 2017, Culloch et al. 2018, SCOS 2020, 2021.
Carrying capacity (K)	K_{Wales}= 7,950 K_{CIS} = 12,000 $K_{CIS+IRE}$ = 16,000 $K_{OSPAR\ 3}$= 81,000	Unknown, so arbitrarily set at 1.5x of the current estimate for the first two options, and 1.25x the current estimate, for the last two, anticipating that these may be nearing carrying capacity
Sex ratio (M:F)	0.7:1	Thomas et al. 2019
Female abundance	N_{Wales}= 3,127 N_{CIS} = 4,794 $N_{CIS+IRE}$ = 7,404 $N_{OSPAR\ 3}$= 58,264	59% of population size given the sex ratio
Pup survival	0.40 0.50 0.60	Based upon range of estimates from SCOS 2018, Thomas et al. 2019
Pup stage (years)	1	SCOS 2020
Juvenile survival	0.90 0.93	Based upon range of estimates from Schwarz & Stobo 2000, SCOS 2018, Sinclair et al. 2020
Juvenile stage (years)	4	Sinclair et al. 2020
Adult survival	0.94	SCOS 2018
Age at first reproduction	5	Sinclair et al. 2020
Reproductive rate	0.84 0.90	Based upon range of estimates from Thomas et al. 2019, Sinclair et al. 2020

Table 18: Grey seal demographic rate scenarios included in PVA, and their resulting mean population growth rates (λ).

Demographic Scenario	**Pup survival**	**Juvenile survival**	**λ (8,125) N_{CIS}**	**λ (12,675) $N_{CIS+IRE}$**	**λ (5,300) N_{Wales}**	**λ (64,854) N_{OSPAR3}**
1	0.40	0.90	1.014	1.004	1.026	1.016
2	0.50	0.90	1.021	1.010	1.036	1.024
3	0.60	0.90	1.026	1.013	1.044	1.029
4	0.40	0.93	1.017	1.006	1.031	1.020
5	0.50	0.93	1.024	1.012	1.041	1.027
6	**0.60**	**0.93**	**1.028**	**1.015**	**1.048**	**1.032**

It is likely that none of the above scenarios precisely match that which the Welsh grey seal population is experiencing, where a steady increase and expansion has been observed at monitored colonies/areas. The nearest to the observed trend is scenario 6. However, pup survival is probably lower than 0.60 due to the high proportion of cryptic breeding sites across much of Wales (Stringell et al. 2014), which may contribute to increased calf mortality in poor weather. The population is almost certainly not a closed one, with potential immigration (as well as emigration) occurring.

3.2.3. Projections from PVA:

The demographic rate scenarios were projected for 10 years with density dependence on reproductive rate. Here, we allowed the reproductive rate to vary between the F_0 values stated in Tables 11-18 and a minimum of 0.1 at carrying capacity (F_K; equal to 0.05 in the female only models) in order to be able to test the influence of using different reproductive rates on population trends. This means that when the population is well below carrying capacity then reproductive rate will be as stated in the relevant table. As the population approaches carrying capacity reproductive rate will gradually be lowered to somewhere between the value in the relevant table and 0.1 at carrying capacity.

It should be noted, however, that values for carrying capacity are unknown, as this is a challenging parameter to establish. Where a time series exists, we have set carrying capacity above the highest estimate we have for abundance during the whole time period, otherwise we have used a carrying capacity of x1.5 the population estimate. Nevertheless, its inclusion produces more realistic projections. While a population can grow exponentially over a period of time, this is not realistic in the long-term. For bottlenose dolphin, harbour porpoise and grey seal, six scenarios were projected and their population growth rates recorded in Tables 12 (harbour porpoise), 14 (bottlenose dolphin), and 18 (grey seal), while just one scenario was run for common dolphin (Table 15) and minke whale (Table

17) due to the higher uncertainty in demographic parameters. We were not able to model Risso's dolphin since we did not have the necessary population parameters from the region.

The outcome for harbour porpoise (λ=0.93-1.07) and bottlenose dolphins (λ=0.98-1.02) ranged from decline to population increase (Figures. 7–8 respectively), with the most extreme trends recorded for harbour porpoise. This indicates that great care should be taken in managing these populations and any additional external pressures, particularly given the uncertainties that exist in demographic parameters. The projections for common dolphin and minke whale both showed population increase with mean population growth rates of 1.01 and 1.02, respectively (Figures. 9-10). For grey seals, five of the six scenarios resulted in a population close to stability regardless of the size of the area used (λ=1.00-1.03) (Figure 11).

Importantly, these are all baseline projections in the absence of additional pressures. Removals and impacts on specific demographic rates will be incorporated in Section 3.2.6 to generate comparative scenarios for assessments

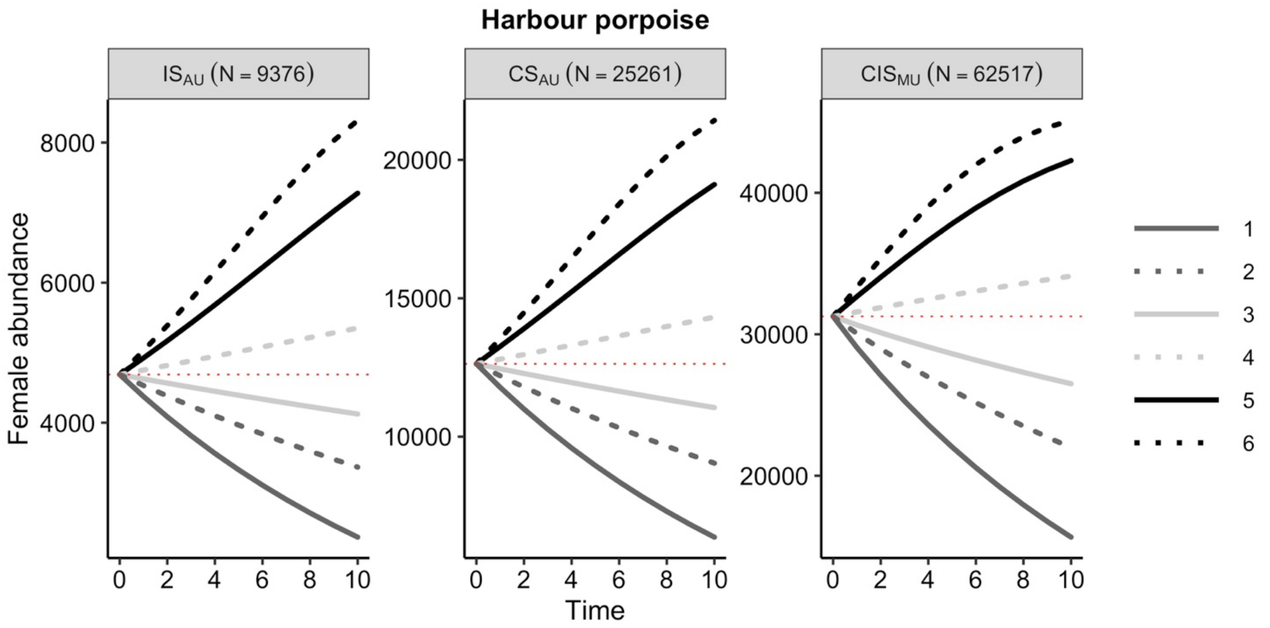

Figure 7: PVA projections of population trends for female harbour porpoise. The lines represent each of the six demographic rate scenarios listed in Table 12. The red dotted line denotes the female starting population.

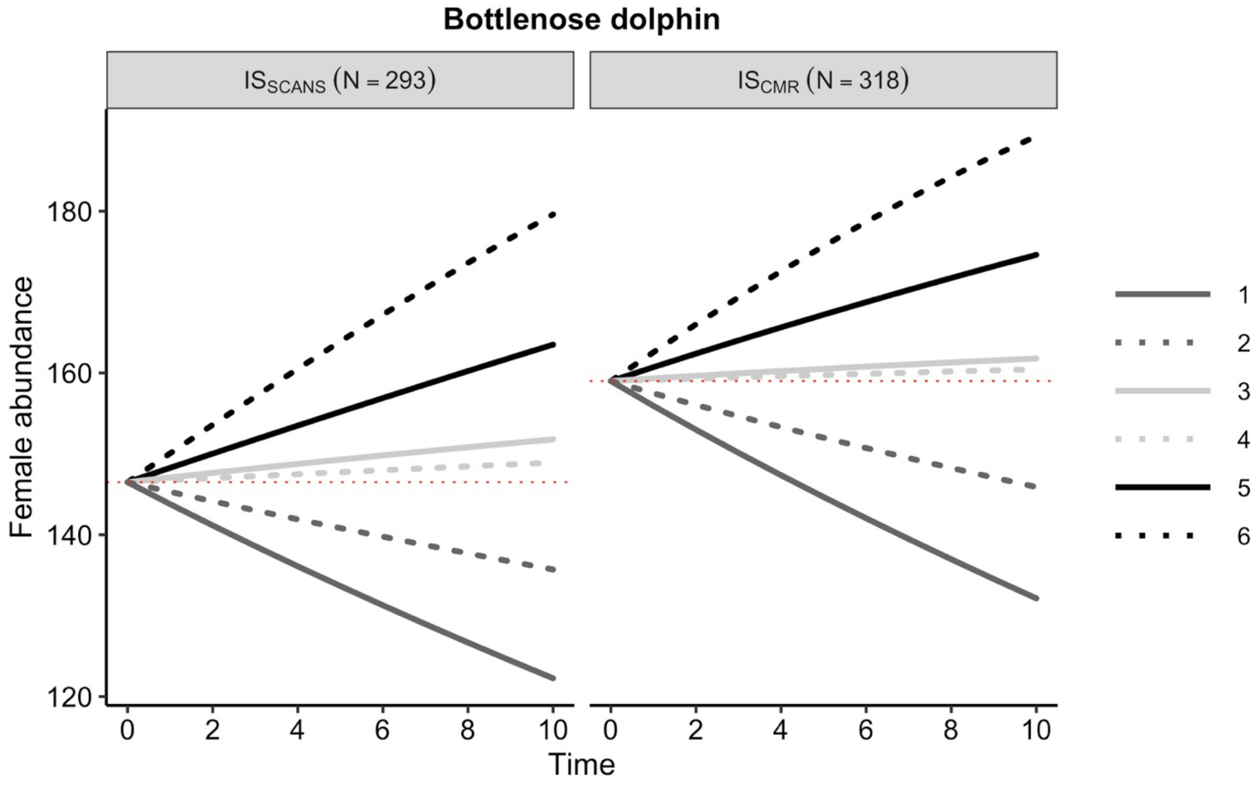

Figure 8: PVA projections of population trends for female bottlenose dolphins using both CMR and SCANS estimates of abundance. The lines represent each of the six demographic rate scenarios listed in Table 14. The horizontal red dotted line denotes the female starting population.

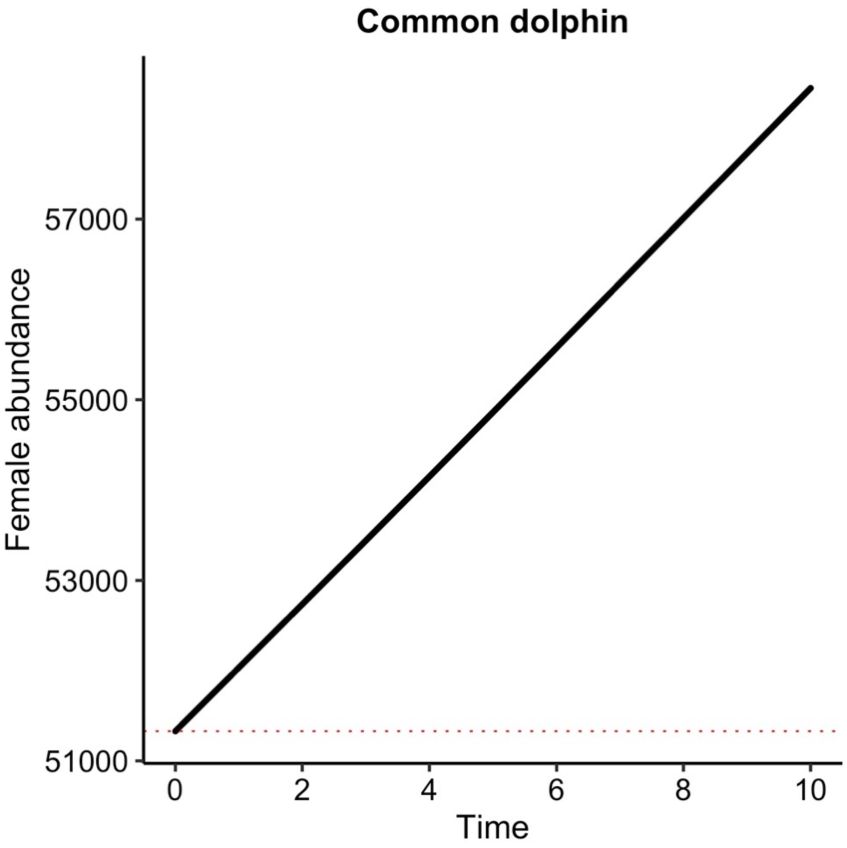

Figure 9: PVA projection of population trend for female common dolphins. The horizontal red dotted line denotes the female starting population. Demographic rates are listed in Table 15.

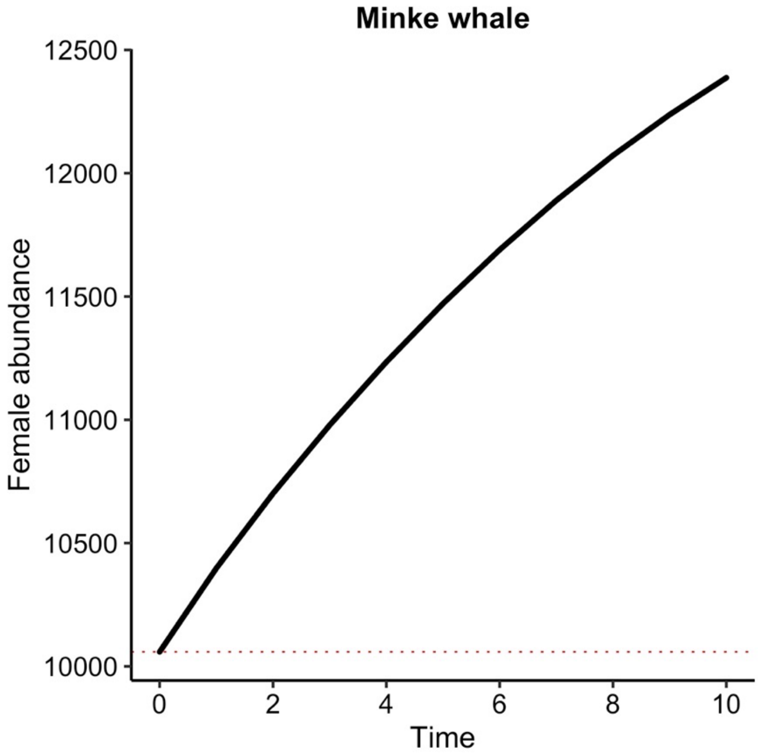

Figure 10: PVA projection of population trend for female minke whales. The horizontal red dotted line denotes the female starting population. Demographic rates are listed in Table 16.

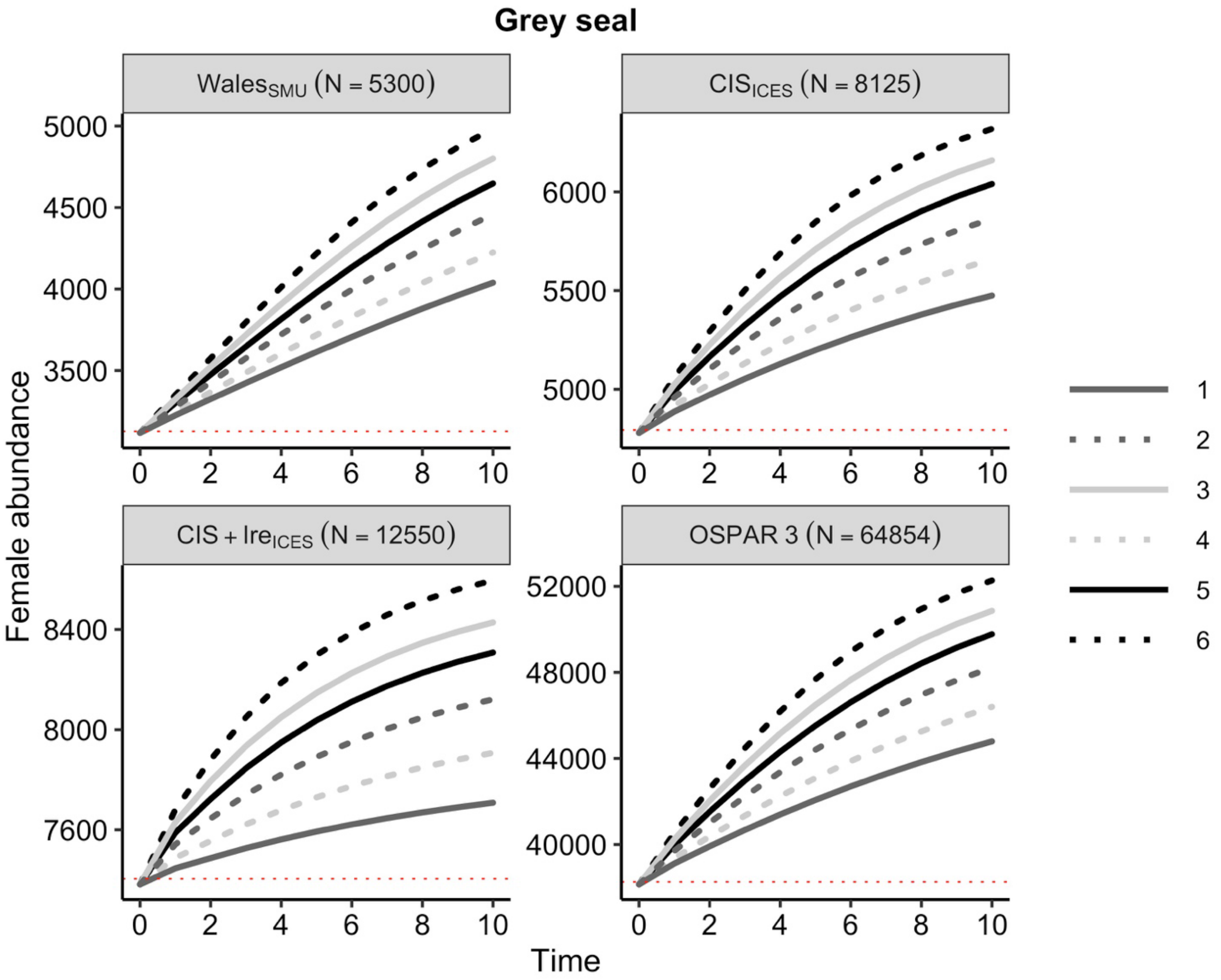

Figure 11: PVA projections of population trends for female grey seals in four alternative proposed management units. The lines represent each of the six demographic rate scenarios that are listed in Table 18. The horizontal red dotted line denotes the female starting population.

3.2.4. Demographic stochasticity:

The PVAs presented so far have been deterministic. However, natural populations are unlikely to behave in such a way. We have therefore projected the demographic rate scenarios to include demographic stochasticity (Figures 12-16). To do this, we ran 100 simulations using the ‘`multiresultm`’ function within the `popbio` package (Stubben & Milligan 2007), which generates multinomial random numbers for state transitions and binomial random numbers for fertilities. Essentially the transition probabilities (G and P) and reproductive rates (F) are treated as random events. Instead of matrix multiplication happening across an entire stage (i.e. calf, juvenile or adult) the probabilities are applied to each individual within each stage. Therefore, the result will vary each time, but not because the demographic rate is different, simply because of the randomness involved in working with probabilities. Demographic stochasticity is therefore clearly more of a problem for small populations. This is also evident in our analysis where stochasticity had a much larger impact on the bottlenose dolphin population trend across all six demographic rate scenarios compared to any of the other species which all have significantly larger population sizes. These demographic stochasticity analyses depicted in Figure 12-18 do not include density dependent reproductive rate and therefore trends may differ slightly from the PVA projections of Figures 7-11.

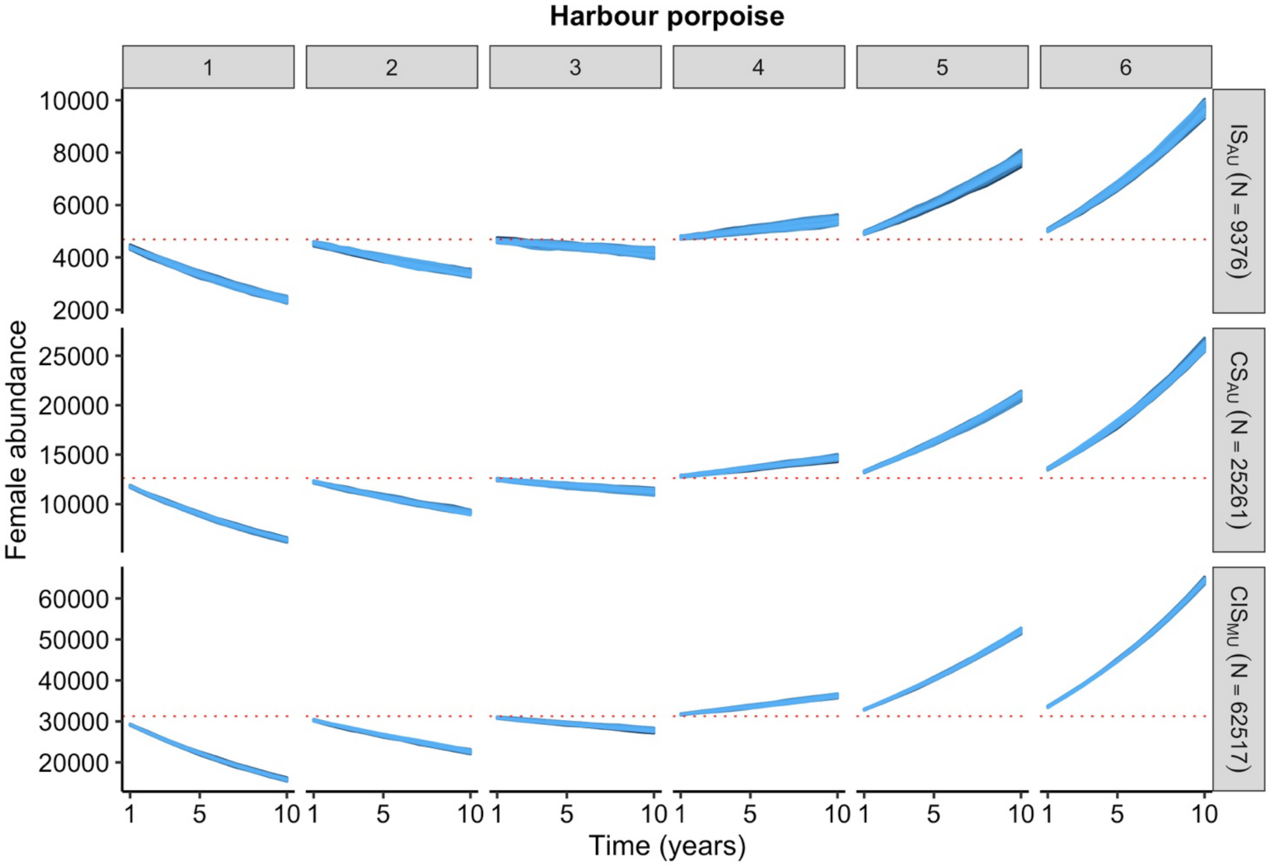

Figure 12: Demographic stochasticity in population trends for female harbour porpoise at the three spatial scales. Stochasticity was modelled using 100 simulations (blue lines) for each of the six demographic rate scenarios (see Table 12).

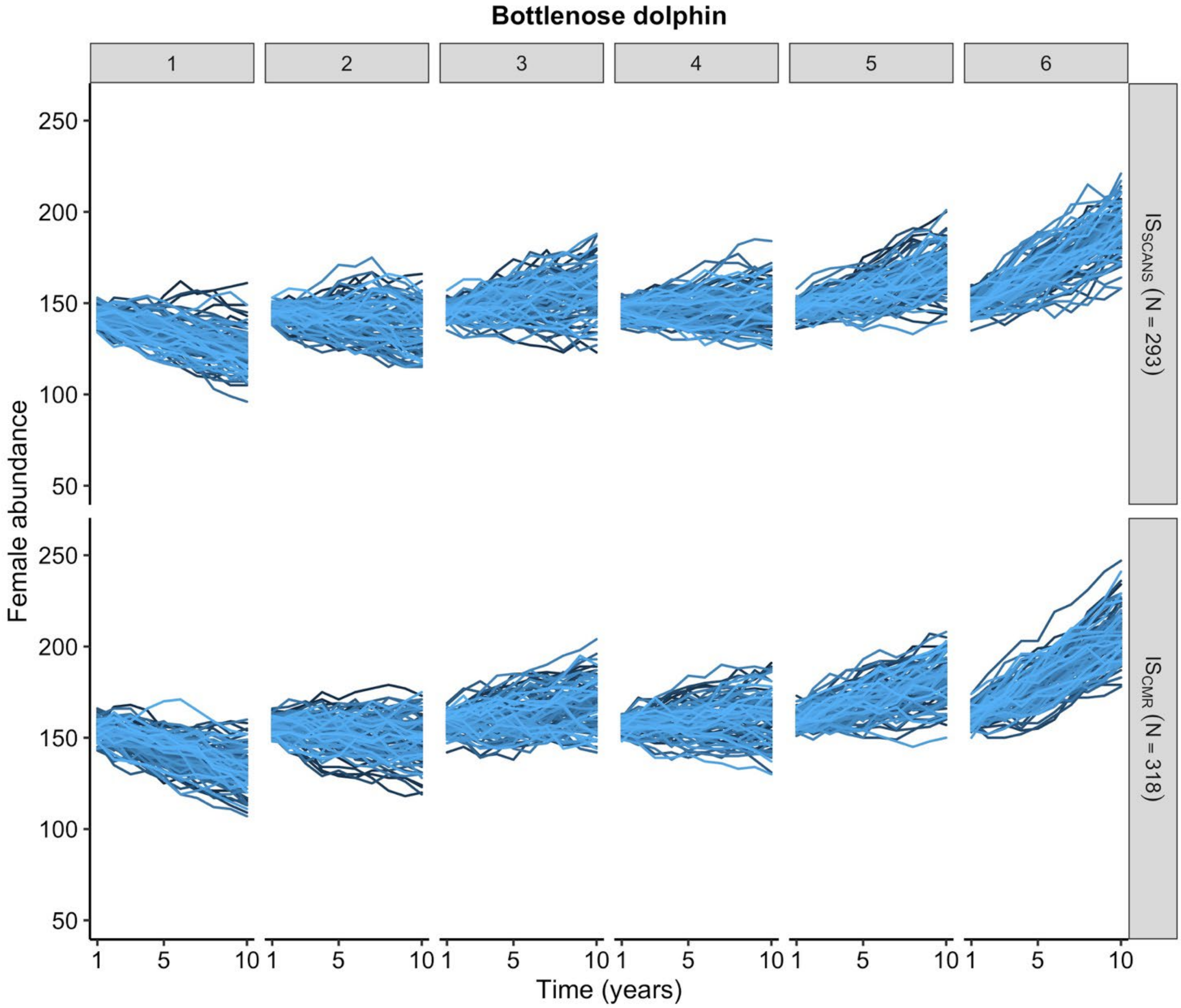

Figure 13: Demographic stochasticity in population trends for female bottlenose dolphins using both estimates of abundance. Stochasticity was modelled using 100 simulations (blue lines) for each of the six PVA demographic rate scenarios (see Table. 14).

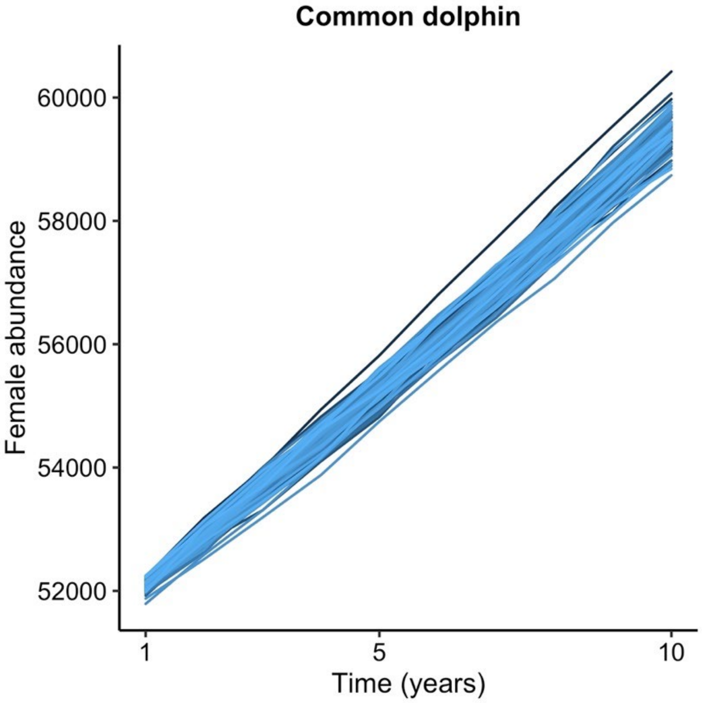

Figure 14: Demographic stochasticity in population trends for female common dolphins. Stochasticity was modelled using 100 simulations. Demographic rates are listed in Table 15.

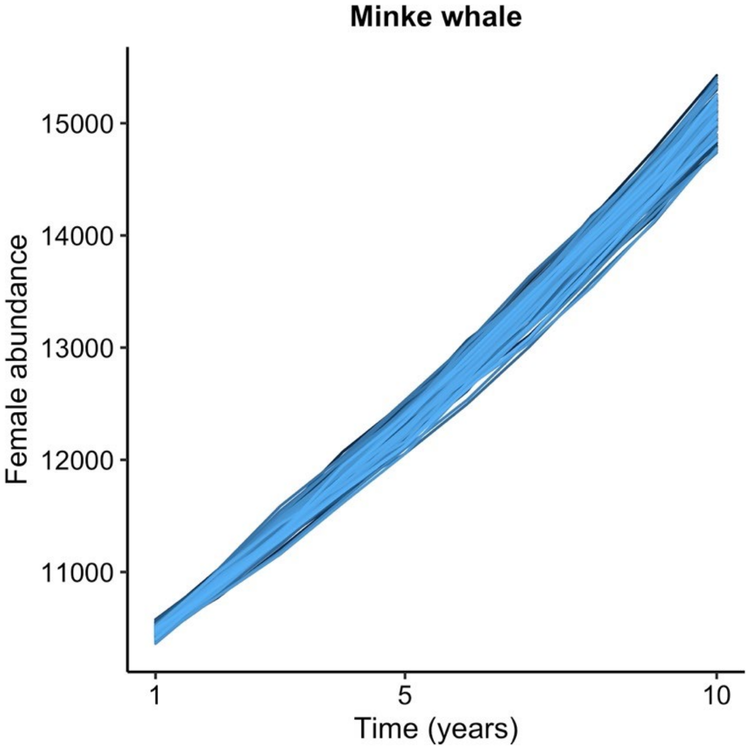

Figure 15: Demographic stochasticity in population trends for female minke whales. Stochasticity was modelled using 100 simulations. Demographic rates are listed in Table 16.

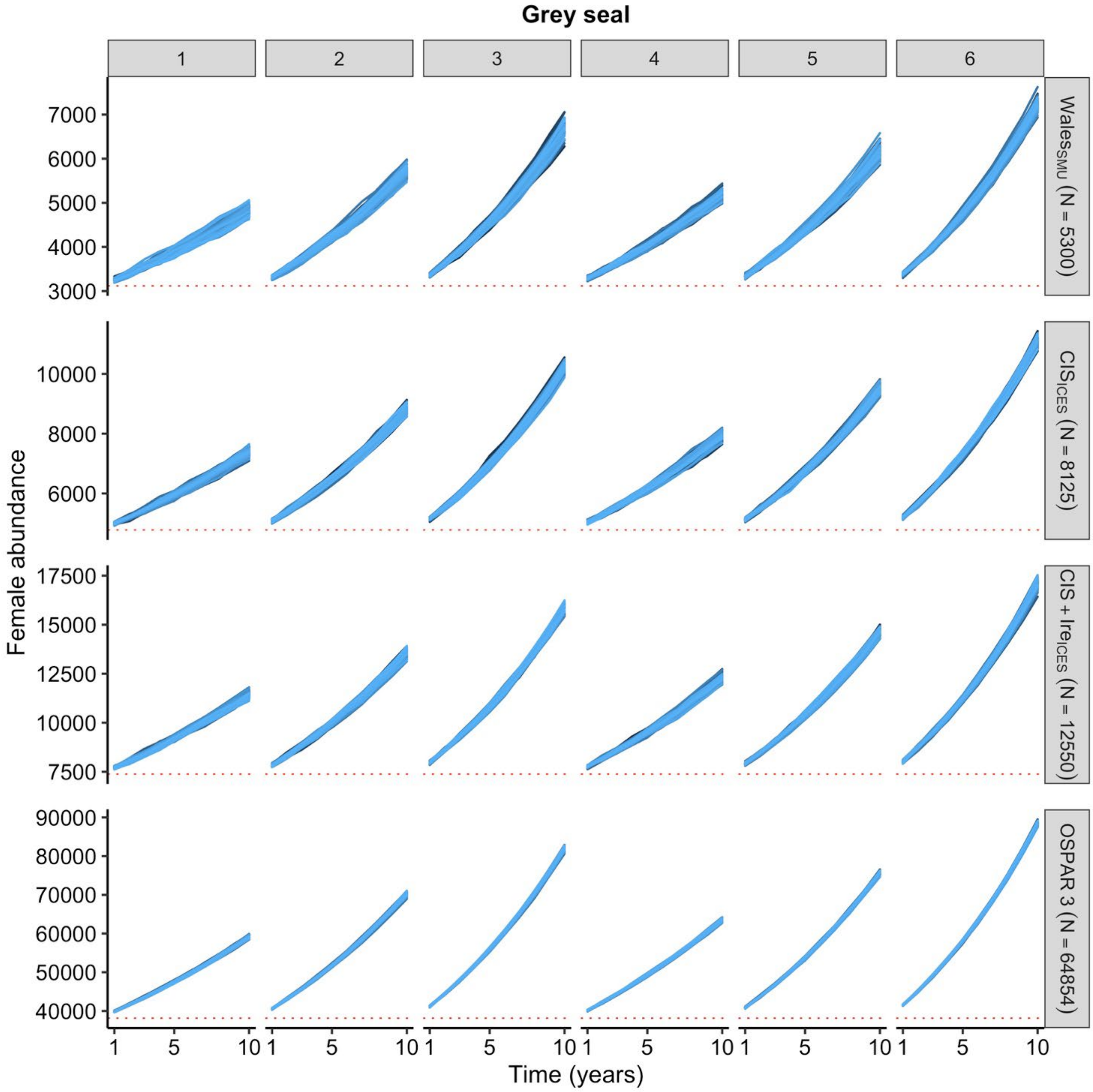

Figure 16: Demographic stochasticity in population trends for female grey seals in four alternative proposed management units. Stochasticity was modelled using 100 simulations (blue lines) for each of the six demographic rate scenarios (see Table 18).

3.2.5. Perturbation (sensitivity) analysis:

Sensitivity analysis models linear relationships between perturbations and population growth. This may perform reasonably well when small perturbations are involved. However, for larger perturbations these become tangents to more complicated, non-linear relationships. Sensitivity analysis can therefore underestimate the amount of change required to reverse long-term population decline (Stott et al. 2012). Perturbation analysis, or transfer function analysis, on the other hand is a tool that models the non-linear relationship between perturbations and population growth rate by changing individual demographic parameters. Perturbation analysis was performed using the transfer function `tfa_lambda` of asymptotic growth from the `popdemo` package (Stott et al. 2012).

The transfer function analysis required three parameters, namely a population matrix, a perturbation structure which is given by two vectors describing the rows and columns within the population matrix, and a range of perturbation magnitude which is the range of values that describe how much to change the individual demographic rates by. The demographic rates involved are those described in Figure 6 and matrix A. This includes the probability of reproducing in year *t* (F) which here does not include density dependence, the probability of surviving year *t* and remaining in the same stage in year *t+1* (P), and the probability of surviving year *t* and transitioning to the next stage in year *t+1* (G), which are specific to the three life cycle stages, calf/pup (c/p), juvenile (j) and adult (a), included in our models. Therefore, G(j,a) refers to the probability that a juvenile (j) survives year *t* and transitions to an adult (a) in year *t+1*. Note that harbour porpoise and grey seal have five states (there is no transition from calf to juvenile as they reach the latter state within year 1) whereas bottlenose dolphin, common dolphin and minke whale have the full six states.

Perturbation magnitude is the amount of change (positive or negative) in the individual demographic rates. For example, if calf survival is 0.60 and the chosen perturbation magnitude ranged from -0.1 to 0.1, this would show the consequence on population growth by varying calf survival between 0.50-0.70. Here, we assigned realistic perturbation magnitudes to each individual demographic parameter. For example, a very large perturbation to adult survival would not be sensible as adult survival tends to be fairly robust in long-lived species. Similarly, we avoided perturbations that would exceed or reach the bounds of 0 and 1 for the individual demographic rates, which meant typically including only small positive perturbations to juvenile and adult survival, and for bottlenose dolphins only relatively small negative perturbations to reproductive rate.

Figures 17-21 indicate the growth rate resulting from perturbations applied to the different demographic rates. Where the lines cross the vertical line at perturbation magnitude=0, indicates the original population growth rate with no perturbations applied. Steeper lines highlight the demographic rates that have the greatest impact on population growth and therefore the ones that would be most beneficial to target (if possible) from a management perspective. For example, if a population was most sensitive to change in juvenile and adult survival, then targeting management of bycatch of these two stage classes would result in the biggest impact on population growth. Shallower lines indicate the demographic rates that have the least impact on the population growth rate. Any scenario entirely located above the horizontal line where λ=1 indicates that the population can withstand some negative perturbation (at least within the predefined range for that particular demographic rate) without seeing a population decline. In contrast, any scenario entirely located below the horizontal line of λ=1 will show continued population decline despite any positive perturbations (e.g. the results of any targeted management measures).

Generally, population growth rates across all species were by far the most sensitive to changes in adult survival (P(a,a)), and the least sensitive to changes in calf/pup survival (P(c,c) and G(c,j)). Population growth rates were relatively sensitive to changes in reproductive rate (F(a,c)), as well as the probability that a juvenile would survive and transition to an adult (G(j,a)).

Harbour Porpoise

For harbour porpoise (Figure 17), there is much more variation in the resulting population growth rates from perturbations across the scenarios compared to the other species, from population growth to significant decline. Great care should there be taken regarding the management decisions for harbour porpoise especially considering the uncertainty surrounding demographic rates and factors effecting the population (and their magnitude). Demographic rate scenarios 5 and 6 could withstand the perturbation magnitude included without causing a decline, whereas for demographic rate scenarios 1 and 2, no positive perturbation would result in population increase (Table 20). Changes in adult survival clearly have the largest effect on population growth rates.

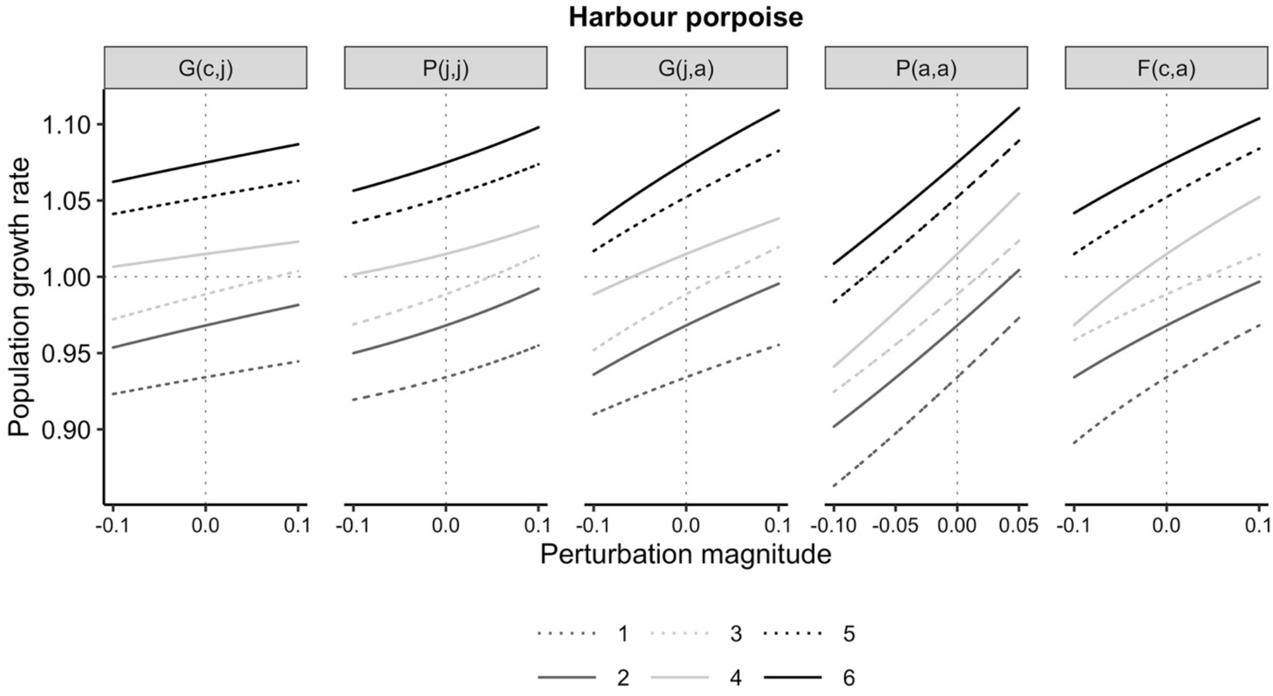

Figure 17: Perturbation analysis results for each of the six harbour porpoise CIS MU demographic rate scenarios (grey scale solid and dotted lines) showing impact of perturbation on the PVA model parameters. G(c,j) is the probability of surviving and transitioning from calf to juvenile, P(j,j) is changes in juvenile survival, G(j,a) is the probability of surviving and transitioning from juvenile to adult, P(a,a) is changes in adult survival, and F(c,a) is changes in reproductive rate.

Bottlenose dolphin

Across the demographic rate scenarios, only a small change to adult survival of bottlenose dolphins would result in significant change to the population growth rate (steepest lines) (Figure 18). Adult survival is therefore the most effective demographic parameter to target from a management perspective. Perturbations, particularly negative perturbations, to reproductive rate (F(c,a)) and the probability of juveniles surviving and transitioning to adults (G(j,a)) also have relatively large impacts on population growth rate. For demographic rate scenario 1 (and 2 to some extent), any positive perturbation to calf survival (P(c,c)) and transition rates (G(c,j)) as well as juvenile survival (P(j,j)) and transition rates (G(j,a)) did not result in population increase. Demographic rate scenarios 5 and 6 could withstand some negative perturbations without the population declining, except when this involved adult survival (P(a,a)), and the probability of juveniles transitioning to adults (G(j,a)).

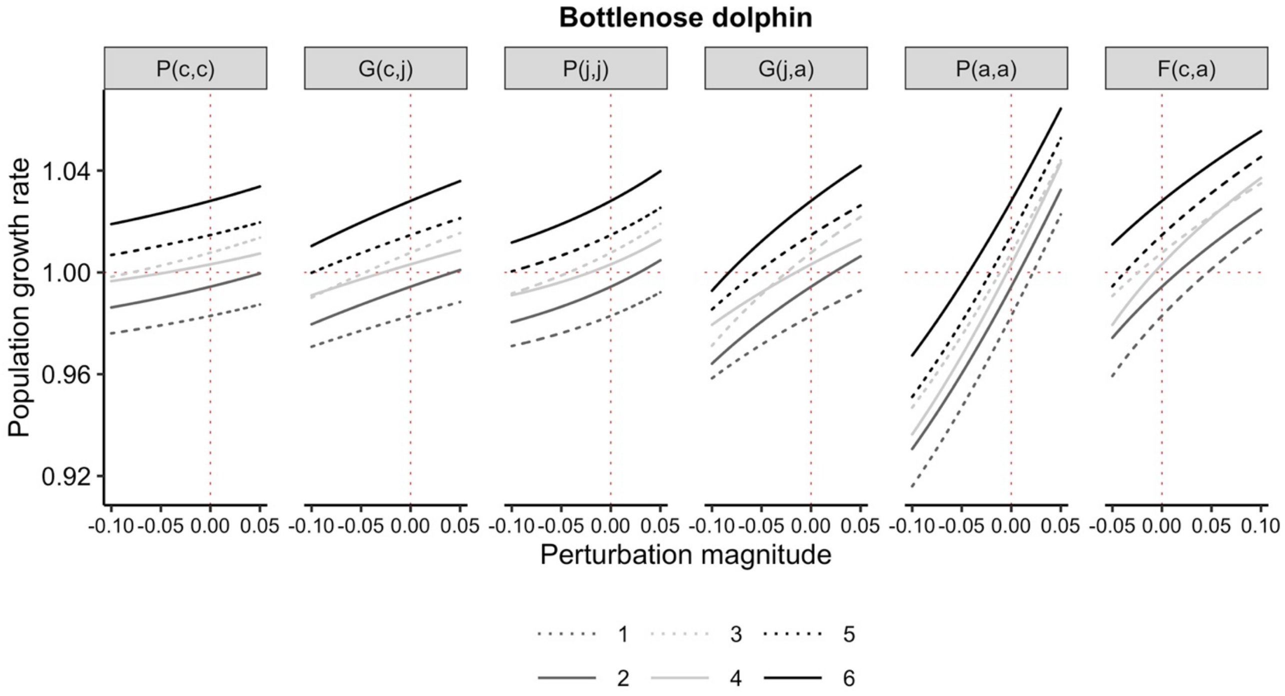

Figure 18: Perturbation analysis results for each of the six bottlenose dolphin demographic rate scenarios (grey scale solid and dotted lines) showing impact of perturbation on the PVA model parameters. The scale on the x-axis differs between plots.

Common dolphin

For common dolphins, the population growth rate was by far most sensitive to changes in adult survival (P(a,a)), and least sensitive to changes in calf survival (P(c,c)) (Figure 19).

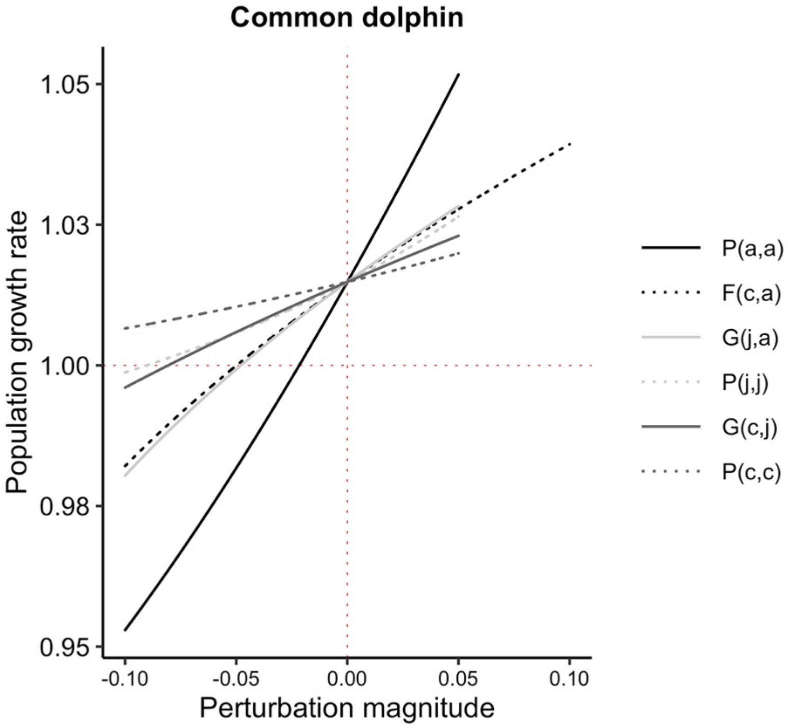

Figure 19: Perturbation analysis results for common dolphins showing impact of perturbation on the PVA model parameters.

Minke whale

For minke whales, the population was by far the most sensitive to both changes in adult survival (P(a,a)), and the probability that juveniles survive and transition to adults (G(j,a)) (Figure 20). Perturbations on calf survival (P(c,c)) and transition (G(c,j)), reproductive rate (F(c,a)) and juvenile survival (P(j,j)) didn't have much of an influence on minke whale population growth rate.

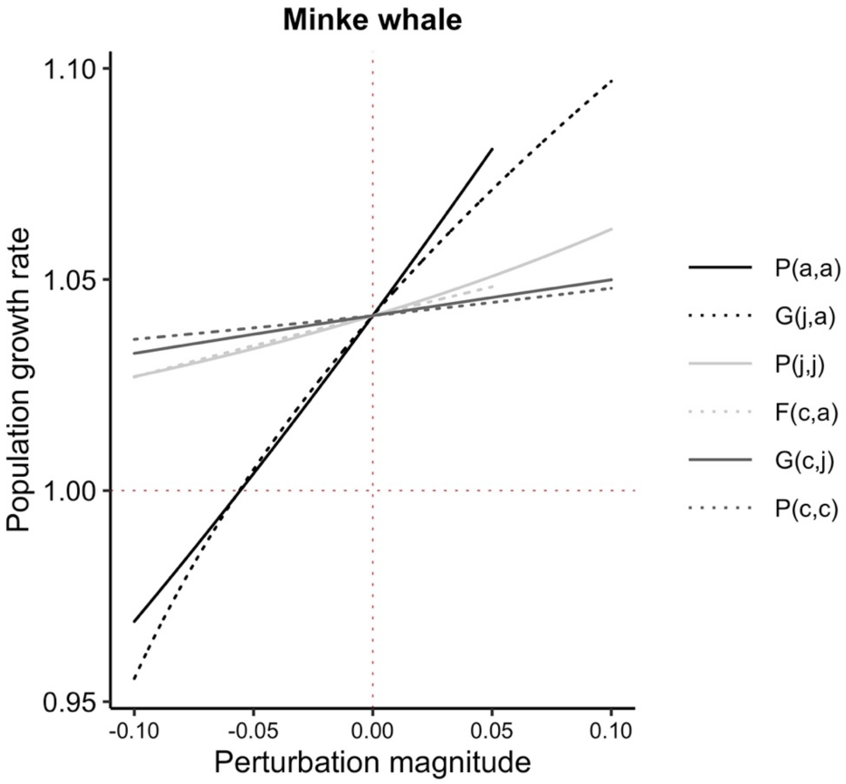

Figure 20: Perturbation analysis results for minke whales showing impact of perturbation on the PVA model parameters.

Grey Seal

Across the current scenarios (Figure 21), the grey seal population could withstand quite significant negative perturbations (up to nearly 0.1 change) to all demographic rates, except adult survival (P(a,a)), where this was only the case for demographic scenarios 3, 5, and 6. The population of grey seals is most sensitive to changes in adult survival, P(a,a), and the probability that a juvenile survives and transitions to an adult, G(j,a).

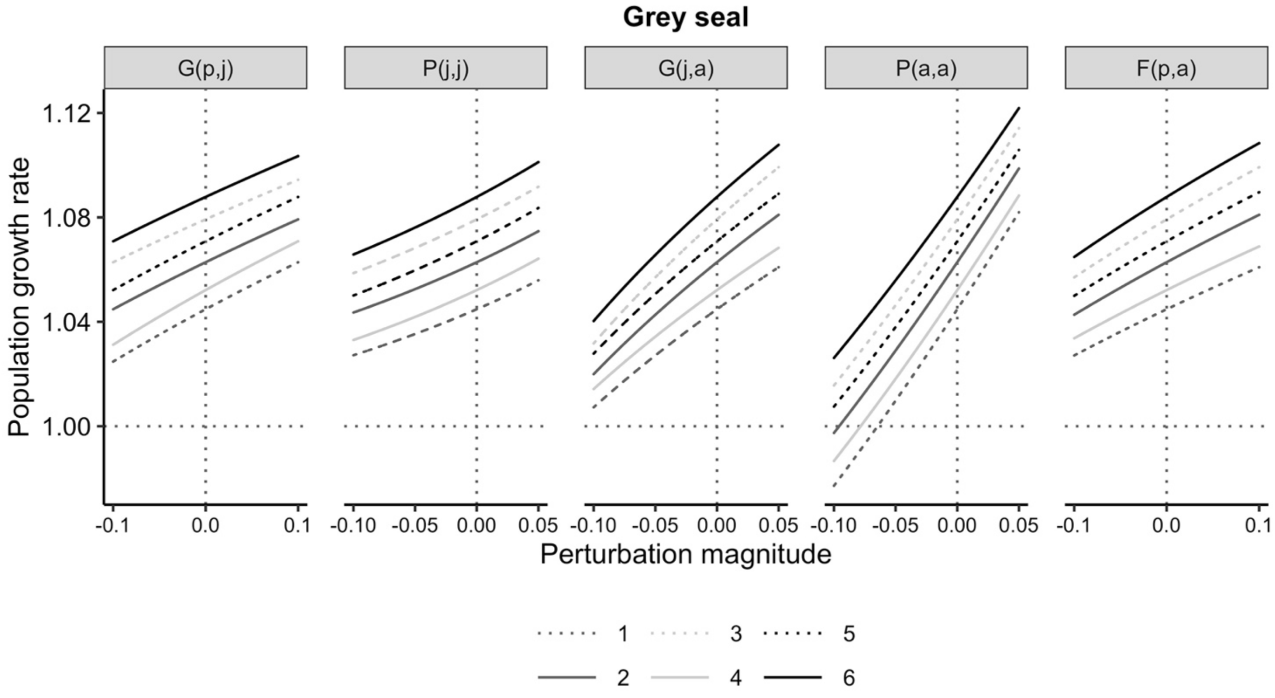

Figure 21: Perturbation analysis results for each of the six grey seal demographic rate scenarios (grey scale solid and dotted lines) showing impact of perturbation on the PVA model parameters.

3.2.6. PVA Impact Scenarios:

Population viability analysis was used to explore the impact of known or predicted key pressures (as described in section 2) to the different species over a period of 10 years. This involved the impact of contaminants on reproductive rate for all species, except minke whales, vessel strike from recreational vessels on bottlenose dolphins and bycatch of harbour porpoise, grey seals and common dolphins. The impact on contaminants on reproductive rate is not yet known for these species. Reproductive females are able to unload some of their contaminant burden to their offspring through milk, which for grey seals and harbour porpoise could occur annually or every other year, but for bottlenose and common dolphins less regularly due to the longer inter-birth interval. Contaminants may therefore have the biggest impact on their first reproduction. However, as this information is not currently possibly to fully quantify, we modelled the impact of contaminants on reproductive rate as a 10% reduction. This suggested reduction in reproductive rate is an arbitrary value as we lack the information to set a more precise value. When more information becomes available, the models can easily be modified to include this. Bottlenose dolphins rarely wash up on the shore so that sample sizes of causes of death are low. Over a ten-year period, three were recorded along the Welsh coast as having died by physical trauma, one of these acute. There is other evidence indicating interactions with high-speed vessels in the region (see section 2.2.5) and so we felt it prudent to include this as a potential source of removals estimated at 0.33 per year as a worst-case scenario.

In order to further interrogate the PBR values (see section 3.1), we also included scenarios removing the number of individuals defined by PBR analysis for all species. For bottlenose dolphin, harbour porpoise and grey seal we also included the number of removals from the

MU thought to result in Adverse Effect of Site Integrity (AEOSI) (largely in relation to potential mortality associated with marine developments) as described in NRW's position statement (NRW 2020). As this only included a few individuals per year, we applied this pressure to juveniles only, as these are more naïve. In the absence of detailed information relating to specific areas, we used the same values for AEOSI and contaminants across all spatial scales. Lastly, we included a scenario of cumulative impacts involving all those pressures described above, except PBR. See Tables 19, 21, and 23 for more details on the species-specific scenarios.

Harbour porpoise

For harbour porpoise, bycatch has the biggest impact on the population, which can only withstand current estimated bycatch levels under demographic rate scenarios 5-6 (Figure 22. In contrast, the potential impact of marine development had the smallest impact on population growth. The population can only withstand the consequences of contaminants and removals suggested from the PBR analysis under demographic rate scenarios 4-6. In terms of the cumulative impacts of bycatch and contaminants, the population can only tolerate these under demographic rate scenarios 5-6 as a result of the combination of higher calf survival and higher reproductive rates. For all other demographic rate scenarios, these impacts result in population decline.

Table 19: Description of impact scenarios for harbour porpoise. All numbers of removals provided are subsequently halved to represent females assuming a 50:50 sex ratio.

Scenario	**Description**
Contaminants	Reduce reproductive rate by 10% (see text on page 90)
Bycatch	For the Irish Sea MU, remove 12 per year; for the Celtic Sea, remove 738 per year; and for CIS MU, remove c. 1,000 individuals per year. This was achieved by selecting a random number each year within the range given whereby 60% of these individuals were juveniles and 40% were adults.
PBR	Remove 72 individuals per year for the Irish Sea MU, 168 individuals per year for the Celtic Sea, and 561 individuals per year for the CIS MU. 60% of these individuals will be juveniles and 40% adults.
Marine developments	Removal of <5 (4 animals chosen) juveniles per year. The suggested maximum removal threshold described in NRW's position statement on AEOSI.
Combined (excl. PBR)	Reduce reproductive rate by 10%, remove between 700-1500 juveniles and adults per year as a result of bycatch, and remove a further 4 juveniles per year as a result of interaction with marine developments.

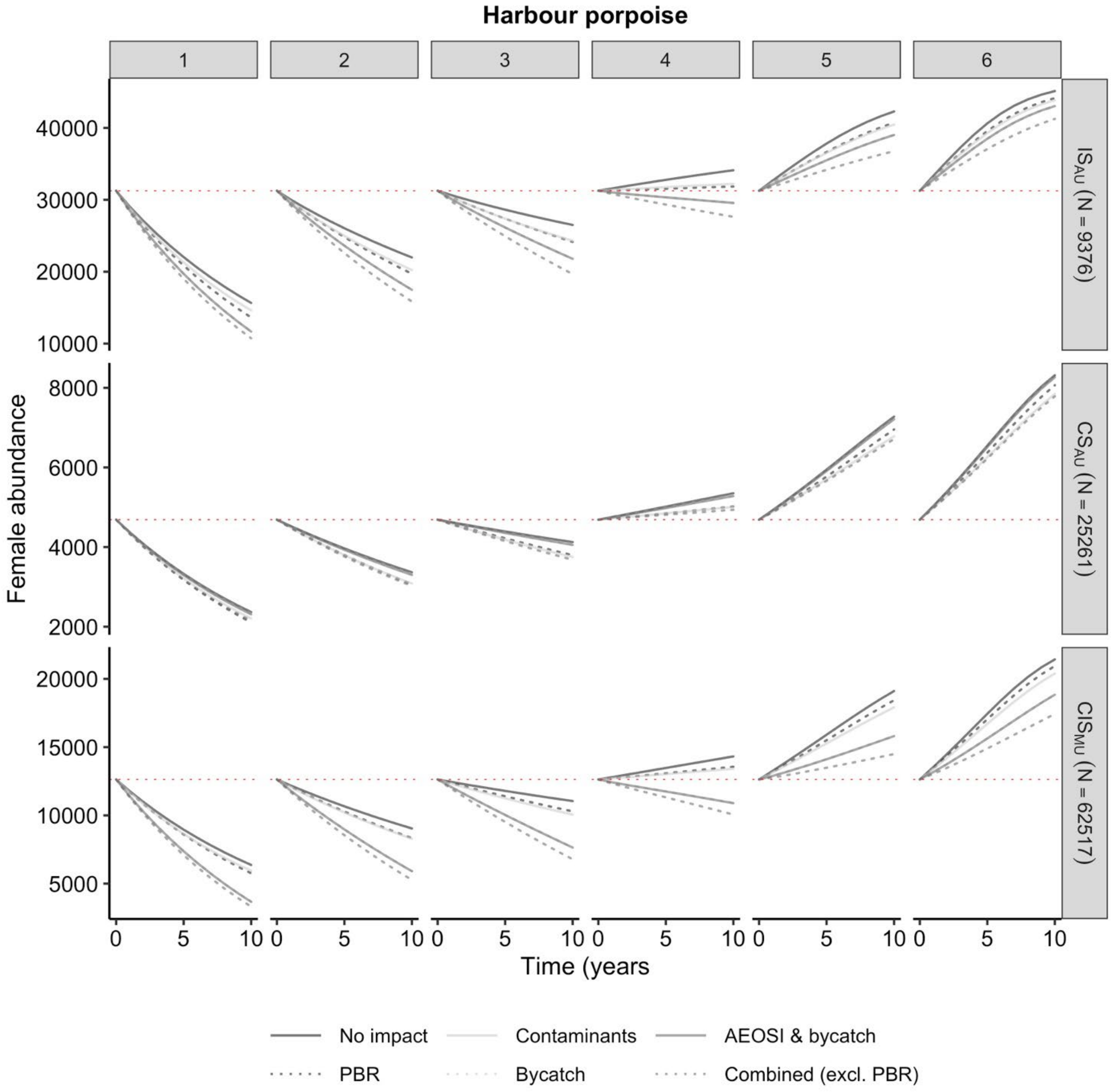

Figure 22: Projected population trends for each impact scenario across each of the six demographic rate scenarios (see Table 19) for female harbour porpoise at the three spatial scales.

Table 20: Population growth rates for harbour porpoise for six different demographic rate scenarios (1-6) (see Table 12) under six different impact scenarios, at three different spatial scales. Values > 1.0 indicate an increase and < 1.0 indicate a decline.

N/spatial area	**Impact scenario**	**1**	**2**	**3**	**4**	**5**	**6**
62,517/CIS_{MU}	No impact	0.933	0.965	0.984	1.009	1.031	1.038
62,517/CIS_{MU}	AEOSI & bycatch	0.906	0.944	0.965	0.994	1.022	1.033
62,517/CIS_{MU}	Contaminants	0.927	0.957	0.975	1.003	1.026	1.035
62,517/CIS_{MU}	PBR	0.920	0.955	0.975	1.002	1.027	1.035
62,517/CIS_{MU}	Bycatch	0.906	0.944	0.965	0.994	1.022	1.033
62,517/CIS_{MU}	Combined	0.899	0.934	0.955	0.988	1.016	1.028
9,376/IS_{AU}	No impact	0.934	0.967	0.987	1.013	1.045	1.059
9,376/IS_{AU}	AEOSI & bycatch	0.932	0.966	0.986	1.012	1.044	1.058
9,376/IS_{AU}	Contaminants	0.927	0.959	0.978	1.007	1.038	1.053
9,376/IS_{AU}	PBR	0.923	0.958	0.979	1.007	1.040	1.056
9,376/IS_{AU}	Bycatch	0.932	0.966	0.986	1.012	1.044	1.059
9,376/IS_{AU}	Combined	0.925	0.957	0.976	1.005	1.037	1.052
25,261/CS_{AU}	No impact	0.934	0.967	0.987	1.013	1.042	1.054
25,261/CS_{AU}	AEOSI & bycatch	0.884	0.927	0.951	0.985	1.023	1.041
25,261/CS_{AU}	Contaminants	0.927	0.959	0.978	1.006	1.036	1.049
25,261/CS_{AU}	PBR	0.925	0.959	0.980	1.007	1.039	1.052
25,261/CS_{AU}	Bycatch	0.884	0.927	0.951	0.985	1.023	1.041
25,261/CS_{AU}	Combined	0.875	0.916	0.940	0.978	1.014	1.033

Bottlenose dolphin

For bottlenose dolphins, using the CMR and IAMMWG (SCANS/ObSERVE) population estimates did not result in a significant difference in terms of the impact of the different pressures. Only demographic rate scenarios 5 and 6 with the higher juvenile and adult survival rates and the two highest reproductive rates could withstand all impact scenarios including the cumulative impacts (Figure 23). Demographic rate scenarios 1 and 2 resulted in (intensified) decline with even just one of these pressures applied to the population, whereas the outcome for scenarios 3 and 4 being more mixed depending on the nature of the impact. Across all scenarios, contaminants have the biggest impact on population growth rates. The bottlenose dolphin population is clearly quite sensitive to even relatively small pressures compared to the other species, e.g. declines noted by removing less than one individual per year, most likely due to the small population size. It is therefore particularly important to be aware of the cumulative pressures this population is likely facing, and especially any removals of juveniles or adults and any reductions in fecundity which is already low due to the multiyear weaning period. We have assigned removals to the juvenile stage since that is where greater mortality occurs. The current population trends suggest that demographic rate scenario 3 may be the most realistic.

Table 21: Description of impact scenarios for bottlenose dolphin. All numbers of removals provided are subsequently halved to represent females assuming a 50:50 sex ratio.

Scenario	**Description**
Contaminants	Reduce reproductive rate by 10% (see text on page 89)
Vessel strike	Remove 3 juveniles over 10 years, equivalent to 0.3 juveniles per year
PBR	Remove 0.8 adult per year for N=293, and 0.9 adults per year for N=318
Marine developments	Remove 2 juveniles over 3 years, equivalent to 0.67 individuals per year. The suggested maximum removal threshold based on a PBR of 0.7 described in NRW’s position statement on AEOSI
Combined (excl. PBR)	Reduce reproductive rate by 10%, remove 0.3 juveniles per year from vessel strike, and another 0.67 juveniles from marine developments

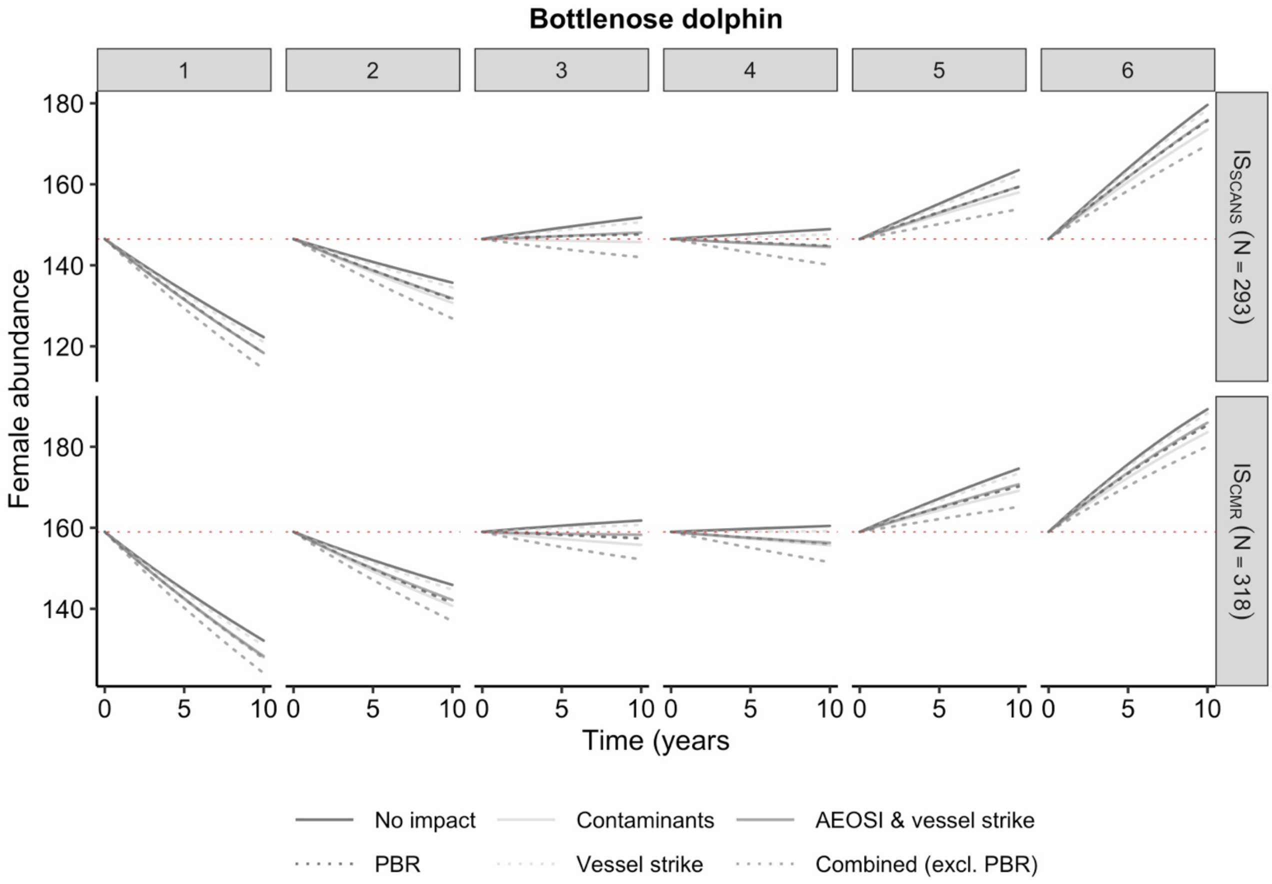

Figure 23: Projected population trends for each impact scenario across demographic rate scenarios 1-6 (see Table 14) for female bottlenose dolphin at two different population sizes in the Irish Sea MU.

Table 22: Population growth rates for bottlenose dolphins at two different abundance estimates in the Irish Sea MU for six different demographic rate scenarios (1-6) (see Table 14) under six different impact scenarios. Values above 1.0 indicate and increase and below 1.0 indicate a decline.

N	**Impact scenario**	**1**	**2**	**3**	**4**	**5**	**6**
293	No impact	0.982	0.992	1.004	1.002	1.011	1.021
293	Vessel strike	0.981	0.992	1.003	1.001	1.010	1.020
293	Marine developments	0.980	0.990	1.002	1.000	1.009	1.019
293	PBR	0.979	0.989	1.001	0.999	1.008	1.018
293	Contaminants	0.978	0.989	0.999	0.999	1.008	1.017
293	Combined	0.976	0.986	0.997	0.996	1.005	1.015
318	No impact	0.982	0.991	1.002	1.001	1.009	1.018
318	Vessel strike	0.981	0.991	1.001	1.000	1.009	1.017
318	Marine developments	0.980	0.990	1.000	0.999	1.008	1.016
318	PBR	0.978	0.988	0.999	0.998	1.007	1.015
318	Contaminants	0.979	0.988	0.998	0.998	1.006	1.014
318	Combined	0.976	0.985	0.996	0.995	1.004	1.013

Common dolphin

Since for common dolphin, we have just one population abundance and a single demographic rate scenario, the findings are not tabulated but presented here in Figure 24. For the area under consideration, we have used a removal rate of 720 (360 females) per year attributable to bycatch, believed to be by far the greatest source of mortality (ICES WGBYC 2020). Note that this value would need to be increased substantially if the area was extended to include the Bay of Biscay, along with a different abundance estimate. Removals from other pressures could not be assessed but are believed to be low, although contaminants may affect reproductive rates, and so a 10% reduction in reproductive rates was applied. The PBR estimated above was 808 (404 female) common dolphin.

For common dolphins, none of the impact scenarios resulted in population decline. Bycatch caused the largest drop in population growth (λ =1.007) followed by PBR ($\lambda = 1.006$), with contaminants ($\lambda = 1.009$) having the lowest impact of them all (Figure 24). For the combined impact scenario $\lambda = 1.002$.

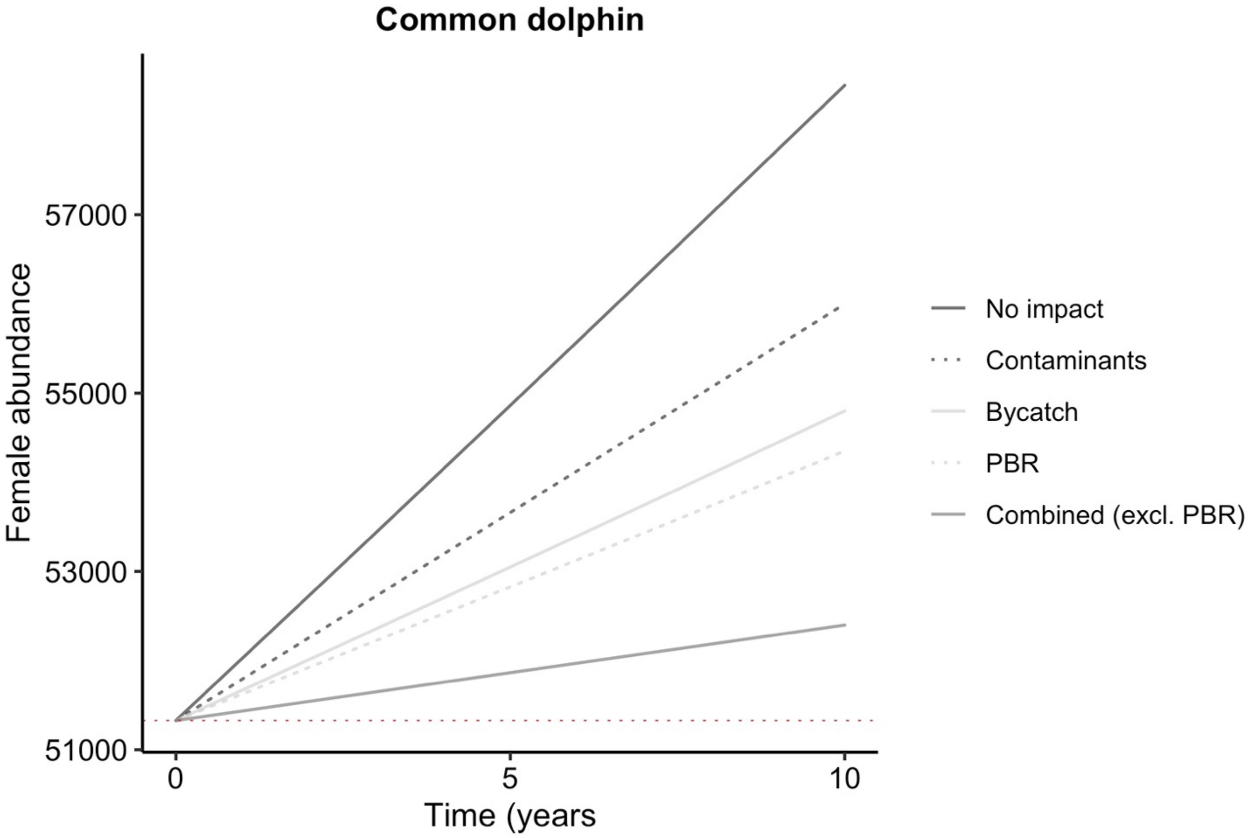

Figure 24: Projected population trends for no impact and each impact scenario (solid and dotted grey scale lines) for female common dolphin.

Minke whale

As with the previous species, we have just one population abundance and a single demographic rate scenario, and so the findings are not tabulated but presented here in Figure 25. At present, too little information exists to be able to recommend levels for any anthropogenic removals in minke whales, although they do suffer bycatch and also can be victims of ship strike. The PBR value used for minke whales was 173 (86.5 females).

For minke whales, PBR resulted in a $\lambda = 1.017$ (Figure 25).

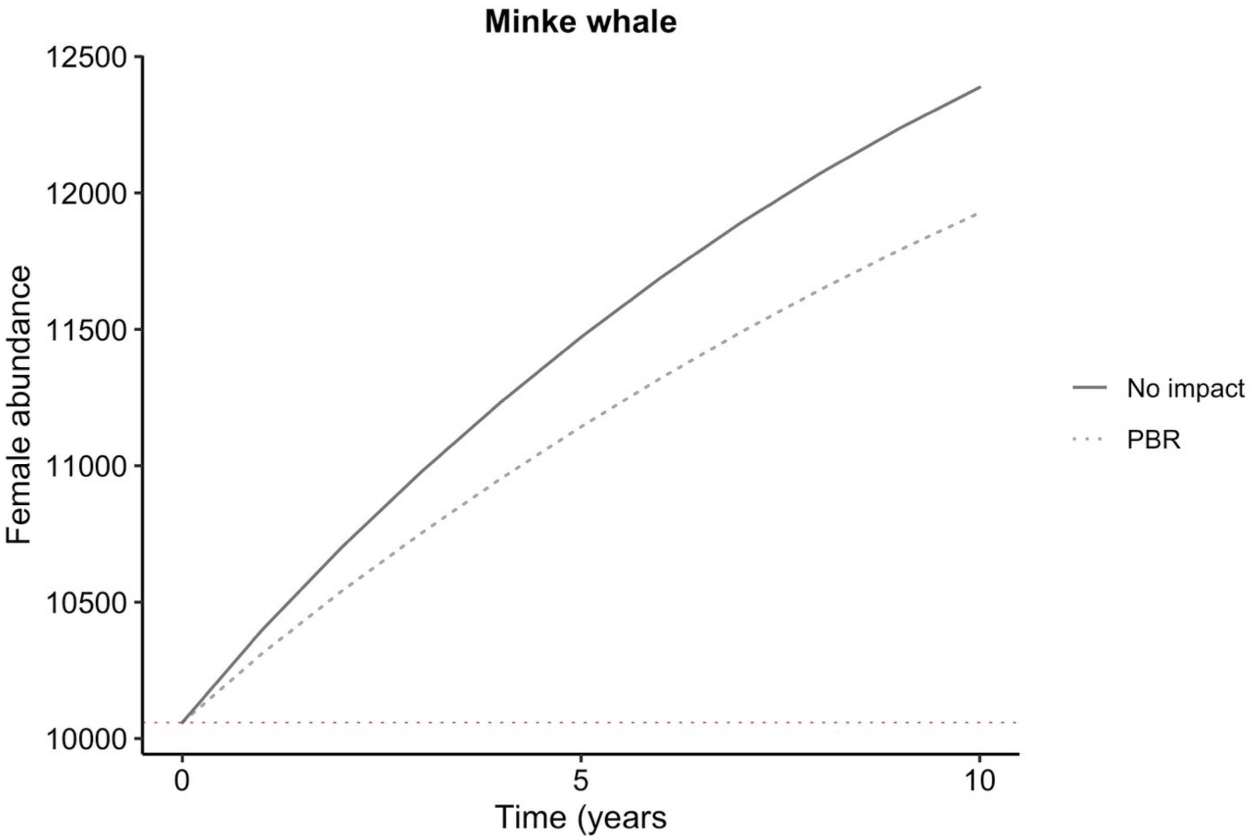

Figure 25: Projected population trends for no impact and the impact of PBR on female minke whales.

Grey seal

Despite all demographic rate scenarios for grey seal showing population increase, certain external pressures had significant impact on the population, particularly when modelling the larger MU area and population size (Figure 26); this was driven by bycatch, PBR limits and, of course, the cumulative impacts. The factors that had the smallest impacts were marine developments and contaminants. For the smaller MU areas (and population sizes) bycatch, PBR levels and cumulative impacts only caused concern under demographic rate scenarios 1 and 4 (and scenario 5 in the case of the CIS MU) which are also associated with the lowest pup survival probability.

Table 23: Description of impact scenarios for grey seals. All numbers of removals provided are subsequently halved to represent females assuming a 50:50 sex ratio.

Scenario	**Description**
Contaminants	Reduce reproductive rate by 10% (see text on page 89)
Bycatch	Remove 3 individuals per year from the Welsh SMU, 339 individuals from CIS MU, 394 individuals from CIS + Ireland MU, and 494 individuals from ICES subarea 7 (OSPAR III area). 60% of these individuals allocated as juveniles and 40% adults.
PBR	Remove 239 individuals from the Welsh SMU, 359 individuals (juveniles and adults) per year for CIS MU, 554 juveniles and adults per year for CIS + Ireland MU, and 2,917 individuals for OSPAR III region. 60% of these individuals will be juveniles and 40% adults.
Marine developments	Removal <10 (9 was chosen) individuals per year across all four MU area options. The suggested maximum removal threshold described in NRW’s position statement on AEOSI.
Combined (excl. PBR)	Reduce reproductive rate by 10%, remove area-specific bycatch of juveniles and adults per year, and remove an additional 9 juveniles per year through interaction with marine developments.

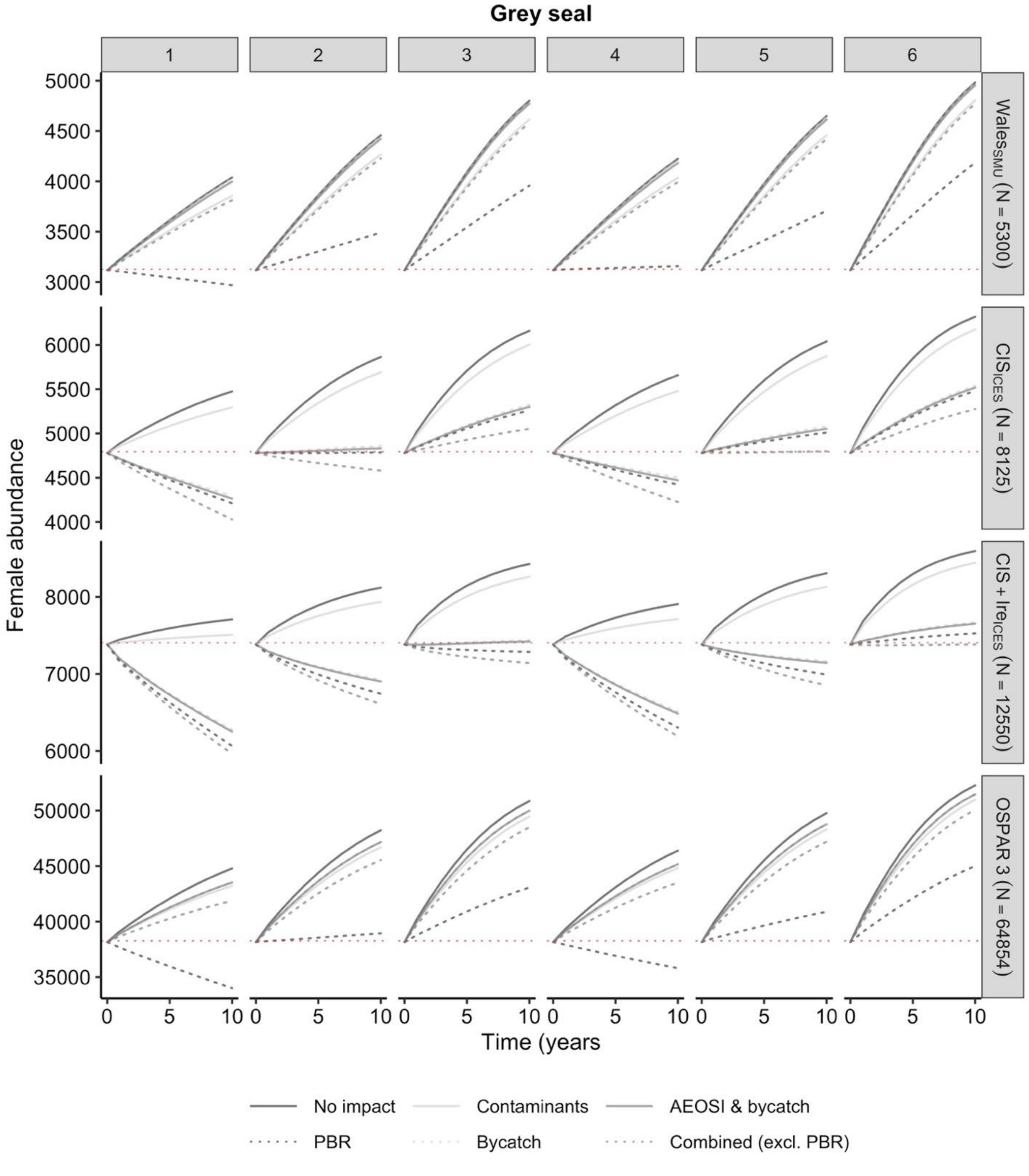

Figure 26: Projected population trends of female grey seal for each impact scenario across each of the six demographic rate scenarios (see Table 23) for the four spatial areas and their population sizes.

Table 24: Population growth rates for grey seal for six different demographic rate scenarios (1-6) (see Table 18) under six different impact scenarios in the four spatial areas and their population sizes. Values above 1.0 indicate an increase and below 1.0 indicate a decline.

N/spatial area	**Impact scenario**	**1**	**2**	**3**	**4**	**5**	**6**
5,300/Wales$_{\mathrm{MU}}$	No impact	1.026	1.036	1.044	1.031	1.041	1.048
5,300/Wales$_{\mathrm{MU}}$	AEOSI & bycatch	1.025	1.036	1.044	1.030	1.040	1.048
5,300/Wales$_{\mathrm{MU}}$	Contaminants	1.021	1.032	1.040	1.026	1.036	1.044
5,300/Wales$_{\mathrm{MU}}$	Bycatch	1.026	1.036	1.044	1.031	1.041	1.048
5,300/Wales$_{\mathrm{MU}}$	PBR	0.995	1.011	1.024	1.001	1.017	1.030
5,300/Wales$_{\mathrm{MU}}$	Combined	1.020	1.031	1.039	1.025	1.036	1.044
8,125/CIS$_{\mathrm{ICES}}$	No impact	1.014	1.021	1.026	1.017	1.024	1.028
8,125/CIS$_{\mathrm{ICES}}$	AEOSI & bycatch	0.989	1.001	1.010	0.993	1.006	1.014
8,125/CIS$_{\mathrm{ICES}}$	Contaminants	1.010	1.018	1.023	1.014	1.021	1.026
8,125/CIS$_{\mathrm{ICES}}$	Bycatch	0.989	1.002	1.011	0.994	1.006	1.015
8,125/CIS$_{\mathrm{ICES}}$	PBR	0.987	1.000	1.010	0.992	1.005	1.014
8,125/CIS$_{\mathrm{ICES}}$	Combined	0.983	0.996	1.006	0.988	1.000	1.010
12,550/CIS+Ire$_{\mathrm{ICES}}$	No impact	1.004	1.010	1.013	1.007	1.012	1.015
12,550/CIS+Ire$_{\mathrm{ICES}}$	AEOSI & bycatch	0.984	0.993	1.001	0.987	0.997	1.004
12,550/CIS+Ire$_{\mathrm{ICES}}$	Contaminants	1.002	1.007	1.011	1.004	1.010	1.014
12,550/CIS+Ire$_{\mathrm{ICES}}$	Bycatch	0.984	0.994	1.001	0.988	0.997	1.004
12,550/CIS+Ire$_{\mathrm{ICES}}$	PBR	0.981	0.991	0.999	0.984	0.995	1.002
12,550/CIS+Ire$_{\mathrm{ICES}}$	Combined	0.979	0.989	0.999	0.983	0.993	1.000

64,854/OSPAR3	No impact	1.016	1.024	1.029	1.020	1.027	1.032
64,854/OSPAR3	AEOSI & bycatch	1.013	1.021	1.027	1.017	1.025	1.030
64,854/OSPAR3	Contaminants	1.013	1.020	1.026	1.016	1.024	1.029
64,854/OSPAR3	Bycatch	1.013	1.022	1.027	1.017	1.025	1.031
64,854/OSPAR3	PBR	0.989	1.002	1.012	0.994	1.007	1.017
64,854/OSPAR3	Combined	1.009	1.018	1.024	1.013	1.022	1.028

3.3. Interim Population Consequences of Disturbance (iPCoD)

The Interim Population Consequences of Disturbance (iPCoD) framework was developed to investigate the population consequences of the effects from exposure to noise (disturbance and auditory injury), primarily from piling activity during offshore wind farm construction (Harwood et al. 2013, King et al. 2015). However, iPCoD can also incorporate removals in the form of mortalities. In this modelling exercise, the same list of demographic rate scenarios were tested for the different population abundances as used for the PVA.

The model generates two future population predictions - one which represents the baseline or un-impacted population and one which represents the impacted population. This is done by incorporating the effects of the expected levels of impact on the vital rates, for example the effect that noise disturbance has on the ability of animals to survive or breed, or the effect that hearing damage from the exposure to noise (in the form of Permanent Threshold Shift, PTS) has on survival and reproduction. Since there is usually little information on the relationship between a given level of impact and the resulting behavioural or physiological changes in individuals, and the effects of such changes on their individual fitness (in terms of reproductive and survival rates), expert elicitation has been used unless empirical data exist.

For predicting the effects of noise disturbance and/or PTS, a day-by-day simulation for up to 1,000 individual animals (the precise number is determined by the size of the population) is performed across the period of predicted disturbance. From this, both the number of animals experiencing disturbance and/or PTS and the amount of disturbance and/or PTS experienced by each of the individuals, by the end of each year, is calculated derived from an estimate of the amount of time an individual is likely to be exposed to the noise and the proportion of the population exposed.

By scaling the numbers up to the total population size, a Leslie matrix model can be created that is used to calculate the future population growth of the impacted population using modified survival and birth rates for those animals that have experienced disturbance and/or PTS. In parallel, the baseline survival and birth rate values available for the population allow a Leslie matrix model to project the future trajectory of the un-impacted population. This is repeated many times (1,000 times is the default, and is the minimum recommended by King *et al* 2015, but this can be changed by the user) and each simulation draws parameter values from statistical distributions describing the uncertainty in the parameters. The distributions of the two trajectories can be compared to demonstrate the size of the long-term effect of the predicted impact on the population as well as demonstrating the uncertainty in predictions.

The framework can also be used to incorporate the number of predicted mortalities per year, e.g. from collisions with marine renewable energy devices, by taking the number of surviving individuals in each year and simply reducing it by the number of predicted removals. However, iPCoD does not differentiate between juveniles and adults.

The advantage of this modelling approach is that non-lethal impacts such as disturbance from underwater noise can be examined. The main limitation is that it requires a great deal of subjective judgment since rarely do empirical data exist. Besides uncertainties within species population parameters, information on the duration for which an individual is exposed to the disturbance (taking account of its movement patterns), the actual impact of the disturbance in terms of reduced energy expenditure and its consequences, and the proportion of the population exposed, rarely exists. The current version also does not incorporate any density-dependence that may be operating.

Below, we use iPCoD to consider the population impact of removals of individuals from the population for harbour porpoise (N=62,517, 9,376, & 25,261), bottlenose dolphin (N=318 & 293), and grey seal (N=5,300, 8,125, 12,550 & 64,854), using the population parameters proposed in Table 1 projecting across 10 years, similar to the PVA. Importantly, we are not attempting to model the impact of disturbance on the population.

3.1. Harbour porpoise:

iPCoD was run assuming that four harbour porpoise individuals are removed every year from any of the three MU areas, as described in NRW's position statement on defining AEOSI (NRW 2020). They were based upon the CIS spatial scale. In this iPCoD analysis (and the PVA analyses above), the number of removals has not been scaled down proportionately to the smaller spatial MU areas modelled. The results indicate virtually no difference in the population trajectories between an un-impacted and an impacted population from this removals value alone. This is almost certainly because the main impact (which we believe is causing the observed decline – see Section 2) comes from bycatch (estimated for the CIS MU at c. 1,000 per year). We have therefore run iPCoD combining potential removals from marine developments – i.e. AEOSI from NRW (2020) - with removals from bycatch (1,000 harbour porpoise per year in CIS MU, 12 in IS AU, and 738 in CS AU) to simulate the impact of bycatch (Figure 27). Bycatch has a significant impact on population growth across all demographic rate scenarios (Table 20).

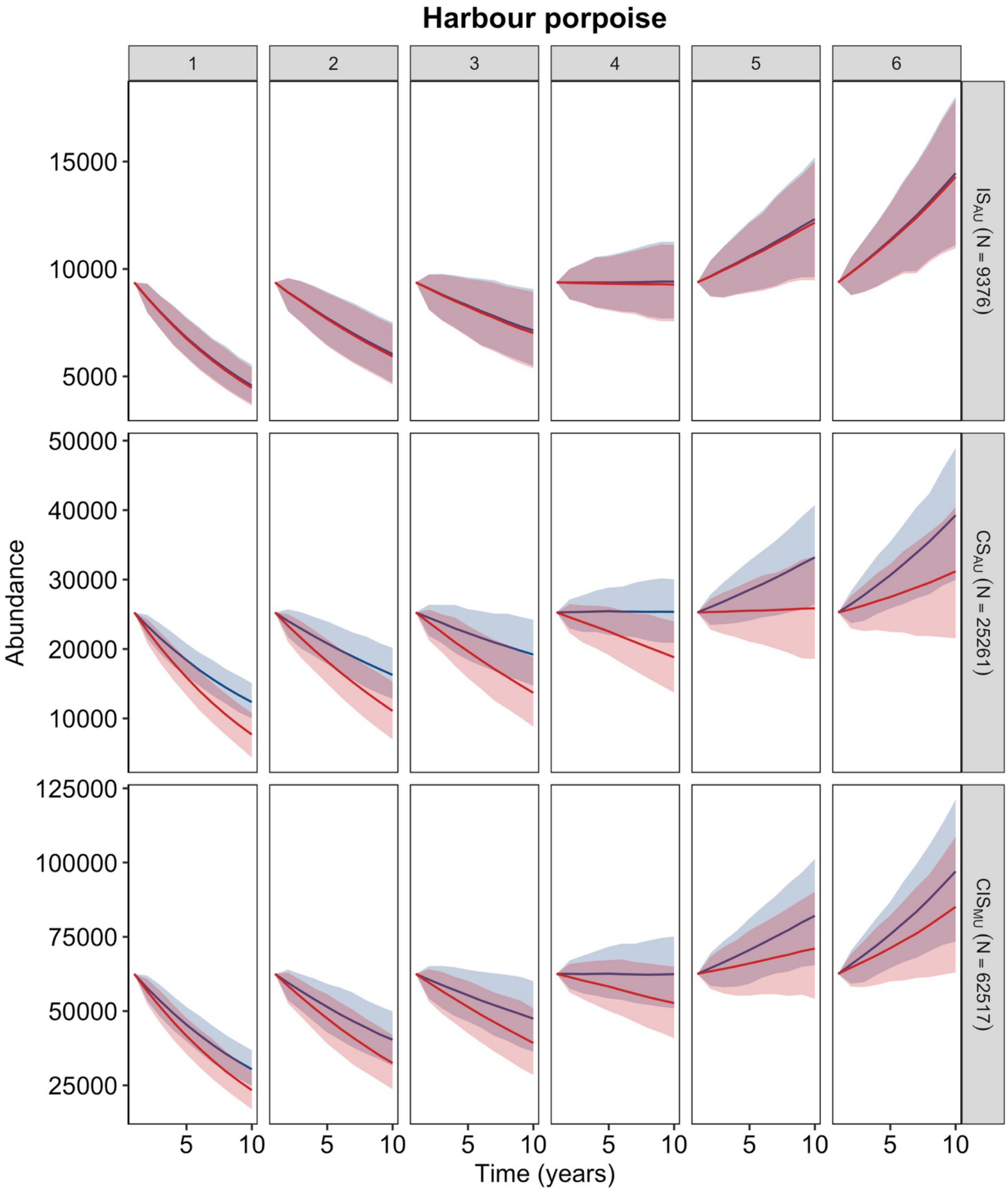

Figure 27: The projected iPCoD population trends for harbour porpoise at three spatial scales for an un-impacted population (top blue line and 95% confidence intervals) and an impacted population (lower red line and 95% CI) following removals from bycatch and marine developments (as described in NRW's position statement on determining AEOSI) for demographic rate scenarios 1-6 (see Table 12).

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Table 25: iPCoD population growth rates (10-year mean) comparing the un-impacted and impacted (AEOSI and bycatch) harbour porpoise populations across demographic rate scenarios 1-6 (see Table 12) at three spatial scales.

N/spatial area	Demographic Scenario	Un-impacted	Impacted
62,517/CIS_{MU}	1	0.923	0.896
62,517/CIS_{MU}	2	0.953	0.930
62,517/CIS_{MU}	3	0.970	0.950
62,517/CIS_{MU}	4	1.000	0.981
62,517/CIS_{MU}	5	1.031	1.014
62,517/CIS_{MU}	6	1.050	1.035
9,376/IS_{AU}	1	0.923	0.921
9,376/IS_{AU}	2	0.952	0.950
9,376/IS_{AU}	3	0.970	0.968
9,376/IS_{AU}	4	1.000	0.999
9,376/IS_{AU}	5	1.031	1.029
9,376/IS_{AU}	6	1.049	1.048
25,261/CS_{AU}	1	0.923	0.876
25,261/CS_{AU}	2	0.952	0.913
25,261/CS_{AU}	3	0.970	0.934
25,261/CS_{AU}	4	1.000	0.968
25,261/CS_{AU}	5	1.031	1.003
25,261/CS_{AU}	6	1.050	1.024

3.2. Bottlenose dolphin:

Here, we have run iPCoD assuming that two bottlenose dolphin individuals are removed over three years (i.e. n=0.67/year based upon a SCANS III estimate of 288 in the Irish Sea MU area) as described in NRW's position statement on determining AEOSI (NRW 2020). The results are shown in Figure 28 for every demographic rate scenario described in Table 14.

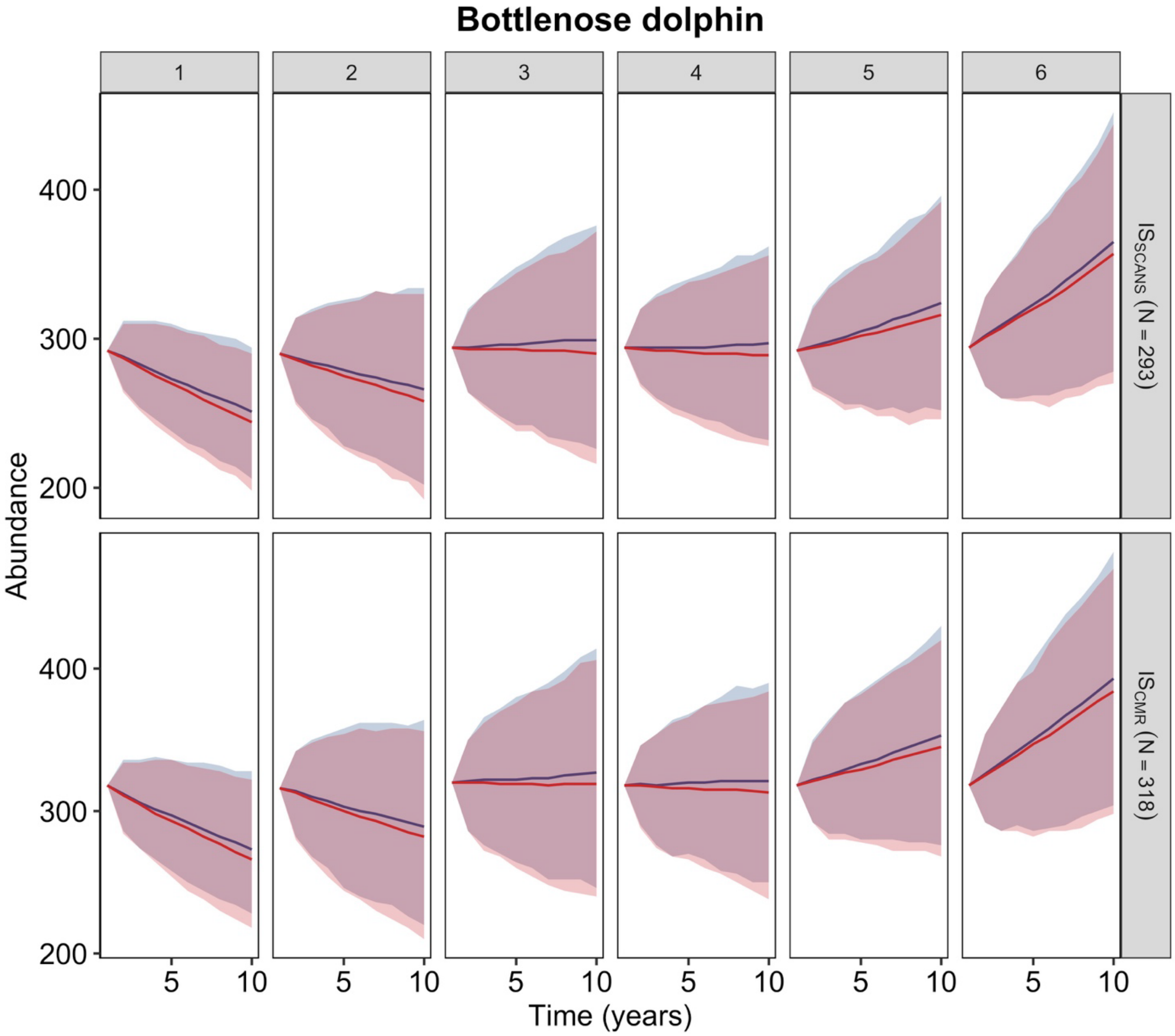

Figure 28: The projected iPCoD population trends for bottlenose dolphins in the Irish Sea MU for an un-impacted population, with a starting population of N=293 (upper graphs) and 318 (lower graphs) (top blue line and 95% confidence intervals) and an impacted population (lower red line and 95% CI) following removals of 2 individuals over 3 years (as described in NRW's position statement on determining AEOSI: NRW 2020) and 3 individuals every ten years from vessel strikes, for demographic rate scenarios 1-6 (see Table 14).

They indicate little change in population trajectories between an un-impacted and impacted population from this pressure (removal value) alone. Table 26 shows the cumulative impacts of vessel strike (3 individuals over 10 years, i.e., 0.3 per year) and removals (from marine developments/AEOSI: 2 individuals over 3 years) compared with an un-impacted population. Just by increasing the removal of individual dolphins per year from essentially 0.67 to 0.97, an impact on population growth rate becomes much more noticeable (Table 26).

Table 26: iPCoD population growth rates (10-year mean) of the unimpacted (with a starting population of N=318 and N=293) and impacted (AEOSI and vessel strike) bottlenose dolphin populations in the Irish Sea MU across demographic rate scenarios 1-6 (see Table 14).

N	**Demographic Scenario**	**Un-impacted**	**Impacted**
318	1	1.024	1.022
318	2	0.990	0.987
318	3	1.002	1.000
318	4	1.001	0.998
318	5	1.012	1.009
318	6	1.024	1.021
293	1	0.983	0.980
293	2	0.990	0.987
293	3	1.002	0.998
293	4	1.001	0.998
293	5	1.012	1.009
293	6	1.024	1.022

3.3. Grey seal:

We ran iPCoD for grey seals on the four starting population sizes of N = 5,300 (Wales$_{SMU}$), 8,125 (CIS$_{ICES}$), 12,550 (CIS+ Ireland$_{ICES}$) and 64,854 (OSPAR III), and assumed 9 grey seal individuals are removed every year from marine developments + bycatch across all four MU areas, following NRW in their position statement on AEOSI (NRW 2020). We did this with and without bycatch. Results indicate very little difference in the population trajectories between an un-impacted and an impacted population from AEOSI removals alone.

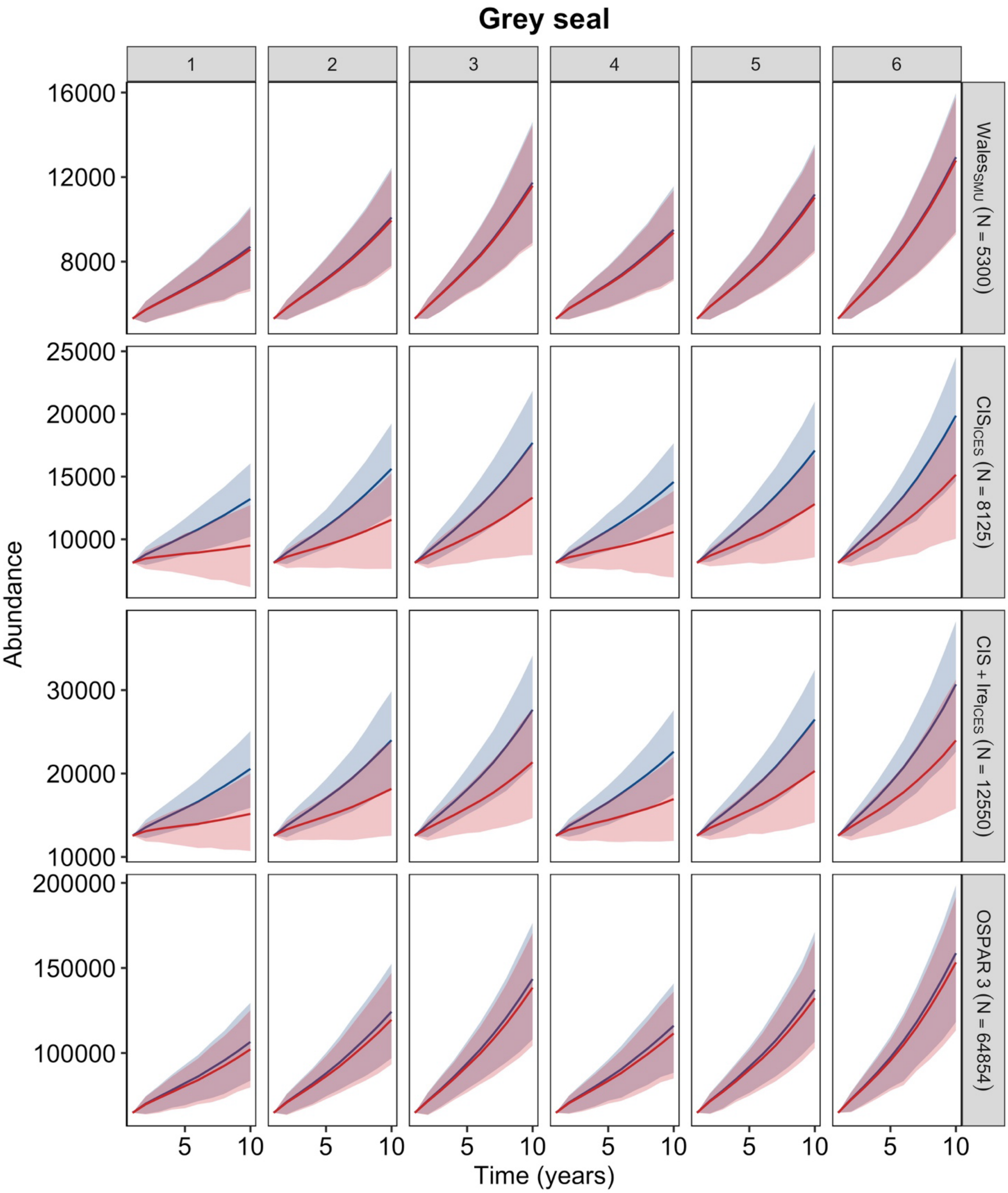

Figure 29: The projected iPCoD population trends for grey seals for an un-impacted population (top blue line and 95% confidence intervals) and an impacted population (lower red line and 95% CI) (9 removals per year (AEOSI plus bycatch) for demographic rate scenarios 1-6 (see Table 18) for the four spatial areas.

Table 27: iPCoD population growth rates (10-year mean) of the un-impacted and impacted (AEOSI plus bycatch) grey seal populations across demographic rate scenarios 1-6 (see table 18) for the four spatial areas.

N/spatial area	**Demographic Scenario**	**Un-impacted**	**Impacted**
5,300/Wales$_{SMU}$	1	1.057	1.055
5,300/Wales$_{SMU}$	2	1.074	1.073
5,300/Wales$_{SMU}$	3	1.092	1.091
5,300/Wales$_{SMU}$	4	1.067	1.065
5,300/Wales$_{SMU}$	5	1.087	1.085
5,300/Wales$_{SMU}$	6	1.104	1.103
8,125/CIS$_{ICES}$	1	1.056	1.018
8,125/CIS$_{ICES}$	2	1.075	1.040
8,125/CIS$_{ICES}$	3	1.090	1.057
8,125/CIS$_{ICES}$	4	1.067	1.030
8,125/CIS$_{ICES}$	5	1.086	1.052
8,125/CIS$_{ICES}$	6	1.104	1.072
12,550/CIS+IRE$_{ICES}$	1	1.056	1.021
12,550/CIS+IRE$_{ICES}$	2	1.075	1.042
12,550/CIS+IRE$_{ICES}$	3	1.092	1.061
12,550/CIS+IRE$_{ICES}$	4	1.068	1.034
12,550/CIS+IRE$_{ICES}$	5	1.087	1.055
12,550/CIS+IRE$_{ICES}$	6	1.104	1.075
64,854/OSPAR III	1	1.057	1.052
64,854/OSPAR III	2	1.075	1.070
64,854/OSPAR III	3	1.092	1.088
64,854/OSPAR III	4	1.067	1.062
64,854/OSPAR III	5	1.087	1.082
64,854/OSPAR III	6	1.105	1.100

The main impact on grey seals for most regions except for the Wales SMU, comes from bycatch (estimated at 3 per year for the Wales SMU, 339 per year CIS$_{\text{ICES}}$ MU, 394 for the CIS+Ireland$_{\text{ICES}}$ MU area, and 494 for ICES Subarea 7 (OSPAR III) – see section 2.6.5), and the combined effect of bycatch and removals from marine developments (AEOSI) shows reduced populations compared to the unimpacted in the two medium/large sized areas (CIS$_{\text{ICES}}$; CIS+IRE$_{\text{ICES}}$) and very little difference in the smallest (Wales SMU) and the largest (OSPAR III) areas (Figure 29). No scenario results in a declining population (Table 27).

4. Conclusions & Recommendations

Population Parameters

The models are as good as the data that go into them, and it is therefore important to stress that even for the species about which we have some reasonable knowledge, there are some critical uncertainties. The most important of these is whether or not the unit of population is appropriate. It is likely that all the species will have some population structure that may create differences in demographic rates even if this is simply the result of isolation by distance. However, if the boundaries applied to an MU are incorrect, and there is movement in or out, this will affect whether the abundance estimate is appropriate, and likewise the observed population trends. Estimating the influence of bycatch, for example, could also be compromised if a significant portion of the population is exposed to it outside the region under consideration.

Of the six marine mammal species covered in this report, the population of bottlenose dolphins in the Irish Sea is likely to be the one with the most restricted distribution, as indicated from photo-ID (Feingold & Evans 2012, 2014, Lohrengel et al. 2017) . It is also the species that we have greatest knowledge within its MU area, but we recognise that there are still uncertainties in a number of its population parameters. Populations of common dolphin and minke whale almost certainly extend beyond the management boundaries proposed by IAMMWG, and this will have implications on estimates of population size and observed trends. Furthermore, both species experience known mortality outside those boundaries (Evans 2020, ICES WGBYC 2021, 2022) (particularly significant for common dolphin), which makes it difficult to determine the impact upon demography within the MUs used.

There is uncertainty around all of the population parameters available. Those estimates that exist come largely from outside the IAMMWG MUs being considered here, and where they do come from within the region, they show variation between areas (e.g., grey seal), have been collected at different time periods (e.g., harbour porpoise), or do not span the entire region (e.g., bottlenose dolphin).

Pressures

The most obvious pressure leading to removals is bycatch, and evidence to date suggests that its influence is greatest upon harbour porpoise, common dolphin, and grey seal (ICES WGBYC 2021, 2022, ICES WKMOMA 2021). Bycatch for all three species occurs for the most part outside the Irish Sea and Welsh waters and so its impact will depend upon the extent to which animals from the Irish Sea travel to those areas; for harbour porpoise this is the Southwest Approaches to the Channel and southwest of Ireland; for common dolphin, it is largely the Bay of Biscay, outside the boundaries set here as the Management Unit; and for grey seal it is around Southwest England (including the Bristol Channel) and off southern Ireland. For those species, the influence of other pressures such as offshore renewables on population demography is likely to be dwarfed by bycatch. That is not likely to be the case, however, for coastal bottlenose dolphin whose population is not only small and largely confined to the Irish Sea but records suggest bycatch risk is negligible (Welsh strandings database, 1990-present; ICES WGBYC, 2020, 2021, 2022).

Vessel strike is also known to cause mortality although at a low level across species, mainly affecting large whales such as fin and sperm whale (Evans 2020). From direct observations (Feingold & Evans 2014, Lohrengel et al. 2017) and post-mortem examination of Welsh strandings (Penrose 2010-20), cases of blunt strike trauma were recorded in bottlenose dolphins, but for other species, it was believed to have only minimal effect at a population level. It is very difficult to quantitatively determine the impact of contaminants on population demography. There is evidence for harbour porpoise and bottlenose dolphin, and possibly also common dolphin, that contaminants reduce fertility rates in females primarily in their first year of breeding (Murphy et al. 2010, 2019; Jepson et al. 2016). We therefore applied an arbitrary 10% reduction in reproductive rates of harbour porpoise and bottlenose dolphin.

Model Assessment

PBR is simplistic in that the user only needs to estimate one value, the population size (N_{min}). Other parameters are largely default values, including an estimated maximum growth rate of 0.04 and 0.12 for cetacean or a pinniped species, respectively, despite there being significant variation in life histories within these groups. PBR also assumes the population is at optimum sustainable growth and does not explicitly take into account the status of the population nor any pressures that it might be experiencing. For grey seals, for example, where the population growth rate might not be at an optimal level but still growing towards carrying capacity, PBR values may be underestimates. Nevertheless, removing the number of individuals suggested by PBR in the PVA resulted in a significant change in grey seal population growth, particularly for the large $CIS+IRE_{ICES}$ area. The choice of the chosen Recovery Factor is somewhat arbitrary, although some rule-based methods or suggestions have been devised (Wade 1998, Dillingham & Fletcher 2011). Additionally, the population estimate must represent the appropriate spatial scale of the population, which is generally uncertain. While it may be an appropriate tool for small discrete populations (such as bottlenose dolphins), it is less appropriate for larger populations unless their demographic rates are favourable. In the case of the grey seal population over the wider area, every scenario including the removals suggested by PBR indicated a resulting decline. Despite bycatch exceeding PBR which should result in a declining population, the grey seal population estimates indicate a continued increase in abundance. A possible reason for this is that the juveniles being captured in fisheries in SW Britain are from colonies outside that region.

Because of data uncertainties, models such as the Removals Limit Algorithm (RLA) and Bayesian state space models will be of limited applicability for some Welsh marine mammal populations (e.g. common dolphin, Risso's dolphin, minke whale) until more detailed information is available, and in particular population trends are available across the most appropriate Management Unit. However, it should be possible to apply RLA to harbour porpoise and grey seal populations once the boundaries of management units are more clearly established. Although long-term trends exist for bottlenose dolphin, these data/estimates are confined to Cardigan Bay and largely within the Cardigan Bay SAC and so do not fully account for movements elsewhere.

Although PVAs used here are deterministic, they offer a useful tool for exploring the impact of using different demographic rates on population growth, which is valuable when borrowing information from other populations. PVA therefore gives some relative indication of the future state of the population under differing demographic rate scenarios which can be evaluated and used in advice, but only when outputs are subject to expert interpretation and critique. If the population growth rate is close to 1 so that any perturbation could lead to a decline, then a precautionary approach limiting anthropogenic removals should be applied. The results of the PVAs are robust as long as users keep in mind that they are driven solely by the demographic parameters that are fed into them. These carry some uncertainty, and hence it is preferable to run multiple demographic rate scenarios in some cases to give an idea of how sensitive the population is to changing the demographic parameters and the range of expected population growth rates. It is difficult to judge which demographic rate scenario is the optimal one as it depends on how much confidence we have in the demographic rates that have been used, especially when some of these have been borrowed from elsewhere. It is therefore important to support and improve work that attempts to estimate more accurate demographic rates relevant to the area of interest. Further stochasticity can be incorporated into PVAs to generate more variability in outcomes (for example accounting for environmental variability). However, this generally requires further data including time series of demographic rates, which only really exist for Cardigan Bay SAC bottlenose dolphins and not for other species.

In this context, PVAs offer more flexibility than other approaches, especially because they allow the exploration of the impact of demographic stochasticity and inclusion of density dependence. The role of stochasticity in affecting demographic outcomes was really only an important consideration for bottlenose dolphins due to their small population size. Furthermore, PVAs allow the assessment of how sensitive the population is to change in the different demographic rates, which is useful for management purposes as it reveals where effort might be most effective. It is well known that for long-lived species with slow life histories that population growth rate is most sensitive to change in adult survival. That was also the case for the species analysed here, and was followed by the probability of a juvenile surviving and transitioning to an adult as well as the reproductive rate in most cases. For example, in our model runs the most influential parameters included any change in juvenile and adult survival for all study species, as well as any change in reproductive rate of bottlenose dolphins. Therefore, managing direct removals, e.g. bycatch, of juveniles and adults would have the largest impact on population growth. Finally, PVAs allow assessment and comparison of the impact of different pressures on population growth. This can reveal some potentially unexpected results – for example, despite grey seals showing positive population projections across scenarios, they are predicted to be sensitive to bycatch.

Notwithstanding the useful role that PVA can play in exploring the potential demographic consequences of anthropogenic removals, it does not provide thresholds of acceptable take in the way that PBR or RLA can do, and thus has not been readily applied to marine mammal populations as a management tool.

iPCoD has been developed to help evaluate sublethal effects such as disturbance from underwater noise, although it also allows users to run models to explore the impact of direct removals. However, it is not possible to specify whether these removals should be juvenile, adult or some combination of life stages. It is also not possible to remove a varying or stochastic number of individuals per year (i.e. where bycatch is likely to be a range of individuals rather than a set value). Nevertheless, iPCoD allows the user the same flexibility as PVA to input different demographic and population parameters relating to the underlying population, potentially allowing the modelling of more complex population dynamics; however, this would need to be explored further. A definite advantage of using iPCoD is that it allows the production of 95% confidence limits around the predicted population trend. But it still appears to be a complex step for purely exploring relatively small impacts of removals around marine developments and is not purpose-built for exploring impacts such as bycatch. In the context of the Irish Sea, for large populations (i.e. excluding bottlenose dolphin), removing only small numbers of animals per year, for example <5 or <10 individuals as defined by NRW (2020) in the context of AEOSI, did not show any impact on the populations of harbour porpoise or grey seal, respectively. But, as expected, it revealed more significant impacts on population growth when including bycatch for these species. When multiple impacts are combined, population growth did appear to be affected in several scenarios for both harbour porpoise and bottlenose dolphin, but not for grey seal.

When comparing results from PVA and iPCoD, it is important to note that iPCoD does not include density dependence. We believe this functionality is planned for iPCoD, and its inclusion would likely result in differences in growth rates in some cases, particularly scenarios showing healthy population increase.

For bottlenose dolphins, there was good agreement between growth rates estimated by the iPCoD and PVA modelling frameworks. It is important to note that for bottlenose dolphins, cumulative impact for the iPCoD model run refers to the combined impact of marine development/AEOSI thresholds and vessel strike. In the PVA for bottlenose dolphins, vessel strike was modelled separately and in combination with impacts of AEOSI.

For harbour porpoise, there was fairly good agreement between the two modelling methods (iPCoD and PVA) in terms of the estimated growth rates for the two main types of removal – bycatch and marine developments/AEOSI thresholds. Here the PVA estimates of growth rates are slightly higher compared to those estimated by iPCoD.

For grey seals, there was less agreement between iPCoD and PVA. Across demographic rate and impact scenarios, the mean growth rates produced by PVA are always lower compared to those produced by iPCoD. Because all demographic rate scenarios for grey seals show population increase, the difference in growth rates between these two modelling methods may be due to the effect of density dependence in the PVA. Furthermore, while iPCoD never predicts a population decline in grey seals, for most MU areas, PVA predicts a decline for demographic rate scenarios 1 and 4 under impact scenarios that include bycatch.

Recommendations

In conclusion, all models come with their own specific pros and cons which needs to be carefully evaluated, particularly when most demographic and population parameters are uncertain. In this context, we would recommend application of PVA due to its relative simplicity and flexibility. We believe PVA is more appropriate here compared to iPCoD, and in most cases there was good agreement between growth rates generated by either modelling method (except for grey seals for reasons described above), which supports the use of the more user friendly and adaptable PVA approach. Although more complexity can be added to PVAs, we did not feel this was important or appropriate in this context as one would be adding complexity to uncertainty, and instead chose to keep the models relatively simple. PVA seems well suited for investigating the effects of anthropogenic removals upon demographic trends, including useful and informative additional features or possibilities including transfer function analysis, exploring demographic stochasticity, and testing various scenarios. PVA could be used in conjunction with PBR, the latter being important in providing a threshold or management trigger which PVA obviously does not do. However, in some cases, including PBR removal limits in PVAs resulted in significant reduction in population growth rates, which may not be the desired or realistic outcome. Therefore, PBR and PVA should be tested together.

There are uncertainties with both PBR and PVA. The accuracy of PBR depends on R_{max} which is somewhat arbitrarily set at 0.04 for cetaceans and 0.12 for pinnipeds. Accuracy also depends upon how good the estimate of N_{min} is, as well as whatever decision is made on the recovery factor. Similarly, PVA is only as good as the demographic rate values we include. A value of using both methods is that one can compare results and if they align with one another and the observed trends we should be able to have more confidence than when they do not agree.

Where there is relatively good information on population parameters, it would be instructive to run RLA as well, now that the software coding issue we experienced has been resolved, particularly since this has been an option used by OSPAR in the bycatch common indicator assessment for harbour porpoise (in the Greater North Sea).

5. References

Aarts G, Brasseur S, Kirkwood R. 2018. Response of grey seals to pile-driving. Wageningen Marine Research (University & Research Centre), Wageningen, The Netherlands. Wageningen Marine Research Report C006/18. 54pp.

Akçakaya HR, Sjögren-Gulve, P. 2000. Population viability analysis in conservation planning: an overview. Ecological Bulletins, 48, 9-21.

Amano M, Miyazaki N. 2004. Composition of a school of Risso's dolphin, *Grampus griseus*. Marine Mammal Science, 20, 152-160. doi:10.1111/j.1748-7692.2004.tb01146.x

Anderwald P, Daníelsdottir AK, Haug T, Larsen F, Lesage V, Reid RJ, Vikingsson G, Hoelzel AR. 2011. Possible cryptic stock structure for minke whales in the North Atlantic: Implications for conservation and management. Biological Conservation*,* 144, 2479-2489.

Anonymous. 2000. Report of the IWC- ASCOBANS Working Group on Harbour Porpoises. Journal of Cetacean Research and Management, 2, 297-305.

Arso Civil M, Cheney B, Quick NJ, Islas- Villanueva V, Graves JA, Janik VM, Thompson, PM, Hammond, PS. 2018. Variations in age- and sex-specific survival rates help explain population trend in a discrete marine mammal population. Ecology and Evolution, 9(1), 533-544. https://doi.org/10.1002/ece3.4772

Arso Civil M, Cheney B, Quick, NJ, Thompson PM, Hammond PS. 2017. A new approach to estimate fecundity rate from inter- birth intervals. Ecosphere, 8(4), e01796. https://doi.org/10.1002/ecs2.1796

ASCOBANS. 2000. Resolution on Incidental Take of Small Cetaceans. ASCOBANS Meeting of Parties 3, Bristol, UK.

Baines ME, Earl SJ, Pierpoint CJL, Poole J. 1995. The west Wales grey seal census. CCW Contract Science Report No. 131. Countryside Council for Wales, Bangor.

Barlow J, Swartz SL, Eagle TC, Wade, PR. 1995. U.S. Marine Mammal Stock Assessments: Guidelines for Preparation, Background, and a Summary of the 1995 Assessments. U.S. Department of Commerce, NOAA Technical Memorandum. NMFS-OPR-6, 73pp.

Baines ME, Evans PGH. 2012. Atlas of the Marine Mammals of Wales. 2nd Edition. Marine Monitoring Report No. 68. Countryside Council for Wales, Bangor.

Beare D, Burns F, Jones E, Peach K, Portilla E, Greig T, McKenzie E, Reid, D. 2004. An increase in the abundance of anchovies and sardines in the north-western North Sea since 1995. Global Change Biology*,* 10, 1209–1213.

Beissinger SR, McCullough DR (editors). 2002. Population viability analysis. University of Chicago Press, Chicago.

Benjamins S, Dale A, Hastie G, Lea M, Scott BE, Waggitt JJ, Wilson B. 2015. Confusion reigns? A review of marine megafauna interactions with energetic tidal features. Oceanography and Marine Biology: An Annual Review, 53, 1–54.

Bergman A, Olsson M. 1986. Pathology of Baltic grey seal and ringed seal females with special reference to adrenocortical hyperplasia: is environmental pollution the cause of a widely distributed disease syndrome? Finnish Game Research, 44, 47-62.

Bloch D, Desportes G, Harvey P, Lockyer C, Mikkelsen B. 2012. Life history of Risso's dolphin (*Grampus griseus*) (G. Cuvier, 1812) in the Faroe Islands. Aquatic Mammals, 38, 250–266.

Bonner WN. 1981. Grey Seal *Halichoerus grypus* Fabricius, 1791. Pp. 111-144. In: Handbook of Marine Mammals, Volume 2: Seals (SH Ridgway & R Harrison, Editors). Academic Press, London & New York.

Boyce M. 1992. Population Viability Analysis. Annual Review of Ecology and Systematics, 23, 481-506.

Boys R. 2015. Fatal interactions between bottlenose dolphins (*Tursiops truncatus*) and harbour porpoises (*Phocoena phocoena*) in Welsh waters. BSc thesis, Bangor University, Bangor.

Buckland ST, Anderson DR, Burnham KP, Laake JL, Borchers DL, Thomas L. (Editors) 2004. *Advanced Distance sampling: Estimating abundance of biological populations*. Oxford University Press, Oxford.

Bull JC, Börger L, Franconi N, Banga R, Lock KM, Morris CW, Newman PB, Stringell TB. 2017. Temporal trends and phenology in grey seal (*Halichoerus grypus*) pup counts at Skomer, Wales. NRW Evidence Report No: 217, 23pp, Natural Resources Wales, Bangor.

Carter MID, Boehme L, Duck CD, Grecian WJ, Hastie GD, McConnell BJ, Miller, DL, Morris CD, Moss SEW, Thompson D, Thompson, PM, Russell, DJF. 2020. Habitat-based predictions of at-sea distribution for grey and harbour seals in the British Isles. St Andrews: Sea Mammal Research Unit. Report to BEIS, OESEA-16-76/OESEA-17-78.

Caswell H. 2001. Matrix population models: construction, analysis, and interpretation, Second edition. Sinauer, Sunderland, Massachusetts, USA.

Cheney B, Graham IM, Barton TR, Hammond PS, Thompson PM. 2018. Site Condition Monitoring of bottlenose dolphins within the Moray Firth Special Area of Conservation: 2014-2016. Scottish Natural Heritage Research Report No. 1021.

Cheney B, Thompson TJ, Cordes L. 2019. Increasing trends in fecundity and calf survival of bottlenose dolphins in a marine protected area. Scientific Reports, 9, 1767.

Christensen I. 1975. Preliminary report on the Norwegian fishery for small whales: expansion of Norwegian whaling to Arctic and northwest Atlantic waters, and Norwegian investigations of the biology of small whales. Journal of Fisheries Research Board, Canada, 32(7), 1083-94.

Christensen I. 1981. Age determination of minke whales, *Balaenoptera acutorostrata*, from laminated structures in the tympanic bullae. Report of the International Whaling Commission, 31, 245-53.

Cook ASCP, Robinson RA. 2017. Towards a framework for quantifying the population-level consequences of anthropogenic pressures on the environment: The case of seabirds and windfarms. Journal of Environmental Management, 190,113-121.

Cooke JG. 1999. Improvement of fishery-management advice through simulation testing of harvest algorithms. ICES Journal of Marine Science, 56, 797-810.

Cooke J, Leaper R, Wade P, Lavigne D, Taylor B. 2012. Management rules for marine mammal populations: A response to Lonergan. Marine Policy, 36: 389-392.

Cosgrove R, Cronin M, Reid D, Gosch M, Sheridan M, Chopin N, Jessopp M. 2013. Seal depredation and bycatch in set net fisheries in Irish waters. Fisheries Resource Series Vol. 10 (2013).

Culloch R, Horne N, Kregting L. 2019. A review of Northern Ireland seal count data 1992-2017: Investigating population trends and recommendations for future monitoring. Report for the Department of Agriculture, Environment and Rural Affairs, Northern Ireland. 78pp.

Deaville R (compiler) 2018. UK Cetacean Strandings Investigation Programme Final report for the period 1st January 2011 - 31st December 2017*.* Zoological Society of London, UK.

Deaville R, Jepson PD. (compilers). 2011. UK Cetacean Strandings Investigation Programme Final report for the period 1st January 2005 - 31st December 2010*.* Zoological Society of London, UK. Available at:
http://randd.defra.gov.uk/Document.aspx?Document=FinalCSIPReport2005-2010_finalversion061211released[1].pdf

De Boer M, Clark J, Leopold MF, Simmonds MP, Reijnders PJH. 2013. Photo-Identification Methods Reveal Seasonal and Long-Term Site-Fidelity of Risso's Dolphins (*Grampus griseus*) in Shallow Waters (Cardigan Bay, Wales). Open Journal of Marine Science, 3, 66-75.

Dillingham PW, Fletcher D. 2011. Potential biological removal of albatrosses and petrels with minimal demographic information. Biological Conservation, 144, 1885-1894.

Duck C, Morris C. 2012. Seals in Northern Ireland: Helicopter survey of harbour and grey seals, August 2011.SMRU Unpublished report to the Northern Ireland Environment Agency.

Duck C, Morris C. 2019. Aerial thermal-imaging surveys of Harbour and Grey Seals in Northern Ireland, August 2018. Report for the Department of Agriculture, Environment and Rural Affairs, Northern Ireland.

Duckett A. 2018. Cardigan Bay bottlenose dolphin *Tursiops truncatus* connectivity within and beyond marine protected areas. MSc Thesis, University of Bangor. 76pp.

European Union. 2004. Council Regulation (EC) No. 812/2004 of 26 April 2004 laying down measures concerning incidental catches of cetaceans in fisheries and amending regulation (EC) No. 88/98. Official Journal of the European Union, Series L 150, 30.4.2004: 12-31. online. http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32004R0812&from=EN. Corrigendum: http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:02004R0812-20070719&from=EN. Summary: http://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=LEGISSUM:l66024&from=EN

Evans PGH. 2012. Recommended Management Units for Marine Mammals in Welsh Waters. CCW Policy Research Report No. 12/1.

Evans PGH. 2013. The Risso's Dolphin in Europe. Pp. 10-24. In: Chen, I, Hartman, K, Simmonds, M, Wittich, A, and Wright, AJ. (eds.) *Grampus griseus* 200th Anniversary: Risso's Dolphins in the Contemporary World. Report from the European Cetacean Society Conference Workshop, Galway, Ireland, 25 March 2013. ECS Special Publication Series No: 54. 108pp.

Evans PGH. 2020. European Whales, Dolphins and Porpoises. Marine Mammal Conservation in Practice. Academic Press, London & New York. 306pp.

Evans, PGH, Hammond PS. 2004. Monitoring Cetaceans in European Waters. Mammal Review 34: 131-156.

Evans PGH, Teilmann J. (editors). 2009. Report of ASCOBANS/HELCOM Small Cetacean Population Structure Workshop. Bonn, Germany: ASCOBANS/UNEP Secretariat. 140pp.

Evans PGH, Waggitt, JJ. 2020. Impacts of climate change on Marine Mammals, relevant to the coastal and marine environment around the UK. Marine Climate Change Impacts Partnership (MCCIP) Annual Report Card 2019 Scientific Review, 1-33.

Evans, P.G.H., Carrington, C., and Waggitt, J. 2021. *Risk Assessment of Bycatch of Protected Species in Fishing Activities*. European Commission, Brussels. 213pp. https://ec.europa.eu/environment/nature/natura2000/marine/docs/RISK MAPPING REPORT.pdf

Feingold D, Evans PGH. 2012, Sea Watch Foundation Welsh Bottlenose Dolphin Photo-Identification Catalogue 2011. CCW Marine Monitoring Report No: 97: 1-262.

Feingold D, Evans PGH. 2014. Bottlenose Dolphin and Harbour Porpoise Monitoring in Cardigan Bay and Pen Llyn a'r Sarnau Special Areas of Conservation 2011-2013. Natural Resources Wales Evidence Report Series No. 4. 124pp.

Fontaine MC, Tolley KA, Michaux JR, Birkun A, Ferreira M, Jauniaux T, Llavona Á, Oztürk B, Oztürk, AA, Ridoux V, Rogan E, Sequeira M, Bouquegneau J-M, Baird SJE. 2010. Genetic and historic evidence for climate-driven population fragmentation in a top cetacean predator: The harbour porpoises in European waters. Proceedings of the Royal Society B: Biological Sciences, 277 (1695), 2829-2837.

Fontaine MC, Thatcher O, Ray N, Piry S, Brownlow A, Davison NJ, Jepson P, Deaville R, Goodman SJ. 2017. Mixing of porpoise ecotypes in southwestern UK waters revealed by genetic profiling. Royal Society Open Science 4:160992. doi.org/10.1098/rsos.160992

Freeman S, Searle K, Bogdanova M, Wanless S, Daunt F. 2014. Population dynamics of Forth & Tay breeding seabirds: review of available models and modelling of key breeding populations (MSQ – 0006). Contract report to Marine Scotland Science.

Genu M, Gilles A, Hammond, PS, Macleod K, Paillé J, Paradinas Aranjuelo J, Smout S, Winship AJ, Authier M. 2021. Evaluating strategies for managing anthropogenic mortality on marine mammals: an R implementation with the package RLA. Frontiers in Marine Science 24 December 2021 | https://doi.org/10.3389/fmars.2021.795953

Hammond, PS, Berggren P, Benke H, Borchers DL, Collet A, Heide-Jørgensen MP, Heimlich S, Hiby AR, Leopold MF, Øien, N. 2002. Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. Journal of Applied Ecology, 39, 361-376.

Hammond PS, Macleod K, Berggren P, Borchers DL, Burt ML, Cañadas A, Desportes G, Donovan GP, Gilles A, Gillespie D, Gordon J, Hiby L, Kuklik I, Leaper R, Lehnert K, Leopold M, Lovell P, Øien N, Paxton CGM, Ridoux V, Rogan E, Samarra F, Scheidat M, Sequeira M, Siebert U, Skov H, Swift R, Tasker ML, Teilmann J, Van Canneyt O, Vázquez JA. 2013. Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. Biological Conservation, 164, 107-122.

Hammond PS, Lacey C, Gilles A, Viquerat S, Börjesson P, Herr H, Macleod K, Ridoux V, Santos MB, Scheidat M, Teilmann J, Vingada J, Øien N. 2017. Estimates of cetacean abundance in European Atlantic waters in summer 2016 from the SCANS-III aerial and shipboard surveys. Available from: https://synergy.standrews.ac.uk/scans3/files/2017/05/SCANS-III-design-based-estimates-2017-05-12-final-revised.pdf

Hammond, PS, Paradinas I, Smout SC. 2019. Development of a Removals Limit Algorithm (RLA) to set limits to anthropogenic mortality of small cetaceans to meet specified conservation objectives, with an example implementation for bycatch of harbour porpoise in the North Sea. JNCC Report No. 628, JNCC, Peterborough, ISSN 0963-8091.

Harding KC, Härkönen T, Helander B, Karlsson O. 2007. Status of Baltic grey seals: Population assessment and extinction risk. NAMMCO Scientific Publications, 6:33-56.

Harwood J, King S. 2014. The Sensitivity of UK Marine Mammal Populations to Marine Renewables Developments. Report number SMRUL-NER-2012-027 (unpublished).

Harwood J, King S, Schick R, Donovan C, Booth C. 2013. A Protocol For Implementing The Interim Population Consequences Of Disturbance (PCoD) Approach: Quantifying And Assessing The Effects Of UK Offshore Renewable Energy Developments On Marine Mammal Populations. Report Number SMRUL-TCE-2013-014. Scottish Marine And Freshwater Science, 5(2).

Hauksson E. 2007. Growth and reproduction in the Icelandic grey seal (*Halichoerus grypus*). NAMMCO Scientific Publications, 6, 153-162.

Hauksson E, Vikingsson G, Halldorsson S, Olafsdottir D, Sigurjonsson J. 2011. Preliminary report on biological parameters for North Atlantic minke whales in Icelandic waters. IWC 2011 SC/63/O15.

IAMMWG. 2015. Management Units for cetaceans in UK waters (January 2015). JNCC Report No. 547, JNCC Peterborough.

IAMMWG. 2021. Updated abundance estimates for cetacean Management Units in UK waters. JNCC Report No. 680, JNCC Peterborough, ISSN 0963-8091.

ICES. 2014. OSPAR request on implementation of MSFD for marine mammals. Special request. Available from: http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2014/Special%20Requests/OSPAR_Implementation_of_MSFD_for_marine_mammals.pdf

ICES. 2018. Bycatch of small cetaceans and other marine animals – review of national reports under Council Regulation (EC) No. 812/2004 and other information. Available from: https://www.ices.dk/sites/pub/Publication%20Reports/Advice/2018/2018/byc.eu.pdf

ICES. 2020. EU request on emergency measures to prevent bycatch of common dolphin (*Delphinus delphis*) and Baltic Proper harbour porpoise (*Phocoena phocoena*) in the Northeast Atlantic. ICES Special Request Advice Northeast Atlantic ecoregions 21pp. Available from: https://www.ices.dk/sites/pub/Publication%20Reports/Advice/2020/Special_Requests/eu.2020.04.pdf

ICES. 2021. OSPAR request to estimate bycatch mortality of marine mammals (harbour porpoise *Phocoena phocoena*, common dolphin *Delphinus delphis*, grey seal *Halichoerus grypus*) within the OSPAR maritime area. ICES Special Request Advice. Northeast Atlantic ecoregions. 6pp. https://ices-library.figshare.com/articles/report/OSPAR_request_to_estimate_bycatch_mortality_of_marine_mammals_harbour_porpoise_Phocoena_phocoena_common_dolphin_Delphinus_delphis_grey_seal_Halichoerus_grypus_within_the_OSPAR_maritime_area/18639914

ICES WGBYC. 2018. Report from the Working Group on Bycatch of Protected Species (WGBYC), 1–4 May 2018, Reykjavik, Iceland. ICES CM 2018/ACOM:25. 128pp.

ICES WGBYC. 2019. Working Group on Bycatch of Protected Species (WGBYC). ICES Scientific Reports. 1: 51. 163pp. http://doi.org/10.17895/ices.pub.5563

ICES WGBYC. 2020. Report from the Working Group on Bycatch of Protected Species (WGBYC) on Special Request in Bycatch Emergency Measures. ICES Scientific Reports.

ICES WGBYC. 2021. Report of the Working Group on Bycatch of Protected Species (WGBYC), ICES Scientific Reports. 3:107. 175 pp. http://doi.org/10.17895/ices.pub.9256

ICES WGBYC. 2022. Report of the Working Group on Bycatch of Protected Species (WGBYC), ICES Scientific Reports. 4:91. 265 pp. http://doi.org/10.17895/ices.pub.21602322

ICES WGMME. 2014. Report of the Working Group on Marine Mammal Ecology (WGMME), 10–13 March 2014, Woods Hole, Massachusetts, USA. ICES CM 2014/ACOM:27. 234 pp.

ICES WGMME. 2019. Working Group on Marine Mammal Ecology (WGMME). ICES Scientific Reports. 1: 22. 133 pp. https://doi.org/10.17895/ECES.pub.4980.

ICES WKEMBYC. 2020. Workshop on Fisheries Emergency Measures to Minimize Bycatch of Short-beaked Common Dolphins in the Bay of Biscay and Harbour Porpoise in the Baltic Sea (WKEMBYC). ICES Scientific Reports. 2: 43. 344pp. Available from: https://www.ices.dk/sites/pub/Publication%20Reports/Forms/DispForm.aspx?ID=36938

ICES WKMOMA. 2021. Workshop on estimation of mortality of marine mammals due to bycatch (WKMOMA). ICES Scientific Reports, 3:106. 95 pp. https://doi.org/10.17895/ices.pub.9257

IWC (International Whaling Commission). 1999. The Revised Management Procedure for baleen whales. Journal of Cetacean Research and Management (JCRM);1(Suppl), 251–8.

Jepson PD, Bennett PM, Deaville R, Allchin CR, Baker JR, Law RJ. 2005. Relationships between polychlorinated biphenyls and health status in harbour porpoises (*Phocoena phocoena*) stranded in the United Kingdom. Environmental Toxicology and Chemistry, 24(1): 238-248.

Jepson PD, Deaville R, Barber JL, Aguilar A, Borrell A, Murphy S, Barry J, Brownlow A, Barnett J, Berrow S, Cunningham AA, Davison NJ, ten Doeschate M, Esteban R, Ferreira R, Foote AD, Genov T, Giménez J, Loveridge J, Llavona A, Martin V, Maxwell DA, Papachlimitzou A, Penrose R, Perkins MW, Smith B, de Stephanis R, Tregenza N, Verborgh P, Fernandez A, Law RJ. 2016. PCB pollution continues to impact populations of orcas and other dolphins in European waters. Nature Scientific Reports 6:18573 | DOI: 10.1038/srep18573.

Jones EL, Hastie GD, Smout S, Onoufriou J, Merchant ND, Brookes KL, Thompson D. 2017. Seals and shipping: quantifying population risk and individual exposure to vessel noise. Journal of Applied Ecology, 54(6): 1930-1940. DOI: 10.1111/1365-2664.12911.

Kendall BE, Fujivara M, Diaz-Lopez J, Schneider S, Voigt J. 2019. Persistent problems in the construction of matrix population models. Ecological Modelling, 406, 33-43.

Kesselring T, Viquerat S, Brehm R, and Siebert U. 2017. Coming of age: - Do female harbour porpoises (*Phocoena phocoena*) from the North Sea and Baltic Sea have sufficient time to reproduce in a human influenced environment? PLOS ONE 12, e0186951. https://doi.org/10.1371/journal.pone.0186951

Kiely O, Lidgard D, McKibben M, Connolly N, Baines ME. 2000. Grey seals: Status and monitoring in the Irish and Celtic Seas. Maritime Ireland/Wales INTERREG Report No. 3.

King SL, Schick RS, Donovan C, Booth CG, Burgman M, Thomas L, Harwood J. 2015. An interim framework for assessing the population consequences of disturbance. *Methods in Ecology and Evolution*, 6(10), 1150-1158.

Lacy RC, Williams R, Ashe E, Balcomb III KC, Brent, JN, Clark CW, Croft, DP, Giles DA, MacDuffee M, Paquet PC. 2017. Evaluating anthropogenic threats to endangered killer whales to inform effective recovery plans. Scientific Reports, 7(1), 14119.

Langley I, Rosas da Costa Oliver T, Hiby L, Stringell TB, Morris CW, O'Cadhla O, Morgan L, Lock K, Perry S, Westcott S, Boyle D, Büche BI, Stubbings EM, Boys RM, Self H, Lindenbaum C, Strong P, Baines M, Pomeroy PP. 2020. Site use and connectivity of female grey seals (*Halichoerus grypus*) around Wales. Marine Biology, 167: 86 https://doi.org/10.1007/s00227-020-03697-8.

Law RJ, Allchin CR. 1994. Organochlorines in the blubber of bottlenose dolphins (*Tursiops truncatus*) and harbour porpoise (*Phocoena phocoena*) from Cardigan Bay (Wales) and the Moray Firth (Scotland). ICES CM 1994/(E+N) 9: 8pp.

Law RJ, Allchin CR, Morris RJ. 1995. Uptake of Organochlorines (Chlorophenyls, Dieldrin, Total PCG and DDT) in Bottlenose Dolphins (*Tursiops truncatus*) from Cardigan Bay, west Wales. Chemosphere, 30 (3): 547-560.

Leaper R, MacLennan E, Brownlow A, Calderan SV, Dyke K, Evans PGH, Hartny-Mills L, Jarvis D, McWhinnie L, Philp A, Read FL, Robinson KP, Ryan C. 2022. Estimates of humpback and minke whale entanglements in the Scottish static pot (creel) fishery. Endangered Species Research, 49, 217-232.

Lohrengel K, Evans PGH, Lindenbaum CP, Morris CW, Stringell TB. 2017. Bottlenose dolphin monitoring in Cardigan Bay 2014-2016. NRW Evidence Report No: 191, Natural Resources Wales, Bangor. Available at: https://naturalresources.wales/evidence-and-data/research-and-reports/marine-reports/marine-and-coastal-evidence-reports/?lang=en

Luck C, Jessopp M, Tully O, Cosgrave R, Rogan E, Cronin M. 2020. Estimating protected species bycatch from limited observer coverage: A case study of seal bycatch in static net fisheries. Global Ecology and Conservation, 24. DOI: 10.1016/j.gecco.2020.e01213

Maclean IMD, Frederikson M, Rehfisch MM. 2007. Potential use of population viability analysis to assess the impact of offshore windfarms on bird populations. BTO Research Report No. 480 to COWRIE. BTO, Thetford.

Mannocci L, Dabin W, Augeraud-Véron E, Dupuy J.-F, Barbraud, C, Ridoux V. 2012. Assessing the impact of bycatch on dolphin populations: the case of the common dolphin in the eastern North Atlantic. *PLoS One*, 7, e32615.

McMath AJ, Stringell TB. 2006. Grey seal pup production in Wales. SCOS Briefing Paper 06/11. Pp. 101-108. *In:* SCOS 2006. Scientific Advice on Matters Related to the Management of Seal Populations: 2006.

Moore JE, Read AJ. 2008. A Bayesian Uncertainty Analysis of Cetacean Demography and Bycatch Mortality Using Age-at-Death Data. Ecological Applications, 18, 1914-1931.

Morgan LH, Morris CW, Stringell TB. 2018. Grey Seal Pupping Phenology on Ynys Dewi / Ramsey Island, Pembrokeshire. NRW Evidence Report No: 156, 22 pp, Natural Resources Wales, Bangor.

Moritz C. 1994. Defining 'evolutionarily significant units' for conservation. Trends in Ecology & Evolution, 9, 373-375.

Morris WF, Doak DF. 2002. Quantitative Conservation Biology: Theory and Practice of Population Viability Analysis. Sinauer Associates, Sunderland, MA.

Morris C, Duck CD. 2019. Aerial thermal-imaging survey of seals in Ireland, 2017 to 2018. *Irish Wildlife Manuals*, No. 111 National Parks and Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Ireland.

Morris RJ, Law RJ, Allchin CR, Kelly CA, Fileman CF. 1989. Metals and organochlorines in dolphins and porpoises of Cardigan Bay, West Wales. Marine Pollution Bulletin, 20(10): 512-523.

Morris WF, Doak DF. 2002. Quantitative conservation biology: Theory and practice of population viability analysis. Sinauer, Sunderland, Massachusetts, USA.

Murphy S, Barber JL, Learmonth JA, Read FL, Deaville R, Perkins MW, Brownlow A, Davison N, Penrose R, Pierce GJ, Law RJ, Jepson, PD. 2015. Reproductive failure in UK harbour porpoises *Phocoena phocoena*: legacy of pollutant exposure? PLoS ONE, 10(7): e0131085. doi:10.1371/journal.pone.0131085

Murphy S, Dabin W, Ridoux V, Morizur Y, Larsen F, Rogan E. 2007. Estimation of Rmax for the common dolphin in the Northeast Atlantic. NECESSITY Contract 501605 Periodic Activity Report No 2 - Annex 8.4.

Murphy S, Evans PGH, Pinn E, Pierce GJ. 2019. Conservation management of common dolphins: lessons learned from the North-east Atlantic. Aquatic Conservation: Marine and Freshwater Ecosystems. DOI: 10.1002/aqc.3212

Murphy S, Law RJ, Deaville R, Barnett J, Perkins MW, Brownlow A, Davison N, Penrose R, Barber JL, Jepson, PD. 2019. Organochlorine contaminants and reproductive implication in cetaceans; A case study of the common dolphin. Pp. 3-38. *In:* M. C. Fossi and C. Panti (editors). Marine Mammal Ecotoxicology: impacts of multiple stressors on population health.

Murphy S, Petitguyot MAC, Jepson PD, Deaville R, Lockyer C, Barnett J, Perkins M, Penrose R, Davison NJ, Minto C. 2020. Spatio-Temporal Variability of Harbor Porpoise Life History Parameters in the North-East Atlantic. Frontiers in Marine Science. 7:502352. DOI: 10.3389/fmars.2020.502352

Murphy S, Pierce GJ, Law RJ, Bersuder P, Jepson PD, Learmonth JA, Addink M, Dabin W, Santos MB, Deaville R, Zegers BN, Mets A, Rogan E, Ridoux V, Reid RJ, Smeenk C, Jauniaux T, López A, Farré JMA, González AF, Guerra A, García-Hartmann M, Lockyer C, Boon JP. 2010. Assessing the effect of persistent organic pollutants on reproductive activity in common dolphins and harbour porpoises. NAFO/ICES/NAMMCO symposium "The Role of Marine Mammals in the Ecosystem in the 21st Century". Journal of Northwest Atlantic Fishery Science, 42, 153-173.

Murphy S, Winship A, Dabin W, Jepson PD, Deaville R, Reid RJ, Northridge SP. 2009. Importance of biological parameters in assessing the status of *Delphinus delphis*. Marine Ecology Progress Series, 388, 273-291. doi:10.3354/meps08129

Nabe-Nielsen J, Sibly RM, Tougaard J, Teilmann J, Sveegaard S. 2014. Effects of noise and by-catch on a Danish harbour porpoise population. Ecological Modelling, 272, 242-251.

Natural Resources Wales (NRW). 2020. NRW's position on determining Adverse Effect on Site Integrity for marine mammal site features in Wales in relation to potential anthropogenic removals (mortality) from marine developments. NRW Position Statement PS013. Available here (Natural Resources Wales / Marine mammal mortality and adverse effect on site integrity)

Niel C, Lebreton J.-D. 2005. Using demographic invariants to detect overharvested bird populations from incomplete data. Conservation Biology, 19, 826–835.

NMFS (National Marine Fisheries Service) 2005. Revisions to Guidelines for Assessing Marine Mammal Stocks. 24pp. http://www.nmfs.noaa.gov/pr/pdfs/sars/gamms2005.pdf

NAMMCO (North Atlantic Marine Mammal Commission) and the Norwegian Institute of Marine Research (IMR). 2019. Report of Joint IMR/NAMMCO International Workshop on the Status of Harbour Porpoises in the North Atlantic. Tromsø, Norway: North Atlantic Marine Mammal Commission. 235pp.

Northridge S, Cargill A, Coran A, Mandleberg L, Calderan S, Reid RJ. 2010. Entanglement of minke whales in Scottish waters: an investigation into occurrence, causes and mitigation. Sea Mammal Research Unit, Final Report to Scottish Government CR/2007/49. 57pp.

Northridge S, Kingston A, Thomas L. 2017. Annual report on the implementation of Council Regulation (EC) No 812/2004 during 2016. Sea Mammal Research Unit, St Andrews. 36pp.

Northridge S, Kingston A, Thomas L. 2018. Annual report on the implementation of Council Regulation (EC) No 812/2004 during 2017. Sea Mammal Research Unit, St Andrews. 34pp.

Northridge S, Kingston A, Thomas L. 2019. Annual report on the implementation of Council Regulation (EC) No 812/2004 during 2018. Sea Mammal Research Unit, St Andrews. 37pp.

Nowacek D, Thorne LH, Johnston DW, Tyack PL. 2007. Responses of cetacean to anthropogenic noise. Mammal Review, 37(2), 81-115.

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

Ó Cadhla O, Strong D, O'Keeffe C, Coleman M, Cronin M, Duck C, Murray T, Dower P, Nairn R, Murphy P, Smiddy P, Saich C, Lyons D, Hiby AR. 2008. An assessment of the breeding population of grey seals in the Republic of Ireland, 2005 . Irish Wildlife Manuals No. 34. National Parks & Wildlife Service, Department of the Environment, Heritage and Local Government, Dublin, Ireland. 60pp.

Olsen E. 2002. Errors in age estimates of North Atlantic minke whales when counting growth zones in bulla tympanica. Report of the International Whaling Commission, 4(2), 185-191.

Olsen E, Sunde J. 2002. Age determination of minke whales (*Balaenoptera acutorostrata*) using aspartic acid recemization technique. Sarsia, 87, 1-8.

Olsen MT, Andersen LW, Dietz R, Teilmann J, Harkonen TH, SiegismundI HR. 2014. Integrating genetic data and population viability analyses for the identification of harbour seal (*Phoca vitulina*) populations and management units. Molecular Ecology, 23, 815–831

Onoufriou J, Brownlow A, Moss S, Hastie G, Thompson D. 2019. Empirical determination of severe trauma in seals from collisions with tidal turbine blades. Journal of Applied Ecology. doi:10.1111/1365-2664.13388.

OSPAR IA. 2017. Harbour porpoise bycatch. OSPAR Intermediate Assessment. https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/biodiversity-status/marine-mammals/harbour-porpoise-bycatch/

Palsbøll PJ, Bérubé M, Allendorf FW. 2007. Identification of management units using population genetic data. Trends in Ecology & Evolution, 22, 11-16.

Penrose RS. 2010-2020. Marine Mammal and Marine Turtle Strandings (Welsh Coast). Annual Reports 2009-19. Marine Environmental Monitoring, Cardigan, Wales.

Pesante G, Evans PGH, Baines ME, McMath M. 2008a. Abundance and Life History Parameters of Bottlenose Dolphin in Cardigan Bay: Monitoring 2005-2007. CCW Marine Monitoring Report No. 61. Bangor: Countryside Council for Wales.

Pesante G, Evans PGH, Anderwald P, Powell D, McMath M. 2008b. Connectivity of bottlenose dolphins in Wales: North Wales photo-monitoring. CCW Marine Monitoring Report No. 62. Countryside Council for Wales, Bangor.

Plön S, Heyns-Viale ER, Smale MJ, Froneman PW. 2020. Life history parameters and diet of Risso's dolphins, *Grampus griseus*, from southeastern South Africa. Marine Mammal Science, 36, 786-801.

Quick NJ, Arso Civil M, Cheney B, Islas V, Janik V, Thompson PM, Hammond PS. 2014. The east coast of Scotland bottlenose dolphin population: Improving understanding of ecology outside the Moray Firth SAC. This document was produced as part of the UK Department of Energy and Climate Change's offshore energy Strategic Environmental Assessment programme.

Reed JM, Mills LS, Dunning Jr JB, Menges ES, McKelvey KS, Frye R, Beissinger SR, Anstett M-C, Miller P. 2002. Emerging Issues in Population Viability Analysis. Conservation Biology, 16(1), 7-19.

Richardson WJ, Greene CR, Malme CI, Thomson DH. 1995, *Marine Mammals and Noise*. Academic Press, San Diego, CA.

Robinson GJ, Clarke LJ, Banga R, Griffin RA, Porter J, Morris CW, Lindenbaum CP & Stringell TB. 2020. Grey Seal (*Halichoerus grypus*) Pup Production and Distribution in North Wales during 2017. NRW Evidence Report No. 293. 54pp. Bangor: Natural Resources Wales.

Robinson KJ, Hall AJ, Debier C, Eppe G, Thomé J-P, Bennett KA. 2018. Persistent Organic Pollutant Burden, Experimental POP Exposure, and Tissue Properties Affect Metabolic Profiles of Blubber from Gray Seal Pups. Environmental Science & Technology 52 (22): 13523-13534. DOI: 10.1021/acs.est.8b04240.

Robinson KP, Sim TM, Culloch RM, Bean TS, Aguilar IC, Eisfeld SM, Filan M, Haskins GN, Williams G, Pierce, GJ. 2017. Female reproductive success and calf survival in a North Sea coastal bottlenose dolphin (*Tursiops truncatus*) population. PLoS ONE 12:e0185000.

Rogan E, Breen P, Mackey M, Cañadas A, Scheidat M, Geelhoed S, Jessopp M. 2018. Aerial surveys of cetaceans and seabirds in Irish waters: Occurrence, distribution and abundance in 2015-2017. Department of Communications, Climate Action & Environment and National Parks and Wildlife Service (NPWS), Department of Culture, Heritage and the Gaeltacht, Dublin, Ireland. 297pp.

Russell DJF, Hastie GD, Thompson D, Janik VM, Hammond PS, Scott-Hayword LAS, Matthiopoulos J, Jones EL, McConnell BJ. 2016. Avoidance of wind farms by harbour seals is limited to pile driving activities. Journal of Applied Ecology, 53, 1642-1652.

Russell DJ, McConnell BJ, Thompson D, Duck CD, Morris C, Harwood J, Matthiopoulos J. 2013. Uncovering the links between foraging and breeding regions in a highly mobile mammal. Journal of Applied Ecology*,* 50(2), 499-509. https://doi.org/10.1111/1365-2664.12048.

Russell DJF, Morris CD, Duck CD, Thompson D, Hiby AR. 2019. Monitoring long-term changes in UK grey seal *Halichoerus grypus* pup production. Aquatic Conservation: Marine and Freshwater Ecosystems. DOI: 10.1002/aqc.3100.

Saavedra C. 2017. Multispecies population modelling of the common dolphin (*Delphinus delphis*), the bottlenose dolphin (*Tursiops truncatus*) and the southern stock of European hake (*Merluccius merluccius*), in Atlantic waters of the Iberian Peninsula. University of Vigo: PhD thesis. DOI: 10.13140/RG.2.2.35434.82881.

Schwarz, C. J., & Stobo, W. T. (2000). Estimation of juvenile survival, adult survival, and age-specific pupping probabilities for the female grey seal (*Halichoerus grypus*) on Sable Island from capture – recapture data, 253(1994), 247–253.

SCOS. 2012. Scientific Advice on Matters Related to the Management of Seal populations: 2011. Special Committee on Seals, UK. 174pp.

SCOS. 2018. Scientific Advice on Matters Related to the Management of Seal populations: 2017. Special Committee on Seals, UK. 154pp.

SCOS. 2020. Scientific Advice on Matters Related to the Management of Seal populations: 2019. Special Committee on Seals, UK. 166pp.

SCOS, 2022. Scientific Advice on Matters Related to the Management of Seal populations: 2021. Special Committee on Seals, UK. 266pp.

Smout S, King R, Pomeroy P. 2019. Environment-sensitive mass changes influence breeding frequency in a capital breeding marine top predator. Journal of Animal Ecology, 2019;00:1–13. https ://doi.org/10.1111/13652656.13128

Sinclair RR, Sparling CE, Harwood J. 2020. Review of demographic parameters and sensitivity analysis to inform inputs and outputs of population consequences of disturbance assessments for marine mammals. Scottish Marine and Freshwater Science Vol 11, No 14. 78pp.

Southall BL, Bowles AE, Ellison WT, Finneran JJ, Gentry RL, Greene CR, Kastak D, Ketten DR, Miller JH, Nachtigall PE, Richardson WJ, Thomas JA, Tyack PL. 2007. Marine mammal sound exposure criteria: Initial scientific recommendations. Aquatic Mammals, 33: 411-522.

Sparling CE, Lonergan M, McConnell B. 2018. Harbour seals (*Phoca vitulina*) around an operational tidal turbine in Strangford Narrows: No barrier effect but small changes in transit behaviour. Aquatic Conservation: Marine and Freshwater Ecosystems, 28, 194–204.

Sparling C, Smith, K, Benjamins S, Wilson B, Gordon J, Stringell T, Morris C, Hastie G, Thompson D, Pomeroy P. 2015. Guidance to inform marine mammal site characterisation requirements at wave and tidal stream energy sites in Wales. NRW Evidence Report No. 82.

Stevens A. 2014. A photo-ID study of the Risso's dolphin (*Grampus griseus*) in Welsh coastal waters and the use of Maxent modelling to examine the environmental determinants of spatial and temporal distribution in the Irish Sea. MSc thesis, University of Bangor. 97pp.

Stewart BS., Leatherwood S. 1985. Minke Whale *Balaenoptera acutorostrata*. Pp. 91-136. In: Handbook of Marine Mammals, Volume 3: The Sirenians and Baleen Whales (S.H. Ridgway & R. Harrison, Editors). Academic Press, London & New York.

Stott I, Hodgson D, Townley S. 2018. *popdemo*: Demographic Modelling Using Projection Matrices. R package version 1.3-0. https://CRAN.R-project.org/package=popdemo

Stringell TB, Millar CP, Sanderson WG, Westcott SM, McMath MJ. 2014. When aerial surveys will not do: grey seal pup production in cryptic habitats of Wales. Journal of the Marine Biological Association of the United Kingdom. 94(6), 1155-1159.

Strong P.G., Lerwill J., Morris S.R. & Stringell, T.B. (2006). Pembrokeshire marine SAC grey seal monitoring 2005. CCW Marine Monitoring Report No: 26; licensed version. 51pp.

Stubben CJ, Milligan BG. 2007. Estimating and Analysing Demographic Models Using the popbio Package in R. Journal of Statistical Software 22:11.

Sveegaard S, Galatius A, Dietz R, Kyhn L, Koblitz JC, Amundin M, Nabe-Nielsen J, Sinding M-HS, Andersen LW, Teilmann J. 2015. Defining management units for cetaceans by combining genetics, morphology, acoustics and satellite tracking. Global Ecology and Conservation, 3, 839-850. doi:10.1016/j.gecco.2015.04.002.

Taylor BL, DeMaster DP. 1993. Implications of non-linear density dependence. Marine Mammal Science 9, 360-371.

Taylor BL, Dizon AE. 1999. First policy then science: why a management unit based solely on genetic criteria cannot work. *Molecular Ecology*, 8, S11-S16.

Taylor BL, Chivers SJ, Larese J, Perrin, WF. 2007. Generation length and percent mature estimates for IUCN assessments of cetaceans. NOAA Administrative Report LJ-07-01. 24pp.

Taylor BL, Wade PR, De Master DP, Barlow J. 2000. Incorporating uncertainty into management models for marine mammals. Conservation Biology, 14, 1243e1252. https://doi.org/10.1046/j.1523-1739.2000.99409.x.

Thomas L, Russell D, Duck C, Morris C, Lonergan M, Empacher F, Thompson D, Harwood. J. 2019. Modelling the population size and dynamics of the British grey seal. Aquatic Conservation Marine and Freshwater Ecosystems, 29(S1), 6-23.

Thompson PM, Wilson B, Grellier K, Hammond, PS. 2000. Combining power analysis and population viability analysis to compare traditional and precautionary approaches to conservation of coastal cetaceans. Conservation Biology 14,1253–1263.

Tregenza NJC, Berrow SD, Hammond PS, Leaper R. 1997. Harbour porpoise (*Phocoena phocoena* L.) by-catch in set gillnets in the Celtic Sea. ICES Journal of Marine Science, 54, 896-904.

Tufto J, Sæther BE, Engen S, Swenson JE, Sandegren F. 1999. Harvesting strategies for conserving minimum viable populations based on World Conservation Union criteria: brown bears in Norway. Proceedings of the Royal Society of London B. Biological Sciences, 266 (1422), 961e967.

Vergara-Peña A. 2019. The Effects of Marine Tourism on Bottlenose Dolphins in Cardigan Bay. PhD thesis, Bangor University.

Vincent C, Fedak MA, McConnell BJ, Meynier L, Saint-Jean C, Ridoux V. 2005. Status and conservation of the grey seal, *Halichoerus grypus*, in France. Biological Conservation 126:62-73.

Vincent C, Huon M, Caurant F, Dabin W, Deniau A, Dixneuf S, Dupuis L, Elder JF, Fremau MH, Hassani S, Hemon A, Karpouzopoulos J, Lefeuvre C, McConnell BJ, Moss SEW, Provost P, Spitz J, Turpin Y, Ridoux V. 2017. Grey and harbour seals in France: Distribution at sea, connectivity and trends in abundance at haulout sites. Deep Sea Research Part II, 141, 294-305. DOI: 10.1016/j.dsr2.2017.04.004.

Wade PR. 1994. Managing populations under the Marine Mammal Protection Act of 1994: a strategy for selecting values for Nmin, the minimum abundance estimate, and Fr, the recovery factor. U.S. Dep. Commer., NOAA, National Marine Fisheries Service, Southwest Fisheries Science Centre Administrative Report LJ-94-19, 26pp.

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

Waggitt JJ, Evans PGH, Andrade J, Banks AN, Boisseau O, Bolton M, Bradbury G, *et al.* 2020. Distribution maps of cetacean and seabird populations in the North-East Atlantic. Journal of Applied Ecology, 57, 253-269. DOI: 10.1111/1365-2664.13525.

Wells RS, Scott MD. 1999. Bottlenose Dolphin *Tursiops truncatus* (Montagu, 1821). Pp. 137-182. In: SH Ridgway & R Harrison (Editors). Handbook of Marine Mammals. Vol. 6: The second book of dolphins. Academic Press, London.

Westcott SM. 2002. The distribution of grey seals (*Halichoerus grypus*) and census of pup production in North Wales 2001. CCW Contract Science Report, no.499, 150 pp.

Westcott SM, Stringell TB. 2003. Grey seal pup production for North Wales, 2002. CCW Marine Monitoring Report, no.5, 57pp.

Westcott SM, Stringell TB. 2004. Grey seal distribution and abundance in North Wales, 2002–2003. CCW Marine Monitoring Report, no.13, 80pp.

Wiig Ø. 1991. Demographic parameters for Norwegian grey seals, *Halichoerus grypus*. Fauna Norvegica, Series A. 12, 25-28.

Williams RS, Curnick DJ, Brownlow A, Barber JL, Barnett J, Davison NJ, Deaville R, ten Doeschate M, Perkins M, Jepson PD, Jobling S. 2021. Polychlorinated biphenyls are associated with reduced testes weights in harbour porpoises (*Phocoena phocoena*). Environment International, [106303]. https://doi.org/10.1016/j.envint.2020.106303

Williams RS, Curnick DJ, Barber JL, Brownlow A, Davison NJ, Deaville R, Perkins M, Jobling S, Jepson PD. 2020a. Juvenile harbor porpoises in the UK are exposed to a more neurotoxic mixture of polychlorinated biphenyls than adults. Science of the Total Environment, 708, 134835 https://doi.org/10.1016/j.scitotenv.2019.134835.

Williams R, ten Doeschate M, Curnick DJ, Brownlow A, Barber, JL, Davison NJ, Deaville R, Perkins M, Jepson PD, Jobling S. 2020b. Levels of polychlorinated biphenyls are associated with toxic effects in harbor porpoises (*Phocoena phocoena*) despite having fallen below proposed toxicity thresholds. Environmental Science and Technology. https://doi.org/10.1021/acs.est.9b05453.

Winship A, Berggren P, Hammond, P. 2008. Management framework to assess the impact of bycatch and recommend safe bycatch limits for harbour porpoise and other small cetaceans. St Andrews: Sea Mammal Research Unit.

Winship AJ. 2009. Estimating the Impact of Bycatch and Calculating Bycatch Limits to Achieve Conservation Objectives as Applied to Harbour Porpoise in the North Sea. Ph.D. thesis. St Andrews: University of St Andrews.

Wood DJ, Koprowski JL, Lurz PW. 2007. Tree squirrel introduction: a theoretical approach with population viability analysis. Journal of Mammalogy, 88(5), 1271e1279.

Øigård TA., Frie AK, Nilssen KT, Hammill MO. 2012. Modelling the abundance of grey seals (*Halichoerus grypus*) along the Norwegian coast. ICES Journal of Marine Science, 69, 1436–1447.

Data Archive Appendix

There are no data products or metadata associated with this project. Other products are archived for internal use on server–based storage at Natural Resources Wales within our Document Management System.

The archive contains:

[A] The final report in Microsoft Word and Adobe PDF formats (EVID-307-1256 and EVID-307-1257 respectively).

[B] A .zip file of the following PBR, PVA and iPCoD R code files (EVID-307-1248):

PBR.R - Code to run PBR for all six marine mammal species

PVA R code:

1_Load_demographic_rates.R - Code to load the demographic rates tested in the PVA for the five marine mammal species

2_Load_matrices.R - Code to load the matrices for each of the scenarios used in the PVA with and without density dependence

3_PVA_projections.R - Code to run and plot the deterministic PVA projections

4_Demographic_stochasticity.R - Code to run and plot the PVA demographic stochasticity simulations

5_Perturbation_analysis.R - Code to run the transfer function analysis and plot the results

6_Impact_projections.R - Code to run and plot the deterministic PVA projections including the impact of different pressures

iPCoD R code:

A folder containing iPCoD instructions (Word), subfolders for Grey Seal (GS), Bottlenose dolphin (BND) and Harbour Porpoise (HP), each containing several R files, and a subfolder for MacOSX containing all these files/folders.

Disclaimer: The associated R code files are considered "as is". No warranty is given for accuracy, completeness, or compatibility with current packages. NRW cannot provide user support and is not liable for any loss or issues resulting from reuse.

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