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Informing decisions on lethal control of great cormorant and goosander in Wales: scenarios from Population Viability Analysis

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Crynodeb Gweithredol

Mae poblogaethau eogiaid (*Salmo salar*) a siwin (*Salmo trutta*) yn dirywio yng Nghymru. Mae mesurau rheoli marwol ar gyfer adar pysgysol (yn benodol y fulfran *Phalacrocorax carbo* a'r hwyaden ddanheddog *Mergus merganser*), ar y cyd â saethu i'w dychryn, yn un cam posib y gellid ei gymryd i gefnogi'r gwaith o warchod salmonidau. Contractiwyd BTO gan CNC i gynnal Dadansoddiad o Hyfywedd Poblogaethau (PVA) i fodelu effeithiau posib senarios gwahanol o ran rheolaeth drwyddedig ar boblogaethau o adar sy'n gaeafu.

Defnyddiodd y modelau amcangyfrifon o boblogaethau mulfrain a hwyaid danheddog yn sgil arolygon o afonydd yng ngaeaf 2021-22 (Taylor et al., 2022). Cynhaliwyd PVAs gan ddefnyddio arffiniau isaf ac uchaf ar gyfer paramedrau demograffig oedd wedi'u cyhoeddi, eu cyfrifo a'u hamcangyfrif ac y cytunwyd arnynt ar y cyd gan Banel Arbenigol a oedd yn cynnwys Uwch wyddonwyr o BTO ac UKCEH. Cafodd pob cyfuniad posib o lefelau paramedrau eu modelu ar gyfer y ddwy rywogaeth, a defnyddiwyd dull samplu gwrthod i gadw setiau 'llinell sylfaen' terfynol y modelau, yr oedd eu hystod o drywyddau o ran poblogaeth yn adlewyrchu'r tueddiadau a arsylwyd ar gyfer poblogaeth y ddwy rywogaeth. Cymhwyswyd senarios o ran mesurau rheoli marwol ar gyfer pob rhywogaeth yn amrywio o ladd 0% i 15% o'r boblogaeth bob blwyddyn. Neilltuwyd canlyniadau'r model o dan bob senario i bum categori, yn ôl a oeddent yn rhagweld a) cynnydd/sefydlogrwydd yn y boblogaeth, b) gostyngiad heb sbarduno statws rhybudd cadwraeth, neu c) gostyngiad digonol i sbarduno newid mewn statws cadwraeth a ddiffinnir fel Rhybudd Arolwg o Adar y Gwlyptir o ddifrifoldeb canolig (Ambr) dros gyfnod byr (5 mlynedd), canolig (10 mlynedd) neu hirdymor (25 mlynedd).

O'r setiau cychwynnol o 62,500 o fodelau fesul rhywogaeth, roedd samplu gwrthod yn cadw digon o fodelau ar gyfer pob rhywogaeth (22,512 ar gyfer mulfrain a 12,208 ar gyfer hwyaid danheddog) i ganlyniadau modelau dilynol gael eu cyflwyno fel tebygolrwyddau. O dan y senario llinell sylfaen (gan gynnwys cyflwyno mesurau rheoli marwol yn gymharol ddiweddar yng Nghymru), roedd y mwyafrif (64.5%) o'r modelau ar gyfer mulfrain yn rhagweld dirywiad yn y boblogaeth, ond dim ond lleiafrif bach (6.3%) o fodelau ar gyfer hwyaid danheddog oedd yn rhagweld y boblogaeth yn dirywio. Ar gyfer y ddwy rywogaeth, roedd lefelau cynyddol o fesurau rheoli marwol yn gysylltiedig â thebygolrwydd cynyddol o ddirywiad yn y boblogaeth, ac yn gysylltiedig â newidiadau negyddol o ran statws cadwraeth dros amserlenni cynyddol fyrrach.

Mae modelau PVA yn tybio bod paramedrau demograffig yn sefydlog dros amser, ac eto mae tystiolaeth yn awgrymu bod nifer o baramedrau demograffig yn newid, yn enwedig ar gyfer mulfrain. Mae cynhyrchiant yn gostwng mewn nythfeydd mulfrain yng Nghymru, ac mae gwaith dadansoddi goroesiad gyda data'r modrwyau o'r nythfa fwyaf sydd gan fulfrain yng Nghymru (Ardal Gwarchodaeth Arbennig Ynys Seiriol) yn awgrymu newid hirdymor mewn cyfraddau goroesi. Os caiff cynhyrchiant sy'n gostwng ei gynnwys yn senarios y modelau, roedd y tebygolrwydd o ddirywiad yn y boblogaeth yn cynyddu'n sylweddol o'i gymharu â modelau oedd yn defnyddio amcangyfrifon o gynhyrchiant sefydlog. Dangosodd dadansoddiadau ychwanegol fod amrywiad ar hap (stocastigrwydd demograffig ac amgylcheddol) fel tywydd garw yn ychwanegu natur anrhagweladwy at ganlyniadau'r modelau.

Mae'r dadansoddiadau hyn yn tynnu sylw at yr angen hanfodol am fonitro priodol ar y poblogaethau er mwyn sicrhau y gellir canfod dirywiadau anrhagweledig mewn poblogaethau a'u hystyried mewn penderfyniadau trwyddedu yn y dyfodol. Mae hyn yn arbennig o bwysig i rywogaethau sydd â thueddiadau demograffig gwaeth, megis mulfrain. Yn Lloegr, mae gwaith blynyddol i fonitro mulfrain yn cael ei ddarparu ar hyn o bryd drwy fynegai sy'n gysylltiedig â'r Arolwg o Adar y Gwlyptir. Gwerthuswyd dull tebyg (gydag addasiadau allweddol ar gyfer sefyllfa Cymru) ar gyfer mulfrain a hwyaid danheddog, er y bydd angen dull mwy cymhleth a phwrpasol o gasglu data ar gyfer hwyaid danheddog.

Executive summary

Populations of Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta*) are declining in Wales. Licensed lethal control, in combination with shooting to scare, of piscivorous birds (specifically great cormorant *Phalacrocorax carbo* ("cormorant") and goosander *Mergus merganser*) is one possible action that may be undertaken to support salmonid conservation. NRW contracted BTO to undertake Population Viability Analysis (PVA) to model the potential impacts of different scenarios of licensed control on wintering bird populations.

Modelling used cormorant and goosander population estimates resulting from the winter 2021-22 river surveys (Taylor *et al*., 2022). PVAs were run using lower and upper bounds for published, calculated and estimated demographic parameters collectively agreed by an Expert Panel consisting of Senior scientists from BTO and UKCEH. All possible combinations of parameter levels were modelled for both bird species, and a rejection sampling procedure applied to retain final 'baseline' model sets whose range of population trajectories reflected the observed population trends for each species. Scenarios of lethal control were applied for each species ranging from 0% to 15% of the population removed per annum. Model outcomes under each scenario were assigned to five categories, according to whether they predicted a) population increase/stability, b) decline without triggering conservation alert status, or c) decline sufficient to trigger a change in conservation status defined as a medium-severity (Amber) Wetland Bird Survey Alert over short (5-year), medium (10-year) or long-term (25-year) timescales.

From the initial sets of 62,500 models per species, rejection sampling retained sufficient models for each species (22,512 for cormorant and 12,208 for goosander) for subsequent model outcomes to be presented as probabilities. Under the baseline scenario (including the relatively recent introduction of lethal control in Wales), the majority (64.5%) of models for cormorant predicted population decline, whereas only a small minority (6.3%) of models for goosander predicted population decline. For both species, increasing levels of lethal control were associated with increasing likelihood of population decline, and associated with negative changes in conservation status over progressively shorter timescales.

PVA modelling assumes demographic parameters are stable over time, yet evidence suggests several demographic parameters are changing, especially for cormorant. Productivity is declining at Welsh cormorant colonies, and survival analysis of ring-recovery data from the largest Welsh cormorant colony (Ynys Seiriol/Puffin Island Special Protection Area) suggests long-term change in survival rates. If declining productivity is included in model scenarios, the likelihood of population decline was substantially increased compared to models using stable productivity estimates. Additional analyses demonstrated that random variation (demographic and environmental stochasticity) such as adverse weather adds unpredictability to model outcomes.

These analyses highlight the critical need for appropriate population monitoring to ensure that unpredicted population declines can be detected and factored into future licensing decisions. This is particularly important for species with poorer demographic trends, such as cormorant. In England, annual cormorant monitoring is currently delivered through an index linked to the Wetland Bird Survey. A similar approach (with critical modifications for the Welsh situation) was evaluated for cormorant and goosander, although goosander will require a more complex and targeted approach to data collection.

1. Introduction

Fish populations in the rivers and still-waters of Wales are an important natural resource, a key part of our freshwater ecosystems that also provide a resource for the enjoyment of recreational angling. However, the current status of Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta*) populations in Welsh rivers is of conservation concern, with Atlantic salmon in particular now considered 'At Risk' or 'Probably at Risk' in all Wales' 23 principal salmon rivers. Both fish species are generally in decline. These declines are driven by a range of pressures operating on both the freshwater and marine phases of their life cycle; known pressures in freshwaters including habitat quality, water quality and the potential impacts of predatory birds (NRW, 2020).

There has been significant research in the UK and internationally into the impacts of fish-eating birds (particularly the two races of great cormorant *Phalacrocorax carbo*, hereafter referred to as "cormorant") on fisheries, and potential strategies for mitigating and managing these impacts. Predation is a universal natural process, and prey choice by generalist predators is primarily a function of prey availability and size. Robust evidence of survival and productivity in predated fish species is lacking, as is robust evidence for population-level impacts of predation by cormorant and goosander *Mergus merganser*. It is, however, recognised that conflicts between fish-eating birds, native fish populations and stocked stillwater fisheries are complex (Cooke, 2021) and involve a range of stakeholders across a spectrum of natural and human made aquatic habitats. These conflicts are also sensitive to factors such as the population dynamics of both birds and fish; change in biotic and abiotic features of the freshwater environments (e.g. bankside and in-stream structures, water quality), and variations in external factors, particularly climatic conditions affecting water flow (Harris *et al.*, 2008).

It is widely accepted that for predation to have an impact at population level it must represent additive mortality for the prey species – i.e. mortality that is not compensated by increased survival, productivity or fitness of the remaining individuals (Carss and Russell, 2022). Salmonid biology suggests that most of their juvenile freshwater life-stages are density-dependent, permitting compensation for predation; but that density-dependence declines as the juvenile fish develop (Harris *et al*., 2008). Mortality in the final (smolt) stage, when juvenile fish migrate downstream to the sea, is not compensated by increased survival or fitness of the remaining individuals, and therefore represents the most vulnerable stage of juvenile development to additive mortality from predation. However, it is difficult to measure predation by generalist bird predators (as distinct from bird presence, bird density or rates of fishing activity by birds) as they are opportunistic generalists; it has also proved similarly difficult to measure the severity of any impact of fish-eating birds on fish populations. Seas, lakes and rivers are complex systems, and it is also difficult to separate the effects of bird predation from other critical factors affecting migratory fish populations (Wiik Vollset *et al.,* 2022).

The management of potential impacts of three fish-eating species of birds (cormorant, goosander and red-breasted merganser *Mergus serrator*) on wild fish populations and stocked fisheries have been the focus of policy reviews in England (Defra, 2013) and in Scotland (Harris *et al*., 2008). In Wales, the majority of licences issued for the purposes of conserving fish populations or for preventing serious damage to fisheries are for the lethal control of cormorant and goosander.

Recognising the complexity of bird-fish interactions, in 2018 Natural Resources Wales ("NRW") established a Fish-eating Birds Advisory Group ("the Advisory Group") in response to concerns about the impact of fish-eating birds - specifically cormorant and goosander - on natural and stocked fisheries in Wales as well as on the conservation status of classified fish features of European sites. The purpose of the Advisory Group is predominately to review relevant evidence and develop recommendations concerning the management of salmonid predation by cormorant and goosander and to contribute to the NRW Salmon Plan of Action (NRW, 2020) to address pressure on salmon and sea trout populations in Welsh rivers. The work of the Advisory Group also forms an integral part of a broad and comprehensive review currently being undertaken by NRW in their approach to regulating the lethal control of wild birds in Wales.

Population status of cormorant and goosander in Wales

A recent survey of overwintering fish-eating bird populations in Wales concluded that approximately 2,894 cormorants (95% confidence interval: 2,580–3,259) overwinter on Welsh rivers, estuaries and stillwaters (Taylor *et al.*, 2022). This number is likely to be substantially higher than the number present during the breeding season, when the salmon smolt run also occurs (Macgregor *et al.*, 2022). At both UK national scale and in Wales, numbers of overwintering cormorants are increasing (Frost *et al.*, 2021). Breeding populations in the UK are increasing over 23-year timescales but stable or slightly declining over 10-year timescales (Harris *et al.*, 2020), possibly reflecting the impacts of increased lethal control in England in recent decades. Breeding Bird Survey trends are not published for cormorant in Wales due to insufficient data, but data obtained for this study (see below) similarly suggests stability or decline over 10-year timescales.

Cormorants in the UK belong to one of two subspecies; the primarily coastal-breeding *P. carbo carbo*, and the 'continental' *P. carbo sinensis*, which more frequently forms inland breeding colonies. Much of the long-term increase in cormorant populations at UK national scale is thought to be attributable to the expansion of *P. c. sinensis*, particularly in England, with coastal colonies of *P. c. carbo* remaining stable or declining over the same time period (Newson *et al.*, 2005). It is therefore important to note that there is, to date, no evidence for the occurrence of inland-breeding cormorants in Wales, and that the Welsh breeding population is therefore currently considered to be coastal-breeding *P. c. carbo* (Pritchard *et al.*, 2021). The status of the overwintering population is less clear, and along with unknown net migration of *carbo* birds is likely to include an unknown but significant proportion of *P. c. sinensis* individuals.

Regarding goosander, the winter survey concluded that approximately 1,460 goosanders (95% confidence interval: 1,223–1,750) overwinter on Welsh rivers, estuaries and stillwaters (Taylor *et al.*, 2022). This number is likely to be reasonably similar to the number present during the breeding season and early in the smolt run, although males appear to depart on moult-migration at around the same time as the smolt run is taking place (Macgregor *et al.*, 2022). Goosander populations are stable or declining at UK national scale (Harris *et al.*, 2020, Frost *et al.*, 2021), but increasing in Wales, where goosander is considered to be colonising (Frost *et al.*, 2021, Pritchard *et al.*, 2021).

Legal framework and Welsh licensing regime for cormorant and goosander

Like all wild birds, cormorant and goosander are protected under the Wildlife and Countryside Act 1981, as amended, ("the Act"). Under section 16(1) of the Act, NRW as the licensing authority may grant licences to carry out lethal control of wild birds for a range of purposes. NRW currently grants two types of licences under section 16(1) of the Act, allowing the killing or taking of wild birds and/or destruction of eggs and nests: specific licences and general licences. Cormorant and goosander are not general licence listed species and therefore any lethal control is authorised under a specific licence. Before granting a licence for any purpose listed in section 16(1) NRW must be satisfied that, as regards that purpose, there is no other satisfactory solution.

There is a legislative requirement in the Wild Birds Directive ("the Directive"), underpinned in UK legislation by the Conservation and Habitat and Species Regulations 2017 (as amended), that there should be no deterioration in the conservation status of wild birds subjected to derogated licenced control from the provisions of Articles 5 and 8 of the Directive. Although the UK is no longer an EU member state subject to the Birds Directive, the terms of the Birds Directive are still relevant under Regulation 9(1) of Conservation of Habitats and Species Regulations 2017. In England, the licensing Authority, in this case Natural England, sets a national upper limit for the number of cormorants that can be killed to ensure that licensed removal does not negatively affect this species' conservation status in England. The Animal and Plant Health Agency (formerly The Central Science Laboratory) model the consequences of the licensing policy on the English cormorant population. There is no comparable approach in England for goosander, nor in Wales for either cormorant or goosander.

Population models

Population viability analysis (PVA) has become a commonly used tool in conservation biology and in the management of threatened or endangered species (Keedwell, 2004). PVA is a general term for demographic predictive models which forecast the robustness of a population to scenarios of risk (e.g. threats or predictive pressures to a population, extinction) comparative to an unimpacted baseline (Beissinger *et al.*, 2006). Currently there is a lack of evidence to inform guidance as to which PVA model (e.g. abundance time-series models, Leslie matrix models, semi-integrated population models) is best suited to the species of the populations being analysed. Although different PVA types can be useful tools for conservation managers, a quality PVA cannot be run without sufficient demographic data (e.g. population estimate) on the target species (Keedwell, 2004). Where sufficient evidence on demographic and environmental variability in demographic parameters exists, stochastic models can be used; otherwise models are fitted deterministically.

Natural England ("NE") commissioned work to develop a PVA modelling framework, applicable to seabirds at a variety of scales (Searle *et al.*, 2019). This framework is a front-end, interactive web application user interface to allow users to set-up, apply and run their own PVA models for seabird species without the need for access to specific software. A key development objective of the modelling tool was to allow users the flexibility to explore population effects in circumstances of a defined impact; in this case impacts of seabird mortality as a consequence of colliding with offshore wind turbines. The modelling tool can be used to assess any type of impact that changes any avian species survival or productivity rate, or as a cull or licensed removal of a fixed size per year (Searle *et al.*, 2019). In other words, the generic nature of the tool is such that it can be applied to other groups of birds.

The NE seabird modelling tool is available to other users under an Open Government Licence.

Aims and scope of this report

An improved understanding of the status of cormorant and goosander populations in Wales is needed to help ensure conservation requirements are met (i.e. maintaining the birds' conservation status) and to inform NRW licensing decisions. It remains vital that a reasonable balance is struck between protecting Welsh fisheries, including salmon conservation, and the conservation status of Welsh cormorant and goosander populations. Population-level impacts of licensed control of cormorant and goosander in Wales are not well understood, with difficulties caused by limited data on demographic metrics (e.g. first-year survival, annual productivity). As the licensing authority in Wales, NRW require predictive models to assess population effects of different licensed control scenarios to inform the option to develop licensing thresholds that are evidence-based. BTO were contracted by NRW to:

i) Develop and test PVA models against a range of impact scenarios (e.g. different levels of licensed control, ranging from no control, through to levels that resulted in negative population change) to determine their effects on wintering populations of cormorant and goosander.
ii) Consider whether it is appropriate for future PVA modelled scenarios to take account of the annual rate of change in the Welsh cormorant and goosander population using appropriate national survey data e.g. Wetland Birds Survey.
iii) Review the cormorant modelling approach used in England and appraise its applicability to Wales to inform future licensed control

The findings from this work will inform a final report and recommendations paper from the Fish-eating Advisory Group to NRW.

2. Methods

2.1 BTO approach

Several concerns were raised and discussed with NRW at the tender stage and during initiation meetings at the start of this work; these are described below and the approach agreed in each case is summarised.

First, implementation and use of the Natural England population viability analysis (NEPVA) tool in its online form as a closed interactive web application is not compatible with BTO internal policies (e.g. BTO Code of Good Scientific Practice; "Development of project materials should be tracked through the use of a version control system"). This is because modelled outputs from the NEPVA web app (i.e. population projections) are dependent upon data inputs that must be manually entered into a web form, with the possibility for undetected typographic errors or incorrect selections from options menus. This process is not fully reproducible and creates substantial concerns around quality assurance. The underpinning R code for the NEPVA app is available through GitHub, therefore BTO carried out this work to an acceptable standard of transparency and accuracy using the R programming environment.

A second, and serious, concern in using the NEPVA tool is its core assumption that the population being modelled is a closed population of breeding individuals. This assumption is violated for the wintering populations being modelled, since breeding birds of both cormorant and (to a lesser extent) goosander in Wales migrate seasonally across country boundaries, and both species are supplemented outside the breeding season by individuals from breeding populations from elsewhere in the UK and continental Europe (Pritchard *et al.,* 2021; Wernham *et al.,* 2002; Hearn, 2015). It is therefore essential to clearly define the population being modelled as a geographically-defined mixed wintering population (in both species). It is also critical that the anticipated seasonal timing of lethal control is clarified prior to the commencement of modelling work. There are several related implications, each of which is discussed below; and each of which also represents an important caveat that increases uncertainty around the outcomes of lethal control.

a) If lethal control takes place when migrant individuals are present (i.e. potentially any time outside the breeding season), there is an unknown (and variable, from year to year) probability that each individual killed belongs either to the resident Welsh breeding population, a population from elsewhere in the UK, or to a continental population. If a continental bird is removed, the effect of this control on recruitment to the following winter's immigrant birds is unclear, but likely to be complex. These probabilities cannot be explicitly modelled using existing data since it is not currently possible to identify wintering individuals of either species to race or breeding population in the field. Extensive colour-marking studies and/or genetic and morphological examination of all killed birds would be required to provide a means of understanding these probabilities.

b) The Welsh population estimates for cormorant and goosander underpinning the population modelling (Taylor *et al.*, 2022) are derived from a census of ten catchments carried out during mid-winter (2020/21), and therefore include both an unknown proportion of the resident breeding population of each species, and an

additional and unknown number of immigrant overwintering birds. If lethal control is intended to take place during mid-winter, the 2021 population estimate is a suitable basis for modelling, with appropriate caution, but still contravenes the NEPVA requirement for a closed population. If lethal control is intended to take place at *any other time of year*, the population size will differ; in the case of cormorant, is likely to be substantially smaller (Macgregor *et al.*, 2022). Using winter population estimates to model sustainable licensed control thresholds to be conducted outside the winter season (e.g. early autumn, or during the smolt-run in spring) is likely to result in an unsustainably high number of birds being controlled, with population-level consequences. On this basis it was confirmed that population modelling will be conducted based on lethal control taking place during the winter months, accepting that the NEPVA modelling assumption of closed populations is violated. For the purpose of modelling, it will be assumed for both species that control mortality is equally likely to affect resident and overwintering individuals.

Thirdly, population viability analyses are parameterised using a range of demographic metrics (e.g. brood size, adult and immature mortality). Where possible, these metrics should be accurate, contemporary and relevant to the population being modelled. For cormorant, the NEPVA package is already populated with such metrics, but it is unclear whether their use would be appropriate since NEPVA's focus during development was on seabirds, and therefore presumably refers to the coastal-breeding *Phalacrocorax carbo carbo* rather than inland-breeding *P. c. sinensis.* Significant differences in at least some key metrics have been identified between Welsh coastal-breeding and English inland-breeding cormorant colonies (Newson *et al.*, 2005). For goosander, data availability is much poorer, with few of the necessary metrics available in peer-reviewed or even grey literature. Given these data constraints, BTO convened an expert panel with relevant expertise in cormorant and goosander population ecology (including Dr David Carss, UKCEH), to agree informed lower and upper bounds for each demographic parameter. In order to properly reflect uncertainty, all possible combinations of agreed lower and upper bounds and quartiles were modelled. Upper control limits will be based on worst-case scenarios (i.e. demographic metrics at the lower bounds agreed by the expert panel). This precautionary approach will minimise the risk of parameter uncertainty or inaccuracy suggesting a higher level of removal than is sustainable.

Finally, the evidence for density-dependence in either species is limited, and inconclusive (Frederiksen and Bregnballe, 2000a). The Welsh goosander population, increasing in both numbers and range, are very unlikely to be subject to density-dependence at present. Cormorant, being colonial, may be subject to density-dependent process at colony level; but such processes are unlikely to operate across the entire Welsh breeding population (in multiple colonies) nor on a wintering population that is both open and a mixture of two ecologically distinct subspecies (*P. c. carbo* and *P. c. sinensis*). Applying a density-dependent model, whereby lethal control results in slight increases in survival or productivity due to reduced competition, might represent ecological reality for either species at some spatial scales; but basing thresholds for lethal control on such a model would risk introducing inappropriate positive bias, since density-dependent processes do not exist in all populations or under all conditions and circumstances. BTO therefore applied a density-independent approach, which complies with the requirement for a cautious approach to modelling the impacts of lethal control at the scale of national populations.

BTO expert workshop

A workshop to discuss and agree demographic modelling parameter bounds was held (over teleconference link) and included Dr Rachel Taylor, Dr Rob Robinson, Dr Niall Burton, Dr Stuart Newson (all BTO staff), with Dr David Carss (UKCEH); and Dr Callum Macgregor (BTO) acting as facilitator. The agreed demographic parameter bounds are presented below, including a brief explanation of the discussion, evidence or expert opinion underpinning each decision; they are also summarised in Table 1.

2.2 Demographic metrics: cormorant

Maximum brood size

Although the NEPVA package defaults to six for cormorant, all participants (including those with substantial experience of monitoring cormorant nests in Wales/the UK) agreed that they had never encountered broods larger than five; and only infrequently larger than four. The panel discussed any potential impacts on the model, since preliminary testing of the modelling environment (Callum Macgregor *pers. comm*.) suggested this variable was only important when fitting stochastic models, and unlikely to influence model outputs when using the expected productivity values. **It was therefore agreed that maximum brood size be set at four (lower bound) and five (upper bound) respectively.**

Age at first breeding

Participants were aware of evidence that individual cormorants may commence breeding at any point between two to six years of age (Newson, 2000). However, all participants agreed that the majority of cormorant are likely to recruit to the breeding population at around three years old; so both modelled **lower and upper bounds were set to three years**.

Productivity

Productivity reported from a range of UK cormorant colonies under the JNCC Seabird Monitoring Programme (SMP) ranges between approx. 1.3–2.4 per pair, with Welsh colonies within this sample ranging from 1.38–2.36. The panel discussed a range of data sources from colonies in the UK (Wernham and Peach, 1999; Newson, 2000; Newson *et al.*, 2005) and abroad (Hogan, 1979; Bregnballe and Gregersen, 1997). Participants agreed that a Welsh national population mean was unlikely to fall at the lower extreme of this range; by comparison, the lowest identified colony-scale mean from studies of a range of North American colonies was 1.63 (Hogan, 1979). However, the panel did not feel able to rule out the upper extreme; therefore, **lower and upper bounds were set at 1.63 and 2.40** respectively.

Adult survival

The best available evidence was drawn from a 20-year capture-recapture study at a single colony of *P. c. sinensis* in Denmark, which found a mean adult survival rate of 0.88, varying annually between 0.74–0.95 (Frederiksen and Bregnballe, 2000b). Panel participants could not agree a narrower range of values within this region, so the **lower and upper bounds were set at 0.74 and 0.95** respectively.

Immature survival

Survival of first-year birds was reported by several studies to be substantially lower than adult survival. Studies in various UK colonies (primarily of *P. c. carbo*) reported first-year survival rates ranging between 0.240–0.455 (Wernham and Peach, 1999), whilst a study of the previously cited Danish colony (*P. c. sinensis*) found mean first-year survival to be 0.58 (annual variation 0.42–0.75; Frederiksen and Bregnballe, 2000b). Participants agreed that studies in continental colonies (*P. c. sinensis*) were likely to report higher rates of first-year survival than is relevant for Welsh colonies of *P. c. carbo*, so the upper extremes of this range were discounted. The **lower and upper bounds were set at 0.24 and 0.58** respectively.

Multiple sources reported that survival of second-year birds was substantially higher than for first-year birds. The Danish study (*P. c. sinensis*) reported second-year birds to have similar survival rates to adults (Frederiksen and Bregnballe, 2000a), but data from UK colonies (Wernham and Peach, 1999) suggest that survival in the second year was around 80% that of adult survival on average (range 72.5–87.5%). Multiplying these values by the agreed lower and upper bounds for adult survival yielded a range of 0.54–0.83, which was approximately in agreement with reported values for second-year survival in Wernham and Peach (1999) of 0.57–0.78. Therefore, the **lower and upper bounds were set at 0.54 and 0.83** respectively.

Deferral rate

Participants agreed that deferred breeding is likely to occur in cormorants, but that no good data is available on its frequency. It was also highlighted that, in practical terms, including deferred breeding in the model would account for a significant proportion of birds commencing breeding at later than three years old (Newson, 2000). In the absence of further evidence, it was agreed to set relatively wide **lower and upper bounds of 0.0 and 0.2** respectively for this parameter.

2.3 Demographic metrics: goosander

Maximum brood size

As for cormorant, it was agreed that accuracy in setting this metric for goosander was of little importance because it is irrelevant for deterministic models, and unlikely to have any influence on stochastic models for species with large broods and low chick survival (a description that applies well to goosander). Various sources of evidence suggested single-mother broods of over ten eggs were possible, but the true maximum is hard to assess due to the frequent occurrence of intraspecific brood parasitism; broods of 14 or more were considered likely to have multiple mothers in some studies (Eriksson and Niittylä, 1985). It was therefore agreed to set **lower and upper bounds of 11 and 14** respectively.

Age at first breeding

All participants agreed that individual goosanders are likely to commence breeding at either two or three years old, and that the true population mean could be either value; therefore, **lower and upper bounds of two and three** respectively were set.

Productivity

No reliable data could be sourced to provide evidence for productivity in goosander. A study of the closely-related red-breasted merganser *Mergus serrator* in Scotland found that productivity was relatively low (0.7–1.8 well-grown ducklings per female) (Marquiss and Duncan, 2008). Panel participants considered the increasing population trend for Welsh goosander and agreed that higher productivity than red-breasted merganser was likely. A single 3-year study from Scotland (Marquiss *et al.*, 1998) provided data on the proportion of broods resulting in at least one well-grown duckling close to independence (25–64%) and the mean size of broods of such well-grown ducklings (5.4–7.4); describing productivity as a function of the two. Assuming the majority of duckling mortality takes place in younger ducklings (such that mean brood size of well-grown ducklings is similar to the number of birds reaching independence), these values yield a range of productivity between 1.35–4.74. Although this was a wide range, participants felt that there was insufficient evidence to support any narrower range of values, so the **lower and upper bounds were set at 1.35 and 4.74** respectively.

Adult survival

Evidence from a study of goosander survival across a range of states and territories in North America found wide variation between 0.1–0.89 (Pearce *et al.*, 2005). Participants agreed that the lower extremes of this range were not relevant to the Welsh situation, since legal hunting of this species is widespread in North America but non-existent in Wales. Based on these data and a review of survival rates in similar-sized species of ducks (BTO Birdfacts website (Robinson, 2005)), participants set **lower and upper bounds of 0.7 and 0.9** respectively.

Immature survival

The only direct evidence for immature survival rates in goosander comes from a single study of eight birds satellite-tagged in Alaska (Pearce and Petersen, 2009), of which just two individuals survived their first winter (i.e. survival = 0.25). For similar reasons to the discussion of adult survival, participants agreed that this value was likely to be lower than the Welsh situation. In the absence of any further evidence, it was proposed to adopt the same lower and upper bounds as for cormorant (0.24–0.58), on the grounds that the two species are not very dissimilar in size, overwinter in similar locations and exploit a largely shared food resource. Evidence from a range of similar-sized ducks, including mallard *Anas platyrhynchos* (first-year survival 0.49–0.55 (Arnold and Clark, 1996; McDougall and Amundson, 2017)), northern shoveler *Spatula clypeata* (first-year survival 0.32 (Arnold and Clark, 1996)), common eider *Somateria mollissima* (first-year survival 0.52 (Bårdsen *et al.*, 2018) or 0.75 (Nicol-Harper *et al.*, 2021)) and the (smaller) Eurasian wigeon *Mareca penelope* (first-year survival 0.22–0.29 (Guillemain *et al.*, 2013)) provided support for the true rate of goosander first-year survival being likely to fall within this range. **Lower and upper bounds were set at 0.24 and 0.58** respectively.

For models with age at first breeding set to three years, survival in second-year birds was assumed to be equal to adult survival (lower and upper bounds of 0.7 and 0.9 respectively).

Deferral rate

As for cormorant, participants agreed that deferred breeding can occur in goosander, but that no good data was available on its frequency. Therefore, it was again agreed to set relatively wide **lower and upper bounds of 0.0 and 0.2** respectively for this parameter.

Table 1. Modelled parameter estimates. Lower and upper bounds for each parameter were agreed by an expert panel, and the median and quartiles between these bounds calculated to generate intermediate values. All possible combinations of parameter estimates in this table were modelled (except quartiles were not modelled for maximum brood size or age at first breeding). In total, 62,500 candidate models per species, including a small amount of duplication for cormorant (because 'age at first breeding' was fixed at 3; i.e. the lower and upper bounds were the same, but both were modelled).

Species	Variable	Lower bound	Lower quartile	Median	Upper quartile	Upper bound
Cormorant	Initial population size (year 2020)	2580	2750	2920	3089	3259
	Maximum brood size	4	-	-	-	5
	Age at first breeding	3	-	-	-	3
	Productivity	1.63	1.8225	2.015	2.2075	2.40
	Adult survival	0.74	0.7925	0.845	0.8975	0.95
	Immature survival (1st year)	0.24	0.325	0.410	0.495	0.58
	Immature survival (2nd year)	0.54	0.6125	0.685	0.7575	0.83
	Breeding deferral rate	0.00	0.05	0.10	0.15	0.20
Goosander	Initial population size (year 2020)	1223	1355	1487	1618	1750
	Maximum brood size	11	-	-	-	14
	Age at first breeding	2	-	-	-	3
	Productivity	1.35	2.1975	3.045	3.8925	4.74
	Adult survival	0.70	0.75	0.80	0.85	0.90
	Immature survival (1st year)	0.24	0.325	0.410	0.495	0.58
	Immature survival (2nd year)	0.70	0.75	0.80	0.85	0.90
	Breeding deferral rate	0.00	0.05	0.10	0.15	0.20

2.4 General modelling parameters applied to both species

Density dependence

The question of using density-dependent or independent models for cormorant or goosander was specifically addressed prior to running any models. The expert group additionally reviewed and discussed this question.

The expert panel participants reviewed the published evidence for cormorant (Frederiksen and Bregnballe 2000), specifically discussing colony and race dynamics of the species. The panel confirmed that this evidence is derived solely from a detailed long-term study of a

single colony of *P. c. sinensis* (in Denmark), and agreed that this was insufficient evidence to robustly conclude that density-dependent processes act either in individual colonies of *P. c. carbo*, or on the population as a whole; the violated assumption of a closed population was also raised here. Cormorant, being colonial, may be subject to density-dependent process at colony level; but such processes are unlikely to operate across the entire Welsh breeding population (in multiple colonies) nor on a wintering population that is both open and a mixture of two ecologically distinct subspecies (*P. c. carbo* and *sinensis*). Consequently, all participants agreed that including density dependence in cormorant models as an *a priori* assumption would introduce a positive bias to all modelled outcomes: predicting lower levels of population decline than would be expected in the absence of density-dependent processes.

The Welsh goosander population, a colonising species increasing in both numbers and range, are very unlikely to be subject to density-dependence at the current time. It should be noted that such processes cannot be ruled out in future, if the population continues to increase or stabilises in the absence of increased mortality, particularly through licensed control.

In summary, inappropriately applying a density-dependent model, whereby lethal control results in slight increases in survival or productivity due to reduced competition, might represent ecological reality at very small spatial scales (colony or reach) for either species; but basing thresholds for lethal control on such a model would introduce inappropriate positive bias, since density-dependent processes do not exist in all populations or under all conditions and circumstances. BTO therefore applied a density-independent approach to both species, which also complies with the stated requirement for a cautious approach to modelling the impacts of lethal control at the scale of national populations.

Constrained productivity

All participants agreed that there was no evidence of second breeding attempts being made after a successful attempt in either cormorant or goosander. All models are therefore constrained by maximum brood size.

2.5 Analytical approach

Modelling environment

Population viability analysis was conducted using the NEPVA R package (Searle *et al.*, 2019), version 4.17. The NEPVA package is designed to underpin a user-friendly Shiny interface that can be used through a standard web browser, and consists of a suite of non-compiled, interrelated R functions; it is not, therefore an R package *sensu stricto*. Instead, the NEPVA suite provides a general-function environment in which to use population viability analyses (PVAs) to assess population-level consequences of anthropogenic activities upon population demographic rates (Searle *et al.*, 2019). Analyses were undertaken using the underlying R packages rather than the Shiny app to facilitate reproducibility, and to allow the rapid, streamlined processing of large numbers of scenarios.

The most recent version of the packages (v4.17) was downloaded into a local directory on 4th October 2021 (from https://github.com/naturalengland/Seabird_PVA_Tool); in order to provide maximum reproducibility, this version of the package has been archived alongside

the R scripts used for analysis, in the BTO's institutional Github repository (https://docs.github.com/en/repositories/creating-and-managing-repositories/about-repositories accessed 11/03/2022).

A range of options are available within the NEPVA package to conduct PVAs in different ways. For the purposes of this study, the following options were agreed upon. Environmental and demographic stochasticity were eliminated from models (by setting model.envstoch = "deterministic" and model.demostoch = FALSE), because it was felt that too little data was available to make accurate estimates of intra- and interannual variability in productivity and survival (especially for goosander). Instead, estimation of model uncertainty was conducted by modelling large numbers of combinations of input parameter levels, and inspecting variation between outcomes (see below). As described above, it was agreed to construct models without density dependence (model.dd = "nodd") and with productivity constrained by brood size (model.prodmax = TRUE). An option to include a number of "burn-in" years to allow age structure within the model to settle was used, because the existing age structure of Welsh populations of cormorant and goosander is unknown; initial trials suggested that a default value of 11 burn-in years (used in several examples in the NEPVA package) was sufficient for age structure to stabilise in all cases. Because models were fully deterministic, there was no need for multiple simulations of each model (sim.n = 1 and sim.seed = NULL).

All models were fitted with initial population values from the Welsh winter census conducted in 2020-21 (Taylor *et al.,* 2022). These yielded lower and upper bounds of 2,580–3,259 for cormorant, and 1,223–1,750 for goosander, respectively. Impacts (of lethal control) were modelled to begin in 2022 and continue annually, with models simulating 25 years of impacts (i.e. ending in 2047). This modelling period is unaffected by the inclusion of pre-2022 model 'burn-in' years, which effectively take place in the lead-up to 2022 without affecting starting population size.

We were unable to get the breeding deferral rate flag "demobase.bskippc" to work as intended. To allow modelling to proceed to schedule, it was determined that the agreed values for deferral rate could be treated as a modifier to per-nest productivity, in order to achieve the same demographic outcome. Therefore, for each model, baseline productivity was multiplied by (1 - deferral rate), and the outcome used for productivity in the model. Since the lower bound for deferral rate was set at 0 for both species, this approach also encapsulated potential scenarios where deferred breeding might have already been factored into the population-level productivity values used by the expert panel to parameterise models (i.e. where population-level productivity was given, being a function of nest-level productivity and deferral rate).

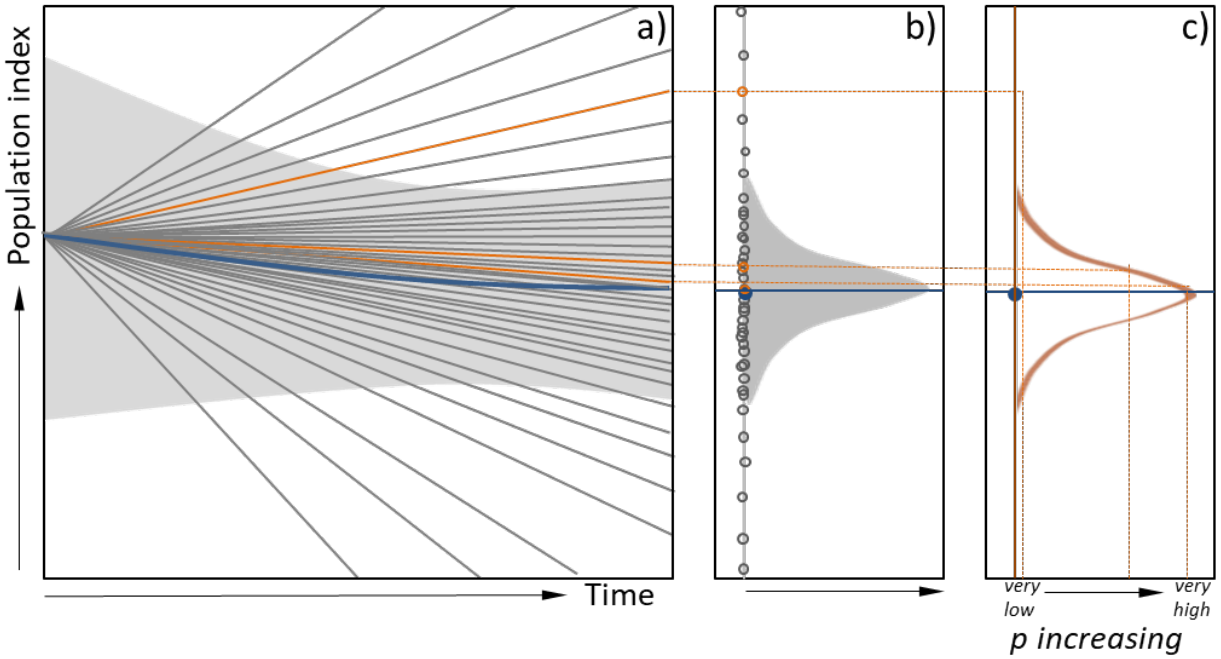
Model construction

The above procedure generated a total of 62,500 possible combinations of parameters for each species, although these sets contained a small number of duplicates within species where rounded values for maximum brood size (mbs) and age at first breeding (afb) were the same across multiple levels. Each model parameter with the exception of maximum brood size and age at first breeding provided five input levels – upper bound, lower bound, mean and quartile values. We modelled all possible combinations of these values for initial population size, productivity, adult survival, first- and second-year immature survival, and deferral rate. To increase modelling efficiency and reduce the occurrence of duplicate

models, we modelled only lower and upper bounds for maximum brood size and age at first breeding.

Removal (i.e. lethal control) was applied as a modifier to survival of all age classes. This is intended to represent the likely real-world situation, since reliably determining the age of cormorants and goosanders in the field (unless in the hand) is challenging. After running the baseline scenario, and considering the population trends of both species, an initial broad subset of models was run (baseline, 5%, 10% and 15% removal, respectively). After considering the significant changes in population trend created by these scenarios and discussing preliminary results with NRW, we further refined the lower-removal model sets by adding all increments of 0.5 between the baseline scenario ("0%") and 6% removal (13 scenarios), in order to provide a higher-resolution range of scenarios predicting outcomes of up to ~50% probability of causing population decline for both species, and also retained the two initial steps of 10% and 15% removal respectively.

Figure 1. Illustration of the rejection sampling approach used to refine model sets. For a hypothetical species BBS trend (panel a); population trend line in blue and 95% confidence intervals in light grey), the full set of population trends created by all combinations of all plausible parameters includes trends leading to every endpoint including rapid expansion and rapid extinction (darker grey lines). In panel b), a cross-section of the modelled endpoint, the bootstrapped 95% confidence interval is presented as a probability distribution of likely population estimates (grey) centred on the BBS trend (blue). Modelled endpoints fall both within and outside this distribution. In panel c) this probability distribution around the species trend is used to assign the probability that any individual theoretical model is included in the 'accepted model set' for the species – three example models (highlighted in orange) describe very low, mid- and very high probabilities of inclusion in the final set.



Some combinations of parameters (e.g. upper bounds used for all three of productivity, adult and immature survival) produced population estimates in the baseline (0% removal) scenario which were clearly not consistent with observed population trends in the focal species (e.g. population explosion, or extremely rapid extinction). Although each parameter within these models was individually plausible, in such cases the overall combination was

not plausible. To narrow down the total set of 62,500 candidate models per species to a set of models that could plausibly explain observed population trends, we calibrated baseline models against population trends from the BTO Breeding Bird Survey (BBS), using Approximate Bayesian Computation (Hartig *et al.*, 2011). We used ten-year trends calculated over the period 2008–18 for all BBS squares in Wales where the focal species was observed (cormorant: 23 squares; goosander: 11 squares), and applied a rejection sampling approach whereby the probability of each model's acceptance was determined by the Probability Distribution Function (PDF) of the 10-year trend estimates, which was approximated by bootstrapping (Harris *et al.*, 2020). This process is presented visually in Figure 1.

This means that models predicting a rate of population change similar to the central BBS trend estimate were relatively more likely to be accepted, and models predicting a rate of change far from the central BBS trend estimate (i.e. the extremes of the PDF) were more likely to be rejected. This yielded a final set of 22,512 plausible accepted models for cormorant, and 12,208 plausible accepted models for goosander. 95% of the accepted models predict population trends falling within the 95% confidence intervals of the observed BBS trends. For cormorant this is -50.89 to +68.32% over 10 years, equating to a population change rate of 0.93–1.05 per year; and for goosander -6.31 to +200.81%, equating to a population change rate of 0.99–1.12 per year).

Declining productivity in cormorants

Change over time in any variable used to parameterise these models will alter the accuracy of their predictions. The observed long-term decline in per-nest productivity reported for colonies of cormorant in Wales is a current example (Cook and Robinson, 2010; JNCC, 2020). This decline is estimated at a rate of 0.027 chicks per nest per year. If this decline continues, models parameterised according to present-day or historical productivity data will provide an overly optimistic forecast of population viability in the long-term. To provide some assessment of the risk presented by this scenario, we approximated the effect of 10 further years of sustained productivity decline by repeating the full modelling procedure for cormorant, using adjusted lower and upper bounds for productivity of 1.36 and 2.13 respectively (i.e. 0.27 less than the lower and upper bounds agreed during the expert workshop). We then selected the same set of 22,512 models for further analysis that, with the agreed range of productivity values, had been accepted during rejection sampling.

Stochastic survival

Stochasticity (e.g. year-to-year variation in survival rates (Frederiksen and Bregnballe, 2000a)) plays a substantial role in shaping long-term population trends. Here, models were fitted deterministically due to a lack of evidence about variance in many of the parameters. However, stochastic effects on population dynamics are generally predicted to make longer-term population projections less reliable (Saether and Engen, 2002), and indeed under some circumstances may increase rates of decline and extinction risk over time (e.g. Saether *et al.*, 1998), such that our deterministic models may represent a relatively optimistic set of outcomes. In effect, this is because a random percentage decline requires a larger percentage increase to recover from: e.g. a 50% decline followed by a 50% increase would result in an overall 25% decline, whereas a 100% increase would be necessary to result in stability in this example.

To explore this possibility, we refitted a single deterministic model with stochasticity incorporated for annual survival in cormorants; and separately with stochasticity incorporated for productivity as well. We selected a demographic parameter set from the accepted models for cormorant that was expected to predict approximate population stability when modelled deterministically, and replaced the three survival parameters (adult, first-year and second-year survival) with modelled estimates based on ring-recovery data (dead recoveries) for a major Welsh breeding colony (see section 2.3.4 for full details of the survival analysis). We fitted this model once deterministically (as above), and also fitted 50 replicates (across two runs) incorporating stochasticity in the parameters. In the first run, 25 replicates were fitted with stochastic variance in survival, set (for each age class) as the standard error around estimated survival from the combined analysis of all ring types fitted post-2006 (Table 2), with all other parameters (including productivity) fixed as for the deterministic model. In the second run, 25 replicates were fitted with stochastic variance in survival as above, and stochastic variance in productivity set as the difference between median and quartiles drawn from the workshop lower and upper bounds (in the absence of evidence for the true variability of this parameter). Stochasticity was modelled as environmental stochasticity (i.e. year-to-year variation in population means) with a beta distribution (the most appropriate statistical distribution for a random continuous variable bounded between 0 and 1) for all parameters. All other parameters were set identically for stochastic and deterministic models. We extracted raw annual population estimates from each model for inspection.

Analysis of model outputs

Waterbird species in the UK are primarily monitored through the Wetland Birds Survey (WeBS), which includes categories of population-decline based on percentage declines in smoothed population trends over defined time periods. Where large (>25%) increases or declines in populations over 5, 10 or 25-year timescales are observed, “WeBS Alerts” are triggered, specific to the severity (“medium” or “high”) and timescale (“short-term”, “medium-term” or “long-term”). This provides a means of categorising modelled population change in response to management in terms of the likelihood of triggering WeBS Alerts at different timescales.

From the subset of accepted models, we used predicted population trends to assess the proportion of models that fell into each of five possible outcomes (which were collectively non-overlapping and all-encompassing). Specifically, we assessed the proportion of accepted models in each removal scenario that predicted: (i) population increase or stability (no change in conservation status); (ii) population decline, below WeBS Alert thresholds; and (iii) population decline triggering a WeBS Alert (i.e. a medium-Alert (Austin *et al.*, 2019)), which was further subdivided into (1) decline triggering a long-term WeBS medium-Alert (i.e. 25% decline over a 25-year period); (2) decline triggering a medium-term WeBS medium-Alert (i.e. 25% decline over a 10-year period); and (3) decline triggering a short-term WeBS medium-Alert (i.e. 25% decline over a 5-year period). Medium-Alerts for decline were used because they represent the most conservative threshold for detecting a decline, but note that high-Alerts can also be triggered by 50% decline over the same timescales, and that positive Alerts signifying rapid population increase are also possible (Austin *et al.*, 2019).

Additionally, we used population estimates generated by each accepted model to calculate 95% Confidence Intervals (CI) for the predicted populations of cormorant and goosander, by taking the relevant quartiles from the full set of population estimates, in each year and

under each removal scenario. These were used to estimate a 95% confidence interval for the number of birds for subsequent removal (by multiplying the 95% CI population estimate by the relevant percentage of the population for removal) in each year, under each scenario, of which the lower confidence interval represents a **conservative maximum** number of birds for removal in each year. Note that removal of this maximum number of birds is only conservative in a limited sense: it is highly likely that the number of birds removed will not accidentally exceed the target population percentage (in the absence of up-to-date population estimates), but under some scenarios, removal of this number of birds may still result in a change in conservation status.

Declining productivity

Using the same set of models that, with unadjusted productivity inputs, had been accepted by rejection sampling, we repeated the above process incorporating ten years of simulated productivity decline as described above. We directly compared the proportions of models predicting each outcome between the two model sets.

Cormorant survival analysis

A long-term study ringing and colour-marking pre-fledging cormorants on Ynys Seiriol/Puffin Island Special Protection Area (SPA), the largest cormorant breeding colony in Wales (Pritchard *et al.*, 2021), offers the possibility to directly estimate regionally-relevant survival rates for this species. This analysis was not part of the review process by the expert panel, since it represents unpublished new analysis of ringing data rather than a peer-reviewed study or the opinion of experts in the absence of other data. The Expert Panel based its lower and upper bounds for cormorant survival in three age classes (first-year, second-year and adult) primarily upon studies of a single Danish colony of *P. c. sinensis* (Frederiksen and Bregnballe, 2000a, 2000b), along with a study of various British colonies (Wernham and Peach, 1999), all of which were published over 20 years ago. Rather than vary our approach between parameters (and species), it was decided to use this dataset to provide an independent test of the procedure of parameter estimation by the Expert Panel.

We used data from the Ynys Seiriol/Puffin Island SPA ringing study to estimate survival of Welsh *P. c. carbo*, in the same three age classes, and critically compared these survival rates to those set by the panel. Under this scheme, since 1982 a sample of approx. 200 (range 105–284) juvenile cormorants have been metal-ringed each year in the nest, typically during late June/early July, at the Ynys Seiriol/Puffin Island SPA colony, Wales (lat: 53.317, lon: -4.028). Every year since 2006 (except 2012), an additional 49–50 birds have been fitted with colour rings (Darvic engraved with a three-letter code, as well as a metal ring). To date, 7,749 birds have been fitted with a metal ring only (of which 2,153 were type A clip-rings, and 5,596 were type B split-rings), and 699 birds have also been fitted with a colour ring (8,448 birds ringed in total). Resightings and recoveries of metal-ringed and colour-ringed birds have been collated and compiled, with the dataset used for analysis up to date to 24th October 2021.

Survival was estimated using the package RMark (as an interface to Mark software v. 9.0). Resightings (= “recaptures”) and recoveries were assigned as alive or dead as appropriate. Assignation to ‘year’ was done using an offset threshold Julian date of 91 (1st April in a non-leap year); this ensured that recoveries of birds that died during their first winter were treated as first-year birds, and so forth (the closest approximation to survival *sensu* the PVA models). A Burnham joint live recapture-dead recovery survival model (Burnham, 1993) was

constructed, with survival and fidelity allowed to vary between age classes and the probabilities of live recaptures and of dead recoveries held constant. Three independent survival models were constructed with this structure, using data from colour-rings (fitted 2006–21), A-type metal clip-rings (fitted 1983–86, 1988–94, plus 10 in 1997), and B-type metal split-rings (fitted 1982–83, 1987, and 1994–2021) respectively, since probabilities of recapture/recovery were expected to vary between ring types (Steve Dodd, *pers. comm*., 2022). In addition, an overall model was fitted for all ring types fitted in the period 2006–21, with probability of live recaptures and dead recoveries modelled as a function of ring type.

3. Results

3.1 Accepted models

In total, the rejection sampling procedure selected 22,512 plausible accepted models for cormorant, and 12,208 plausible accepted models for goosander. Models were accepted or rejected based on their predicted rate of population change, in proportion to the Probability Distribution Function (PDF) for ten-year trends in Wales (based on BBS data). Differences between the two species' model sets were shaped by this process, such that they primarily reflect differences in the species' trends rather than outcomes of selection of different parameters.

The disparity in the number of accepted models should not be interpreted as an indication that the procedure performed better or worse for either species: in both cases, the outcome is a set of accepted models that accurately represents the probability distribution of each species' population trends. The important consideration here is not the number of models *per se,* but that the accepted model set is representative of the statistical distribution of likelihood for that species' observed population trend. It is also clear that fewer models were selected for goosander because the wider bounds set by the Expert Panel for goosander parameters (relative to cormorant) generated a broader spread of model outcomes, meaning that proportionally fewer goosander models fell in the plausible middle-ground described by the population trend. For both species, the accepted models were drawn from the central peak of the distribution of all models (Figs 2a–d).

The vast majority of accepted models for goosander (93.7%) predicted positive population trends in the baseline (0% removal) scenario, reflecting that the majority of the PDF for goosander trends was similarly positive (c.f. 95% confidence interval: -6.31 to +200.81%). By comparison, accepted models for cormorant were more evenly split between positive and negative trends, with the majority (64.5%) predicting population decline in the baseline scenario.

Retrospective examination of demographic parameters

We examined the relative frequency of use in accepted models of each of the five possible levels for each parameter (excluding initial population size, which did not influence the rates of population change used during rejection sampling, and both maximum brood size and age at first breeding, for which only two levels were employed). Selection of different parameter levels was somewhat variable. The rate of selection for each level was expected to be proportional to the rate at which models were accepted from the initial set of 62,500

models. However, in some cases, accepted models were much more likely to use certain levels of a parameter than others. This can be considered to provide some feedback as to the likely true value of each parameter. However, the modelling approach did not fit models with a full range of prior values, and therefore is insufficient to treat the mean value for each parameter as a true posterior. Consequently, although a useful perspective on demographic metrics, these mean values are not recommended for use as defined parameters in future modelling.

For the majority of variables, a lack of clear pattern provides support for the true parameter value lying between the lower and upper bounds selected by the expert panel; although an alternative interpretation would be that the parameter in question has little or no influence on the population outcome. Very frequent selection of an intermediate parameter level provides clearer evidence for the true parameter value lying within the range selected by the expert panel.

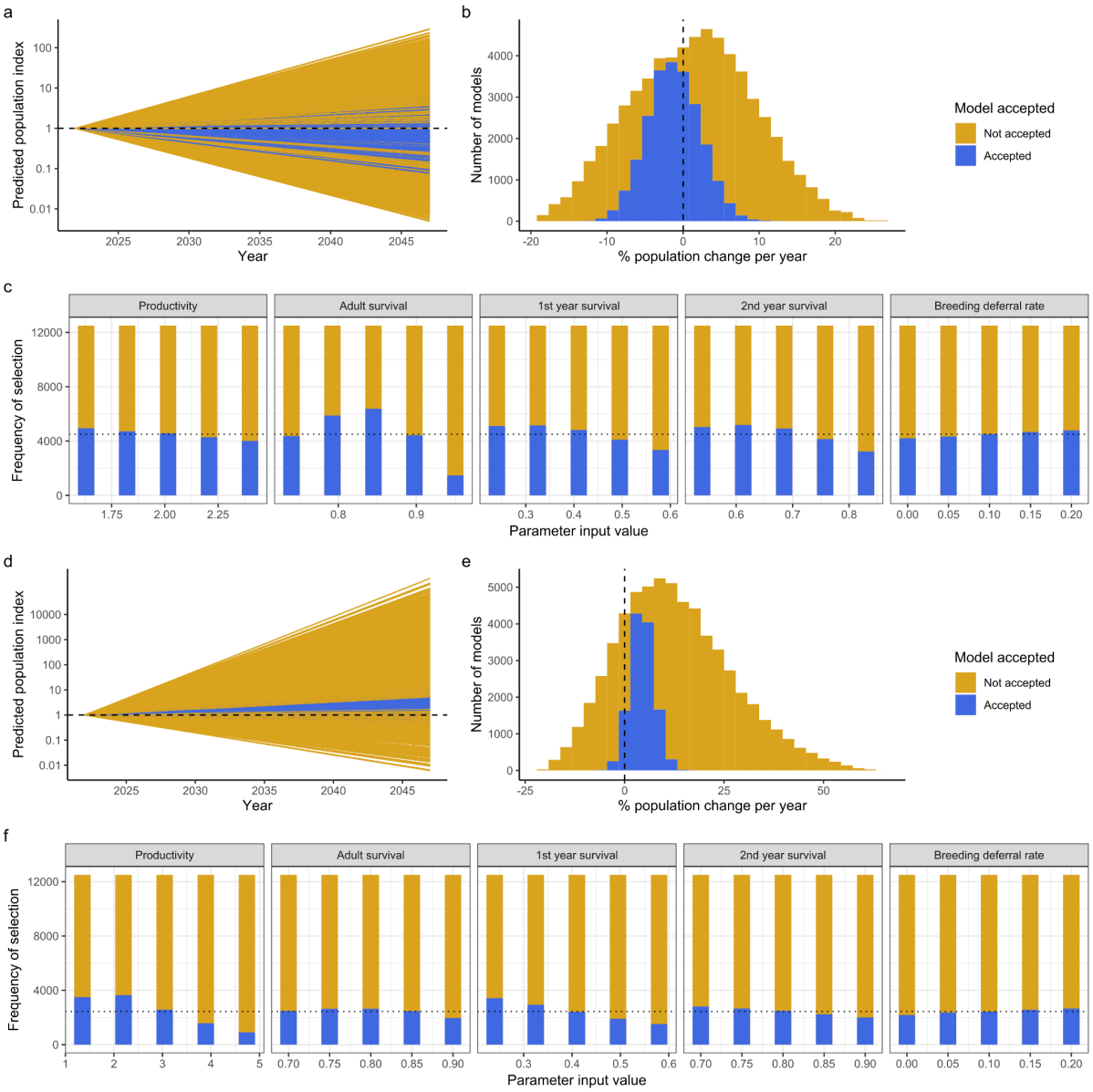
Among cormorant models, a large proportion of accepted models used the median level for adult survival (0.845, see Table 1) and correspondingly few used the lower and upper bounds. Both first- and second-year survival showed weaker patterns of high selection frequency for their respective lower quartile values (Fig. 1e).

For goosander models, the strongest pattern was high selection frequency of lower values for productivity (especially the lower quartile, 2.1975; Fig 2f). A relatively clear pattern for frequent selection of the lower bound for first-year survival in goosander (0.24) is the only instance suggestive of a true value potentially falling outside the modelled range (in this case, below it); nonetheless, some 12.4% of accepted models selected the *upper* bound (0.58) for this parameter (Table 2), so population trends in goosander can be plausibly explained by a large number of models incorporating relatively high rates of first-year survival.

Table 2. Parameter selection frequencies. Number of times each parameter level was used in accepted models, and percentage of total accepted models. Each parameter level was tested in a total of 12,500 models, forming 20% of the initial model set of 62,500. Geometric mean of all parameter values used in accepted models is also presented, except breeding deferral rate, for which arithmetic means are presented because 0 was used as a parameter level. Note, the modelling approach used is insufficient to treat the mean value for each parameter as a true posterior for use as the parameter value in future modelling.

Species	Variable	Geometric mean	Lower bound	Lower quartile	Median	Upper quartile	Upper bound
Cormorant	Productivity	1.98	4943 (22.0%)	4709 (20.9%)	4576 (20.3%)	4276 (19.0%)	4008 (17.8%)
	Adult survival	0.83	4371 (19.4%)	5879 (26.1%)	6374 (28.3%)	4423 (19.6%)	1465 (6.5%)
	Immature survival (1st year)	0.38	5102 (22.7%)	5149 (22.9%)	4814 (21.4%)	4101 (18.2%)	3346 (14.9%)
	Immature survival (2nd year)	0.66	5031 (22.3%)	5182 (23.0%)	4922 (21.9%)	4145 (18.4%)	3232 (14.4%)
	Breeding deferral rate	0.10	4208 (18.7%)	4334 (19.3%)	4525 (20.1%)	4662 (20.7%)	4783 (21.2%)
Goosander	Productivity	2.33	3502 (28.7%)	3645 (29.9%)	2579 (21.1%)	1574 (12.9%)	908 (7.4%)
	Adult survival	0.79	2485 (20.4%)	2641 (21.6%)	2643 (21.6%)	2475 (20.3%)	1964 (16.1%)
	Immature survival (1st year)	0.36	3426 (28.1%)	2936 (24.0%)	2418 (19.8%)	1915 (15.7%)	1513 (12.4%)
	Immature survival (2nd year)	0.79	2805 (23.0%)	2664 (21.8%)	2491 (20.4%)	2243 (18.4%)	2005 (16.4%)
	Breeding deferral rate	0.10	2181 (17.9%)	2368 (19.4%)	2435 (19.9%)	2570 (21.1%)	2654 (21.7%)

Figure 2. Selection of accepted demographic models by rejection sampling from 62,500 candidates per species. The full set of 62,500 models per species, incorporating all possible combinations of input parameters, is depicted in orange, with accepted models (those selected by rejection sampling, by comparison with the Probability Density Function for observed ten-year trends in BBS squares in Wales) depicted in blue. (a and d) Rates of change relative to a starting index of 1. (b and e) Number of models predicting different rates of population change. (c and f) Number of models utilising each input level for each parameter. Models are shown separately for cormorant (a–c) and goosander (d–f). Dotted lines indicate population stability (change index =1, % change =0).



3.2 Analysis of model outputs

Cormorant

The baseline scenario represents the model sets that match the observed population trends of cormorant and goosander; for cormorant this includes both the birds controlled under licence in Wales, and the unknown number similarly controlled in England (Pritchard *et al*., 2021, SCAN Ringing Group *pers comm.* 2022). Under this baseline scenario, 51.9% of accepted models predict rates of population decline consistent with a change in conservation status (i.e. triggering a WeBS alert), with just over a third of accepted models (35.5%) predicting no change in conservation status (population stability or increase; Fig. 3, Table 3). As the proportion of the population to be removed annually increased, so a larger proportion of accepted models predicted a decline resulting in change in conservation status, and a correspondingly smaller proportion predicted population increase. Under the 15% removal scenario, 99.8% of accepted models predicted population decline resulting in a change in conservation status (of which 96.2% of accepted models predicted a WeBS alert would be triggered after only five years).

Uncertainty in modelled population estimates increased over time (Appendix 1: Table S1), and showed considerable uncertainty within a relatively short period of time, driven by the wide range of plausible baseline trends (against which rejection sampling was conducted) including models of both increase and decline (Fig. 1). Consequently, uncertainty around the estimated number of cormorants to be removed in each year under each scenario also increased over time (Appendix 1: Table S2), but the conservative maximum number of cormorants to be controlled (i.e. the lower confidence interval of this estimate) decreased under all scenarios to below 10 birds per year after 25 years (Appendix 1: Table S3).

Goosander

Under the baseline scenario, the vast majority of accepted models (93.7%) predict continued population increase. As for cormorant, increasing the proportion of the population to be removed annually resulted in a larger proportion of models predicting a change in conservation status (declines triggering WeBS Alert). (Fig. 3). Under the 15% removal scenario, 93.2% of accepted models predicted a decline resulting in such a change in conservation status (Table 3).

Similarly to cormorant, uncertainty in modelled population estimates increased substantially over time, but unlike cormorant, many scenarios had relatively stable lower confidence intervals with rapidly-widening upper confidence intervals (Appendix 1: Table S4). This meant that, although the conservative maximum estimated number of goosanders to be removed in each year under each scenario also decreased under all scenarios except 0.5% removal, it did so relatively more gradually in lower-removal scenarios (Appendix 1: Tables S5–6).

Table 3. Percentage of models (from the subset of plausible models) that predict each of five possible outcomes: (i) population increase or stability (no change in conservation status); (ii) population decline, below WeBS Alert thresholds; and (iii) population decline triggering a WeBS Alert, which was further subdivided into (1) decline triggering a long-term WeBS medium-Alert (25% decline over a 25-year period); (2) decline triggering a medium-term WeBS medium-Alert (25% decline over a 10-year period); and (3) decline triggering a short-term WeBS medium-Alert (25% decline over a 5-year period).

Species	% population removed per year	% models predicting population increase	% models predicting population decline below alert thresholds	% models predicting change in conservation status (to alerted decline)	Of which, % models predicting a WeBS alert triggered after…		
					25 years	10 years	5 years
Cormorant	0	35.5	12.6	51.9	19.2	22.6	10.1
	0.5	31.6	12.6	55.8	18.4	24.8	12.6
	1	28.1	12.1	59.8	18.6	26.4	14.8
	1.5	24.5	11.4	64.1	18.8	27.9	17.5
	2	21.2	10.4	68.4	18.8	28.9	20.7
	2.5	18.2	10.0	71.8	18.3	29.7	23.8
	3	15.3	9.4	75.3	17.4	30.2	27.7
	3.5	13.0	8.4	78.6	16.4	30.9	31.2
	4	11.1	7.5	81.4	15.7	30.7	35.0
	4.5	9.3	6.4	84.3	14.2	30.8	39.3
	5	7.8	6.0	86.6	13.4	29.7	43.5
	5.5	6.4	4.7	88.8	12.1	29.0	47.7
	6	5.4	4.1	90.5	10.5	28.1	52.0
	10	0.8	1.0	98.2	3.5	13.9	80.8
	15	0.1	0.1	99.8	0.3	3.3	96.2

Species	% population removed per year	% models predicting population increase	% models predicting population decline below alert thresholds	% models predicting change in conservation status (to alerted decline)	Of which, % models predicting a WeBS alert triggered after…		
					25 years	10 years	5 years
Goosander	0	93.7	3.5	2.8	2.2	0.6	0.0
	0.5	91.9	4.5	3.6	2.8	0.8	0.0
	1	90.1	5.3	4.6	3.3	1.2	0.0
	1.5	87.5	6.4	6.1	4.4	1.7	0.1
	2	84.2	7.9	7.9	5.5	2.2	0.2
	2.5	81.1	9.2	9.7	6.7	2.9	0.2
	3	77.3	10.1	12.5	8.6	3.7	0.3
	3.5	73.8	10.9	15.3	9.9	4.9	0.5
	4	69.4	11.9	18.7	11.8	6.2	0.8
	4.5	65.3	12.3	22.4	13.8	7.5	1.2
	5	60.7	13.5	25.8	14.4	9.8	1.5
	5.5	55.8	14.0	30.2	16.4	11.7	2.1
	6	51.4	14.3	34.3	17.5	13.9	2.8
	10	19.5	11.0	69.5	20.4	31.0	18.1
	15	3.6	3.2	93.2	9.1	26.9	57.2

3.3 Declining productivity in cormorant

The alteration of only one input parameter, productivity, by a relatively minor degree, equivalent to 10 years of change at a known rate (Cook and Robinson, 2010), had a strong effect on model outcomes (Figure 4). Across all removal scenarios, approximately an additional 15% of previously-accepted (i.e. plausible when fitted with unaltered productivity) models predicted decline consistent with a change in conservation status, with an equivalent amount fewer predicting population increase (Appendix 1: Table S7). Under this hypothetical case study, the 10% removal scenario had 99.6% of models predicting decline consistent with a change in conservation status, with 99.2% of models under the 15% scenario predicting a WeBS alert within five years (the most severe outcome).

Figure 3: Percentage of accepted models predicting five possible outcomes under each modelled removal scenario. Outcomes under which a WeBS alert would be triggered over short-term (dark red), medium-term (light red), or long-term (orange) timescales are all collectively consistent with a change in conservation status for the species concerned. Blue shading represents models predicting a population remaining stable or increasing; yellow shading indicates a population declining without triggering a WeBS alert during the modelled time period.

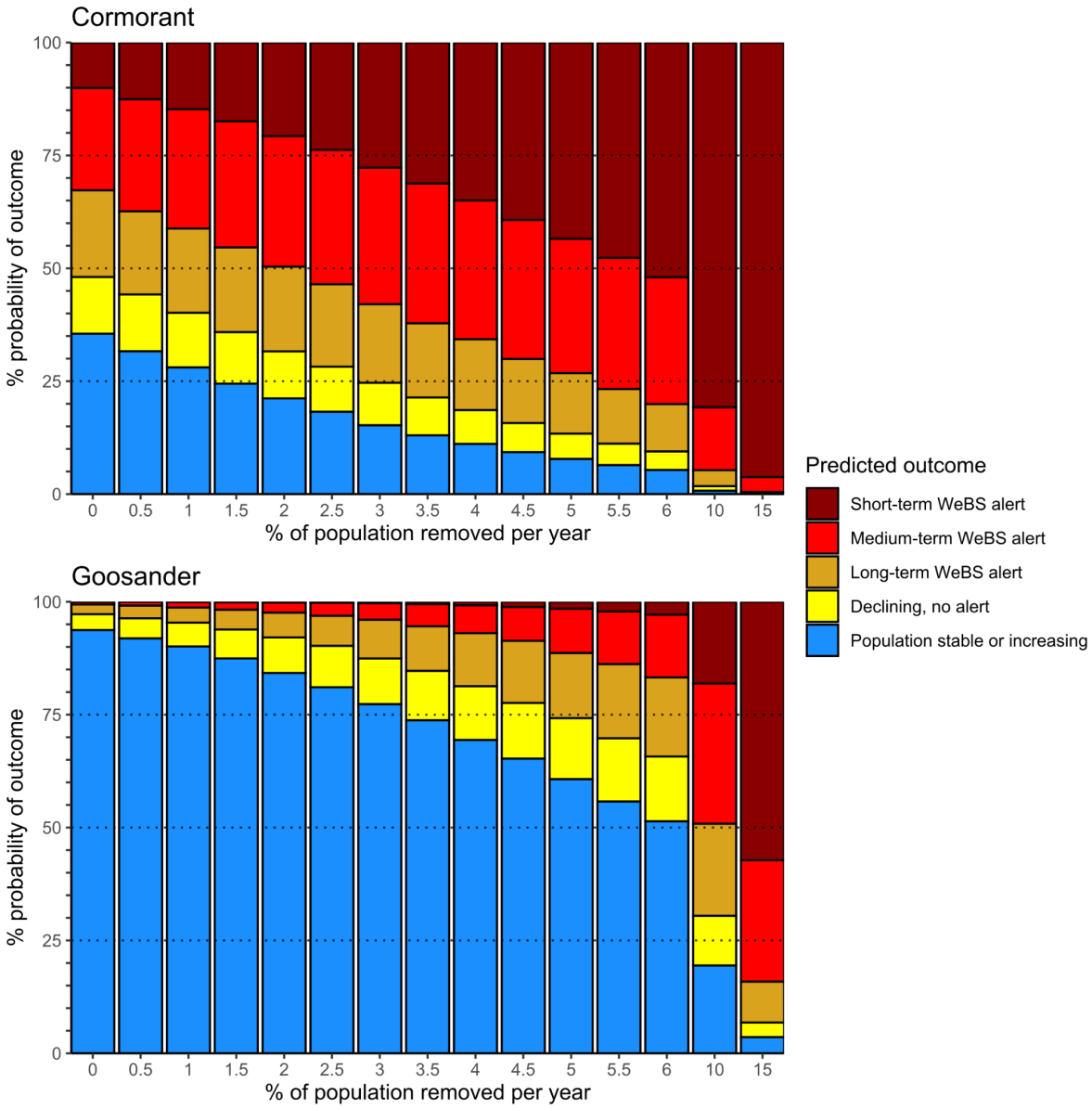
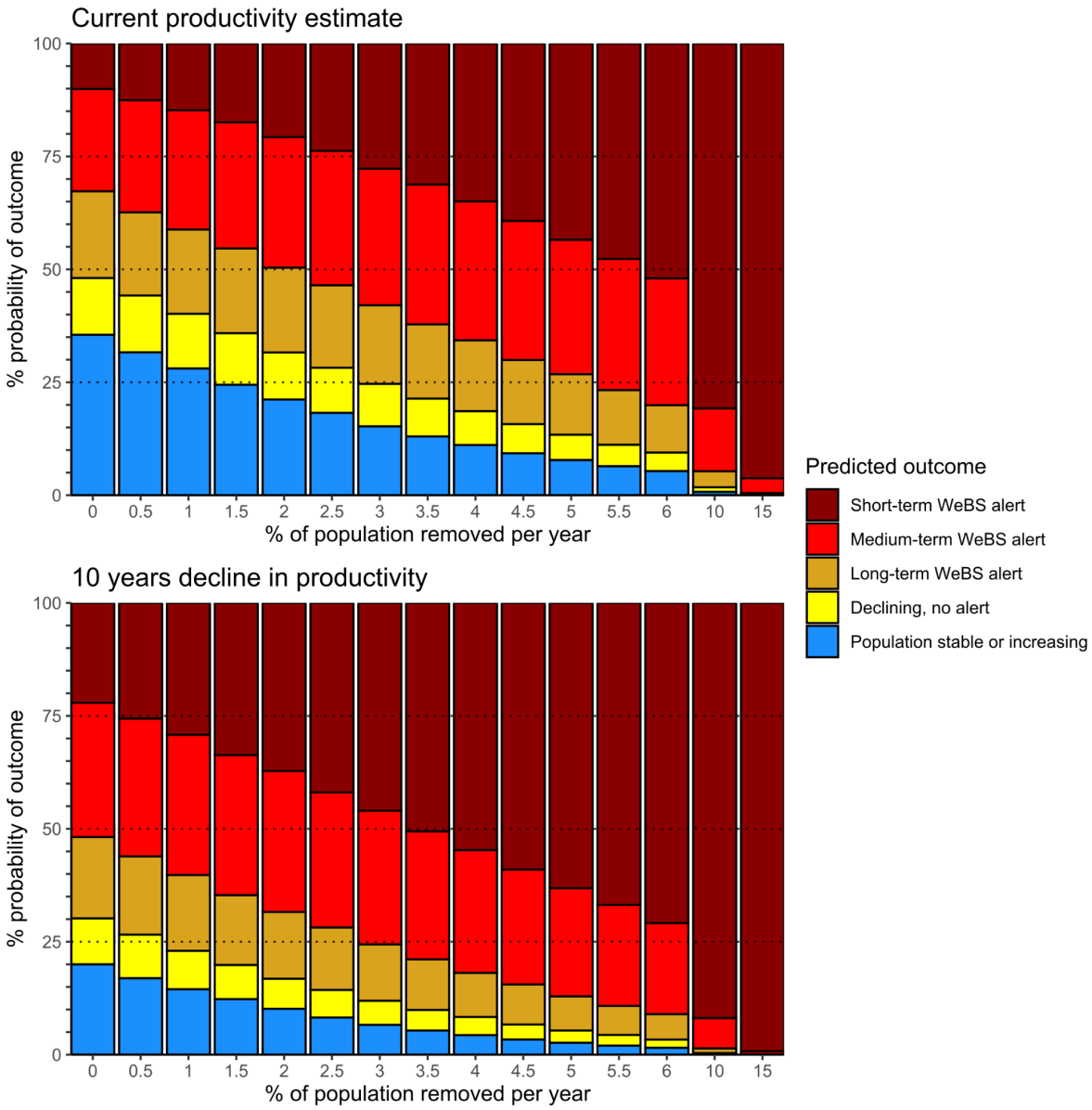


Figure 4. Percentage of accepted cormorant models predicting five possible outcomes under each modelled removal scenario, compared to a paired set of models with input productivity reduced in line with a 10-year decline in productivity at known rates. Outcomes under which a WeBS alert would be triggered over short-term (dark red), medium-term (light red), or long-term (orange) timescales are collectively consistent with a change in conservation status for the species concerned. Blue shading represents models predicting a population remaining stable or increasing; yellow shading indicates a population declining without triggering a WeBS alert during the modelled time period.



Interpretation of figures 3 and 4

A ‘change in conservation status’ was defined as predicted population decline sufficient to trigger a WeBS Alert during the period used in these models (25 years); these are shaded dark-red, red and amber. If the predicted decline is gradual enough not to trigger a WeBS Alert during the 25 year modelling period, (yellow shading), or if the model predicts either

stability or population increase (blue shading) this does not constitute a predicted change in conservation status (as defined here) within the 25 year modelling period.

Given the number of models in each scenario and the selection method employed, these outputs can be interpreted as a set of probabilities for the different outcomes. Under the baseline scenario (including current Welsh rates plus additional English impacts of lethal control) there is a greater than 50% probability the cormorant population will decline such that the species' conservation status changes (WeBS Alert triggered). Such an Alert might be over 25 years (amber shading, 19%), ten years (red shading, 23%) or five years (dark red shading, 10% probability) (Table 3 and Figure 3).

Under the model set taking into account the known declines in cormorant productivity in Wales and projecting these into the future, this probability of a change in conservation status in cormorant rises to 70%, even in the absence of any additional lethal control licensed in Wales (Figure 4).

Considering the modelled scenario where lethal control of cormorant is licensed at 3.5% of the annual population estimate, 79% of plausibly-structured demographic models predict a change in conservation status (WeBS Alert triggered) within the 25y timescale of the models. Such an Alert status change might be over 25 years (amber shading, 31%), ten years (red shading, 31%) or five years (dark red shading, 16% probability) (Table 3 and Figure 3). If declining productivity is included in the model design, the summed probability of a change in Alert (conservation) status (amber plus red plus dark-red shading) rises to 90% over the 25 years modelled.

3.4 Cormorant survival analysis

All three independent analyses of cormorant survival from the Ynys Seiriol/Puffin Island SPA colony, based on re-sightings and recoveries of ringed birds (colour-rings and two types of metal-ring), agreed well with the lower and upper bounds set by the expert panel for cormorant survival across all three age classes (Table 1). Colour-ringed birds, the cohort with the highest likelihood of resighting/recovery but also the temporally-shortest ringing scheme, provided extremely strong agreement with the modelled bounds. The two metal-ring datasets provided by SCAN Ringing Group (Steve Dodd, *pers. comm.* 2022) each made more precise estimates of survival than the colour-ring dataset, due to their longer data run; but produced significantly different estimates of 1st-year survival. Apparent survival was marginally higher among birds fitted with type A clip-rings up until 1994 than among birds fitted with type B split-rings, mostly after 1994. Analysing colour rings produced slightly lower estimates (especially at the lower confidence interval) than either metal ring type, whilst analysing type B metal and colour rings together for the recent period 2006–21 (excluding data from metal rings fitted in 2005 or earlier) produced the lowest survival estimates of any model (Table 4). One possible interpretation is that this indicates a reduction in survival rates among more recent cohorts.

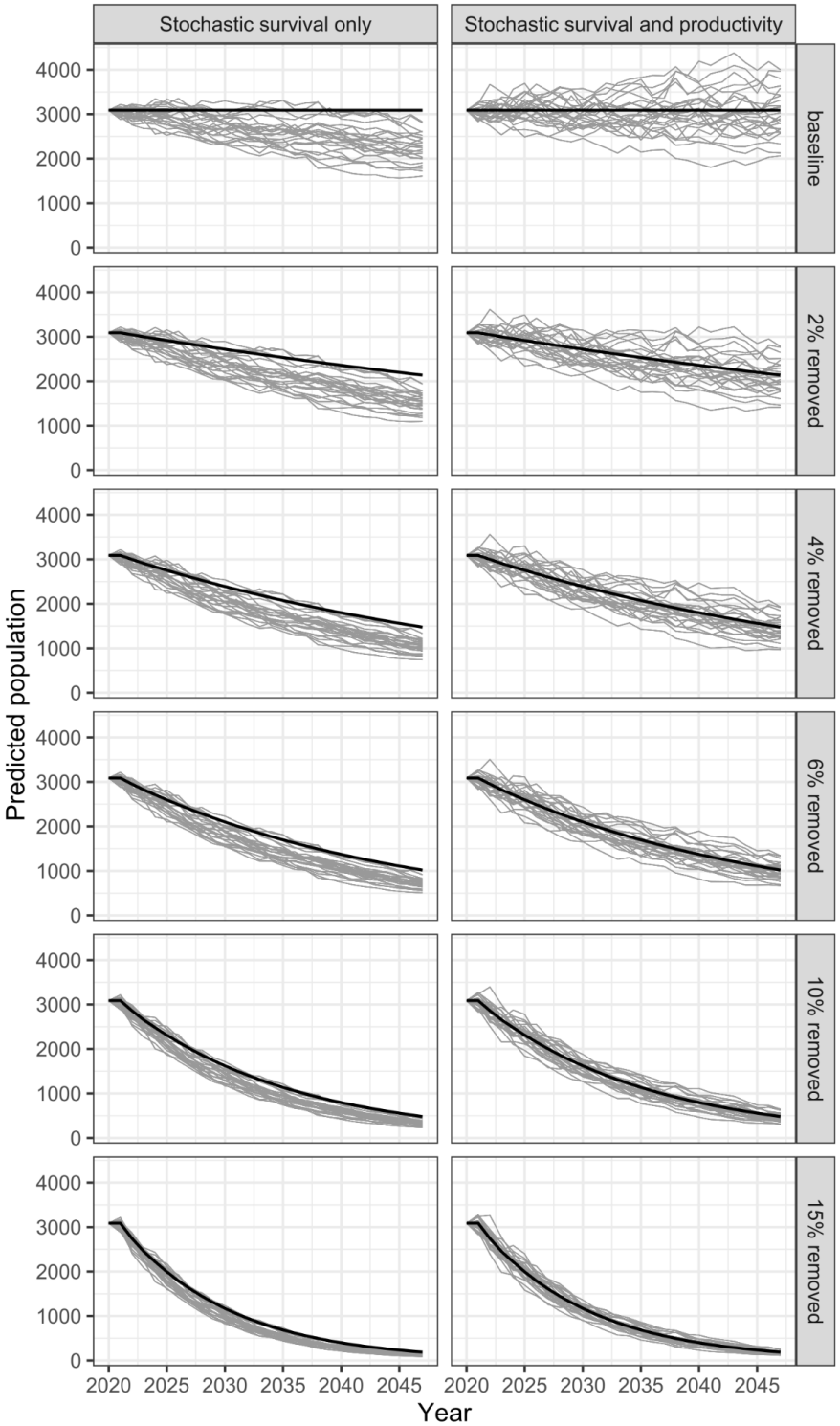
Table 4. Estimated cormorant survival for birds ringed as juveniles in the Ynys Seiriol/Puffin Island SPA colony, north Wales. Independent estimates are made using different ring types because the probability of recovery/resighting is likely to vary between ring types. The lower and upper bounds for cormorant survival, as set by the expert panel, are also presented for comparison.

Source (model/dataset)	Time span	Birds ringed	Statistic	1st year immature	2nd year immature	Adult
Lower and upper bounds from expert panel	'Current'	-	Lower/upper bounds	0.24–0.58	0.54–0.83	0.74–0.95
Survival model – colour-ring data	Rings fitted 2006–2021	699	Mean (+/- s.e.)	0.44 (0.07)	0.66 (0.08)	0.80 (0.04)
			95% CI	0.32–0.58	0.49–0.79	0.72–0.86
Survival model – metal-ring data, type A	Rings mostly fitted 1983–1994	2153	Mean (+/- s.e.)	0.55 (0.03)	0.78 (0.04)	0.84 (0.02)
			95% CI	0.49–0.62	0.70–0.85	0.81–0.87
Survival model – metal-ring data, type B	Rings mostly fitted 1995–2021	5596	Mean (+/- s.e.)	0.44 (0.03)	0.66 (0.04)	0.85 (0.02)
			95% CI	0.38–0.49	0.58–0.74	0.80–0.89
Survival model – all ring types combined	Rings fitted 2006–2021	3700	Mean (+/- s.e.)	0.38 (0.04)	0.62 (0.05)	0.78 (0.03)
			95% CI	0.31–0.45	0.51–0.71	0.71–0.84

3.5 Stochastic survival in cormorants

The majority of the set of 25 models incorporating environmental stochasticity in cormorant survival only, predicted smaller populations than a deterministic model using the mean value of survival (Fig. 5). By comparison, the set of 25 models incorporating environmental stochasticity in both survival and productivity were spread approximately evenly above and below the corresponding deterministic model. This emphasizes that stochastic effects, beyond simply making long-term population projections potentially unreliable, may cause populations to decline more than predicted by deterministic models such as those used herein; but that the extent of this will depend on the relative importance of stochasticity to different demographic parameters and the ability of the population to respond positively when a stochastic pressure is relaxed.

Figure 5. Stochasticity may lead to more varied, and potentially more negative, cormorant population outcomes than predicted by deterministic models. Comparison of a single deterministic model (black line) to 25 models incorporating stochasticity (grey lines) in cormorant survival only, and 25 models incorporating stochasticity in both cormorant survival and productivity, with all other parameters held the same.



4. Discussion

4.1 Modelling approach used in the present study

Many different methods have been used to provide modelled predictions of population change under management scenarios; and their accuracy (and usefulness) depends upon the accuracy of their input parameters. For both cormorant and goosander, some uncertainty exists around the demographic variables used to parameterise the demographic models. Parameters were therefore reviewed by an expert workshop consisting of senior conservation science staff from BTO and UKCEH. For cormorant, most available evidence considered was based on studies of continental European colonies of *Phalacrocorax carbo sinensis*, whereas the entire Welsh breeding population is thought to consist of *P. c. carbo* (Pritchard *et al.*, 2021). There is some evidence that the two subspecies may have different demographic rates (Newson *et al.*, 2005), although it is possible these differences may be exacerbated by differences in breeding location within the UK rather than the subspecies' innate biology. For goosander, very little evidence could be found for any of the demographic parameters, and the panel considered evidence from a range of similarly-sized ducks (especially common eider *Somateria mollissima*) and even from cormorant. It is therefore possible that some parameters set by the panel are inaccurate. However, the panel deliberately set wide lower and upper bounds for variables where there were serious concerns about accuracy, so the true parameter values are very likely to lie within these bounds for most parameters. Inspection of the frequency of parameter selection in models accepted by the rejection sampling protocol (Fig. 2c, f) supports this for all parameters except (possibly) first-year survival in goosanders, which could be close to or even lower than the lower bound. Reassuringly, survival analyses conducted for cormorants from the Puffin Island breeding colony, north Wales, provided good support for the lower and upper bounds set by the panel for cormorant survival across all three modelled age classes (first-year, second-year and adult).

4.2 Population estimates

Starting population estimates for the two species were taken from Taylor *et al.* (2022), as the most recent and comprehensive survey of cormorant and goosander wintering populations (and catchment distributions) available. The authors compared those estimates favourably to predicted population estimates (from the APEP report series) and described the level of statistical confidence in the estimates.

The population modelling and lethal control impact modelling reported in this study uses measures of percentage change and control, rather than absolute numbers of birds. This approach is similar to the approach of most bird population monitoring activity, where monitoring generates a *change metric* (trend) accompanied by periodic population estimates. Population change, expressed as an annual % change, leads to the same outcome over time irrespective of the size of the starting population. The absolute size of a population size becomes a significant factor when the modelled scenario is converted to a figure (number of birds) for the purpose of the licensing authority. Examples of this conversion are included in the supplementary tables in Appendix 1 (S1, 2 and 3) which include 95% confidence intervals derived from the population estimates for both species.

4.3 Population change data sources

Breeding Bird Survey (BBS) trends

The accuracy of the Approximate Bayesian Computation (ABC) rejection sampling approach to select models that predicted plausible rates of population change by comparison to the latest ten-year trends from the BTO Breeding Bird Survey (BBS) depends on the accuracy (and relevance) of the BBS trends themselves. BBS trends are not published at Welsh national scale for either species because these birds are recorded in too few BBS squares (cormorant: 23 1km squares; goosander: 11 1km squares, compared to the BBS reporting threshold of 30 squares). Therefore, there is considerable uncertainty around the population trend estimates themselves. This uncertainty was retained during the ABC rejection sampling procedure, which selected models by comparison to the probability distribution function of the 10-year trend estimates. As a consequence, some 36% of baseline models for cormorant (i.e. those without additional lethal control) predicted population stability or increase, even though the central estimate of the BBS trend was a substantial (17%) decline over ten years. Similarly, 6% of goosander baseline models showed population decline, even though the BBS central estimate was a 50% increase over ten years and goosander is considered to be colonising Wales (Pritchard *et al.*, 2021). This highlights a key challenge associated with using BBS data to monitor riverine and coastal birds in Wales; that BBS is a monitoring survey primarily designed to monitor common terrestrial breeding birds and to be representative of habitat coverage (land-use types) at the national scale. The coastal distribution of breeding cormorant in Wales, and the riverine association of breeding goosander (along with their complex migratory behaviour) makes both species a poor fit with the structural design of BBS.

Theoretically, increasing BBS coverage is a possible approach to meeting the challenge of monitoring population change in cormorant and goosander. The additional survey effort required would, however, be very significant because of the way the survey is designed (to be nationally representative). The underlying survey structure is a stratified random allocation sample. In order to add sufficient additional cormorant and goosander 'positive' squares without biasing the entire national BBS survey, a very much larger number of squares would need to be selected *and surveyed for at least five years*, such that the required additional goosander and cormorant data was collected. Finally, for a colonial breeder such as cormorant, trends would be reported "in brackets" under BBS, because counts are more over-dispersed than for an average species (i.e. it is harder to randomly sample colonies with the BBS survey design than to randomly sample nests/territories for an evenly-distributed species).

Wetland Bird Survey (WeBS) trends

BBS is a survey of *breeding* birds, whereas this tender required PVA modelling to be applied to winter population estimates and assumed birds will be removed only in winter. BBS is not a winter survey, and trends are not expected to be representative of trends in overwintering birds of migratory species. The Wetland Bird Survey (WeBS) offers an alternative dataset for which primary data collection takes place during winter months (Frost *et al.*, 2021). The ten-year population trend for goosander in Wales estimated by WeBS is a 44% increase - very similar to that estimated by BBS (50% increase). However, WeBS also estimates a 22% increase over ten years in cormorant, contrasting sharply with the 17% decline in the breeding population. This apparent disagreement likely arises from the mixed populations

present in winter - a breeding population in decline but net immigration in winter increasing, as a result of positive breeding trends in the source populations elsewhere in the UK and continental Europe. It highlights a number of challenges that a programme of licensed lethal control must overcome, since at present there is little evidence available to understand the proportion of the Welsh overwintering cormorant population that also breeds in Wales (especially within the Ynys Seiriol/Puffin Island SPA); and also since it is impossible to know at the time of removal whether any individual bird is resident or overwintering, or derived from which subspecies.

WeBS does not provide sufficient coverage of habitats occupied by wintering goosander to make reliable inferences about the goosander population trend in Wales. Only 6% of the goosander winter population estimate made by Taylor *et al*. (2022) was in the estuarine habitats that are a primary focus for WeBS counts. Therefore, to maintain a consistent methodology across both species for the present study, BBS was considered to be the most relevant available monitoring dataset. Long-term and robust population monitoring is particularly critical if lethal control of birds is applied, and consideration should be given in future to a strategy for updating population estimates as populations begin to diverge from the census estimate of 2021. That survey may provide a starting point for methodologies appropriate to wintering and breeding goosander as well as river-wintering cormorant (Macgregor *et al.*, 2022; Taylor *et al.*, 2022). Coastal breeding cormorant are monitored under the Seabird Monitoring Programme, but little data exist on inland breeding cormorant in Wales and it is assumed this is negligible.

4.4 Limitations of predictive modelling

The impact of stochasticity

A key limitation of this modelling approach was that the limited evidence base and associated variance in the majority of demographic parameters necessitated deterministic modelling; despite understanding the potential importance of stochasticity in determining long-term population outcomes. Our comparisons between a single case study deterministic model and matched model sets incorporating stochasticity in (a) survival only and (b) survival and productivity demonstrate the way in which unpredicted outcomes can arise simply by variation around a known mean. Almost all models incorporating stochasticity in survival alone predicted worse outcomes than the deterministic model (Fig. 4). These models may represent something of a worst-case scenario, since they preclude the possibility that a year of unusually high productivity might offset a year of unusually low survival; and indeed, models incorporating stochasticity in both survival and productivity were evenly dispersed around the deterministic model. Stochasticity (i.e. year-to-year variation) in survival has been clearly demonstrated in cormorants for both adult and immature age classes (Frederiksen and Bregnballe, 2000a, 2000b), and is most likely driven by variable winter conditions, with extreme climatic events causing high mortality (as has been demonstrated in the closely related shag *Phalacrocorax aristotelis* (Acker *et al.*, 2021)). It is less clear whether stochastic effects on productivity occur in cormorant, but multiple potential drivers of such effects exist, including (but not limited to): variable abundance of nest predators e.g. brown rat *Rattus norvegicus* at breeding colonies, variable levels of disturbance from humans, variable food availability, and indeed extreme climatic events, as for survival. An additional layer of complexity is that density-dependence (excluded from this study for reasons described above) could also offset the impacts of stochasticity in survival ***at a single colony***, by driving increased productivity as a response to low population density

or *vice versa*. Ongoing population monitoring would be necessary to ensure that unpredicted population declines or increases driven by stochastic effects can be detected and accounted for when making licensing decisions.

Comparison with reality

For modelled population projections such as those presented here, it is important to consider how close an approximation of reality is being modelled. A key area of some uncertainty in this regard is what the 'baseline' scenario, sometimes described as 0% removal, actually represents (and therefore, what each modelled increment above this baseline also represents). This is because the ABC rejection sampling approach utilised ten-year population trend estimates for the period 2008–2018, during which licensed lethal control of cormorant and goosander was *already occurring*. Throughout the entire ten-year period, a number of cormorants belonging to the Welsh breeding population were shot under license each year in England (clearly evidenced in ring recoveries from the Puffin Island ringing study). In the second half of this ten-year period, starting in winter 2013–14, licensed 'lethal scaring' of both species also took place in Wales. On average, 43 (+/- s.d. 17) cormorants and 27 (+/- s.d. 12) goosanders were shot per year until the end of winter 2017–18 (the end of the period over which trends were calculated), and this activity has continued since. This means that the population trends from which our final model set was selected already include the influence of a certain level of removal.

One possible interpretation is that the baseline scenario represents current levels of removal, with increments above this representing "additional removal on top of present levels". However, two considerations refute this interpretation. First, the recent introduction of licensed lethal control in Wales – midway through the ten-year period for which population trends are available – means that it is too early to tell whether the introduction of licensed control in Wales has affected the long-term trajectory of population trends for either species. If affected, it is unlikely that trends estimated for 2008–18 (including data from a number of years prior to the introduction of licensed control in Wales) will yet accurately reflect a new trajectory; but instead represent some intermediate within a process of transition. Only when population trend estimates are available for a ten-year period in which licensed lethal control took place in all years, will it be possible to safely consider such a baseline scenario as a true model of the population response to that level of lethal control. Second, the influence of English lethal control is clearly significant and likely applies to at least all the North Wales colonies including the Ynys Seiriol/Puffin Island SPA colony, but remains undocumented and unmeasurable. This contributes considerable uncertainty in the actual number of birds removed from the Welsh population each year, and also increases the vulnerability of Welsh cormorant to changes in English licensing.

Given the impossibility of enumerating the true level of lethal control operating on Welsh cormorant and goosander at present, the conservative approach is therefore to consider this modelled (transitional) baseline as representing 0% removal in Wales, and the lethal control scenarios as additional to, and proceeding from, that baseline.

Expressing trends in demographic parameters

One key limitation of the NEPVA approach and the resulting model set is that demographic parameters do not change over time. However, productivity in Welsh colonies of cormorants has already been observed to be declining (Cook and Robinson, 2010). Related concerns were raised by members of the expert panel around cormorant survival; which concern led

to our analysis of ringing data from the nationally-important Puffin Island colony. This as-yet unpublished analysis indeed provides some evidence suggestive of a change in survival over time. For goosander, no data exists with which to assess change over time in any demographic parameters, but such changes are equally possible.

Our analysis of a (hypothetical future) scenario in which cormorant productivity is decreased, by an amount equivalent to ten years at the rate recently observed in Wales (Cook and Robinson, 2010), reveals the potentially serious implications for model accuracy that arise from any *change* in demographic parameters over time, especially if such changes led to the true parameter value falling outside our modelled values. Modelling to include the impacts of parameter change over time would be computationally and analytically extremely challenging, and could not be performed using the NEPVA approach as required by the NRW contract.

This concern is significant, but can be addressed by ensuring sufficient and robust population monitoring for both species. Monitoring, or at least episodic resurvey, would highlight if the population of either species exceeds either the upper or lower expected 95% confidence intervals of the populations predicted using static parameters.

Age and sex bias in lethal control

Another assumption of our modelling is that an individual removed from the population by licensed lethal control is likely to be an adult or an immature bird ***in proportion with the stable age structure of the population***. In practice, this is not necessarily the case. The limited available data suggest that immature birds are more likely to be removed among cormorants; and potentially the reverse is true for goosanders (S. Newson, D. Carss, pers. comm.). Sex was not considered in our models at all, so it is effectively assumed, likewise, that controlled individuals are equally likely to be male or female. Again, limited data suggests that this is not the case for goosander (D. Carss, pers. comm.). The potential implications for our model outcomes of any severe violations of these assumptions are unclear, but the realised proportions of adults vs. immatures and males vs. females removed would depend on a complex range of factors that determine which demographic groups are most likely to be foraging at particular types of sites, and which groups are most vulnerable to shooting when they are present. These assumptions are even more likely to be violated (with potentially severe consequences) if licensed control were extended outside the winter period used in the 2021 population census and modelling (November to February).

Geographic bias in lethal control

Models of this type also assume that each individual is equally likely to be removed regardless of its geographic location within Wales, and that the impacts of its removal on the Welsh population are also equal (with an additional underlying assumption, known to be violated, that the population is closed).

Understanding the relative spatial sensitivity of the model is important given both the north-west-coastal bias in breeding cormorant within Wales, and the protected status of the key breeding colony in north Wales. Ynys Seiriol/Puffin Island SPA currently holds the largest breeding colony of cormorants in Wales (Pritchard *et al.*, 2021) and may represent almost 50% of the population. The island's designation as a Special Protection Area includes this colony as a designated feature, with a vision that the island should "continue to provide

nesting habitat for at least 1% of the NW European (Atlantic) population of cormorant *Phalacrocorax carbo*" (CCW, 2008).

It is, however, clear from cormorant ringing recovery data that birds originating from the designated Puffin Island colony experience lethal control, since recoveries described as 'shot under license' are annually reported from England and elsewhere (SCAN Ringing Group, *pers. comm.* 2022). These reports, alongside live sightings of colour-marked birds, might be used in future to investigate dispersal and seasonal migratory movements from this important colony, particularly as they are relevant to licensing the lethal control of cormorants in the different Welsh rivers. More detailed investigations of dispersal direction and distance from Puffin Island using data from the existing ringing/colour-marking scheme, potentially supplemented by expansion of ringing/colour-marking to other colonies or more refined and less biased methods such as GPS-tracking of cormorants may help to answer multiple questions about dispersal in cormorants, such as: (i) where do Welsh breeding adult cormorants winter; (ii) where do young Welsh cormorants disperse to (distance and direction from the natal colony); (iii) where do cormorants overwintering in Wales arrive from; and (iv) what proportion of cormorants overwintering in Wales belong to Welsh, UK or continental populations. Such information could allow licensing to be appropriately sensitive to the potential impacts on the Ynys Seiriol/Puffin Island SPA's cormorant feature.

Interpreting these results

The use of the lower and upper bounds for each parameter to generate thousands of candidate models (62,500 per species) followed by an ABC rejection sampling approach overcomes many of the concerns described above, because it contributes a dispersed "risk scenario" for each percentage removal increment, rather than a single-model outcome. This approach represents a considerable improvement over the single-deterministic-model approach of the Shiny-app NEPVA model, and provides an improved understanding of the breadth of risk associated with uncertainty around input parameters and population trends. For instance, most removal scenarios have a non-zero probability of all outcomes from the most positive, "population stable or increasing", to the most negative, "short-term WeBS alert" (Fig. 5); but the distribution of probabilities between the possible outcomes varies substantially between removal scenarios.

To provide a clarifying example, under the baseline scenario for cormorant (Table 3), it is already more probable that a decline leading to a change in conservation status will take place (51.9% probability) than population stability/increase (35.5% probability); the most likely timescale for a WeBS alert is 10 years (22.6% probability). By comparison, under the 5% removal scenario, it is highly probable (86.6%) that a decline leading to a change in conservation status will take place, with the most likely timescale for a WeBS alert being only 5 years (43.5% probability). Under the 15% removal scenario, it is almost certain (96.2% probability) that a WeBS alert will be triggered within 5 years.

Similarly for goosander, under the baseline or 'business as usual' scenario, it is highly probable that the population will continue to increase (93.7% probability); but nonetheless, a decline leading to a change in conservation status cannot be ruled out (2.8% probability). Under the 5% removal scenario, population increase remains the most probable outcome (60.7% probability) but there is a much larger chance of a decline leading to a change in conservation status (25.8% probability), with the most likely timescale for a WeBS alert being 25 years (14.4% probability). Under the 15% removal scenario, a decline leading to a change

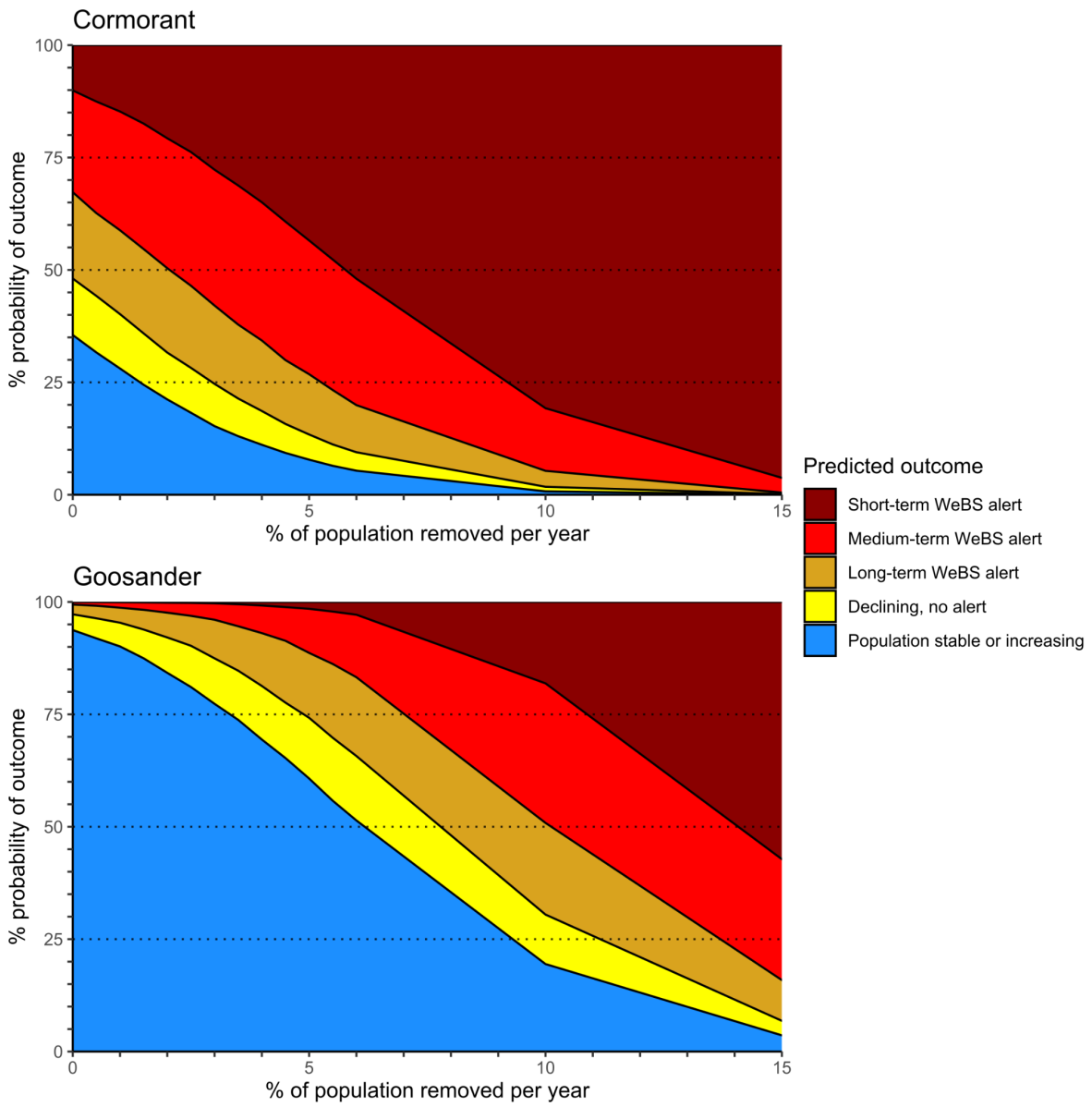
in conservation status is the most probable outcome (93.2% probability); with the most likely timescale for a WeBS alert being only five years (57.2% probability).

Considering the balance of probabilities in this way may inform the 'conservative' setting of removal thresholds. It should be noted, however, that each removal scenario has been modelled separately. Although it is tempting to interpolate between modelled thresholds, this would provide inaccurate results, because the relationships between inputs (percentage removal) and responses (probability of each outcome) for each species are not linear. An intermediate scenario should be modelled independently in order to fully understand the likely outcomes of, for example, 0.25% or 12% removal.

The 95% confidence intervals for population estimates in each year, under each scenario (see Appendix 1; cormorant: Table S1; goosander: Table S4), are translated into a conservative maximum number of individuals removed in each year, under each scenario (see Appendix 1, Tables S3 and S6; which represent the ***lower confidence intervals*** from Tables S2 and S5). For both species, this conservative maximum number of birds that can be removed decreases year-on-year, because scenarios of population decline form the majority of accepted baseline models for cormorant (see Appendix 1, Table S1) and cannot be entirely ruled out for goosander (see Appendix 1, Table S4).

Uncertainty, in the form of widening confidence intervals, increases over time – as should be expected for population projections from a census baseline without annual re-survey. Both this increasing uncertainty, and possibly the annual decrease in the number of individuals represented by proportional licensed control (i.e. using a percentage of predicted population for each species) could be reset if the models indicating population decline can be ruled out or reduced through future resurvey or appropriately enhanced population monitoring (e.g. repeats of the population census by Taylor *et al.,* 2022).

Figure 5. Risk surfaces for outcomes of different levels of removal (based on Table 3).



Ongoing monitoring and / or resurvey

Ongoing population monitoring is critical, given that the long-term projections of these models contain a large degree of uncertainty and rely upon a number of assumptions, some of which are known or considered likely to be violated. Questions therefore exist as to what shape ongoing monitoring should take, and its desired frequency; and also whether monitoring effort should be concentrated on breeding or wintering birds.

Some degree of annual monitoring is possible through existing surveys, notably Atlases, SMP, BBS, WBBS and WeBS. Annual trends for breeding birds are primarily produced from BBS, but its terrestrial methodology is a poor fit for either cormorant or goosander, and

coverage for both species is also poor and below the threshold for inclusion in standard BBS Reports in Wales. WBBS is a better fit for goosander, being targeted for riverine birds; but, again, coverage in Wales is insufficient for the production of short-term trends with confidence intervals adequate to monitoring the impacts of annual lethal control. Similarly, cormorant are arguably better-monitored by SMP (breeding) and WeBS (wintering), but with related concerns around coverage in Wales, especially in habitats where conflicts with salmonids are likely to occur.

It is likely, therefore, that additional structured, targeted monitoring will be required for both species. Population viability modelling is performed on a 'virtual' breeding population, so despite the focus of lethal control policy being applied to wintering birds, the key datasets will include monitoring impacts on breeding birds (which in Wales includes breeding cormorant as an SPA feature). At a minimum, this should consist of surveys designed to capture good data on breeding populations on a regular basis, to monitor impacts on both species' conservation status in Wales: e.g. cormorant colony counts and riverine surveys for breeding goosander. Additional monitoring of overwintering populations is also desirable, given that control is expected to take place during winter; for example, periodic repeats of the 2020–2021 full population census for Wales (Taylor *et al.*, 2022) would provide much greater confidence about changes in overwintering populations than can be provided by WeBS alone. Given that the lower confidence interval for population estimates in both species roughly halves over the first ten modelled years under many removal scenarios (specifically, under low- to medium-removal scenarios for cormorant, and under high-removal scenarios for goosander), a sensible upper limit for the resurvey interval would be 10 years - such that a fit-for-purpose resurvey of cormorant and goosander populations in Wales is conducted *at least* once every ten years whilst licensed lethal control is ongoing.

Predation from fish-eating birds such as cormorant and goosander is one of a host of pressures in both freshwater and marine environments known to be playing a role in salmonid declines; including habitat loss and alteration, river pollution, climate change, invasive non-native species, and overharvesting (Cooke, 2021; Wiik Vollset *et al.,* 2022). In the wider context of the conservation trade-off between fish-eating birds and declining salmonids in Wales, it is therefore ethically important that on-going monitoring activity be maintained not only of the birds (in order to ensure that policy interventions are proportionate and not driving unacceptable conservation impacts), but also of the salmonid populations whose conservation status is at present unfavourable. Evidence for the effectiveness or otherwise of bird removal in promoting salmon recovery is lacking. With well-designed monitoring of the impacts of lethal and non-lethal bird control on both bird and fish populations it should be possible in future to confirm whether removing fish-eating birds makes any contribution to salmonid resilience or recovery.

4.6 Review of the cormorant modelling approach used in England

Background to and summary of the current English approach

The current system for determining the number of licenses to be issued for shooting cormorants in England has been in place since 2004, when Defra increased the maximum number of cormorants that could be shot each year from 500 to 2000 (with scope for a further 1,000, up to a total of 3,000, birds to be shot "for a short period") (Heydon, 2008). Each year since then, Natural England has set a threshold for the number of cormorants to be shot, which is generally similar to the original threshold of 2,000 (Defra, 2011). This threshold is set based on the outputs of annual population modelling conducted by FERA (Food and Environment Research Agency; now superseded by the Animal & Plant Health Agency, APHA), using cormorant overwintering population data from WeBS (Defra, 2013).

The specific modelling approach employed in England was peer-reviewed (Smith *et al.*, 2008) and has since been fully critically appraised (Austin and Burton, 2014). The authors assessed the critical assumptions and limitations of the Index, which are at least as applicable in Wales as in England:

The WeBS-Defra Index allows for the fact that the Core Count WeBS Index scheme is not proportionally representative of the habitats frequented by cormorant and their potentially different trends (and those of uncovered habitats). The authors note that distributional compensation is provided only by data from the DWS survey of a single winter (2002/2003); and that it is assumed that

i) WeBS relative habitat coverage remains constant over time
ii) Relative habitat use by cormorants remains constant over time.

The second of these two assumptions was considered to be poorly supported by evidence in England (c.f. no further DWS survey since 2002/2003 but considerable population change over the same period) and is likely to be less well supported for Wales, owing to overall poor DWS coverage, regional differences in coverage, and the intrinsic differences between the race composition of Welsh and English wintering cormorant populations. The authors discuss the limitations of DWS coverage, sources of potential bias and alternative approaches to the critical question of ongoing survey in considerable and relevant detail.

It may be informative to consider the application and critical assessments made of this modelling approach as it has been applied in England. Using this annual population modelling approach, evidence has been published indicating that culling up to 2,000 birds a year would result in decline (below a 1996–2000 reference population level) led to a downwards adjustment in the annual threshold below 2000, with a presumption against making large adjustments in the threshold to minimise the impact of changes on individual fisheries. Modelling work in 2007 suggested that the population would decline by more than 5% if 2,000 birds were removed, and that approximately 500 birds could be removed with population stability (Heydon, 2008). This modelling exercise was considered to provide evidence that licensed removal of cormorants had been a significant factor in population declines observed between 2003 and 2007 in England. The threshold was immediately adjusted to 1,800 for the 2007–08 season – the authors bearing in mind that the number of birds reported as having been shot was typically considerably lower than the number licensed to be shot - ranging between 33.2–72.2% of the license limit (Heydon, 2008).

The wintering cormorant population in England during the reference period (1996-2000) was structurally and demographically different from the current Welsh wintering population: as has been discussed previously in this report. The outcomes of modelling work by Heydon (2008) and Smith et al (2008) highlighted the potential for lethal control of >5% of a population previously growing at 4-6% per annum, to drive a consequent decline above WeBS Alert thresholds. Population trends and lethal control limits in England were subsequently reappraised such that this modelling approach is no longer in use, being superseded by the WeBS-DEFRA Annual Cormorant Index (see below).

WeBS-DEFRA Annual cormorant Index

From 2004–13, FERA's modelling to monitor annual population changes in cormorant and modify annual thresholds for removal accordingly (Smith *et al.*, 2008) was supported by the standard WeBS annual cormorant Index alone. Use of the standard WeBS index drew criticism, partly because confidence limits cannot be attached to the WeBS index (because there is no standard-sized sampling unit), and partly because bias in habitat coverage within WeBS might affect outputs (Austin and Burton, 2014); for example, if some habitats which are heavily used by overwintering cormorants have low coverage within WeBS. A new methodology, the WeBS-DEFRA Annual cormorant Index, was devised to overcome these issues (Chamberlain *et al.*, 2013). An additional benefit of the new index was that it provides a direct estimate of population size in each year (as opposed to simply being an index of change from previous year(s).

The WeBS-DEFRA Annual cormorant Index starts with the same data that underpins the standard WeBS cormorant Index, and combines it with other data sources to generate an estimate of the total number of cormorants in the wider landscape. Data from WeBS are used both to estimate the number of cormorants in habitats covered by WeBS, and also to estimate the rate of population change from previous years. Other data sources (specifically, the Dispersed Waterbirds Survey (DWS) for inland populations and the Non-estuarine Waterbird Survey (NEWS) for coastal populations) are used to estimate the number of birds at other sites, bootstrapped (using random sampling with replacement) to generate confidence intervals. The latter estimates are then adjusted according to the rate of population change estimated from WeBS, and added to population estimates from WeBS sites, to generate a final estimate of the total abundance of overwintering cormorants in England.

Although the WeBS-DEFRA Annual cormorant Index was seen to be a significant improvement on the status quo (Austin and Burton, 2014), it nonetheless has its own set of caveats and assumptions. Firstly, the DWS was undertaken only once, in the winter of 2002–03, and is assumed to be representative of habitat use by cormorants in all winters; no other evidence is available against which to assess the robustness of this assumption. Coverage of the DWS was also acknowledged to be below what was desired, and not spatially even, with particularly poor coverage in some areas of England; so it may not be fully representative of all habitats or all regions. The NEWS requires a similar assumption about being representative in all years, but has been repeated on three occasions (in the winters of 1997–98, 2006–07, and 2015–16 (in addition to the preceding Winter Shorebird Count in 1984–85), giving greater confidence in this assumption. A further assumption is made that the relative habitat coverage within WeBS, relative to coverage in the wider countryside as represented by DWS, has remained constant over time; this assumption is reasonably defensible because the WeBS methodology limits the influence of sites entering and leaving

the dataset over time (Austin and Burton, 2014). A final, critical, assumption is that population change observed within WeBS sites is representative of, and reflected within, sites covered by DWS and NEWS (such that populations away from WeBS sites change by the same proportion as those within them). No suitable data exist with which to assess the robustness of this assumption.

Applying this approach in Wales

The criticisms levelled at the original WeBS Index approach, and the caveats presented by Austin and Burton (2014), apply with particular significance to the Welsh cormorant population. The two most important assumptions made under the English approach are that a) WeBS provides a robust change metric for the cormorant population over time, and b) the DWS data provide a reliable means of understanding distribution and habitat use in cormorant. A very comprehensive consideration of these assumptions was made by Austin and Burton (2014). In Wales a more recent distributional survey dataset is available, which informs a much deeper understanding of each species' winter distribution and provides an opportunity to understand the limitations of the DWS data in Wales.

A single survey carries the risk of interpreting distribution in a single year as consistent with distribution in all years. Cormorants are mobile, and as generalist predators subject to active human disturbance (scaring and shooting) their distribution is responsive both to patterns of human behaviour and to prey availability, as well as wider pressures such as weather and climate. Assuming distribution in all years, especially for a species that has undergone colonisation and race-composition change in recent decades, need to be tested; and resurvey is the most appropriate tool for testing assumptions. An alternative to resurvey (for Wales) might be a formal comparison between the distributions provided by DWS and those generated in the winter survey 2020–21, although there will be difficulties in comparing surveys using such distinct methodologies and time periods, and with very different spatial coverage.

Simply extending the use of a model designed for English populations carries a second risk, in that the ratio of *P. c. carbo* to *P. c. sinensis* cormorants differs between England and Wales. Welsh breeding cormorant are almost exclusively coastal-breeding *P. c. carbo*; tree-nesting (commonly seen in *P. c. sinensis* colonies) has ***never been recorded in Wales*** (Pritchard *et al.*, 2021). The wintering population is an unknown and likely annually variable ratio of Welsh breeding *P. c. carbo*, migrant *P. c. carbo* and *P. c. sinensis* breeding birds from elsewhere in the British Isles (especially more northerly regions), and continental *P. c. sinensis* birds. This population level structural difference will likely affect the relative coastal/riverine/inland distribution of wintering cormorant in Wales, and certainly requires an amendment to the population estimation (and trend analysis) DWS-distribution correction factor currently being applied in England. WeBS coverage is itself biased (in terms of habitat) relative to the distribution of cormorants, although the inclusion of NEWS data (to improve coverage of coastal non-estuarine *P. c. carbo* birds) and stillwaters data would improve its applicability to the Welsh population.

The final major caveat applicable to Welsh cormorant relates to the impacts of weather (and, to a lesser extent, long-term population dynamics) on a population of unknown racial composition. There is evidence that the Welsh breeding population (as previously stated, primarily coastal-breeding carbo) is at best stable and may be declining under a range of ecological pressures. The addition of overwintering *P. c. sinensis* birds could lead to a

situation where a decline in breeding *P. c. carbo* cormorant is effectively masked from detection by an increasing number of migratory *P. c. sinensis* birds. This dynamic would be exacerbated in winters with particularly poor weather conditions, when weather-related mortality (particularly in juvenile and sub-adult cormorant) could be masked by increased seasonal migration from northern UK and the European mainland. A wintertime programme of licensed lethal control that has a substantial impact on breeding populations, disguised by an increasing overwintering migrant population, could ultimately lead to sink population dynamics applying to cormorant in Wales.

Population lethal control modelling for goosander

No similar modelling approach has been developed for calculating and reviewing lethal control of goosander in England. We might, however, consider the cormorant approach as an exemplar, and evaluate its application to goosander based on our understanding of the species' ecology and distribution in Wales.

The 2020–21 winter survey (Taylor *et al.*, 2022) offers a (river and stillwater) distribution survey that could be used in place of DWS data as a basis for population estimation (c.f. the demographic modelling in the present report). However, the major annual resurvey used in that model is WeBS; and WeBS coverage for goosander in Wales is poor owing to the species' riverine distribution. Less than 4% of the estimated goosander population was found in (primarily tidal estuarine) WeBS datasets, with the remainder recorded on rivers and stillwaters (Taylor *et al.*, 2022). WeBS-monitored stillwaters were not included in that estimation process and would help to some degree to alleviate this problem, but nonetheless the annual winter monitoring data for goosander is likely to be poorly representative of the population and provide a weak understanding of real trends. Likewise, annual survey data for the breeding population of goosander (i.e. from the BBS) needs to be interpreted with care, owing to (a) low coverage (only 11 BBS squares in Wales regularly recording goosander); (b) unsuitable BBS targeting, both in space (largely not riverine) and in time (poor temporal overlap with the goosander breeding season); and (c) seasonal spatial dynamics within goosander (both sexes undertaking non-synchronous moult-migrations to differing extents).

4.7 Developing a parallel approach for Wales: data and modelling requirements

Given the caveats presented above for both species, it seems inadvisable to simply apply the English cormorant model to Wales (and impossible to do so for goosander, since no such model exists). It may, however, be possible to develop a parallel approach for Wales. The requirements for data and modelling in order to do so differ between the two species.

Cormorant

Most of the necessary major data components to construct a parallel index to the WeBS-Defra Annual cormorant Index are available for Wales (Table 5). Despite the caveats surrounding WeBS, the 2020–21 survey estimated that over half of the overwintering Welsh cormorant population is found in estuarine sites monitored by WeBS (Taylor *et al.*, 2022); so WeBS represents a suitable source of annual winter population data. However, if a derivative of WeBS is used as a measure of annual cormorant population change in Wales, additional survey effort may be appropriate to confirm that removal based on winter data and modelling is not having an unexpected deleterious effect on the breeding population (especially at Ynys Seiriol/Puffin Island SPA).

Data on usage of habitats not covered by WeBS can be derived from the 2020–21 survey (Taylor *et al.*, 2022) for rivers and stillwaters, replacing or supplementing DWS, and by NEWS for coastal habitats.

Resurvey of riverine and stillwater distributions (with population re-modelling) is indicated in future, to enable testing of assumptions around the relative distribution of cormorants between habitats, and to confirm that the population estimate from the WeBS-derived model remains accurate. It might be possible to conduct this resurvey with a carefully stratified design to reduce effort (e.g. by taking a subsample of rivers or conducting a stratified resampling procedure) rather than surveying every river section twice. Alternatively, resurveys could be timed to integrate with, and fill gaps between, other repeat population censuses conducted in Wales, such as the BTO Bird Atlas (last conducted for the period 2007–11, on an approximately 20-year cycle); Seabird Census (just completed for the period 2015–21, on an approximately 15-year cycle); and/or Avian Population Estimates Panel (last completed in 2020, on an approximately seven-year cycle).

Goosander

For goosander, WeBS covers such a small fraction of the observed distribution in Wales (even if stillwaters were included) as to be an uncertain basis for annual population monitoring (Taylor *et al.*, 2022). An alternative approach would be required.

Methods to produce hybrid population estimates using multiple data sources (e.g. BBS/WeBS + BirdTrack) are under investigation, and could become an option in the future. More immediately, a rolling programme of targeted river surveys could offer a means to simultaneously meet the needs for both annual population monitoring and repeats of the 2020–21 survey. However, the modelling in this report, combined with the status of goosander as a colonising species in Wales, suggests that annual population review may not be quite as critical as it is for cormorant, since there is a relatively much lower probability of causing a decline leading to a change in conservation status for most removal scenarios modelled. A parallel example is provided by the initial removal threshold for cormorant in England (~5% or 2000 birds) which has been subject to annual modelling but required only minor changes since its introduction in 2004; due to the continued increase in the English cormorant population over this period (Frost *et al.*, 2021). With a conservative control threshold, periodic resurvey at sensible intervals may be sufficient for long-term impact assessment; for instance, conducting a bespoke survey every 20 years to alternate with Bird Atlas updates would give revised population estimates approximately every ten years.

Table 5. Summary of requirements for a parallel approach to the WeBS-DEFRA Annual Cormorant Index in Wales.

General requirement	Data source used in England	Availability and suitability in Wales	
		Cormorant	Goosander
Robust initial population estimate	Smith *et al*. (1998) model based on WeBS	Taylor *et al*. (2022) BTO Rivers & Stillwaters survey	Taylor *et al*. (2022) BTO Rivers & Stillwaters survey
Robust annual population [change] survey	WeBS	WeBS: suitable, with caveats	WeBS: unsuitable BBS: probably unsuitable (or with heavy caveats) BirdTrack + WeBS or BBS: requires development
Habitat use modifier for non-WeBS sites	DWS (inland) + NEWS (coastal)	BTO Rivers & Stillwaters survey (inland) + NEWS (coastal)	BTO Rivers & Stillwaters survey (inland) + NEWS (coastal)
Demographic model	Fera model	Requires development	Requires development

5. Conclusions

This report models the possible outcomes of licensed removal of fish-eating birds (specifically, cormorant and goosander) in Wales. By incorporating estimation of uncertainty around demographic parameters and current population trends, we highlight the variable probability of different outcomes, ranging from population increase or stability to decline leading to change in conservation status, arising as a consequence of different removal scenarios. The modelling reveals that cormorant carries a higher probability, and therefore greater risk, of negative outcomes; including under scenarios of low (baseline) removal. By comparison, negative outcomes for goosander cannot be ruled out entirely but are less probable.

Statistical uncertainty surrounding modelled outcomes should be borne in mind. Models include a number of assumptions, some of which are possible or even known to be violated in reality; for example, the assumption that demographic parameters will not change over time is called into question by the observed decline in cormorant productivity in Wales (Cook and Robinson, 2010). Additionally, models do not incorporate stochastic effects, raising the possibility for unexpected outcomes to arise. Ongoing population monitoring will be required to ensure that unexpected population changes beyond the confidence intervals of the modelled predictions can be swiftly detected and incorporated into future planning.

In England, this population monitoring requirement for cormorant is currently delivered through the Wetland Bird Survey, and a derived WeBS-DEFRA Annual Cormorant Index. Lack of knowledge around several key assumptions of this index is an obstacle to simply applying the same index to Wales for cormorant (and no such index exists for goosander). The development of a parallel model would be feasible for cormorant and much of the necessary data already exists. Lack of appropriate annual monitoring data is a much more significant obstacle for goosander, where additional bespoke surveys will likely be required to ensure sufficient ongoing monitoring. However, since the probability of negative outcomes is substantially lower for goosander under most removal scenarios, it may not be essential that such monitoring takes place on an annual basis.

Key recommendations and caveats

Ongoing population monitoring of cormorant and goosander is essential to ensure that observed population trends remain within the confidence intervals of modelled outcomes at the relevant level of intervention, especially in the light of uncertainty around changing demographic parameters and unpredictable stochastic effects.

On-going population monitoring of salmonids is also essential, to demonstrate a link between lethal control of fish-eating birds and positive outcomes for fish.

Full resurvey of overwintering cormorant and goosander should take place at least every ten years whilst lethal control is on-going.

Further research would be appropriate to investigate and understand:

- The **extent of predation** of salmon by cormorant and goosander during the smolt run, when mortality from predation may not be compensated by increased salmon survival or productivity.

- **Dispersal of cormorants** from the Ynys Seiriol/Puffin Island SPA colony, and the likelihood that lethally controlled cormorants in different catchments in Wales (and elsewhere) originate from this protected colony.
- **Response and dispersal** behaviour of cormorant and goosander displaced by non-lethal predation management activity, such that non-lethal activities can be better targeted and made as effective as possible

All modelling work assumes that lethal control will take place during the winter, and results are not valid if any lethal control is conducted at other times of year. If lethal control is applied at other times of year, it should be underpinned by **additional modelling work** as the consequences for population trends in both species, but particularly cormorant, could be severe.

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Appendices

Appendix 1: Supplementary Tables

Table S1. Cormorant population estimates – 95% confidence intervals of population estimates made in each year, under each removal scenario, by all plausible models.

Year	Baseline	0.5%	1%	1.5%	2%	2.5%	3%	3.5%	4%	4.5%	5%	5.5%	6%	10%	15%
2021	2449–3348	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2022	2311–3460	2302–3447	2293–3434	2283–3422	2274–3408	2264–3395	2255–3380	2245–3368	2235–3355	2225–3343	2215–3330	2206–3317	2196–3304	2121–3203	2027–3077
2023	2168–3598	2150–3571	2131–3545	2114–3518	2096–3492	2079–3466	2062–3440	2044–3416	2027–3390	2009–3367	1991–3342	1974–3317	1958–3292	1825–3094	1665–2861
2024	2022–3764	1997–3724	1973–3683	1949–3642	1924–3601	1900–3560	1876–3523	1851–3486	1828–3446	1806–3408	1782–3369	1759–3332	1737–3295	1566–3016	1369–2699
2025	1874–3953	1843–3895	1813–3838	1782–3780	1753–3724	1726–3673	1698–3620	1669–3569	1641–3514	1613–3462	1586–3410	1559–3360	1533–3312	1337–2945	1119–2547
2026	1744–4153	1706–4077	1670–4004	1634–3931	1599–3857	1566–3789	1535–3721	1502–3654	1469–3587	1438–3521	1409–3459	1379–3395	1350–3335	1138–2882	910–2402
2027	1608–4364	1569–4272	1532–4180	1494–4087	1456–3999	1419–3913	1385–3830	1351–3745	1317–3662	1284–3580	1252–3503	1219–3426	1187–3357	965–2820	741–2269
2028	1489–4597	1446–4477	1406–4362	1363–4248	1325–4142	1289–4040	1251–3946	1214–3845	1177–3745	1143–3651	1107–3560	1075–3469	1043–3381	820–2760	601–2148
2029	1379–4833	1334–4693	1289–4564	1246–4431	1205–4302	1166–4177	1125–4060	1087–3939	1050–3832	1016–3725	982–3624	950–3516	918–3415	695–2705	487–2030
2030	1273–5092	1225–4932	1180–4774	1137–4621	1095–4470	1054–4328	1015–4185	977–4052	940–3922	905–3801	871–3676	838–3560	806–3447	589–2656	394–1925
2031	1176–5366	1129–5180	1082–4996	1037–4817	996–4647	955–4477	915–4319	877–4171	841–4026	806–3882	773–3743	740–3604	708–3477	499–2614	319–1824
2032	1084–5645	1036–5430	990–5221	946–5020	901–4823	862–4631	824–4451	787–4284	752–4119	718–3956	684–3807	653–3654	623–3512	423–2568	260–1728
2033	1000–5953	952–5695	908–5462	860–5235	819–5026	781–4820	742–4615	705–4418	671–4229	638–4047	606–3870	576–3700	547–3546	358–2513	210–1639
2034	926–6290	878–5998	829–5720	784–5456	743–5217	703–4982	666–4754	631–4538	597–4335	566–4130	537–3940	506–3759	479–3582	303–2466	171–1559
2035	854–6643	806–6314	760–6003	715–5703	673–5418	635–5162	599–4915	566–4664	533–4439	502–4219	473–4017	446–3817	420–3618	256–2421	138–1476
2036	789–6999	742–6646	694–6292	653–5959	611–5643	574–5347	539–5068	506–4803	475–4552	447–4305	419–4083	393–3857	368–3651	217–2377	112–1401
2037	726–7392	681–6985	636–6600	593–6227	555–5880	519–5545	483–5242	452–4939	425–4656	397–4393	370–4142	346–3910	322–3685	184–2338	90–1331
2038	669–7810	624–7347	582–6904	542–6502	505–6115	470–5754	435–5403	406–5082	379–4781	352–4487	327–4214	305–3960	282–3734	155–2297	72–1264
2039	618–8243	572–7717	531–7229	494–6778	458–6363	425–5962	393–5584	364–5235	337–4896	312–4581	289–4291	268–4025	247–3775	131–2261	58–1196

2040	569–8690	527–8102	487–7596	450–7075	415–6623	383–6178	354–5768	326–5393	300–5022	277–4687	255–4369	235–4091	216–3817	110–2222	47–1136
2041	525–9169	484–8546	446–7945	409–7398	376–6875	346–6408	319–5957	291–5538	267–5139	245–4788	225–4458	206–4146	189–3871	93–2186	38–1079
2042	486–9661	445–8979	407–8318	373–7719	341–7153	312–6626	285–6143	260–5691	237–5271	216–4899	198–4538	181–4216	165–3918	78–2153	30–1024
2043	448–10190	409–9454	371–8722	339–8048	308–7424	281–6860	256–6334	232–5846	211–5408	192–4997	175–4619	158–4277	144–3961	66–2116	24–973
2044	412–10778	373–9904	338–9119	308–8396	279–7716	253–7111	230–6545	208–6020	188–5535	170–5108	154–4708	139–4343	125–4000	55–2078	19–923
2045	379–11382	342–10387	309–9552	279–8754	252–8016	228–7355	206–6749	185–6209	167–5680	151–5224	136–4803	122–4422	110–4049	46–2039	15–875
2046	348–12002	313–10953	282–9994	254–9123	228–8338	206–7609	185–6984	165–6383	149–5842	134–5341	119–4908	107–4494	96–4098	39–2006	11–828
2047	322–12647	288–11515	258–10454	231–9507	207–8685	186–7878	165–7225	148–6579	132–6002	118–5463	105–5006	94–4567	84–4152	32–1971	9–788

Table S2. Number of individual cormorants to be removed in each year under each scenario – 95% confidence intervals for the estimated value of n% of the population in each year, based on population estimates in Appendix 1, Table S1.

Year	Baseline	0.5%	1%	1.5%	2%	2.5%	3%	3.5%	4%	4.5%	5%	5.5%	6%	10%	15%
2021	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2022	-	11–17	22–34	34–51	45–68	56–84	67–101	78–117	89–134	100–150	110–166	121–182	131–198	212–320	304–461
2023	-	10–17	21–35	31–52	41–69	51–86	61–103	71–119	81–135	90–151	99–167	108–182	117–197	182–309	249–429
2024	-	9–18	19–36	29–54	38–72	47–89	56–105	64–122	73–137	81–153	89–168	96–183	104–197	156–301	205–404
2025	-	9–19	18–38	26–56	35–74	43–91	50–108	58–124	65–140	72–155	79–170	85–184	91–198	133–294	167–382
2026	-	8–20	16–40	24–58	31–77	39–94	46–111	52–127	58–143	64–158	70–172	75–186	81–200	113–288	136–360
2027	-	7–21	15–41	22–61	29–79	35–97	41–114	47–131	52–146	57–161	62–175	67–188	71–201	96–282	111–340
2028	-	7–22	14–43	20–63	26–82	32–101	37–118	42–134	47–149	51–164	55–178	59–190	62–202	82–276	90–322
2029	-	6–23	12–45	18–66	24–86	29–104	33–121	38–137	42–153	45–167	49–181	52–193	55–204	69–270	73–304
2030	-	6–24	11–47	17–69	21–89	26–108	30–125	34–141	37–156	40–171	43–183	46–195	48–206	58–265	59–288
2031	-	5–25	10–49	15–72	19–92	23–111	27–129	30–145	33–161	36–174	38–187	40–198	42–208	49–261	47–273
2032	-	5–27	9–52	14–75	18–96	21–115	24–133	27–149	30–164	32–178	34–190	35–200	37–210	42–256	39–259
2033	-	4–28	9–54	12–78	16–100	19–120	22–138	24–154	26–169	28–182	30–193	31–203	32–212	35–251	31–245
2034	-	4–29	8–57	11–81	14–104	17–124	19–142	22–158	23–173	25–185	26–197	27–206	28–214	30–246	25–233
2035	-	4–31	7–60	10–85	13–108	15–129	17–147	19–163	21–177	22–189	23–200	24–209	25–217	25–242	20–221
2036	-	3–33	6–62	9–89	12–112	14–133	16–152	17–168	19–182	20–193	20–204	21–212	22–219	21–237	16–210
2037	-	3–34	6–66	8–93	11–117	12–138	14–157	15–172	17–186	17–197	18–207	19–215	19–221	18–233	13–199
2038	-	3–36	5–69	8–97	10–122	11–143	13–162	14–177	15–191	15–201	16–210	16–217	16–224	15–229	10–189
2039	-	2–38	5–72	7–101	9–127	10–149	11–167	12–183	13–195	14–206	14–214	14–221	14–226	13–226	8–179
2040	-	2–40	4–75	6–106	8–132	9–154	10–173	11–188	12–200	12–210	12–218	12–225	12–229	11–222	7–170
2041	-	2–42	4–79	6–110	7–137	8–160	9–178	10–193	10–205	11–215	11–222	11–228	11–232	9–218	5–161
2042	-	2–44	4–83	5–115	6–143	7–165	8–184	9–199	9–210	9–220	9–226	9–231	9–235	7–215	4–153
2043	-	2–47	3–87	5–120	6–148	7–171	7–190	8–204	8–216	8–224	8–230	8–235	8–237	6–211	3–145
2044	-	1–49	3–91	4–125	5–154	6–177	6–196	7–210	7–221	7–229	7–235	7–238	7–240	5–207	2–138
2045	-	1–51	3–95	4–131	5–160	5–183	6–202	6–217	6–227	6–235	6–240	6–243	6–242	4–203	2–131
2046	-	1–54	2–99	3–136	4–166	5–190	5–209	5–223	5–233	6–240	5–245	5–247	5–245	3–200	1–124
2047	-	1–57	2–104	3–142	4–173	4–196	4–216	5–230	5–240	5–245	5–250	5–251	5–249	3–197	1–118

Table S3. Conservative maximum number of cormorants to be removed in each year under each scenario – lower confidence interval for the estimated value of *n*% of the population in each year, based on population estimates in Appendix 1, Table S1.

Year	Baseline	0.5%	1%	1.5%	2%	2.5%	3%	3.5%	4%	4.5%	5%	5.5%	6%	10%	15%
2021	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2022	-	11	22	34	45	56	67	78	89	100	110	121	131	212	304
2023	-	10	21	31	41	51	61	71	81	90	99	108	117	182	249
2024	-	9	19	29	38	47	56	64	73	81	89	96	104	156	205
2025	-	9	18	26	35	43	50	58	65	72	79	85	91	133	167
2026	-	8	16	24	31	39	46	52	58	64	70	75	81	113	136
2027	-	7	15	22	29	35	41	47	52	57	62	67	71	96	111
2028	-	7	14	20	26	32	37	42	47	51	55	59	62	82	90
2029	-	6	12	18	24	29	33	38	42	45	49	52	55	69	73
2030	-	6	11	17	21	26	30	34	37	40	43	46	48	58	59
2031	-	5	10	15	19	23	27	30	33	36	38	40	42	49	47
2032	-	5	9	14	18	21	24	27	30	32	34	35	37	42	39
2033	-	4	9	12	16	19	22	24	26	28	30	31	32	35	31
2034	-	4	8	11	14	17	19	22	23	25	26	27	28	30	25
2035	-	4	7	10	13	15	17	19	21	22	23	24	25	25	20
2036	-	3	6	9	12	14	16	17	19	20	20	21	22	21	16
2037	-	3	6	8	11	12	14	15	17	17	18	19	19	18	13
2038	-	3	5	8	10	11	13	14	15	15	16	16	16	15	10
2039	-	2	5	7	9	10	11	12	13	14	14	14	14	13	8
2040	-	2	4	6	8	9	10	11	12	12	12	12	12	11	7
2041	-	2	4	6	7	8	9	10	10	11	11	11	11	9	5
2042	-	2	4	5	6	7	8	9	9	9	9	9	9	7	4
2043	-	2	3	5	6	7	7	8	8	8	8	8	8	6	3
2044	-	1	3	4	5	6	6	7	7	7	7	7	7	5	2
2045	-	1	3	4	5	5	6	6	6	6	6	6	6	4	2
2046	-	1	2	3	4	5	5	5	5	6	5	5	5	3	1
2047	-	1	2	3	4	4	4	5	5	5	5	5	5	3	1

Table S4. Goosander population estimates – 95% confidence intervals of population estimates made in each year, under each removal scenario, by all plausible models.

Year	Baseline	0.5%	1%	1.5%	2%	2.5%	3%	3.5%	4%	4.5%	5%	5.5%	6%	10%	15%
2021	1233–1884	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2022	1244–2031	1240–2024	1234–2015	1229–2008	1224–2001	1219–1994	1214–1986	1209–1979	1203–1971	1199–1964	1194–1957	1189–1948	1183–1942	1144–1883	1093–1809
2023	1252–2200	1242–2184	1233–2169	1223–2154	1213–2138	1203–2124	1193–2108	1184–2093	1175–2078	1165–2063	1156–2048	1146–2034	1137–2020	1063–1906	975–1775
2024	1256–2395	1241–2371	1228–2346	1214–2323	1199–2299	1186–2275	1170–2251	1155–2227	1142–2202	1128–2179	1114–2157	1101–2134	1088–2113	988–1951	869–1764
2025	1255–2609	1235–2575	1216–2542	1197–2506	1178–2474	1160–2439	1143–2407	1125–2374	1109–2342	1092–2310	1075–2280	1059–2249	1041–2219	913–1994	770–1751
2026	1251–2843	1227–2795	1204–2753	1181–2704	1159–2663	1136–2623	1116–2580	1095–2537	1072–2498	1050–2455	1029–2415	1012–2375	991–2336	843–2044	679–1741
2027	1247–3110	1219–3048	1190–2989	1164–2928	1137–2871	1111–2817	1084–2760	1059–2709	1037–2655	1012–2605	989–2556	966–2506	944–2456	777–2106	602–1736
2028	1242–3407	1211–3333	1177–3251	1145–3172	1113–3098	1085–3026	1054–2963	1025–2897	997–2832	970–2766	945–2708	919–2648	893–2587	716–2159	530–1732
2029	1234–3736	1195–3641	1158–3542	1123–3449	1090–3358	1059–3267	1024–3186	992–3108	959–3022	932–2943	904–2872	877–2802	850–2730	657–2226	466–1732
2030	1225–4099	1183–3981	1142–3862	1103–3750	1065–3641	1027–3529	993–3429	960–3328	928–3229	895–3143	864–3053	833–2969	804–2876	598–2292	410–1737
2031	1218–4498	1169–4351	1124–4211	1082–4074	1040–3943	1004–3815	962–3698	926–3583	892–3461	858–3351	825–3247	793–3147	761–3045	548–2368	359–1739
2032	1210–4936	1160–4770	1111–4597	1062–4432	1018–4273	977–4132	935–3988	896–3855	857–3726	819–3592	784–3470	750–3347	717–3228	500–2441	312–1735
2033	1200–5422	1145–5216	1093–5013	1044–4818	995–4634	949–4467	905–4302	864–4146	824–3998	784–3845	746–3693	712–3547	680–3405	458–2511	273–1732
2034	1191–5951	1129–5711	1075–5471	1025–5245	973–5039	924–4833	875–4640	831–4458	789–4289	749–4109	711–3927	676–3767	641–3600	419–2599	240–1737
2035	1178–6549	1116–6259	1057–5992	1001–5740	947–5489	895–5245	847–4999	801–4794	757–4587	716–4389	677–4183	642–3997	607–3817	383–2692	209–1741
2036	1163–7228	1098–6875	1035–6552	976–6249	919–5943	869–5669	821–5401	774–5159	728–4922	685–4692	644–4465	609–4243	573–4053	350–2778	183–1746
2037	1151–7952	1083–7545	1014–7172	953–6814	895–6492	843–6170	794–5847	746–5554	701–5273	656–5005	614–4758	575–4524	540–4302	320–2865	160–1753
2038	1135–8755	1064–8281	1000–7846	933–7428	873–7046	815–6659	765–6293	715–5953	669–5645	625–5339	584–5055	546–4787	509–4545	291–2969	140–1759
2039	1117–9642	1048–9102	980–8570	914–8105	852–7647	793–7207	736–6794	687–6397	639–6051	596–5697	556–5398	518–5089	481–4803	265–3072	122–1775
2040	1104–10635	1028–10005	957–9400	891–8840	829–8314	771–7829	712–7340	660–6891	612–6485	568–6101	528–5742	492–5404	454–5085	242–3176	106–1781
2041	1093–11661	1011–10962	937–10284	868–9641	806–9030	748–8464	690–7925	635–7430	586–6957	541–6537	501–6119	463–5730	429–5373	221–3288	92–1790

2042	1081–12851	992–12016	919–11263	850–10506	783–9829	723–9163	666–8557	611–7978	562–7468	515–6999	475–6514	440–6099	404–5709	201–3408	80–1795
2043	1070–14161	979–13156	902–12287	829–11451	762–10677	698–9958	642–9266	589–8603	538–8000	494–7503	453–6970	417–6485	382–6052	183–3532	70–1803
2044	1057–15565	965–14483	881–13425	807–12486	738–11585	677–10808	620–10044	567–9283	516–8607	472–8025	431–7451	394–6917	360–6397	168–3655	61–1806
2045	1044–17210	949–15870	864–14697	787–13623	719–12608	654–11706	596–10853	543–10011	494–9244	451–8598	410–7974	374–7391	340–6828	152–3767	53–1806
2046	1031–18913	934–17427	849–16082	768–14823	696–13727	632–12687	573–11725	522–10809	474–9939	429–9190	390–8530	353–7874	319–7263	139–3893	46–1809
2047	1019–20816	920–19100	833–17616	752–16191	677–14923	611–13774	554–12645	501–11654	453–10712	409–9878	369–9096	334–8384	301–7705	127–4028	39–1822

Table S5. Number of individual goosanders to be removed in each year under each scenario – 95% confidence intervals for the estimated value of *n*% of the population in each year, based on population estimates in Appendix 1, Table S4.

Year	Baseline	0.5%	1%	1.5%	2%	2.5%	3%	3.5%	4%	4.5%	5%	5.5%	6%	10%	15%
2021	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2022	-	6–10	12–20	18–30	24–40	30–49	36–59	42–69	48–78	53–88	59–97	65–107	70–116	114–188	163–271
2023	-	6–10	12–21	18–32	24–42	30–53	35–63	41–73	47–83	52–92	57–102	63–111	68–121	106–190	146–266
2024	-	6–11	12–23	18–34	23–45	29–56	35–67	40–77	45–88	50–98	55–107	60–117	65–126	98–195	130–264
2025	-	6–12	12–25	17–37	23–49	29–60	34–72	39–83	44–93	49–103	53–114	58–123	62–133	91–199	115–262
2026	-	6–13	12–27	17–40	23–53	28–65	33–77	38–88	42–99	47–110	51–120	55–130	59–140	84–204	101–261
2027	-	6–15	11–29	17–43	22–57	27–70	32–82	37–94	41–106	45–117	49–127	53–137	56–147	77–210	90–260
2028	-	6–16	11–32	17–47	22–61	27–75	31–88	35–101	39–113	43–124	47–135	50–145	53–155	71–215	79–259
2029	-	5–18	11–35	16–51	21–67	26–81	30–95	34–108	38–120	41–132	45–143	48–154	51–163	65–222	69–259
2030	-	5–19	11–38	16–56	21–72	25–88	29–102	33–116	37–129	40–141	43–152	45–163	48–172	59–229	61–260
2031	-	5–21	11–42	16–61	20–78	25–95	28–110	32–125	35–138	38–150	41–162	43–173	45–182	54–236	53–260
2032	-	5–23	11–45	15–66	20–85	24–103	28–119	31–134	34–149	36–161	39–173	41–184	43–193	50–244	46–260
2033	-	5–26	10–50	15–72	19–92	23–111	27–129	30–145	32–159	35–173	37–184	39–195	40–204	45–251	40–259
2034	-	5–28	10–54	15–78	19–100	23–120	26–139	29–156	31–171	33–184	35–196	37–207	38–216	41–259	36–260
2035	-	5–31	10–59	15–86	18–109	22–131	25–149	28–167	30–183	32–197	33–209	35–219	36–229	38–269	31–261
2036	-	5–34	10–65	14–93	18–118	21–141	24–162	27–180	29–196	30–211	32–223	33–233	34–243	35–277	27–261
2037	-	5–37	10–71	14–102	17–129	21–154	23–175	26–194	28–210	29–225	30–237	31–248	32–258	32–286	24–262
2038	-	5–41	10–78	13–111	17–140	20–166	22–188	25–208	26–225	28–240	29–252	30–263	30–272	29–296	21–263
2039	-	5–45	9–85	13–121	17–152	19–180	22–203	24–223	25–242	26–256	27–269	28–279	28–288	26–307	18–266
2040	-	5–50	9–94	13–132	16–166	19–195	21–220	23–241	24–259	25–274	26–287	27–297	27–305	24–317	15–267
2041	-	5–54	9–102	13–144	16–180	18–211	20–237	22–260	23–278	24–294	25–305	25–315	25–322	22–328	13–268
2042	-	4–60	9–112	12–157	15–196	18–229	19–256	21–279	22–298	23–314	23–325	24–335	24–342	20–340	12–269
2043	-	4–65	9–122	12–171	15–213	17–248	19–277	20–301	21–320	22–337	22–348	22–356	22–363	18–353	10–270
2044	-	4–72	8–134	12–187	14–231	16–270	18–301	19–324	20–344	21–361	21–372	21–380	21–383	16–365	9–270
2045	-	4–79	8–146	11–204	14–252	16–292	17–325	19–350	19–369	20–386	20–398	20–406	20–409	15–376	7–270
2046	-	4–87	8–160	11–222	13–274	15–317	17–351	18–378	18–397	19–413	19–426	19–433	19–435	13–389	6–271
2047	-	4–95	8–176	11–242	13–298	15–344	16–379	17–407	18–428	18–444	18–454	18–461	18–462	12–402	5–273

Table S6. Conservative maximum number of goosanders to be removed in each year under each scenario – lower confidence interval for the estimated value of *n*% of the population in each year, based on population estimates in Appendix 1, Table S4.

Year	Baseline	0.5%	1%	1.5%	2%	2.5%	3%	3.5%	4%	4.5%	5%	5.5%	6%	10%	15%
2021	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2022	-	6	12	18	24	30	36	42	48	53	59	65	70	114	163
2023	-	6	12	18	24	30	35	41	47	52	57	63	68	106	146
2024	-	6	12	18	23	29	35	40	45	50	55	60	65	98	130
2025	-	6	12	17	23	29	34	39	44	49	53	58	62	91	115
2026	-	6	12	17	23	28	33	38	42	47	51	55	59	84	101
2027	-	6	11	17	22	27	32	37	41	45	49	53	56	77	90
2028	-	6	11	17	22	27	31	35	39	43	47	50	53	71	79
2029	-	5	11	16	21	26	30	34	38	41	45	48	51	65	69
2030	-	5	11	16	21	25	29	33	37	40	43	45	48	59	61
2031	-	5	11	16	20	25	28	32	35	38	41	43	45	54	53
2032	-	5	11	15	20	24	28	31	34	36	39	41	43	50	46
2033	-	5	10	15	19	23	27	30	32	35	37	39	40	45	40
2034	-	5	10	15	19	23	26	29	31	33	35	37	38	41	36
2035	-	5	10	15	18	22	25	28	30	32	33	35	36	38	31
2036	-	5	10	14	18	21	24	27	29	30	32	33	34	35	27
2037	-	5	10	14	17	21	23	26	28	29	30	31	32	32	24
2038	-	5	10	13	17	20	22	25	26	28	29	30	30	29	21
2039	-	5	9	13	17	19	22	24	25	26	27	28	28	26	18
2040	-	5	9	13	16	19	21	23	24	25	26	27	27	24	15
2041	-	5	9	13	16	18	20	22	23	24	25	25	25	22	13
2042	-	4	9	12	15	18	19	21	22	23	23	24	24	20	12
2043	-	4	9	12	15	17	19	20	21	22	22	22	22	18	10
2044	-	4	8	12	14	16	18	19	20	21	21	21	21	16	9
2045	-	4	8	11	14	16	17	19	19	20	20	20	20	15	7
2046	-	4	8	11	13	15	17	18	18	19	19	19	19	13	6
2047	-	4	8	11	13	15	16	17	18	18	18	18	18	12	5

Table S7. Percentage of models (from the subset of plausible models) that predict each of five possible outcomes, under a hypothetical scenario where cormorant productivity has undergone 10 years of decline at a known rate (Cook and Robinson 2010). The difference to the percentage of models predicting each outcome within the main analysis (Table 3) is given in brackets.

Species	% population removed per year	Max. number removed per year (at outset)	% models predicting continued population increase (change from current productivity models)	% models predicting population decline below alert thresholds (change from current productivity models)	% models predicting change in conservation status to alerted decline (change from current productivity models)	Of which, % models predicting a WeBS alert triggered after…		
						25 years	10 years	5 years
Cormorant	0	0	20.0 (-15.5)	10.2 (-2.4)	69.8 (+17.9)	18.0	29.7	22.1
	0.5	12–16	16.9 (-14.7)	9.6 (-3)	73.4 (+17.6)	17.3	30.5	25.6
	1	25–32	14.5 (-13.6)	8.5 (-3.6)	77.0 (+17.2)	16.8	31.1	29.2
	1.5	38–48	12.3 (-12.2)	7.6 (-3.8)	80.1 (+16.0)	15.5	31.0	33.7
	2	51–65	10.2 (-11.0)	6.7 (-3.7)	83.2 (+14.8)	14.8	31.2	37.2
	2.5	64–81	8.2 (-10.0)	6.1 (-3.9)	85.7 (+13.9)	13.8	29.9	41.9
	3	77–97	6.6 (-8.7)	5.3 (-4.1)	88.1 (+12.8)	12.5	29.6	46.0
	3.5	90–114	5.3 (-7.7)	4.6 (-3.8)	90.1 (+11.5)	11.2	28.3	50.6
	4	103–130	4.3 (-6.8)	4.0 (-3.5)	91.6 (+10.2)	9.8	27.2	54.7
	4.5	116–146	3.4 (-5.9)	3.3 (-3.1)	93.3 (+9.0)	8.9	25.4	59.0
	5	129–162	2.6 (-5.2)	2.7 (-3.3)	94.6 (+8.0)	7.6	23.9	63.2

	5.5	141–179	2.0 (-4.4)	2.4 (-2.3)	95.6 (+6.8)	6.4	22.4	66.9
	6	154–195	1.5 (-3.9)	1.8 (-2.3)	96.7 (+6.2)	5.6	20.1	70.9
	10	258–325	0.2 (-0.6)	0.2 (-0.8)	99.6 (+1.4)	1.0	6.7	91.9
	15	387–488	0.0 (-0.1)	0.0 (-0.1)	100.0 (+0.2)	0.1	0.7	99.2

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