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An Assessment of Sufficiency of Data Availability on UK Waterbird Harvests for Accurately Estimating the Scale and Sustainability of Harvest of AEWA-Listed Waterbird Populations

(Research and Review Report)

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Please note that this report was republished in November 2025 (version 1.1), to address some minor corrections to Table 1. Tufted Duck (North-west Europe (winter) population), Mallard (North-west Europe population), and Greylag Goose (North-west Europe / South-west Europe population) were incorrectly listed as huntable in Guernsey. Note that Tufted Duck is not included on Guernsey's game list, and therefore its hunting is prohibited under the Animal Welfare Ordinance (2012); hunting of Mallard is restricted to only cross-bred or hybrid Mallards; and all Greylag Goose hunted are a feral population.

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The authors declare no financial or personal interests which may be considered as actual or potential competing interests in the delivery of this report. MBE is employed by BASC, the UK's largest representative body for shooting sports.

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Evidence Quality Assurance:
This report is compliant with JNCC's Evidence Quality Assurance Policy.

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Summary

A number of waterbird species that are legally huntable in the UK are listed on Table 1 of the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA), to which the UK is a Contracting Party. As such, the UK has an obligation to ensure that any use of these species is sustainable. In order to establish if hunting is sustainable, Parties are required to understand the numbers of birds hunted – so-called "hunting bags". Currently, UK hunters voluntarily record data on hunting bags, but there is no systematic, mandatory, or enforceable system to collect this information. It is therefore important to understand whether the current data collection on hunting bags is sufficient to reliably determine whether the harvest of AEWA-listed species and populations is sustainable.

This desk-based study describes the various scenarios in which waterbirds are currently hunted in the UK. It examines the available data on hunting bags for AEWA-listed species populations across the UK and relevant Crown Dependencies and Overseas Territories and reviews the sufficiency of evidence for the size and composition of hunting bags. It includes an assessment of the robustness of recent assessments of the scale (Aebischer 2019) and sustainability (Ellis & Cameron 2022) of UK waterbird hunting, which are reliant on understanding the accuracy of hunting bag and population estimates, and identifies limitations in the underlying data. It evaluates other modelling options available to generate sustainable harvest estimates. It concludes by identifying gaps in the current evidence base that prevent the accurate estimation of the scale of harvest of AEWA-listed bird populations in the UK and suggests options for filling them, with an assessment of their advantages and disadvantages.

Schemes for collecting data on UK waterbird harvest

We evaluated four current methods of collecting harvest data (the Game & Wildlife Conservation Trust (GWCT)'s National Gamebag Census (NGC); the Crown Estate's wildfowling returns; the British Association for Shooting & Conservation's Wing Survey; and a formal but irregular large scale, nationwide assessment of hunting participation and behaviour (The Public and Corporate Economic Consultants [PACEC] 2006 & 2014 surveys and the associated forthcoming Value of Shooting report). We also describe the potential to obtain data informally from commercial game shoots, industry surveys or private hunting records.

Adequacy of waterbird harvest data collection in the UK

In the UK, there is at least some collection of harvest data which covers all legally huntable species, but the data is partial, limited, probably biased, and is not always publicly available. The harvest data we do have is at a species level only and cannot be disaggregated to the AEWA population level. Assessments of Mallard harvests are confounded by the release and harvest of perhaps several million of these birds annually, with little ability to distinguish wild from reared birds, and these birds comprise the majority of all waterbirds harvested in the UK. There is no data collected on crippling or unretrieved harvest, and limited data on age and sex of harvested birds. The current data is likely submitted by a very limited subset of Guns, and it is not known how representative those that do submit data are of the total population of UK Guns. The current data schemes provide only partial geographic coverage of UK waterbird harvest; consequently, this skewed coverage is unlikely to provide a representative and accurate national harvest assessment. A key knowledge gap therefore is that while we have estimates, we currently do not know accurately the number harvested of any waterbird species in the UK and our current margins of error are often of an order of magnitude.

Most harvest data collection schemes collect data on a regular enough basis to allow triennial estimates of harvest, but to replicate the Aebischer (2019) approach, access to privately held data would need to be negotiated, and regular updates of the total harvest estimates (e.g. from PACEC/Value of Shooting surveys) are required, or a new method developed.

Sufficiency of the available data for assessing the sustainability of UK waterbird harvest

None of the current methods for estimating harvest are likely to provide a full and accurate representation of wildfowl harvest in the UK. In addition, our current estimates of waterbird population sizes in the UK and across their flyways are poor with large confidence intervals. We currently lack robust age and sex specific life-history data for many UK waterbird species, and we have a poor understanding of productivity and survival of waterbirds in a UK context.

There is poor or no data about how we might expect waterbird populations to change under future scenarios of climate, environment and human behaviour. Crucially, we lack any information about the hunting behaviour and attitudes of Guns that may allow an understanding of how they might adjust their behaviour under different future scenarios.

Consequently, accurately and precisely modelling the sustainability of UK waterbird harvest is difficult. However, the lack of "perfect" information on harvest and life history variables is not a barrier to starting models to assess the sustainability of harvest for waterbirds – if uncertainty is considered. The currently collected data on harvest, combined with population estimates, are sufficient to allow for an initial assessment of the sustainability of the harvest.

Reliability of a recent assessment of the sustainability of UK waterbird harvest

We found that the conclusions of Ellis and Cameron (2022) are reliable within the limitations of the best available data at the time and the assumptions they made. The biggest limitations of Ellis and Cameron were assuming that populations and harvest were static and in not exploring the effect of assuming different risk appetites. However, for most species, the level of harvest is small compared to the populations and so even in the presence of large uncertainties and limitations in existing assessments, we can be confident that harvest is likely to be sustainable. For species where harvest represents a greater proportion of the population (including Mallard, Wigeon, Teal and Greylag Goose) the uncertainty around both populations and harvests means we can be less certain that harvest is sustainable, but we can estimate the degree of uncertainty. Generally, the uncertainty in our assessments of sustainability increases as harvest (as a proportion of the population) increases.

Modelling approaches for assessing the sustainability of waterbird harvest

We found that, using the available harvest data, three approaches for modelling sustainable harvest (Potential Excess Growth [PEG], Prescribed Take Level [PTL], and matrix-based Population Viability Analysis) produced qualitatively similar outcomes. Generally, we found that the Sustainable Harvest Index (SHI) for most species is highly susceptible to the Management Objective (F_{obj}) (the proportion of the maximum sustainable yield deemed to be politically acceptable to harvest) and the Safety Factor (F_s) (an index of the risk that populations will decline if environmental factors change).

Conclusions on the current status of data on the sustainability of UK waterbird harvest

Our review indicates that the *status quo* is not providing a sufficient quantity and quality of data to model accurately and precisely the current level of waterbird harvest in the UK. However, we also conclude that the lack of "perfect" information on life history variables, population sizes or harvest bags is not a barrier to starting to use basic models to estimate the sustainability of harvest, which can be refined as data quantity and quality are increased.

The absence of robust and representative harvest data, coupled with uncertainty over waterbird population size estimates, means that our models have broad confidence intervals, so that determining sustainability of harvest, especially for commonly harvested species, is difficult. The rearing and release of perhaps several million Mallard annually mean that harvested Mallard are likely to be largely composed of farmed rather than wild birds, meaning that the harvest of this species is also likely to be sustainable, but the releases may affect the wild population in other ways.

Recommendations to improve assessment of the sustainability of waterbird harvest in the UK

To determine more reliably the sustainability of the harvest of AEWA waterbird species, our key recommendation is that the quantity and quality of data relating to waterbird harvest and Gun behaviour are markedly improved. We suggest a number of options as to how this might be achieved. Our preferred approaches to achieve this are, in order of priority:

1) To develop and implement a national reporting scheme to provide detailed and accurate harvest records by species, age, sex, harvest type and, for Mallard, whether the birds are wild or released. We believe that a voluntary approach, backed by shooting and conservation organisations should be tried initially, resorting to a more costly and perhaps less accepted mandatory approach if there is not a marked increase in reporting within a few years.
2) To conduct a series of smaller scale desk-based analyses of existing data sets (British Trust for Ornithology ringing data, PACEC/Value of Shooting surveys of Gun attitudes and behaviour, Animal and Plant Health Agency Poultry Register) to increase data quantity and quality on which models can be based. These may contribute only incremental improvements to current models but are likely to be of relatively low cost.
3) To develop and validate prospective, rather than simply retrospective, models of harvest sustainability that can account for likely changes in waterbird ecology and life history and Gun behaviour.

Novel data collection methods and model validations are only feasible with the cooperation of the shooting community, and we recommend that concerted efforts are made to engage its members in this endeavour by deriving and demonstrating benefits to all stakeholders. We note that some of these recommendations have been made repeatedly over the past 20 years (Parrot *et al*. 2003; Aebischer & Harradine 2007) but have not yet been enacted.

Continued poor data risks leading to inaccurate or sub-optimal decisions being made about waterbird harvests and this fails all stakeholders. Overall, by improving data quality, primarily by increasing coverage and representation, we are confident that, using current modelling approaches, it is possible to better determine the sustainability of the harvest of AEWA waterbird species, satisfying the obligations of the UK to the agreement.

Contents

Glossary

Adaptive harvest: a framework for making objective decisions about the number of individuals in a population that can be removed (harvested) in the face of incomplete information. It is adaptive because it considers new knowledge as harvest and population data is updated and thus predicted and observed population data can be compared.

Density dependence: a force that affects the size of the population of individuals in response to the density of the population, given as the number of individuals found in a unit area.

Driven game shooting: Where gamebirds are induced to fly towards a line of Guns, disturbed and directed by a line of people and dogs, typically being shot at while they are flying towards the Gun.

Ecological carrying capacity: the number of individuals/population size that can be supported in a given area, determined by the availability of resources in the area.

Flight pond: an open area of fresh water or marsh used by wildfowl in the evening, when they fly from their daytime resting places on estuaries or large water bodies in order to feed.

Gamebird: a common term to describe bird shot for sport, including pheasants, partridge, grouse and wildfowl and, in the UK, waterbirds.

Guns: throughout the report, we will use the term "Guns" or "a Gun" to refer to waterbird hunters. In the UK, "hunting" or "hunter" has a strong association with mounted fox/stag hunting. The term "shooter" has acquired pejorative connotations. The term "wildfowler" typically describes those shooting wild duck and geese below the high tidemark, implicitly excluding other waterbird hunting scenarios. "Guns" is a term commonly used to describe people who shoot gamebirds, and we extend this to cover shooting of waterbirds. We capitalise it to distinguish it from the shotguns/firearms used in shooting.

Harvest: shooting, killing and recovering a gamebird.

Hunter recall: the accuracy and precision with which a Gun reports their harvest. This may be affected by accidental misreporting due to misidentification or forgetting, or deliberate misreporting due to prestige (not wanting to admit to having failed to shoot any birds) or non-compliance (not wanting to admit to having shot a larger bag than permitted or shooting prohibited species).

Popharvest: a modelling computer package developed to investigate population effects of harvest activity.

Potential Excess Growth: the number of individuals which may be removed from a population under idealised circumstances without causing a population decline.

Prescribed Take Level: the maximum number of individuals that can be removed from a population while allowing it to reach or maintain equilibrium to a certain level of its carrying capacity.

Shot count: the number of shots fired by a Gun during a shoot.

Shoot: (noun) the location, perhaps an estate or farm where organised game shooting occurs

Stage specific population sizes: the number of individuals of particular ages or life-history stage in a population

Stage-structured models: A modelling approach that represent one of two different biological mechanisms: (1) natural biological stages and (2) continuous growth or development. Examples of the former include the various instars in arthropod growth, and of the latter where stages are used to simplify the representation of otherwise continuous growth following hatching in birds (e.g. chick, juvenile, adult).

Sustainable Harvest Index: the ratio of the harvest to Potential Excess Growth (see above), such that an index of one means that 100% of the Potential Excess Growth is harvested and the population would be predicted to remain static.

Walked up game shooting: where birds are disturbed and induced to fly by the approach of the Gun themselves, typically being shot at while they are flying away from the Gun.

Waterbirds: in this report, we use the term waterbird to refer to the ducks, geese, waders and rails that are legally harvested by shooting in the UK and which are listed in Table 1 of the AEWA Action Plan.

Wildfowl: a common term to describe a subset of gamebirds that are predominantly aquatic, which typically includes waterbirds. In the USA, the term Waterfowl is commonly used synonymously.

In the text, throughout the report, we will refer to waterbirds by their British common names, (e.g. Teal, rather than Common Teal; or Pochard, rather than Common Pochard), unless we are directly contrasting them with similarly named birds. In Tables and Figures, we use their longer common names as used in AEWA Table 1 (where applicable) and their scientific names so that the Tables can be read independently without ambiguity.

List of Abbreviations

Acronym	Description
AEWA	African-Eurasian Migratory Waterbird Agreement
APEP	Avian Population Estimates Panel
APHA	Animal and Plant Health Agency
BASC	British Association for Shooting & Conservation
BTO	British Trust for Ornithology
GWCT	Game & Wildlife Conservation Trust
HOST	Height of Ordinary Spring Tide
NGC	National Gamebag Census
PACEC	Public and Corporate Economic Consultants
PEG	Potential Excess Growth
PTL	Prescribed Take Level
PVA	Population Viability Analysis
SHI	Sustainable Harvest Index

1. Introduction

1.1. Aims and rationale for this report

A number of waterbird species that are legally huntable in the UK are listed on Table 1 of the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA), to which the UK is a Contracting Party. As such, the UK has an obligation to ensure that any use of these species is sustainable. To establish if harvest is sustainable, Parties are required to understand the numbers of birds hunted – so-called “hunting bags”. Currently, UK hunters voluntarily record data on hunting bags, but there is no systematic, mandatory, or enforceable system to collect this information. It is therefore important to understand whether the current data collection on hunting bags is sufficient to reliably determine whether the harvest of AEWA-listed species and populations is sustainable.

The aims of this report are:

1. To review and assess the sufficiency of evidence for the size of UK hunting bags of waterbird populations listed in AEWA Action Plan Table 1.
2. To identify any important evidence gaps and suggest options for filling them.
3. To review the reliability of a recent assessment (Ellis & Cameron 2022) of the sustainability of UK waterbird hunting and the robustness of the methods used.

The review will consider the harvest of waterbirds by legal means (hunting), but harvest that is permitted via licenced take and/or falconry is outside the scope of this report.

1.2. Which AEWA-listed waterbird populations are legally hunted in the UK?

For the UK, Isle of Man and Guernsey, three species of goose, 10 species of duck, four species of waders and two species of rail listed in AEWA Action Plan Table 1 can be shot in at least one country or Crown Dependency/territory (Table 1).

There is no legal hunting of any species listed on Table 1 of the AEWA Action Plan in Jersey, Gibraltar, or St Helena, Ascension and Tristan da Cunha.

Table 1. The huntable status of species and populations of waterbirds listed in Table 1 of the AEWA Action Plan throughout the UK and its Crown Dependencies and territories. There are no legally huntable species from Table 1 of the AEWA Action Plan in Jersey, Gibraltar, St Helena, Ascension and Tristan da Cunha.

Species or subspecies listed in AEWA Table 1	**Population listed in AEWA Table 1**	**England & Wales**	**Scotland**	**Northern Ireland**	**Isle of Man**	**Guernsey**
Greylag Goose *Anser anser anser*	Iceland/ UK & Ireland	X	X	X	-	-
Greylag Goose *Anser anser anser*	North-west Europe/ South-west Europe	-	-	-	-	-
Pink-footed Goose *Anser brachyrhynchus*	East Greenland & Iceland/ UK	X	X	X	-	-
Greater White-fronted Goose *Anser albifrons albifrons*	North-west Siberia & North-east Europe/North-west Europe	X	-	-	-	-
Common Goldeneye *Bucephala clangula clangula*	North-west & Central Europe (winter)	X	X	X	-	-
Common Pochard *Aythya ferina*	North-east Europe/North-west Europe	X	X	X	-	-
Tufted Duck *Aythya fuligula*	North-west Europe (winter)	X	X	X	-	-
Greater Scaup *Aythya marila marila*	Northern Europe/Western Europe	-	-	X	-	-
Northern Shoveler *Spatula clyptea*	North-west & Central Europe (winter)	X	X	X	-	-
Gadwall *Mareca strepera strepera*	North-west Europe	X	X	X	-	-

Species or subspecies listed in AEWA Table 1	**Population listed in AEWA Table 1**	**England & Wales**	**Scotland**	**Northern Ireland**	**Isle of Man**	**Guernsey**
Eurasian Wigeon *Mareca penelope*	Western Siberia & North-east Europe/North-west Europe	X	X	X	X	-
Mallard *Anas platyrhynchos*	North-west Europe	X	X	X	X	-
Northern Pintail *Anas acuta*	North-west Europe	X	X	X	-	-
Common Teal *Anas crecca crecca*	North-west Europe	X	X	X	X	-
Eurasian Golden Plover *Pluvialis apricaria altifrons*	Northern Europe/Western Europe & North-west Africa	X	X	X	-	-
Eurasian Woodcock *Scolopax rusticola*	Europe/South & West Europe & North Africa	X	X	X	X	X
Common Snipe *Gallinago gallinago gallinago*	Europe/South & West Europe & North-west Africa	X	X	X	X	X
Jack Snipe *Lymnocryptes minimus*	Northern Europe/South & West Europe & West Africa	-	-	X	-	-
Common Moorhen *Gallinula chloropus chloropus*	Europe & North Africa	X	X	-	-	-
Common Coot *Fulica atra atra*	North-west Europe	X	X	-	-	-

1.3. Types of waterbird harvest in the UK

The harvest of waterbirds in the UK occurs in four distinct scenarios (Table 2). These take place in different habitats, involve different levels of management, demand different methods of encountering and engaging with the birds, and often involve different sets of Guns, likely exhibiting differing general attitudes to hunting and conservation, and consequently resulting in markedly different scales of harvest both per visit and overall.

These four situations of shooting waterbirds have very different implications for our understanding of harvest of waterbirds, the subsequent effects on their populations and any proposed methods to improve accurate harvest data collection. Consequently, when developing improved monitoring of waterbird harvest, it is important to clearly understand and demark these situations and address the data and participants for each situation explicitly. These key differences between situations comprise: differences in the species encountered; differences in the numbers of waterbirds harvested; differences in the oversight or ownership of shooting rights; differences in the number of Guns involved; differences in the attitudes of the Guns involved; differences in the ratio of released Mallard to wild Mallard harvested.

We briefly describe the four situations below. There is some blurring of boundaries between the four situations – Guns on a driven day may end the day hidden around a pond at dusk to await the arrival of roosting birds; or Guns situated in coastal areas above the high tide may target birds moving inland with tidal movements; or some Guns may informally drive waterbirds towards an ambush at an inland pond during a less formal day of shooting – but consideration of these different hunting situations is useful. Throughout the rest of the report, we will use the terms defined below, acknowledging that they are imperfect descriptors.

1.3.1. Wildfowling

This activity occurs in coastal areas including grazing and salt marsh, mud flats and estuaries, typically below the Height of Ordinary Spring Tide (HOST) but can also include inland areas (such as the Ouse Washes and the Somerset Levels). This land is commonly Crown Estate owned foreshore although peripheral areas where wildfowling can occur may be privately owned and/or leased. Shooting of huntable wildfowl (ducks and geese) below the HOST is permitted September 1 to February 20 in England, Scotland and Wales) while that above the HOST, these birds can be harvested September 1 to January 31. In other UK countries, territories and dependencies, the tidal range is irrelevant to hunting seasons. Shooting on them is generally practiced by Guns operating alone or in pairs/small groups and the style of shooting generally involves the Gun ambushing naturally flighting birds whose movements are determined by tidal and diurnal patterns. Guns may entice birds within range by using decoy models or giving calls. In England and Wales this activity is often organised via clubs. In Scotland there are many more areas where access to the shore for wildfowling is via access rights, but there are also some clubs and several permitted public access coastal wildfowling schemes (e.g. Montrose Basin).

1.3.2. Inland Waterbird Shooting

This activity occurs above the HOST and thus typically occurs on private land. Habitats can include coastal areas bordering marshes, flats and estuaries, but it also involves ponds, rivers and arable land. Some landowners or keepers may deliberately construct attractive habitats to draw in more waterbirds. This form of shooting may involve individual Guns, or more organised groups who collectively surround a single, or neighbouring, attractive features. Our defining feature of this sort of shooting is that the Gun ambushes waterbirds as they arrive at, leave or fly over these features. For inland wetlands-based shooting the users may attempt to attract birds within range using decoy models or giving calls, but it is more common that feed is provided, or sanctuary provided through predator control. For inland field-based shooting, natural food concentrations and behaviours may be exploited where users attempt to attract birds within range using decoy models or giving calls, for example targeting geese arriving to feed on crops in fields.

1.3.3. Shooting of wildfowl as part of driven game shoots (Driving Wild Birds)

This activity occurs as part of a larger organised shooting event where Guns are accompanied by other people (Beaters) who may flush out and drive birds towards the Guns and people (Pickers up) who search for and retrieve birds that have been shot but not fallen close to the Guns. Typically, such driven shoots target gamebirds (pheasant, partridge or grouse) with any waterbirds that are encountered as part of the drive being additional harvest. These can include Woodcock and duck encountered in wet woodlands, Common Snipe driven off wet field margins or Golden Plover and/or Common Snipe disturbed on moorland. However, in some situations, Guns may deliberately set out to drive Woodcock and/or Common Snipe, searching out wet woodland or marshes for this purpose. This form of shooting usually involves groups of approximately 6–10 Guns shooting at multiple locations (drives) during a day. Birds are deliberately driven out of their residence and over the guns. Landowners/keepers may manage their land to attract waterbirds to the general area or specific places on predetermined drives, or to increase the natural population to increase harvest during those drives. This may be achieved by creating attractive habitats, supplementary feeding for ducks, the provision of breeding sites or the control of predators to increase breeding success or attractiveness to safe areas.

1.3.4. The artificial rearing and release of Mallard for driven shooting (Driving Released Birds)

Large numbers of waterbirds, specifically Mallard, are driven from ponds over groups of Guns in organised, formal shoots, as in Driving Wild Birds. However, when such drives specifically target Mallard, most of these birds will have been artificially reared and then released to provide sufficient, predictable quarry. The Mallard are reared and released in late summer onto sites, usually ponds, where regular food is provided and predators excluded. The Mallard are imprinted to either return to their home pond or fly to another one nearby on the same shooting estate where food and protection are regularly provided. These flocks of released Mallard can attract wild Mallard and other ducks which may also be shot when the pond is driven. This form of shooting again involves groups of approximately 6–10 Guns shooting at multiple locations (drives) during a day. On some shoots, the entire day may focus on shooting Mallard from a series of such ponds, while on others a pond may provide just one drive during the day. Landowners may manage their land to retain released Mallard and this may serve to also attract wild waterbirds.

Table 2. Summary of the different scenarios in which waterbirds in the UK are harvested.

Scenario	**Wildfowling**	**Inland Waterbird Shooting**	**Driving Wild Birds**	**Driving Released Birds**
Typical species harvested	All geese Eurasian Wigeon *Mareca penelope* Common Teal *Anas crecca* Mallard *Anas platyrhynchos* Northern Pintail *Anas acuta* Common Goldeneye *Bucephala clangula* Northern Shoveler *Spatula clyptea*	All geese Mallard *Anas platyrhynchos* Common Teal *Anas crecca* Gadwall *Mareca strepera* Eurasian Wigeon *Mareca Penelope* Tufted Duck *Aythya fuligula* Common Pochard *Aythya ferina*	Mallard *Anas platyrhynchos* Common Teal *Anas crecca* Gadwall *Mareca strepera* Eurasian Woodcock *Scolopax rusticola* Common Snipe *Gallinago gallinago* Eurasian Golden Plover *Pluvialis apricaria* Common Moorhen *Gallinula chloropus* Common Coot *Fulica atra* Jack Snipe *Lymnocryptes minimus*	Mallard *Anas platyrhynchos* (may include wild birds attracted by reared conspecifics)
Location of shooting	Foreshore and coastal areas	Freshwater ponds, rivers, wetlands and agricultural land	Lowland and upland areas including woodland, arable, improved and unimproved grassland and montane habitats	Lowland and upland areas with high densities of ponds/lakes
Order of magnitude of total annual harvest (Aebischer 2019)	Tens of thousands	Tens to Hundreds of thousands	Hundreds of thousands	Million

Scenario	Wildfowling	Inland Waterbird Shooting	Driving Wild Birds	Driving Released Birds
Order of magnitude of annual participants (Public and Corporate Economic Consultants 2014)	Tens of thousands	Tens of thousands	Hundreds of thousand	Hundreds of thousands
Current methods of data collection	British Association for Shooting & Conservation Wing Survey; Crown Estate Wildfowling Returns	British Association for Shooting & Conservation Wing Survey	Game & Wildlife Conservation Trust National Gamebag Census	Game & Wildlife Conservation Trust National Gamebag Census

2. Current methods and schemes for collecting harvest data

2.1. Overview of current methods and schemes for collecting harvest data

There is no statutory collection of waterbird harvest data in the UK or any of its Crown Dependencies and territories. Harvest data can be collected both formally (through ongoing established schemes or surveys by organisations and their members) and informally (through appeals to individuals or clubs for private records, or collation of publicly available information not specifically intended for this purpose). There are three current schemes operating in the UK that can provide indications of harvest of waterbirds, specifically the relative proportions of different species harvested annually: (Section 2.2) the Game & Wildlife Conservation Trust's (GWCT) NGC, (Section 2.3) the Crown Estate Wildfowling Returns and (Section 2.4) the BASC Wing Survey. There is a dataset arising from a formal but irregular large scale, nationwide assessment of participation in hunting which may inform attempts to estimate absolute, species and population numbers of waterbirds harvested (The Public and Corporate Economic Consultants [PACEC] 2006 & 2014 surveys and the associated forthcoming Value of Shooting report [Section 2.5]). In addition to these four data sources, we also describe the potential to obtain data informally on the scale and geographical distribution of waterbird harvests from advertising commercial game shoots or sporadic surveys (Section 2.6) or studies of Guns, shoots and game managers which may contribute to attempts to estimate absolute, species and population numbers of waterbirds harvested (Section 2.7). Finally, we describe data that may be collected informally and privately by individual Guns, clubs or shooting estates that may be used to assess absolute and/or relative harvest of waterbirds (Section 2.8).

These formal schemes and methods are all specific to the UK. We are not aware of any coordinated or formal schemes to record harvest of waterbirds in the Isle of Man or Guernsey, although individual Guns, clubs or shooting estates on these islands may collect and hold private records. There is no legal hunting of any species on Table 1 of the AEWA Action Plan on Jersey, Gibraltar, St Helena, Ascension or Tristan da Cunha (the last four of which are those UK Overseas Territories to which AEWA applies) and so no attempt has been made to review harvest data collection in these territories.

Harvest data at a national level is the product of actions by individual Guns who may be hunting individually or as part of organised shoots. Therefore, data on harvests can be obtained from submissions by, or recordings of the hunting activity of, individual Guns, or it can be obtained at a higher level from submissions by, or recordings of, commercial shoots. Current schemes rely on the participation and cooperation of Guns and/or shoots.

2.2. The National Gamebird Census (NGC)

This is a voluntary scheme, managed by the GWCT. It collects data annually from approximately six hundred sites across the UK with over 1,800 sites having submitted data since 1961, when it was established (Aebischer 2019). The distribution of sites covers all the UK (Appendix 1). The NGC aims to provide a central repository of game records from shooting estates across the UK. It includes long term data series for 24 huntable bird species (and a further 11 "pest" bird species). This scheme has been maintained over many years, and with the addition of historical records from a smaller number of estates, harvest patterns for some species may be derived spanning multiple decades. The data is held by the GWCT, with published species-specific bag estimates available for 2004, 2012 and 2016. Further information about the scheme is available at the NGC website.

2.2.1. Limitations

Participation in the NGC is self-selecting and there is an over-representation of large shoots (see below), such that wildfowling clubs and those Guns engaged in less formal Inland Waterbird Shooting or roaming guides who target Common Snipe and Woodcock (Driving Wild Birds) are likely to be under-represented.

Analysis of the geographic representativeness of the NGC published in Aebischer and Harradine (2007) (Appendix 1) shows relatively low representation from Wales and Northern Ireland. The annual returns from the 600 shoots represent a sample of ~ 10% of the estimated number of organised shoots in the UK (Madden 2021). The shoots participating in the NGC differ from those reporting to the Animal and Plant Health Agency (APHA) Poultry Register (obligatory for those shoots releasing birds for shooting), with those participating in the NGC being an over-representation of "large" shoots and an under-representation of "small" shoots (as defined in Madden 2021). Nationally, small shoots make up around 71% of those in the UK, while medium shoots comprise 19% and large shoots comprise approximately 10%. This means that data from the NGC will likely underrepresent the behaviour of the majority of Guns but will capture an indication of the highest levels of harvest occurring at these large shoots, at least for some driven harvest. Because released Mallard are more likely to be shot during driven shoots, with the great majority of the ducks harvested in that method being released Mallard (1.3.4), the NGC may tell us relatively little about the harvest of wild Mallard.

The data is skewed in terms of species representation towards the more commonly harvested or widespread species. Harvest data for Mallard, Teal, Woodcock and Common Snipe are available from an average of more than 100 sites/year (Appendix 1, Table A1.1), but for other species, harvest data is available from fewer than 50 sites/year (apart from for Moorhen at 65 sites/year).

The accuracy of the data is contingent on the identification skills of the individual Gun or keeper. In the USA, Guns demonstrated a wide range of identification abilities (Wilson & Rowher 1995; Ahlers & Miller 2019), but for geese shot in Denmark, Guns demonstrated better in-hand identification abilities with a range from 74% to 99% (Christensen *et al*. 2017a). We are not aware of similar tests of identification skills in the UK.

The data from the NGC can only provide an index or relative measure of harvest of each species. This data has been used to estimate absolute harvest levels by combining them with total bag records collected in the PACEC surveys (see below) (Aebischer 2019).

2.2.2. Data type

The data that might be used in assessing harvest levels may be inferred from the data collection form that the NGC is based on (Annual bag record guidelines). It includes:

- The area of the shoot from which the harvest was obtained.
- The number of gamekeepers engaged in managing the shooting.
- The period covered by each submission.
- The numbers of each species that were shot (with options to report that additional unknown numbers of the species were also killed or present).
- The opportunity to report the sex and age structure of the harvest. This is explicitly directed towards gamebirds and not waterbirds but presumably could be adapted to include wild waterbirds.

- The number of birds that have been reared and released, explicitly including Mallard.
- The number of driven and walked up days during which the harvest was conducted. This is critical to account for harvest effort.

2.3. Crown Estate wildfowling returns

The Crown Estate owns approximately 55% of all foreshores (mean high water to mean low water) in the UK. Prior to 1994 there existed a general agreement between BASC and the Crown Estate which allowed open access to wildfowling (largely through preexisting club structures) on the English, Welsh and Northern Irish foreshore. From 1994 onwards wildfowling leases on the Crown Estate were instead managed through the Joint Group on Wildfowling and Conservation on Tidal Land which required clubs to submit annual bag returns as a condition of their leases.

BASC acted, and continues to act, as the administrator for the annual returns and consequently has annual club summaries of the number of visits and number of quarry harvested per club from 1994/95 to date and the Crown Estate makes the annual summary data publicly available. This data only covers England, Wales and Northern Ireland, because Scotland retains a public right of recreation (including wildfowling) on the foreshore. The Crown Estate data is also supplemented by some clubs who submit additional bag return data to BASC either for analysis for private landlords or for information. The land that clubs can access is often a patchwork of Crown Estate, private and public landlords, with some areas owned by the clubs themselves. This data can be used to provide an estimate of the total harvest of wildfowl by wildfowlers by estimating the bag per visit per species on each club each year and scaling this up to national estimates. This approach is currently under peer review.

2.3.1. Limitations

Records from the Crown Estate wildfowling returns are likely to be incomplete. Although providing harvest returns is a lease condition and thus should provide complete and accurate data, there is little if any enforcement of this and it is unknown how reliable the returns are or who holds and shares data from each lease. They depend on individual wildfowlers reporting accurate harvest returns and the clubs collating and returning them, presenting several opportunities for error, data “noise” or omission to enter the chain of information.

Records from the Crown Estate wildfowling returns are likely to only represent a very small proportion of the waterbirds harvested in the UK. A recent estimate put the coastal wildfowl harvest at 1% of all duck hunting mortality in the UK (Ellis & Cameron 2022), certainly a maximum of 10%, leading to a fair question of how effective and efficient the regulatory focus on coastal wildfowling is when it misses out 90%+ of wildfowl harvest mortality, which largely occurs inland. In the 1980s the BASC shooting survey estimated that the total annual duck harvest across the UK was 980,000. This survey was also based on combining the NGC and WSS data (Aebischer & Harradine 2007). At that time 105,000–120,000 of those ducks were estimated to be shot on the foreshore, representing 11.4% of the total duck harvest. With increases in Mallard releases for commercial inland shooting and declines in wildfowling participation over time it is no surprise that estimates of the relative foreshore harvest have declined (see Section 1.3.4).

Records are also likely to represent only a small proportion of Guns. In total, 64 clubs have submitted at least two years of non-zero data between 1994/95 and 2022/23 with an annual average of 24.8 in England, 9.6 in Wales and 4.4 in Northern Ireland. The number of unique wildfowlers who visit these clubs and so contribute data at least once, varies year by year, but in the last 15 years it has peaked at 842 in 2010/11 with a low of 457 in 2020/21. In December 2012 Ellis (2014) estimated that BASC had 146 affiliated wildfowling clubs with 8,237 unique members of which 25% (2,059) went wildfowling at least once per season. So perhaps 10% of wildfowlers contribute data to the Crown Estate, and these comprise a much smaller proportion of the total number of Guns harvesting waterbirds in the UK annually.

The Crown Estate wildfowling returns only cover the foreshore (and perhaps some parcels of associated coastal land) and therefore are likely to reflect wildfowling activity but will under-represent – or more likely entirely omit – Inland Waterbird Shooting and driven wildfowl harvest. Furthermore, wildfowling is likely to target different species than inland duck and goose shooting and is commonly the subject of regulatory controls and limits (such as the framework consenting shooting on Sites of Special Scientific Interest) which may involve limits on numbers of visits or bags per species).

Despite the focus on coastal waterbirds, very small numbers of several species are reported, making robust extrapolations difficult. For 12 species, fewer than 100 birds/year are reported as harvested across the Crown Estate (Appendix 2). This is likely a reflection of low overall harvest for these species at the national level. Any estimates of national wildfowl harvest are dependent on PACEC estimates of total wildfowling, with the associated problems of this approach described below. There is no such harvest data from Scotland, as access is public (see above).

The correct identification of harvested birds is again dependent on Gun skill (see Section 2.2.1). The opportunity for misidentification in reporting is more likely in coastal or wetland wildfowling (the site of Crown Estate wildfowling return records) where greater diversity of species and conditions occur but numerically account for many fewer birds (potentially as low as less than 2% of UK duck harvest).

2.3.2. Data available

The data that might be used in assessing harvest levels may be inferred from the data collection form used for the Crown Estate wildfowling returns, available from Annex H in Wildfowling Lease Procedures 2009. It includes:

- Club/Association ID
- Gun ID
- Site ID
- Null return data
- Date
- Area visited
- Time (hours) spent on foreshore
- The numbers of each species that were shot
- Number of cartridges fired
- Any comments

2.4. BASC Wing Survey

The BASC Wing Survey – identifying age/sex of certain specimens – involves Guns voluntarily collecting and submitting either actual wings, or – recently – digital photos of wings of waterbirds that they have shot, along with other information. There is no stipulation to identify where the birds were shot so it could include waterbirds shot during wildfowling, Inland Waterbird Shooting and driven shoot scenarios. The survey is open to Guns from across the UK. The survey initially ran from 1965 to 2002. It focused largely on Teal and Wigeon and excluded waders, geese and Mallard. The BASC Wing Survey was restarted in 2017 and expanded to include all huntable geese, waders and ducks. Although participation has varied annually the survey historically received several thousand wings per year. Since restarting the survey in 2017 the number of submitted wings steadily grew, but COVID and avian influenza affected hunters' ability to harvest ducks and reduced their willingness to send wings in for analysis. Various initiatives are ongoing to increase awareness of and participation in the BASC Wing Survey and a flyway-scale survey. Limited summary data exists back to 1965, and full data is available from 1986 onwards (Appendix 3).

2.4.1. Limitations

This data are useful for understanding the demographic selectivity of wildfowl harvest, but they are less useful for estimating the harvest mortality or total bag. Participation in the BASC Wing Survey is self-selecting. Due to the physical effort of removing and storing wings, there is likely to be an underrepresentation of large harvest bags from any one location or individual. The level of detail on harvest location is limited (County only) and data on harvest effort is absent. The mean number of wings submitted annually for most species is small. For only two species, Teal and Wigeon, are a mean of more than 100 wings submitted annually (Appendix 3, Table A3.2). There is, especially recently, limited coverage outside England.

2.4.2. Data available

The data that might be used in assessing harvest levels may be inferred from the collection instructions. They include:

- An entire wing – enabling aging and sexing of at least some species
- Date of harvest
- County of harvest

2.5. PACEC/Value of Shooting Surveys

PACEC (Public and Corporate Economic Consultants) conducted surveys of shooting providers and participants in 2011/12 (PACEC 2014). These responses permitted PACEC to make estimates (relevant to this report) of the reported numbers of people engaged in game shooting of different types, the time that they spent participating in different shooting types, and the total harvest that they achieved. Further, the survey also reports estimates of the total number of gamebirds and wildfowl shot in 2012/13, being: Duck (species unspecified) – 1,000,000; Goose (species unspecified) – 110,000; Woodcock – 160,000; Snipe and other waders – 110,000. These total harvest figures were a key component in the harvest estimates by Aebischer (2019) when combined with proportions of species in the overall hunting bag derived from the NGC (see Section 3.9.1).

The PACEC survey was aimed at both participants and shooting providers and thus provides an insight into the demographics and behaviour of individual Guns as well as estates or operators who offer bird shooting in the UK. The analysis was based on responses from 16,234 in 2011/12 (PACEC 2014). It is difficult to estimate the response rate because multiple distributions were conducted from various partner organisations, and it is likely that individuals who were members of multiple organisations received multiple invitations. The survey was similar to one conducted in 2006 so patterns of change could be explored (PACEC 2006). There is a new similar survey – the Value of Shooting survey – which was published in 2024 (Cognisense 2024), but due to non-disclosure agreements, the data for the most recent survey was not available to us at the time this project was undertaken.

The survey gives insight into shooting of waterbirds under four types of shooting. These comprise coastal wildfowling (duck/goose/wader shooting on foreshore) – similar to wildfowling; Inland duck (e.g. flight ponds/marshes) and goose shooting – similar to Inland Waterbird Shooting; Walked-up, predominantly game shooting (including duck); and driven, predominantly game shooting (including duck). These last two categories are defined by the style of shooting rather than the quarry type and do not separate out the harvest of wild waterbirds from those that have been reared and released.

The key results from the 2014 survey pertinent to this report are shown in Appendix 4.

2.5.1. Limitations

Because the PACEC/Value of Shooting surveys were designed to cover shooting sports in their entirety, it is sometimes difficult to separate out material specific to waterbird shooting (as opposed to all gamebird shooting). This is especially notable when considering data relating to “Driven Game Shooting” and “Walked up shooting”.

As is common with surveys, participation was voluntary and so the participants are self-selecting and self-describing. The authors discuss response rates and make attempts to adjust weighting for this (See Appendix A in PACEC 2014). They also acknowledge that there was missing data in sufficient quantities for the authors to attempt inference of missing data (See Appendix A in PACEC 2014).

There are several indications that at least some of the data that is likely to be important in understanding waterbird harvest is unrepresentative, overestimated or at least has changed markedly in the last decade. For example, the aggregate harvest figures for released game (pheasants, partridge and duck) total 18.4 million birds shot. At a harvest efficiency of 33% (see Robertson *et al*. 2017), this suggests that around 55 million birds were released whereas Madden (2021) using 12 different approaches generated estimates of mean release numbers at 43 million. For example, the harvest figure for Woodcock was 160,000, whereas McNicol *et al*. (in press) estimate annual harvest to be 62,000–100,000. If the PACEC 2014 harvest figures are consistent overestimates of perhaps 25–35%, then extrapolations for species specific waterbird harvest may also be overestimated. The forthcoming Value of Shooting report may clarify, correct or update these figures using the same method. Alternatively, refinements using other approaches to provide confirmation may be appropriate (Dobson *et al*. 2020).

From the perspective of ongoing monitoring of waterbird harvest, two key limitations of the PACEC reports are: their timing, being conducted about 8–9 years apart, meaning that they cannot provide up-to-date information; and data accessibility, which is held by the commissioning organisations. It was made clear to participants that the survey was supported by the shooting community and that the data might be useful to produce a case to support the activity. There is a risk that if the data were used for monitoring, especially by statutory bodies, future trust in this survey format may be eroded and new data collection may be compromised.

2.5.2. Data available

The data that might be used in assessing harvest levels changes somewhat between surveys (2006, 2014, 2024), but it appears that there is consistent monitoring of:

- The proportion of participants engaged in different types of (waterbird) shooting and their levels of activity.
- Annual harvest data aggregated to “Duck”, “Goose”, “Woodcock”, “Snipe and other waders”.
- The land area over which shooting takes place.

From these responses, extrapolations can be made, accounting for unbalanced samples, to generate national or regional level values.

2.6. Guns on Pegs website

Guns on Pegs is a commercial advertising site where shoots looking to let days or attract syndicate members can advertise. This website lists several hundred shoots from around the UK, with each one providing some basic information about the quarry that they offer and their approximate location, and most also provide details expected bag size and the form of shooting practiced. It separates quarry and includes categories of “Woodcock”, “Snipe”, “Geese” (presumably shot inland with guides), “Duck” (presumably released Mallard/driven duck), and “Wildfowl” (presumably wildfowling/Inland Waterbird Shooting) but there is no clear distinction between these last three categories. This data source has been used previously to explore the release and shooting of gamebirds (Madden & Sage 2020; Madden 2023).

2.6.1. Limitations

Because this website focuses on commercial shoots, it likely under-represents the more common small and informal shoots. It provides no record of what birds are harvested, only what a typical day’s bag might be expected. It provides no breakdown of species. The current waterbird quarry classifications are ambiguous.

2.6.2. Data available

The data that might be used in assessing harvest levels may be inferred from the adverts posted at Guns on Pegs Shoot Search which includes:

- Approximate location (County level resolution)
- Broad category of quarry

2.7. Guns on Pegs Surveys

The website Guns on Pegs (see above) has conducted a survey of individual Guns (Game Shooting Census) and shooting providers (Shoot Owner Census) annually since 2013. In 2017, the Game Shooting Census included 12,143 respondents and the Shoot Owner Census included responses from 652 shoots. The focus of the census is on the economic spend, management at the shoot, patterns of behaviour and expectations of Guns and shoots, although in each year there may be questions about attitudes or opinions about specific shooting topics, some of which are related to waterbirds (see McNicol *et al*. in press). Some historic census data is publicly available (see examples in Madden 2021), but the raw data is private. A summary of changes in some of the survey results over the past decade is available in the article How has shooting changed in the last decade?

2.7.1. Limitations

The data are private, held by the participating organisations. Published data does not provide information about harvest of specific quarry. The survey is distributed by all shooting organisations in the UK, but participants are more likely to be Guns taking part in driven game shooting and may be more likely to be Guns who use Guns on Pegs to purchase shooting commercially and shoot owners who sell their shooting, thus a non-representative sample.

2.7.2. Data available

Publicly, the only useful data relate to broad trends in overall harvest levels and Gun demographics. Location data is absent or very crude. Attitudinal data including some relating to waterbirds has been collected and may be collected in the future on request, but this is not publicly accessible.

2.8. Private Game Books

Individual Guns or shoots commonly record their shooting activities for personal information or to inform future local management decisions. These records may take the form of game books that a Gun or shoot will maintain across years, or game cards that a shoot issues to Guns who have shot there for the day, with both sources including information about the date, site, bag, number of Guns and other details. Such material is private (although there have been gamebooks and collections of gamecard published, such as Debrett’s book or articles in the journal The Field, etc.). Such records have been used to understand patterns of gamebird harvest in the UK (McNicol *et al*. in press). They may cover all forms of shooting and there is no reason to expect geographical or other biases. Because they are personal records, they are likely to be reasonably accurate with no incentive to falsify records that were not intended for public viewing.

2.8.1. Limitations

The obvious limitation is that these records are private and access to them would be voluntary, introducing self-selection. Additional self-selection comes in the form of those Guns and shoots meticulous enough to maintain these records. Currently, there is no publicly available collection of these records to use in analyses.

2.8.2. Data available

Because this is not a centralised scheme or survey, the data that might be used in assessing harvest levels is going to vary according to the individual recorder. Our inspection of example private records shows that typically they record date, site and numbers of each species as a minimum. Some records include number of Guns, weather, shot count and harvest split down by sub-locations/drives.

3. Adequacy of waterbird harvest data collection in the UK

3.1. Is harvest data collected for all legally huntable AEWA Table 1 waterbird populations?

In the UK there is at least some collection of harvest data which covers all legally huntable species (see Section 2) but, as we describe, the data is partial, limited, probably biased, and is not always publicly available to researchers or statutory authorities. The harvest data we do have is at a species level only and cannot be disaggregated to the AEWA population level. For example, we have estimates of the harvest of Greylag Goose, but no estimates for the harvest of North-west Europe and Icelandic populations of Greylag Goose There is no harvest data collected for huntable species on the Isle of Man or Guernsey, and the remaining Crown Dependencies and Overseas Territories do not allow hunting of any waterbird species.

A key knowledge gap therefore is that while we have estimates, we currently do not know accurately the number harvested of any waterbird species in the UK and our current margins of error are often of an order of magnitude. To use harvest data to assess the sustainability of harvest levels we also require population estimates.

3.1.1. The problem of released Mallard

The assessment of the number of wild birds harvested is unusually complicated for Mallard. Mallard are commonly reared and released for shooting in the UK. The mixing of wild and reared birds, the almost complete inability to discriminate them in the hand or at large, their differential patterns of survival and movement, and the large numbers of reared birds involved means that the practice of releasing them is likely to distort our ability to understand the harvest levels and population dynamics of Mallard in the UK, and more widely across the flyway. More specifically, it affects our understanding of the relative proportion of the total Mallard harvest that is made up of wild versus released birds, although some estimates are possible (Ellis & Cameron 2022). There is no evidence that the practice is utilised in other UK dependencies or territories.

Annually, 2.6 million Mallard (range 0.9–6.0 million) are estimated to be released in the UK (Madden 2021) at around 17% of shoots that release gamebirds. Data from the National Gamebag Census (NGC) reveals that releases have been increasing, with a change of up to 2016 of +590% (332–1,086) since 1966, 90% (30–215) since 1991 and 34% (6–67) since 2004 (Madden & Sage 2020). Mallard are typically released at inland sites and due to low dispersal (see below) are more likely to be shot there. The most recent estimate of UK Mallard harvest was 940,000 in 2016 (Aebischer 2019) and it has been subsequently estimated that 1% of harvest took place on the coast where Mallard are unlikely to be released so the birds shot there are likely (but not definitely) wild (Ellis & Cameron 2022). Another way to estimate harvest of released birds is to assume harvest rates are similar to those of other released gamebirds. If 35% of the 2.6 million released birds are subsequently shot, as reported for pheasants in Great Britain (Robertson *et al*. 2017), then as many as 910,000 of the 940,000 Mallard bag will be released birds, as opposed to wild Mallard.

Crucially, where Mallard are released at high densities and shot in large numbers during driven shooting, it is likely that they are harvested at proportionately higher rates than wild birds at the same site. Released Mallard remain on ponds as Guns approach and take up shooting positions, whereas wild Mallard fly from the pond when disturbance begins. Consequently, pilot data indicates that at ponds where Mallard are released and shot during

Driving Released Birds, 100% of the harvest is of released, rather than wild, birds (T. Cameron pers. comm., from pilot in 2023/24 season at two sites).

Released and wild Mallard also exhibit markedly different survival and movement which further complicates modelling of their populations and response to harvest. Released farmed Mallard (in Europe and North America) survive less well than wild-born birds (Champagnon *et al*. 2016; Brakhage 1953; Dunn *et al*. 1995; Söderquist *et al*. 2021). It is unclear whether these differences in survival are the result of genetic effects, differential maternal effects or differences in early-life conditions that (fail to) prepare the birds for adult life in terms of learning foraging or predator avoidance skills from parents. Dispersal and recovery distances also differ, with only 4% of farmed-released Mallard recovered over 3 km from their Swedish release site, whereas 91% of wild-caught ringed Mallard were recovered over 3 km from their ringing sites (Söderquist *et al*. 2021; Söderquist *et al*. 2024). A study of movement ecology of released Mallard in the UK is currently underway and will be complete in 2025/26 (M. Ellis, T. Cameron, pers. comm).

This report will not attempt to discriminate wild-born from reared Mallard when considering national patterns of harvest. They mix freely in the wild, interbreed and are surely included together in population counts and harvest records. They cannot be discriminated in flight and so Guns are not able to be selective in their harvest. Very few shoots tag their released birds, so it is not possible to reliably discriminate the origins of shot birds, even in the hand. A comprehensive review of the release and harvest of Mallard in the UK, their ecological effects and differences in morphology and behaviour between released and wild Mallard is given in Holt *et al*. (2024).

3.2. Is data collected for all forms of legal harvest (other than that of taking birds under licence) of AEWA Table 1 waterbird populations?

We are not aware of other significant forms of legal harvest of waterbirds other than shooting in the United Kingdom. Netting and trapping is now illegal (see: Prohibited or unauthorised methods to capture or kill) and falconry is either licenced (so falls outside the remit of this review) or is responsible for likely negligible levels of harvest. Therefore, the datasets that we describe above concerning shooting of waterbirds represent all the available data collections pertaining to significant forms of legal harvest.

3.3. Are the right types of data on harvested birds collected?

The current (i.e. voluntary) schemes operating in the UK collect information on the numbers and/or proportions of each of species harvested, under different circumstances, as well as the age and sex of harvested species. See Section 2 for full details of what types of data are collected by each scheme or method.

Note that whilst it is desirable to collect the fullest possible data on harvest (exact numbers and species of birds harvested, age and sex ratios of the harvest, timing of the harvest through the season and measures of hunter effort), there is a limit to the utility of such data in the absence of similarly complete life history data from the full population. The current best data on harvest and populations already allows for an assessment of the sustainability of harvest with clearly defined uncertainties (Ellis & Cameron 2022; Johnson *et al*. 2024).

3.3.1. Are data on crippling and/or unretrieved harvest collected?

We are not aware of any data collection on crippling or retrieval rates from UK hunters, but the confidence intervals around the estimates of harvest are large enough that they likely already capture the additional mortality from unretrieved wounded birds. In contrast, there are a number of studies of crippling and wounding loss in the USA (see e.g. Van Dyke 1981; Norton & Thomas 1994; Ellis & Miller 2022; Ellis *et al*. 2022) or mainland Europe (see e.g. Noer *et al*. 2007; Clausen *et al*. 2017; Liljebäck *et al*. 2023).

Once shot at, a bird can be either hit or missed. If hit, the shot may be immediately lethal, the bird may recover, or it may die subsequently. In any case it is those birds which die and are not retrieved that constitute the wounding loss that needs to be accounted for as unrecorded harvest. However, it is often those birds which were hit and survived (and are carrying embedded shot) that are measured through the crippling rate. Whilst these measures are interlinked, they are not the same and the relationship between wounding loss and crippling rates likely varies with hunting scenario.

It is likely that the wounding loss rates are also highly variable between different types of waterbird hunting. For example, in an inland Driving Released Birds shoot (where most birds hit will be released Mallard), perhaps just two birds may not be retrieved by the team of pickers up and dogs that accompany the Guns during the harvest of 100 birds, representing a 2% crippling rate. In contrast, during coastal wildfowling, a single wild bird that was hit but not retrieved as part of a flight that totalled a harvest of two other birds would represent a 33% crippling rate. It may be appropriate to estimate losses from driven shoots and treat them differently from wildfowling and Inland Waterbird Shooting losses, applying them appropriately when estimating average cripple loss rates to wild birds from across current published studies.

Similarly, even though there are estimates of wounding loss and crippling rates outside of the UK, there are several key differences in hunting techniques, traditions and practices between the UK, mainland Europe and USA. For example, in the USA where daily bag limits operate, a Gun may not expend time and effort in retrieving a crippled bird at the cost of reduced hunting time when they could “fill their bag”. In the UK where coastal wildfowling shooting typically produces fewer than two birds per visit (Ellis 2014), Guns may make greater efforts to collect any potential harvest. The traditions of driven waterbird shooting are much rarer in mainland Europe and the USA than in the UK, so waterbirds may be encountered in a wider range of directions, angles and distances than in driven shooting in the UK and thus may be more likely to be only partially hit and crippled. Likewise, driven shooting in the UK is accompanied by many working dogs trained to retrieve dead and wounded game, so one would expect crippling losses to be minimised compared to losses in expansive wetlands with sole hunters and small teams with fewer dogs. These differences mean one should be cautious about basing estimates of mortality on crippling or retrieval rates in the UK on European or US data.

Assessing crippling rates is not easy. Birds caught for ringing purposes may be scanned for shot (e.g. Noer *et al*. 2007; Liljebäck *et al*. 2023) or shot birds can be examined for old shot wounds but this cannot account for birds that are crippled and die unretrieved thus not contributing to harvest bags. Similarly, measuring the prevalence of embedded shot in live birds only measures the proportion that have survived, with the proportion that died unknown. Guns may be asked to report perceived crippling rates or birds known to have been downed but not retrieved (Ellis & Miller 2022), but this is susceptible to biases including deliberate under-reporting or a failure to observe non-fatal pellet strikes on flying birds. Trained, independent observers might accompany Guns and record non-fatal pellet-strikes (e.g. Anderson & Sanderson 1979) but again, unless the fate of individual birds is followed it is not possible to determine whether the struck bird goes on to die soon afterwards unretrieved or if it survives and continues to contribute to population dynamics.

Despite these problems with the currently available data on crippling, the values that we do have would provide preliminary suitable estimates to build into more precautionary estimates of estimates of hunting mortality.

3.3.2. Is data on age/sex of harvested birds collected?

The BASC Wing Survey provides both age and sex specific information, and such surveys have been shown to be able to provide useable information on the demographic selectivity of harvest in the past (Fox *et al*. 2016; Christensen *et al*. 2017b). The information the BASC Wing Survey provides is most useful when compared to other hunting independent information on age and sex structure of hunted waterbirds, for example from ringing data. By comparing the Wing Survey and ringing data we could assess if harvest rates are a function of quarry availability, or whether there is selectivity in the hunting pressure.

Whilst age (adult or juvenile) data is collected for all species, the collection of sex data is limited to sexually dimorphic species and therefore is not available for Greylag Goose, Pink-footed Goose, White-Fronted Goose, Golden Plover, Woodcock, Common Snipe, Coot or Moorhen. Another limitation is that age and sex ratios from harvested birds are known not to be representative of the wider population, though they can be adjusted to be so (Holopainen *et al*. 2018). This is more problematic because models of the sustainability of harvest can accommodate missing sex data with relatively little loss of output accuracy and precision, but age data is more important because they are stage-based (see Section 2.5.4 for more detail).

3.4. What proportion of hunters submit data on harvest, and is this reliably known?

The number of Guns shooting waterbirds in the different scenarios is poorly known. PACEC (2014) estimates that 280,000 Guns shoot driven game including duck, 150,000 shoot walked up game including duck, 75,000 shoot inland duck and geese (Inland Waterbird Shooting), and 28,000 engage in coastal wildfowling. These estimates are not exclusive, so one Gun may engage in all four types of shooting. This will make an estimation of the proportion of Guns submitting data difficult to assess, even if we have accurate data on individual submissions to the current data schemes.

An estimated 22–41% of members of wildfowling clubs who hunted at least once per year submitted returns to the Crown Estate wildfowling returns. However, this accounts for just a few hundred Guns. Therefore, accurate returns represent less than 1% of all potential harvest effort according to minimum numbers of Guns estimated in PACEC (2014). They also likely only represent Guns and their harvest from the wildfowling scenario. No margins of error are presented in PACEC (2014), so it is not possible to determine the reliability of this proportion.

Individual Guns make submissions to the BASC Wing Survey. Since 2020, fewer than 1,000 wings have been submitted annually, suggesting this may represent a maximum number of participating Guns. Again, this represents a very small proportion of estimated waterbird Guns in the UK and as above, the reliability of a proportion estimate based on these numbers is low.

Harvest data may also be obtained at the shoot, rather than individual Gun level. The NGC typically receives annual harvest data from several hundred shoots, which comprise at most 5–10% of shoots in the UK. This data would likely reflect driven harvests and perhaps to a lesser extent Inland Waterbird Shooting. Without an accurate estimate of the total number of shoots or shooting venues or providers in the UK, it is not possible to determine the reliability of the proportion of shoots that contribute to the NGC.

3.5. Are there any gaps in representation of Guns who submit harvest data?

Because the actual numbers of Guns engaged in different scenarios is unknown, it is difficult to accurately assess the representation of submitted harvest data. It is likely that there is reasonable quality and representation of data from those engaged in wildfowling via the Crown Estate wildfowling returns and BASC Wing Survey datasets. There is the potential for reasonable quality and representation of data for those engaged in Driving Wild/Released Birds from those shoots submitting data to the NGC, but this is not publicly available. Data on Inland Waterbird Shooting is likely to be poorly represented.

Further, the data that is currently available is self-selecting, so it is unknown how representative it is of the activity, harvest, attitudes or competence of other Guns. Non-responses to surveys of hunters distorts understanding of harvest levels and hence models of population trends (Aubry & Guilmain 2019). We are not aware of any surveys of activity, attitudes or competence relevant to waterbird shooting by Guns in the UK from which representativeness could be calculated. Such work could be initiated and supported by membership shooting organisations (e.g. BASC).

3.6. Are there any difficulties in ensuring the continued participation of hunters in existing recording schemes?

Participation in voluntary recording schemes by Guns and shooting estates shows signs of decline. From 1986–2000, the mean number of wings submitted annually to the BASC Wing Survey was 2,766, but from 2000 to present it was 849, a three-fold decline (Appendix 3). More generally, there has been a decline in participation in voluntary game record schemes in the UK since the 1960s. The number of estates contributing bag data to the NGC scheme fell from around 700 in 1961 to around 450 in 1988 (Tapper 1992). The number of estates contributing bag data to the GWCT Partridge Count Scheme fell from just under 600 in 1961 to around 100 in 2002 (Aebischer & Baines 2008), although this may be linked to the decline in population and range of the grey partridge over that period. The causes of these declines in participation are unknown.

In the UK, Guns, and other members of the shooting community, have traditionally been hard to reach when requests for information or collaboration with data collection have been made. It is notable that studies of game managers and Guns conducted by academics and non-shooting organisations tend to be limited to a few tens to hundreds of respondents (e.g. Cox *et al*. 1996: n = 274; Oldfield *et al*. 2003: n = 65; Howard & Carroll 2001: n = 38; Swan *et al*. 2020: n = 20; Newth *et al*. 2019: n = 30). In contrast, surveys by organisations involved in shooting tend to engage far higher levels of participation (e.g. Piddington 1981: n = 626; Guns on Pegs Game Shooting Census 2017: n = 12,143; Guns on Pegs Shoot Owner Census 2017: n = 652; PACEC 2006: n = 2,096; PACEC 2014: n = 16,234).

This may be due to poor communication between Guns and researchers. Game shooting is described by Hillyard (2007) as a ‘total institution’: that is, it has its own membership, rules, traditions and language. This can make it a relatively closed world to researchers, difficult to access and because of the technicalities of the operations and the language used to described them, difficult to either collect data from respondents or even construct intelligible questions to ask of them. Researchers would generally need to overcome these barriers in order to engage with game managers, Guns and other shoot participants in order to access study sites where releasing and shooting occurs.

Additionally, or alternatively, Guns may be reluctant to participate because of suspicions that any data that they provide would be used to restrict their sport. There is a pervasive belief among the shooting community that whenever data is made publicly available it is “used against them”, such as the removal of the ability to hunt a species. The growing polarity of the debate about the legitimacy of game shooting and releases, exacerbated by social media and campaigning organisations and celebrities, is likely to make game managers and Guns more circumspect about engaging with researchers who are unknown to them and/or with unknown motivations. Positive engagement between statutory agencies and the hunting community will help to build trust and hence improve harvest data quantity and quality.

3.7. What is the geographic coverage of data collection, and is this sufficient to give an accurate picture of the national scale of harvest?

Currently, the Crown Estate wildfowling returns provide no data from Scotland. The BASC Wing Survey provides little data outside England. The NGC has little data from Wales and Northern Ireland. Consequently, and in conjunction with the limitations described in Section 2, this skewed coverage of harvests is likely to be insufficient to provide a representative and accurate national harvest assessment. None of the Crown Dependencies or UK Overseas Territories currently operate a formal collection of harvest data, so these areas are not covered.

3.8. Is the frequency of data collection sufficient?

Most of the harvest data collection schemes described in this report collect data on a regular enough basis to allow triennial estimates of harvest for AEWA reporting purposes, but regular updates of the total harvest estimates (e.g. PACEC/Value of Shooting) are required, or a new method developed. It should also be noted that to replicate the Aebischer (2019) approach to estimating waterbird harvest, access to NGC data needs to be negotiated with the GWCT.

The Crown Estate wildfowling returns, BASC Wing Survey, NGC and Guns on Pegs Shoot Owner Census and Game Shooting Census are conducted annually. The PACEC/Value of Shooting report is conducted sporadically (2006; 2014; 2023). The Crown Estate wildfowling returns, BASC Wing Survey and NGC provide proportional/relative data for species harvest, whereas the PACEC/Value of Shooting reports provide estimates of total harvest. Therefore, an estimate of absolute harvest for each species gained using this method, or a variant based on current data sets that indicate the species composition of proportion of harvest, is highly dependent on PACEC/Value of Shooting surveys, which to date have been revised every 8–9 years. This situation could be improved with a truly representative national survey of waterbird harvest. However, it should be noted that this additional effort may not provide a harvest estimate that would be as, or more robust than our current population estimates, which would also be required to assess the sustainability of harvest.

3.9. Can we use the existing data schemes to reliably estimate UK waterbird harvest?

All the current datasets reporting harvest levels that we considered have multiple serious limitations and biases. Each differs from the others to the extent that integrating them will be difficult. For example, each typically considers only a single form of waterbird hunting, which restricts its geographic, taxonomic and societal coverage. This is problematic because the various AEWA waterbird species occupy different habitats and are shot by different means. For example, we believe that 90+% duck mortality is not occurring on the coast (where the more detailed BASC Wing Survey and Crown Estate wildfowling returns are focussed). Each is based on self-selected participation, which introduces a variety of biases. Furthermore, all records apart from those in the BASC Wing Survey are susceptible to misidentification of the quarry by the Guns.

The derivation of harvest estimates from mandatory and voluntary data, where response rates and sample selection may be unknown or skewed, and where there are opportunities for response errors (forgetting data, misclassifying quarry, deliberate misreporting of harvest etc) can result in inaccurate and imprecise estimates. These errors may become compounded when multiple datasets are combined to derive single estimates. However, robust bag estimates can be generated using traditional survey and statistical methods, and a thorough explanation of different approaches is given in Aubry *et al*. (2020), with a visual summary of how different data collection schemes might be subject to particular errors shown in their figure 4 of that paper.

3.9.1. How have waterbird harvests been estimated previously, and were the results reliable?

Aebischer (2019) combined the NGC and PACEC/Value of Shooting datasets to derive estimates of national bag sizes. He achieved this by splitting up the aggregate bags reported by Guns and shooting providers in two large national surveys (PACEC 2006, 2014) by species, in relation to their proportions in the NGC. These species-level values for the 2004 and 2012 seasons were then used to calibrate the changes in indices for each species obtained from the NGC and so calculate total nationwide bag estimates for the 2016 season.

The methods used by Aebischer (2019) are statistically robust, but we have no way to assess their accuracy. Aebischer's methods are reliant on both the national estimates of total waterbird harvests (produced by PACEC 2014) and the estimates of the proportion of the bag comprising each species (produced by the NGC) being accurate and representative of shooting in the UK. Given the self-selected nature of both components of the estimation framework (see Section 2.2 and Section 2.5) there is likely to be a significant component of sampling error, but the extent and effect of this cannot be assessed with the currently available information. It is likely that a greater understanding of the uncertainties in the NGC data could be modelled using Bayesian approaches (Lindstrom & Bergqvist 2020), but this is outside of the scope of the current study.

We would suggest that the confidence intervals presented in Aebischer (2019) likely capture the "true" harvest of most of the more commonly shot species, but that the central estimates for these species are likely an overestimate, especially for species such as Teal and Wigeon. This is due to a combination of an overestimate of the total aggregated harvest by PACEC and a likely disproportionate representation of driven game shoots and commercial duck shooting in the NGC sample. For species such as the geese, Pintail, Pochard and Goldeneye, the estimates are likely underestimates due to the NGC underrepresenting coastal wildfowlers, recreational hunters using personal flight ponds, inland goose shooters and commercial goose guides. However, this may be offset somewhat by the likely overestimate of total waterbird harvest by PACEC.

4. Assessing the sustainability of waterbird harvest in the UK

4.1. Is sufficient data available to allow assessment of the sustainability of UK waterbird harvest?

Robust harvest estimates can be generated using traditional survey and statistical methods. However, this demands that the data is representative and unbiased. This is possible (see Aubry *et al*. 2020), but the currently available survey data has not met these requirements or been interpreted whilst rigorously accounting for these biases. Crucially, the confidence of harvest estimates is increased by having a representative sample of hunters in the survey population, which requires good starting knowledge of wider population – this knowledge for the UK hunting community does not currently exist in either the private or state sector. Finally, complete knowledge of harvest is only of maximum utility when accompanied by similarly complete knowledge of waterbird populations. Whilst any increase in knowledge will be useful there is a need to balance the investment in maximising both population and harvest data accuracy, rather than focusing on one in isolation.

The lack of "perfect" information on harvest and life history variables is not a barrier to starting models to assess the sustainability of harvest for waterbirds – as long as uncertainty is taken into account. The currently collected data on harvest, combined with population estimates, are sufficient to allow for an initial assessment of the sustainability of the harvest. For most waterbird species, the scale of harvest is orders of magnitude smaller than the population and so this level of uncertainty is largely irrelevant. For those species where the harvest represents a greater proportion of the population (such as Mallard, Teal and Wigeon), gathering more accurate and precise data on harvest and Gun behaviour would reduce the margin of uncertainty around assessments of sustainability but the margins would still be large due to the wide confidence intervals around population estimates. Such improved data would allow for more accurate assessments of sustainability with greater confidence.

The status and limitations of the data currently available for assessing the sustainability of waterbird harvest in the UK are described below. For all these parameters, more important than estimates of the average parameters is an estimate of the variation or range that these parameters could take.

4.1.1. Data on harvest sizes and their accuracy

None of the methods for estimating harvest in the UK (see Section 2) are likely to represent a full and accurate picture of the diversity of harvest in the UK. For a review of the reliability of the methods used to estimate national hunting bags by Aebischer (2019), refer to Section 3.9.1.

We are not aware of any published data to assess the reliability of the estimates of Aebischer 2019, but they are supplied with large confidence intervals which likely include the true harvest estimates. The authors are examining other ways to test the reliability of the estimates of the species-specific breakdown of the UK national bag using novel self-disclosure datasets and anonymous reporting.

It is important to recognise that the harvest data estimated by Aebischer (2019) represented a snapshot of the sustainability of waterbird harvest based on data from 2016. In reality, waterbird populations will change, both because of factors relating to harvest and other ecological factors, and because of changes in the behaviour of Guns as they adjust their harvest to opportunity, changed legislation or self-regulation.

Interpreting data on the harvest of Mallard is complicated by the inclusion of released birds that cannot be distinguished from wild birds. It is unclear how corrections might be applied because we know little about how wild and released Mallard interact (Section 3.1.1) or are encountered in different shooting scenarios (Section 1.3). Released birds could be marked and separated from unmarked, wild, birds in harvest returns. How practical, or ethical, it would be to mark several million Mallard each year is uncertain.

Whilst there is data available on wounding losses (birds hit, but unretrieved and which subsequently die) in the USA and Denmark (Ellis & Miller 2022; Clausen *et al*. 2017; Holm & Haugaard 2012), there is no such data available for the UK (see Section 3.3.1). It is likely that the wounding loss rate of wild birds in the UK is much lower than in the USA due to cultural and practical differences. Given the inaccuracy in both the population and harvest estimates for many waterbirds, an additional harvest of up to 10% of the estimated harvest rate is unlikely to make a substantial difference to assessments of sustainability for most species, but it is certainly a knowledge gap to be addressed in the future.

4.1.2. Population size data and its accuracy

Our current estimates of waterbird population sizes in the UK are poor, with large confidence intervals. They are also conducted relatively infrequently, making triennial reporting to AEWA difficult. Only half (7/14) of the current assessments of the waterbird species populations in the UK are based on actual count data (AEWA Conservation Status Report 8) which would enable a large portion of the population to be likely to be detected, while the other half rely on range of estimates and assumptions. Population estimates in other territories or dependencies are likely to be even less robust, lacking the established British Trust for Ornithology (BTO) surveys (Nagy *et al*. 2020) with, for example, only 19% of population estimates from AEWA Conservation Status Report 8 based on censuses, and 50% based solely on expert opinion and only 20% of population trend quality being assessed as “good”. Our simulations (Appendix 5, Table A5.3) of how population estimates might vary depending on the assumptions about encounter and detectability of different species produces errors of up to 50% of the mean population size. Furthermore, the lack of coordinated assessments of migratory waterbird populations across their flyways results in additional uncertainty about populations in the UK, thus complicating models of harvest sustainability.

Semi-regular population assessments are undertaken by the Avian Population Estimates Panel (APEP), and although the latest waterbird estimates are reported from 2020 (APEP 4; Woodward *et al.* 2020) the data represents a five-year mean peak over the period 2012/13–2016/17. For large, aggregating species occurring in open habitats, for example geese, the population estimates are likely to approach a complete census. However, there is evidence that for Icelandic Greylag Goose these counts may be substantial underestimates (Frederiksen *et al*. 2004; F. Johnson pers. comm.). For species that are more cryptic in nature, less aggregated, or more patchily distributed (including Teal and Mallard) the population estimates are likely to be inaccurate, and in many cases probably represent an undercount (Nagy *et al* 2022).

Information on the accuracy of population estimates is a key component of assessing the sustainability of harvest. Population estimates for the UK, including the latest estimates (APEP4), report the reliability of population estimates, but we are not aware of any attempt to quantify the impact of reliability on confidence. For most harvested duck species wintering in the UK, they are estimated as having a low reliability score (e.g. Mallard and Teal), whereas goose estimates are reported to approach a census.

The population estimate of Mallard in the UK is distorted by the inclusion of released birds that cannot be differentiated in the field from wild birds. Marking of released birds may improve counts, although leg bands are not visible underwater. Alternatively, targeted studies of the dispersal of birds from release sites may improve confidence that birds surveyed at sites of different distances from shooting release points are, or are not, likely to have been released.

Waterbird populations are, of course, susceptible to mortality via factors other than hunting, such as habitat changes, weather, fluctuations in predator populations, etc. For some of these species in the UK we have few reliable or up to date specifically UK based measures of how these factors affect survival or population size. Furthermore, because many of these species are migratory, their populations are shaped by events and conditions (including those factors that pertain to the UK listed above) operating outside the UK about which we may have little information.

4.1.3. Age- and sex-specific data

To model waterbird populations, and to best model their harvest, certain parameters are required and ideally, they would be at least age specific. Relevant age specific information in birds includes data provided from ringing and field-based studies where birds are classed as either juveniles or adults. These data are collated and managed by the BTO.

Age-specific parameters are more important than sex specific information as models can be adapted to make use of single sex data with little loss of accuracy, but the equivalent is not true for age data, since demographic models are effectively stage-based. Many models rely on accurate estimates for the intrinsic maximum rate of population increase (R_{max}) under idealised circumstances, such as a lack of density dependence or additional sources of mortality (including hunting). Such data are rarely available but can be estimated in a variety of ways or derived from recorded survival rates. Using the data sources described below and other European studies using ringing data, we can obtain suitable estimates for juvenile and adult survival to allow modelling of waterbird populations (e.g. matrix or discrete models, see Section 5.2.2).

There are very few current or geographically relevant estimates of life history characteristics for the wider populations of waterbirds, with most estimates more than 50 years old, or from outside the UK. Whilst the BTO collates survival data as part of its ringing scheme, much of this is old, may not be nationally representative and, certainly for hunted species, implicitly includes the effect of hunting mortality on survival. Ringing effort of waterbirds in the UK has been in decline for some time, but it can provide contemporary estimates of age and sex specific survival for several hunted waterbirds; some of this work is in progress (T. Cameron pers. comm.). Rarer species are relatively poorly represented in the ringing data and so their life history data remains less robust. We are not aware of any recent analyses of UK waterbird data with the express intention of deriving robust life-history measures which may be used in modelling harvest sustainability.

Sex specific, and to a lesser extent age specific assessments of live flocks of birds can be and are done but these are not common. They tend to be done as part of a particular research call (i.e. as was done for Pochard in the UK ((Brides *et al*. 2017)), and it has been done for other waterbird species to examine juvenile to adult ratios in the field (Holopainen *et al*. 2018). More regular field assessments of juvenile to adult ratio tend to be estimates for geese and swans which are part of the regular Goose and Swan Monitoring Programme censuses hosted by the BTO (and previously by Wildfowl and Wetland Trust).

Although models (such as popharvest – see Section 5) may allow for the estimation of idealised survival from species body mass based on a fitted relationship across many avian taxa, this does not account for any real-life ecological conditions and so is likely to over-estimate R_{max} in real-world conditions. In our experience, estimates of R_{max} derived from body mass are significantly lower than those derived from recorded survival.

Underestimating survival will potentially lead to overestimating the sustainability of harvest levels. Therefore, efforts should be made to improve and expand Life History data relating to age-specific parameters, most simply and obviously by a re-evaluation of the BTO ringing records.

Age and sex-specific life history data for Mallard is complicated because the movement and survival of released birds differ from that of wild conspecifics (Section 3.1.1), yet the two groups cannot be differentiated in the field meaning that the survival rates or dispersal distances of released mallard might skew the parameters used in models intended to reflect wild populations. A careful partitioning out of data may be possible, or targeted studies of known (i.e. marked) wild or released Mallard in the UK may provide more accurate data.

4.1.4. Data on breeding productivity

Breeding productivity is another important parameter required for modelling of waterbird populations. Productivity estimates for most of these species are poorly understood and require investment in field studies in their breeding range in northern and eastern Europe and Russia. Currently, models rely on proxies such as body mass or a "living rate" or alternatively require borrowing life history information for the same or similar species in other continents (i.e. North America).

For some species, their productivity within the UK may be artificially increased by habitat management efforts by Guns trying to improve their sport (e.g. predator control, provision of nesting tubes etc; for example, Balser *et al*. 1968), meaning that population dynamics of these species under different scenarios where shooting levels, and thus associated management, are increased, decreased or cease, are hard to predict.

As discussed above, we may gain some information that could be used for productivity estimates from field-based juvenile to adult ratios obtained from observations or BTO ringing records. The reason this is particularly useful for larger geese and swans is that their juveniles tend to stay in a family unit well into the autumn migration so the number of "fledged" juveniles per adult female can be counted. This information is not always possible to obtain for ducks and waders, but flock-based assessments of those taxa can be made early in the autumn migration and staging areas (Brides *et al*. 2017; Holopainen *et al*. 2018), or from larger field-based catches of waterbirds so that birds can be aged in the hand. Very few large field-based catches of waterbirds occur in the UK or in northern Europe. Either way, both of these approaches require investment and must assume that the juvenile to adult ratio has not changed between breeding areas and the UK, which is unlikely to be true, especially for hunted species where juveniles are known to be more susceptible to harvest, leading to change in the proportion of juveniles down the flyway (Guillemain *et al*. 2010; Guillemain *et al*. 2013). Breeding data might also be available from the BTO nest record scheme that records the progress of individual nests. An up-to-date analysis of UK waterbird nest records may improve breeding parameter values in models of harvest sustainability. Obviously, this is only possible for the species and populations that breed in the UK.

Another approach for obtaining breeding productivity data is cohort or site-based studies of waterbirds in breeding areas. The key data is number of fledged young per female, such that incorporating females without young allows a more accurate assessment of average breeding productivity (i.e. many females do not breed or fail to breed in a given year). While the number of fledged ducklings per female is the key metric required to model productivity, this could be assessed from data on nesting propensity (likelihood that a female initiates a clutch), nest success (the proportion of nests that successfully hatch), the initial brood size of hatched clutches and duckling survival to fledging. Obviously, this requires research investment. Several such studies of breeding ecology in northern countries already exist for wintering waterbirds, some of which winter in the UK (e.g. Mustonen & Kontanen 2019;

Pöysä *et al*. 2018), and for their close taxonomic relatives in North America. "Borrowing" some of these parameters from studies in North America could be suitable, for example nesting propensity and clutch size, as these are likely to be driven by fundamentals of biology and will be similar between Eurasian Teal and the (North American) Green-Winged Teal for example. However, due to differences in habitat and land-uses, predator communities and predator control, nest success and duckling survival are likely to be very different between North America and Europe.

As for age and sex-specific life history data (see above), wild and released Mallard differ in their breeding productivity (Section 3.1.1), making models of harvest sustainability for this species more difficult to interpret or rely on without an update to the knowledge in this area. New research by the authors that began in 2023/24 should improve our knowledge in this area.

4.1.5. Data to allow projection of the sustainability of waterbird harvest under future conditions

While models can provide indications of the sustainability of harvest in current conditions, changes in climate, environment and human behaviour are likely to affect outcomes in future scenarios. At present, there is poor or no data about how we might expect waterbird populations to change in the future. Shifting flyways (Nagy *et al*. 2022), changing land use (Van Eerden *et al*. 2005) and altered breeding seasons (Both & te Marvelde 2007) all change migration, breeding and land use behaviour which will likely affect future waterbird populations overwintering in the UK, and in addition, such climate change also makes it harder to assess population size (Fox *et al*. 2019). Data, relevant and specific to hunted UK waterbirds, on how populations might respond to environmental change is needed to improve models of harvest sustainability. This might be obtained by a careful review of existing literature, and/or additional targeted data might need to be collected.

Gun behaviour may also change in future years, either due to self-restraint (McNicol *et al*. 2024) or targeted action plans (Noer *et al*. 2007). We are aware of almost no data from the UK on the quarry selection, hunting behaviour and attitudes of Guns and landowners that may allow an understanding of how Guns might adjust their behaviour under different future scenarios. The lack of this data makes it difficult to model harvest sustainability under future scenarios that differ from the current conditions.

4.2. How has the sustainability of UK waterbird harvest been assessed previously, and were the results reliable?

An initial assessment by Ellis and Cameron used a novel tool (popharvest – Eraud *et al*. 2021) to rapidly assess the sustainability of waterbird harvest in the UK. The popharvest tool can work with limited data (as is the case in the UK), but the more complete the data, the more accurate the assessment. The main limitations to the data in the UK are described in Section 4.1.

The conclusions of Ellis and Cameron (2022) are reliable within the limitations that they were based on, using the best available data at the time and considering the assumptions they made. The uncertainties around population sizes, harvest levels and life history characteristics means that the uncertainty in assessments of sustainability increases as harvest (as a proportion of the population) increases. For most species, the level of harvest is small compared to the population and so even in the presence of large uncertainties we can be confident that harvest is likely to be sustainable.

For species where harvest represents a greater proportion of the population (including Mallard, Wigeon, Teal and Greylag Goose), the uncertainty around both population and harvest means we can be less certain that harvest is sustainable, but we can estimate the degree of uncertainty. For this reason, we urge caution in interpreting the findings of the model for species apparently identified as being subject to unsustainable harvest. We suggest that these cases highlight those species where there is the greatest need for more information to refine estimates of population, harvest and life history characteristics, and that an adaptive harvest approach would be a precautionary way to achieve this.

Using the best available evidence, our models suggest that harvest of Mallard, Teal, Greylag Goose, Gadwall and Woodcock could be unsustainable, yet population trends are increasing or stable for some of these species (Teal, Gadwall, migratory Woodcock) and known, or thought, to be underestimates for others (Greylag Goose, Mallard, Teal). This does not invalidate the models, nor undermine the importance of existing data collection, but is simply a function of compounding the uncertainty from three separate data sources. On the other hand, for species where we have calculated a very low Sustainable Harvest Index (SHI), we can be confident that harvest is unlikely to be unsustainable. Under these scenarios the highest realistic estimates of harvest do not exceed sustainable levels for even the lowest modelled population estimates meaning that these estimates would have to be out by several orders of magnitude for harvest to be unsustainable.

The initial assessment of Ellis and Cameron (2022) is subject to several assumptions which are outlined in Table 3, but by far the biggest limitation was that it represented a snapshot of the sustainability of waterbird harvest based on 2016. In reality, waterbird populations will change, both because of factors relating to harvest and other ecological factors, and because of changes in the behaviour of Guns as they adjust their harvest to opportunity, changed legislation or self-regulation. Therefore, future assessments should allow for consideration of a range of population and harvest scenarios.

Table 3. Assumptions made by Ellis and Cameron (2022), their reliability and the impact that the assumption is likely to have on the resulting estimates of harvest sustainability.

Assumption	Reliability of assumption	Impact
Population estimates are accurate	Low	Overestimates would bias estimates of sustainability high
Harvest estimates are accurate	Very low	Overestimates would bias estimates of sustainability low
Life history data (survival, age at first breeding and living rate) are accurate	Low/Medium	See Ellis and Cameron (2022) for worked examples
There were no additional wounding losses (or that they were at least captured within the confidence intervals provided by Aebischer's harvest estimates)	Medium	Additional, unaccounted mortality from wounding would result in artificially reduced estimates of mortality and so bias estimates of sustainability high

Assumption	Reliability of assumption	Impact
Derivation of source of migratory vs resident populations of wild birds that are harvested is accurate (currently based on population estimates)	Very low	See impacts for inaccurate estimation of harvest or population (but note this is irrelevant for those populations comprised only of resident birds)

5. Modelling approaches for assessing the sustainability of waterbird harvest

5.1. Potential Excess Growth (PEG) models

The initial assessment by Ellis and Cameron used a novel tool (popharvest – Eraud *et al*. 2021) to rapidly assess the sustainability of waterbird harvest in the UK by applying Potential Excess Growth (PEG) models to estimate a Sustainable Harvest Index (SHI). The PEG is the number of individuals which may be removed from a population under idealised circumstances without causing a population decline. The SHI is a simple ratio of the harvest to PEG, such that an SHI of one means that 100% of the PEG is harvested and the population would be predicted to remain static. An SHI of 0.5 means 50% of the PEG is harvested and an SHI of two means 200% of the PEG is harvested and the population would therefore likely decline. The popharvest tool can work with limited data (as is the case in the UK), but the more complete the data, the more accurate the assessment. The main limitations to the data in the UK are described in Section 4.

5.1.1. Choice of appropriate safety factors

The popharvest PEG model includes the ability to specify a safety factor (F_s) that reflects the risk that policy makers and/or legislators may be comfortable taking, and which can range from zero to one but usually does not exceed 0.5. For any given population, the model estimates the number of individuals required for the population to remain stable accounting for survival and reproductive rates in ideal conditions. Any individuals produced above those required to maintain a stable population form the PEG and may be harvested without reducing the population. The safety factor (F_s) is a simple multiplicative adjustment to reduce the “allowable” take from the PEG. For example, an F_s of one allows for the harvest of 100% of the PEG and an F_s of 0.1 allows for a harvest of 10% of the PEG. A high F_s permits harvest, but risks population declines if unexpected changes to the environment or population occur (e.g. outbreaks of HPAI or unusually poor conditions), whereas a low F_s restricts harvest but is more robust to unexpected population or environmental changes. The SHI is the harvest divided by the PEG adjusted by the F_s, such that with an F_s of 0.1 an SHI of 0.5 means that the harvest is 50% of 0.1*PEG (e.g. 5% of the PEG).

The choice of F_s is not a straight-forward scientific question but one dependant on factors such as the desired overall population size, the “desirability” of the species to society (both positively as one enjoyed by, for example, birdwatchers or because it provides an ecological service, and negatively if it is perceived as a pest, perhaps causing agricultural damage) or its conservation status, for example based on the species’ global IUCN Red List category. Ellis and Cameron (2022) chose their safety factors (F_s) based on species’ global IUCN Red List categories as suggested by the popharvest authors (Eraud *et al*. 2021). Whilst this simplified presentation for the purposes of the paper, it also muddied the water between science and policy and is a limitation of their paper (which will be addressed later in this report). A more in-depth consideration of an appropriate safety factor would require consultation with stakeholders and regulators on the level of risk that society is willing to accept for each species. We suspect basing decisions on IUCN Red List status is not always appropriate. In later sections (e.g. Section 5.2.1) we present the SHI across all Fs so that regulators can choose their own risk level. We conclude that there cannot be a “one-size-fits-all solution” that will work in all cases.

For species listed on Table 1 of the AEWA Action Plan, target population sizes may be best defined with Favourable Reference Values (FRVs), or Favourable Reference Ranges (FRRs). This approach is currently being used by AEWA’s European Goose Management Platform for some goose populations (Defining favourable reference values for the populations of the barnacle goose). In this situation populations above their FRR could have large F_s (of 0.6–1) that would allow for the take of a substantial proportion of the PEG and so would likely drive population decline. Populations within their FRR could have an intermediate F_s (F_s = 0.3–0.5) depending on population stability or confidence in population and harvest estimates. Finally, those populations below the FRR for which harvest is still desirable could have low F_s (F_s = 0.1–0.3). However, a more robust approach may be to use the Prescribed Take Level (PTL) model within popharvest, which we discuss below (see Section 5.2.1).

The effect on SHI of adjustments to F_s can be calculated simply without the need to rerun analysis. For example, a change from F_s = 0.5 to F_s = 0.1 would lead to an SHI five times larger such that under the same conditions $SHI_{Fs0.5}$ = 0.1 is equivalent to $SHI_{Fs0.1}$ = 0.5. A worked example of the relationship between SHI and F_s is given in Appendix 4.

5.2. Alternative modelling approaches for assessing the sustainability of waterbird harvest

Here, we compare two alternatives to the PEG model used by Ellis and Cameron to assess the sustainability of waterbird harvest: the Prescribed Take Level (PTL) option in popharvest; and matrix-based population viability analysis.

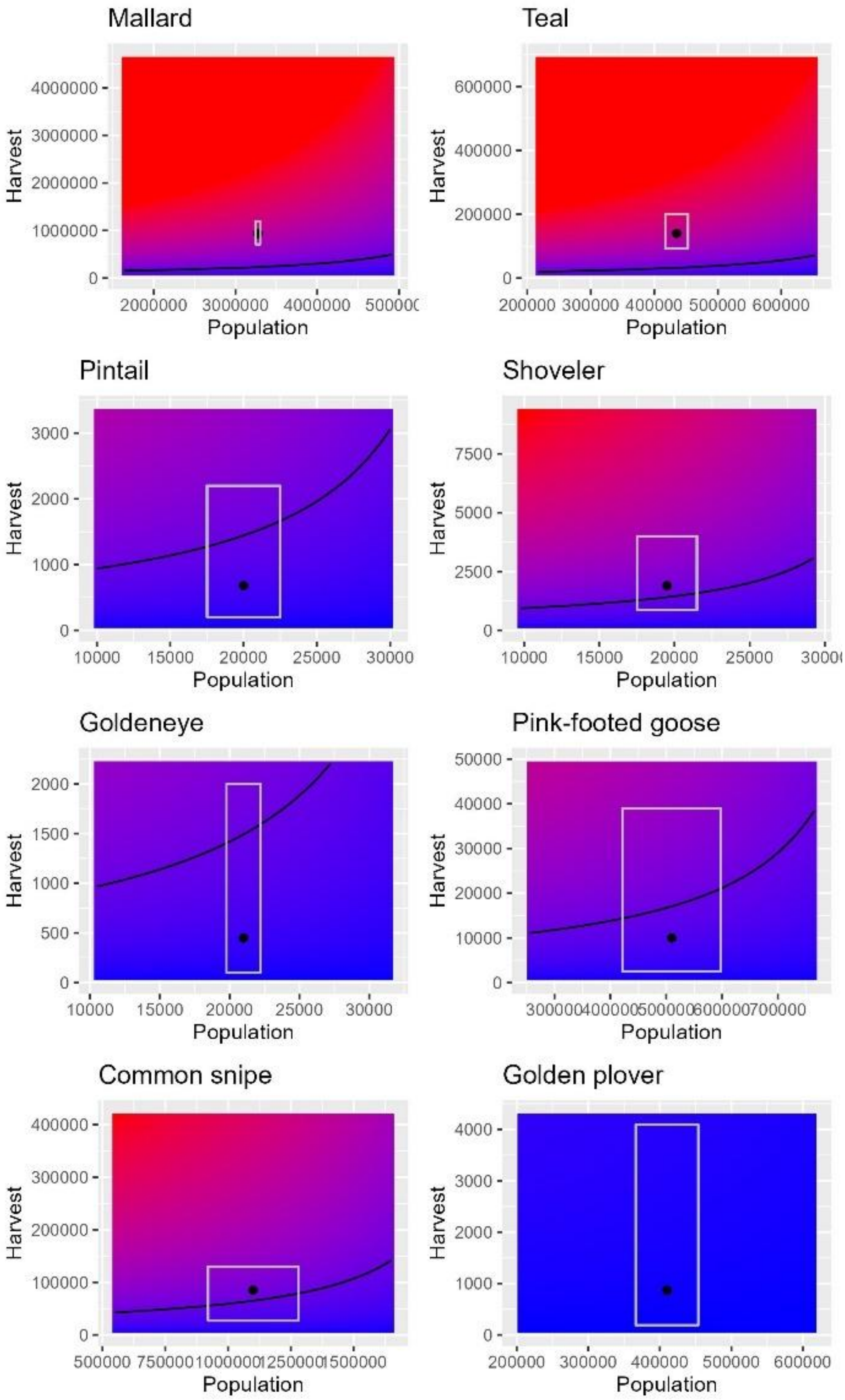
5.2.1. Use of Prescribed Take Level models in popharvest

Ellis and Cameron (2022) used the PEG model in popharvest to assess the sustainability of harvest in the UK. However, recent literature suggests that the PTL option in the same modelling package is a more robust approach (Johnson *et al*. 2024). Given the developments in the literature since Ellis and Cameon (2022), and the clearer distinction between social and biological risk, we recommend that future assessments of sustainability use PTL models.

PTL models use many of the same underlying calculations and assumptions as PEG models but allow for much clearer consideration of ecological assumptions and social appetite for risk through the setting of a management objective (F_{obj}). In the PTL models, an F_{obj} of one represents a management objective to allow harvest of 100% of the maximum sustainable yield. In the absence of other ecological pressures this is an ecologically sustainable strategy which would hold the population at an equilibrium below its carrying capacity but, in reality, is unlikely to be sustainable due to annual fluctuations in the realised survival and productivity (Ludwig 2001). For levels of F_{obj} close to zero only a small proportion of the maximum sustainable yield may be harvested, and under such situations the population would be expected to be brought into equilibrium at a level closer to ecological carrying capacity.

As with the PEG model used in Ellis and Cameron (2022) the principal outputs of the PTL models are a probability that the harvest is unsustainable (specifically, the proportion of simulations which resulted in an SHI greater than one), and the SHI) itself. An SHI of less than one indicates that the harvest is lower than the management objective, and an SHI of greater than one indicates that the harvest is above the management objective. Note that, unlike with the PEG models, this does not mean that an SHI above one will necessarily lead to population decline, it simply means that the level of harvest is greater than that targeted with the management objective. For large stable or increasing populations such a take would likely still be ecologically sustainable and would still allow population growth.

Given the uncertainties in population sizes, harvests and life history characteristics we estimated the SHI for all waterbird species on AEWA Action Plan Table 1 that are hunted in the UK using the PTL approach for values of F_{obj} from zero to one and assuming that population sizes can vary by ±50% and bags can vary from zero to five times the current best estimates. To ensure we captured the uncertainty in current estimates of life history data we modelled scenarios ("low" and "high") using variables known to produce the minimum and maximum values for the SHI and averaged them. Full details on the methods and complete results are given in Appendix 5, and a summary of the results is given in Figure 1, below.


Mallard
Harvest
Population
Teal
Harvest
Population
Pintail
Harvest
Population
Shoveler
Harvest
Population
Goldeneye
Harvest
Population
Pink-footed goose
Harvest
Population
Common snipe
Harvest
Population
Golden plover
Harvest
Population

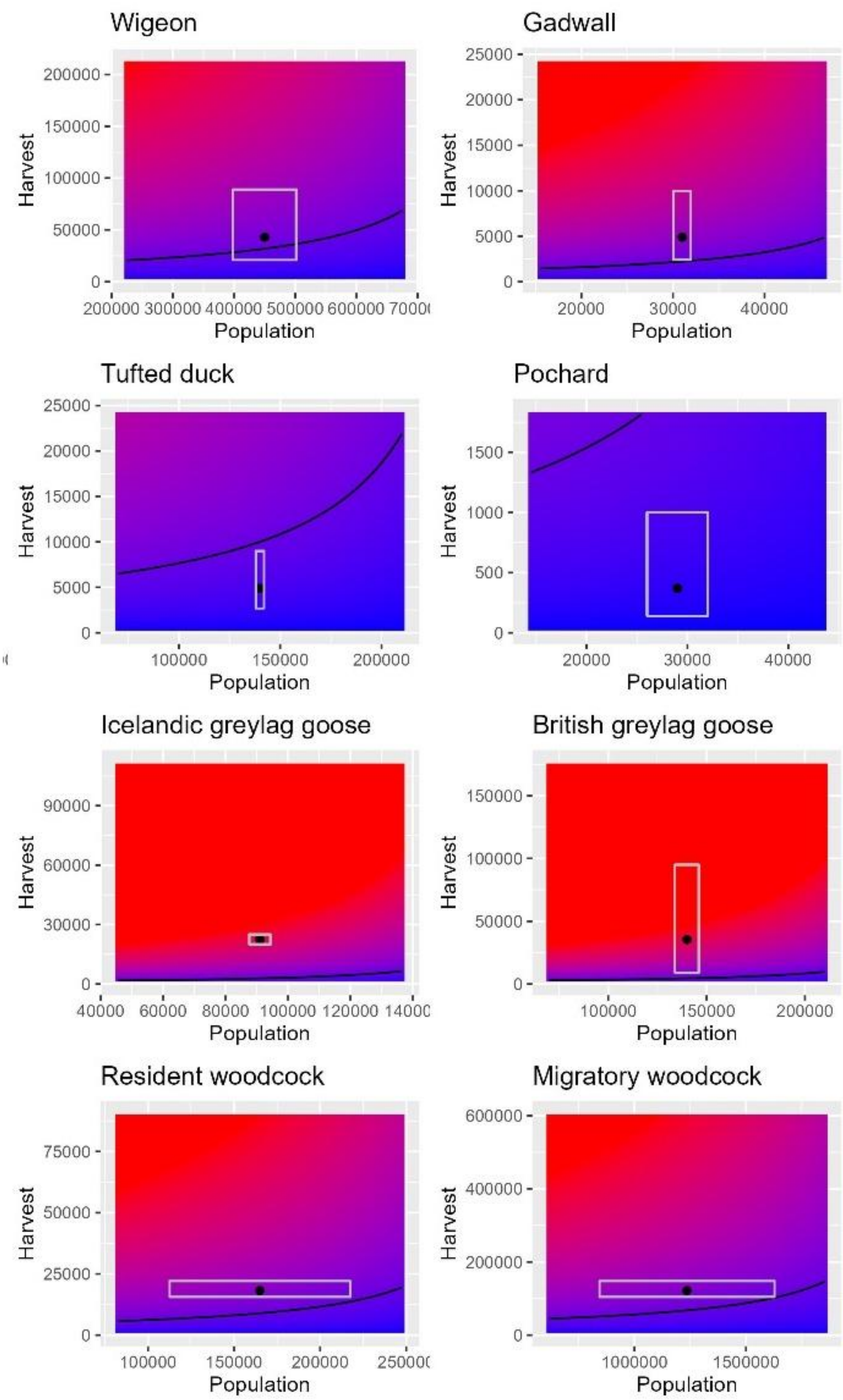
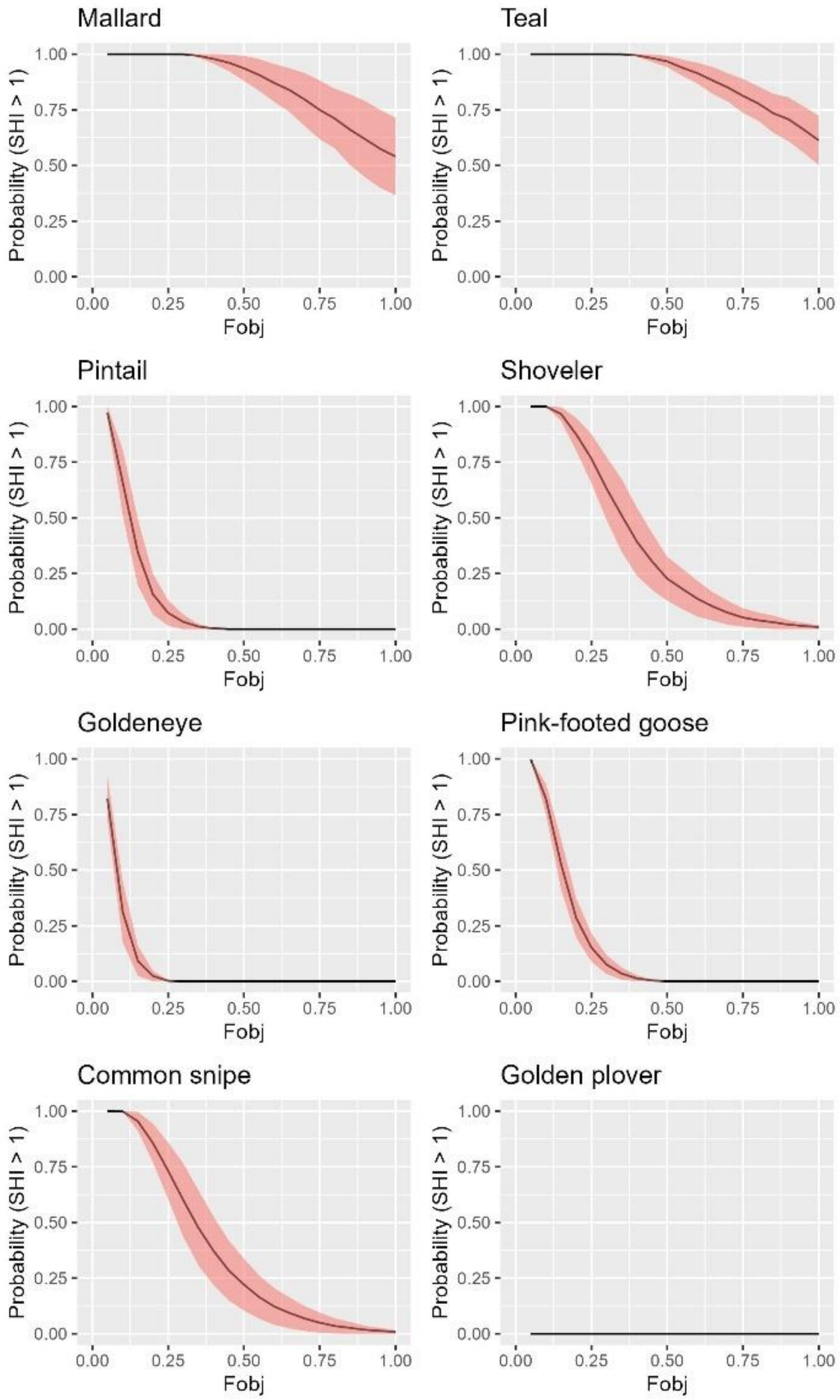


Figure 1. Prescribed Take Level (PTL) application of Sustainable Harvest Index (SHI) for AEWA Action Plan Table 1 species under varying harvest and population levels with an example management objective (F_{obj}) of 0.5. The SHI is indicated from blue (SHI < 1) to red (SHI > 1) with the threshold (SHI = 1) shown with a black line. Combinations of population and harvest above the black line (shown in increasingly red colours) exceed the management objective and may not be sustainable. The current best estimates of population size and harvest are shown with a black dot, and their confidence intervals are shown as the grey box.

Uncertainties around harvest and population sizes of waterbirds confound the assessment of the sustainability of their recreational harvest. This is further complicated by the lack of a clear framework for deciding on appropriate appetites for risk or target population sizes (e.g. FRRs) when setting F_{obj}. To address this, we restricted the models to 0.5–1.5 times the current best estimates for both population and harvest under the low and high scenarios and estimated the proportion of combinations of population and harvest which would result in an SHI above one (harvest exceeds management objective) (Figure 2). For example, for Golden Plover, the probability that harvest exceeds the management objective never occurs within this set of limitations, whereas for Greylag Goose, harvest is almost certain to exceed the proposed management objective at almost all levels of management objectives. For Goldeneye, harvest is likely to exceed the management objectives if these are very restrictive, but when these are more liberal, they are unlikely to be exceeded. This approach likely covers the realistic range of possible harvests and population sizes over the short term but is a gross simplification and compounds many of the assumptions, especially around the accuracy of harvest and population estimates. However, it does provide a useful overview of the likely level of sustainability for each species within a reasonable range of population and harvest estimates, with an emphasis on whether there is a risk of unsustainable harvest, and this is done without making assumptions of the socio-politically acceptable risk level. Overall, our PTL based results are largely the same as those based on PEG published previously (Ellis & Cameron 2022).


Mallard
Teal
Pintail
Shoveler
Goldeneye
Pink-footed goose
Common snipe
Golden plover
Probability (SHI > 1)
Fobj
0.00
0.25
0.50
0.75
1.00

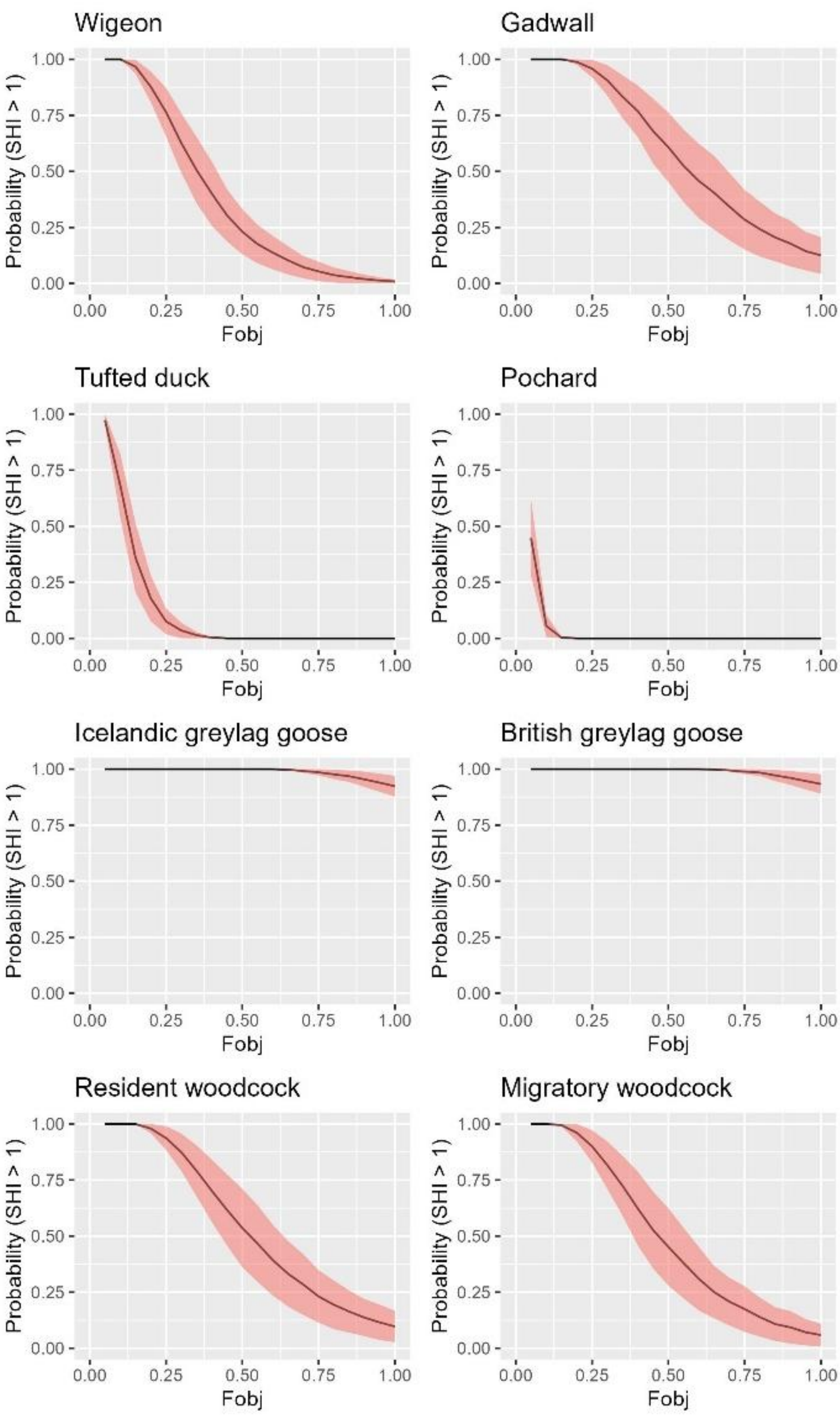


Figure 2. The proportion of simulations where current estimates of harvest and population (±50%) resulted in a Sustainable Harvest Index (SHI) > 1 (harvest exceeds management objective), for values of management objective (F_{obj}) from 0 (restrictive harvest – take minimal surplus in the maximum sustainable yield) to 1 (liberal harvest – take all surplus in the maximum sustainable yield). The (pink) shaded area represents the extent of the “low” and “high” modelling scenarios, and the solid black line is a simple average of the two scenarios to aid presentation.

As a rough guideline we suggest the criteria in Table 4 for assessing the sustainability of harvest under varying harvest and population levels. This approach allows for a clearer capturing of the uncertainty around population and harvest levels but still assumes that the “true” values lie within ± 50% of current best estimates. For most species the model bounds are likely to be valid, but for those species with low reliability population scores (Common Snipe, Woodcock, Mallard and Teal) and those where the division of resident (or released) and migratory populations is unclear (Mallard, Woodcock and Greylag Goose) we simply don’t know if the assumptions are valid, and, if not, in which direction the bounds should be moved.

Importantly, this approach however does not always account for observed trends in the real world. For example, our model suggests that harvest of Gadwall may be unsustainable (category 2*). However, the breeding population of Gadwall has increased by 129% in the past 10 years, and the wintering population by 130% and 16% over the last 25 and 10 years respectively (Burns *et al*. 2020). Similarly, our model suggests that Teal harvest is likely to be unsustainable (category 1*), but Teal show increases in the short-term breeding and short and long-term wintering trends (Burns *et al*. 2020). As discussed in Ellis and Cameron (2022), there are beginning to be indications of a slowing in population growth rate for Teal, but we may also be underestimating both the true population size, and the rate of turnover. Either our harvest estimates are too high, our population estimates for this species are inaccurate and too low, or Gadwall and Teal can sustain higher harvest rates than 50% of maximum productivity more readily than we know. An immediate focus should be on new research of breeding productivity of these species and an updated analysis of juvenile and adult survival from existing ringing data.

The assessment for the harvest of Greylag Goose as being unlikely to be sustainable is complicated by the lack of knowledge of the proportion of resident and migratory species in the harvest (of both the Icelandic and North-west Europe populations), as well as significant uncertainty over the true population size, with estimates suggesting the population could be 2–4 times greater than our current best estimates (Frederiksen *et al*. 2004; F. Johnson pers. comm). Certainly, this would be consistent with our results as a doubling of the population would halve the SHI (to approximately 1 when $F_{obj} = 1$) which would suggest that Greylag Goose harvest was approximately 100% of the maximum sustainable yield.

The assessment for the harvest of Mallard as being unlikely to be sustainable does not account for the fact that the majority of Mallard in the UK wintering population and the majority of the harvest bag includes many of the 2.6 million Mallards released for rear-and-release shooting of this species in the UK. Even if only 35% of the 2.6 million released birds are subsequently shot in this form of shooting (as is reported for harvest rates for other lowland driven gamebirds in the UK – Robertson *et al*. 2017) then as many as 910,000 of the 940,000 Mallard bag will not represent wild Mallards (resident and migratory) and our concern over overharvesting of wild populations would dissipate.

Table 4. Suggested categorisation of the risk of unsustainable harvest for AEWA Table 1 species. Author concerns about data quality or observed contradiction with population trends (e.g. teal) mean these categorisations marked with an asterix (*) should be treated with caution.

Category	Species (population estimate reliability)
1. Species where more than 50% of simulations had an SHI > 1 when F_{obj} = 0.5	Icelandic Greylag Goose *Anser anser anser* (2)* Mallard *Anas platyrhynchos* (3)* Common Teal *Anas crecca crecca* (2)* Resident Greylag Goose *Anser anser anser* (2)*
2. Species where the bounds of the low and high scenarios included a 50% probability of SHI > 1 at F_{obj} = 0.5	Gadwall *Mareca strepera strepera* (1)* Resident Eurasian Woodcock *Scolopax rusticola* (3)* Migratory Eurasian Woodcock *Scolopax rusticola* (3)*
3. Species where fewer than 50% of simulations resulted in an SHI > 1 when F_{obj} = 0.5	Eurasian Wigeon *Mareca penelope* (1) Northern Pintail *Anas acuta* (1) Northern Shoveler *Spatula clyptea* (1) Tufted Duck *Aythya fuligula* (1) Common Pochard *Aythya ferina* (1) Pink-footed Goose *Anser brachyrhynchus* (1) Common Goldeneye *Bucephala clangula clangula* (2) Common Snipe *Gallinago gallinago gallinago* (3) Eurasian Golden Plover *Pluvialis apricaria altifrons* (2)

Population estimate reliability (From Musgrove 2011): 1 – an estimate based on good-quality counts of a large proportion of the individuals involved; 2 – an estimate which is heavily based on count data but for which these data have had to be extrapolated to a large degree; 3 – an estimate which is not strongly based on actual count data and/or for which large assumptions have had to be made.

5.2.2. Matrix-based population viability analysis

The data used in the modelling approach of Ellis and Cameron (2022) and the amended approach used above (Section 5.2.1), can also be used to parameterise stage-structured models such as matrix models. In a matrix model the waterbird life history is fully represented such that survival and reproductive values of each life history stage class are multiplied against stage specific population sizes to generate population growth rates which are projected over time to assess the probability that the population will grow or otherwise decline towards extinction – a population viability analysis (PVA). For this reason, in a matrix model, the more realistic life history and population structure that is attempted to be recreated in the model, requires more data for parameterisation.

To compare the PEG/PTL approach to that obtained through a matrix model based PVA we created a two-stage matrix model for European Teal using the best available data in the literature. A full exploration of these results is given in Appendix 6. Briefly, using data from the literature on the survival rate and productivity for Teal in PTL, we found a near 50:50 probability that harvest was unsustainable and the population would decline. However, when we used popharvest to estimate survival and productivity (as Ellis & Cameron 2022) the PVA matrix model predicted that the harvest was much more likely to be sustainable. This confirms what we already know about the PEG/PTL modelling approach, that when combined with specific F_{obj} it is already a precautionary approach for estimating harvest sustainability of data-poor wildlife populations. Finally, the matrix-based PVA models were used to explore the sensitivity of waterbird population growth to changes in productivity and harvest, moving away from maximum productivity used by the PEG/PTL models. Here we found, using PVA models, that Teal populations were more likely to decline under current estimates of UK harvest when we assumed lower productivity values. This supports a common result in waterbird demographic studies that population growth, and responses to harvest are more strongly coupled to annual productivity than effects on adult survival.

The PVA matrix-based models were no more numerically challenging than the PEG/PTL approach but with many more parameters are much more contingent on suitable data to parameterise those values. Even for Eurasian Teal, a very common breeding species across northern Europe about which we have probably the most detailed and accurate life-history data for UK populations, we had to rely on some studies from North America and a closely-related but different species – Green-winged Teal – to gain appropriate values to represent productivity, nest survival and adult survival.

In conclusion, models are only as good as the data on which they depend, and while PVA approaches provide alternative methods to model wildlife population dynamics and predict their response to harvest, they are not necessarily a more robust or better approach than simpler models. Using adult Teal survival values from North American and mainland European field studies in a PVA model approach we found a near 50:50 likelihood of unsustainable harvest with very high effects of hunting on adult Teal survival. If instead we built a PVA using the optimum survival of Teal as per the ‘popharvest’ approach and modified that survival using the UK hunting mortality from Aebischer (2019) as used in the PEG approach of Ellis and Cameron (2022): the likelihood of unsustainable harvest of teal by UK hunters was closer to zero. This suggests that the use of F_{obj} targets such as in ‘popharvest’ models are more precautionary than using matrix models in a population viability approach to determine whether harvests are likely to be sustainable or not.

We recommend the use of popharvest PTL models to the UK statutory conservation agencies as decision tools for wild bird harvesting in the United Kingdom. ‘Popharvest’ models can operate while relying on less data. Despite being simple models, they have been found to be as functional and as representative as more complex models that seek greater realism (e.g. matrix and PVA approaches). They appear to be more precautionary than other approaches. Perhaps most importantly, they have been accepted as appropriate tools for introducing Adaptive Harvest Management approaches to new and data poor systems (Johnston *et al*. 2024) while also being adopted by the European Commission for those same reasons (Cruz-Florez *et al*. 2024).

6. Conclusions on the sufficiency of the current data for assessing the scale and sustainability of waterbird harvest in the UK

6.1. Key findings

Currently, **our confidence in the ability to accurately estimate the scale or sustainability of waterbird harvests in the UK declines as the estimated proportion of the population harvested increases**. Our ability to obtain these estimates depends on four broad factors: the current population size of the species; the relevant life-history parameters of the species; the harvest level for the species; and the susceptibility of the species to other random non-hunting factors that influence survival and reproduction, both in the UK and across their flyways. For some of these species in the UK we have few reliable or up to date specifically UK based measures, especially those for survival or accurate population size. Furthermore, because many of these species are migratory, their populations are shaped by events and conditions outside the UK about which we may have little information. Were more robust data to exist, then **there already exists a number of suitable modelling approaches that may be used to make predictions about population changes under different hunting scenarios**.

Accurate data on harvest is very sparse, and of all the limitations this is the most significant knowledge gap. All the current harvest datasets that we considered have multiple serious limitations and biases. Each differs from the others to the extent that integrating them will be difficult. In addition to poor data on harvest levels, we also currently lack any information about wounding and retrieval rates, adding further error. However, the scale of the confidence intervals around both harvest and population likely dwarf the impact of both wounding rates and any misidentification. Harvest data is currently commonly provided by individual Guns, and so a better understanding of their representative behaviour, and the variance amongst them, under current and future conditions would provide greater confidence when extrapolating harvest levels from the records of individual Guns. Likewise, for records provided by shoots, rather than individuals, confidence in models of harvest sustainability would be improved by understanding the typical harvests of such shoots and the variance among them, both currently and under possible future scenarios.

Our **current estimates of waterbird population sizes in the UK are poor, with large confidence intervals**. They are also conducted relatively infrequently, making triennial reporting to AEWA difficult. Given the efforts and funding already expended on surveying and estimating UK bird populations, it is unlikely that this component of the calculations can be improved markedly without a fundamentally different surveying approach.

The **relevant life-history parameters of a species that are necessary to calculate population dynamics (survival and reproductive success) are poorly understood for UK waterbirds** but as is the case for the current estimates of population sizes, the lack of “perfect” data is not a barrier to beginning models and interventions based upon them within an adaptive harvest framework. The survival estimates that we do have are old, of questionable UK-relevance and likely to be confounded by ongoing harvest. For key harvested species such as Mallard, Teal, Wigeon and several geese, there exists appropriate ringing return information for these parameters to be estimated – subject to appropriate investment in analysis. Productivity estimates for most of these species are poorly understood and require investment in field studies in their breeding range in northern and eastern Europe and Russia. Deliberate further collection of more detailed and representative data on specific life history parameters is likely to be time-consuming and costly, but such data will gradually accumulate through projects perhaps unrelated to harvest that take these measures as part of ecological or conservation studies of these species. Despite these potential barriers, there is suitable data on productivity for some ducks and geese on the AEWA List and significant ringing data on which to estimate waterbird survival from natural and hunting mortality, but it is currently unanalysed.

For the most commonly shot quarry – Mallard – it is currently difficult to disentangle the harvest of released Mallard, which may total many hundreds of thousands, from that of wild populations or to understand how these released birds, which may dwarf natural populations, are concentrated, survive, disperse and contribute to estimates and dynamics of wild populations.

Crucially, **we lack any information about the quarry selection, hunting behaviour and attitudes of a community of diverse Guns and landowners that may allow an understanding of how Guns might adjust their behaviour under different future scenarios**. Collecting such data may be possible through the participation of Guns and landowners in surveys or monitoring. To achieve this, it would be necessary to develop either social norms or enforceable legislation through which accurate, regular and detailed harvest reports are collected in a standardised manner. This is not a trivial undertaking, given that waterbird hunting is often a lone activity and may be practiced by people with distrust or scepticism towards the collection of data that could be used to restrict their activity. Below we make some suggestions of how this might be achieved (see Section 7.2). However, given the current very poor quality of data available to estimate the bag-size and sustainability of such take for AEWA-listed species, any such data improvements may bring marked benefits, especially if similar improvements can be made to population estimates.

The combination of the uncertainties around population sizes, harvest levels and life history characteristics means that the uncertainty in assessments of sustainability increases as harvest (as a proportion of the population) increases. For this reason, **we urge caution in interpreting the findings of our models for species apparently identified as being subject to unsustainable harvest. We suggest that these cases highlight those species where there is the greatest need for more information in order to refine estimates of population, harvest and life history characteristics, and that an adaptive harvest approach would be a precautionary way to achieve this.** Using the best available evidence, our models suggest that harvest of Mallard, Teal, Greylag Goose, Gadwall and Woodcock could be unsustainable, yet population trends are increasing or stable for some of these species (Teal, Gadwall, migratory Woodcock) and known, or thought, to be underestimates for others (Greylag Goose, Mallard, Teal). This does not invalidate the models, nor undermine the importance of existing data collection, but is simply a function of compounding the uncertainty from three separate data sources. On the other hand, for species where the level of harvest is small compared to the population, we can be confident that harvest is unlikely to be unsustainable. Under these scenarios the highest realistic estimates of harvest do not exceed sustainable levels for even the lowest modelled population estimates meaning that these estimates would have to be out by several orders of magnitude for harvest to be unsustainable.

6.2. Key knowledge gaps

For any waterbird species harvested in the UK we currently do not know accurately the number taken and our current margins of error are often an order of magnitude. To improve model estimates, it would be helpful to have accurate information on waterbird harvest, at the level of individual species, collected annually and accessible in as near to real time as possible to permit rapid modelling to facilitate adaptive harvest management. To address this, a record system is necessary that Guns will reliably comply with, has known boundaries of error (e.g. in terms of bag size, species identification and time of harvest) and is accessible in near to real time. To comply with AEWA obligations, this harvest data should be available to produce updated model population estimates at least triennially. If there is an additional requirement for models to be predictive either so that longer term projections of these population dynamics are possible, or so that potential proposed changes to harvest behaviour can be evaluated, then additional or more nuanced data is also required. In order to produce such models, we require inputs from a range of robust, up-to-date and representative data sets and currently these are unavailable.

We outline below, in order of priority, eight key knowledge gaps arising from our review of available data. We provide more explicit detail on the methods suggested to address the knowledge gaps in Section 7.

6.2.1. The size of annual waterbird harvest (either in total or at the level of an individual Gun) in the UK

This could be addressed through improved coordination of current data collection.

In addition, and to broaden our understanding of harvests in different hunting scenarios and habitats, a stratified sampling survey of Guns and their behaviour should be conducted, perhaps with the assistance of shooting organisations. This will permit future modelling attempts to refine the harvest data available and so improve confidence in the estimates. This might best be conducted in conjunction with, and supported by, hunting organisations.

As a longer-term data source to inform future descriptive and predictive models, a formal bag return program should be instituted with all waterbird harvest outings being recorded and collated (see Section 7.1.1)

6.2.2. An accurate breakdown of species being harvested individually or collectively across the range of waterbird hunting scenarios

This could be addressed using the stratified sample survey described above (Knowledge Gap 1), which could be extended to include data about species shot.

As a longer-term data source to inform future descriptive and predictive models, a formal bag return program should be instituted with all waterbird harvests being recorded and collated (see Section 7.1.1)

6.2.3. An understanding of the harvesting behaviours, preferences, opportunities or decisions of individual Guns

The stratified sample survey described above (Knowledge Gap 1) could be extended to include self-reported and/or observational data about harvest behaviour such as whether they may be “density-dependent”, deliberately practicing self-regulation when quarry numbers are low, or simply refraining from going out shooting when the effort is not matched by the anticipated reward. These surveys also could be extended to include self-reported and/or observational data about Guns’ responses to either legislation (in terms of likely compliance, shifting quarry species, or changes in associated habitat management) or new voluntary self-report schemes that may be devised.

6.2.4. Accurate data about the percentage of waterbirds that are not reported in harvest records because they are wounded

This could be addressed through targeted studies of wounding and retrieval rates of Guns shooting in different scenarios in the UK, either through self-report surveys (see Knowledge Gap 1), or, better, by independent observers accompanying Guns while hunting.

6.2.5. Accurate and representative records of the sex and age of harvested birds (especially those which cannot easily be sexed based on morphological features).

This data could be incorporated in the survey scheme outlined in Knowledge Gap 1. However, the accuracy depends on the identification skills of the Guns or shoot managers collecting the data. An assessment of the identification skills of Guns could be obtained during the observational surveys outlined in Knowledge Gaps 3–5, as could independent aging and sexing conducted by trained recorders during these observational surveys

6.2.6. An understanding of productivity and survival of waterbirds in a UK context

This could be addressed through analysis of current BTO ringing records, or through detailed, species specific ecological field studies (costly).

6.2.7. An understanding for many species of what proportion of the UK population travels across which areas as part of their annual cycle, as well as integrated, up-to-date and representative data about, particularly, the human factors that the birds may encounter *en route* including local hunting or disturbance pressures.

This information gap should be addressed through an AEWA work plan aimed at understanding the sustainability of harvest at the flyway level, likely to start in 2025, as well as a European Commission project with similar aims due to report in late 2024 (M. Guillemain pers. comm.).

6.2.8. An understanding of the contribution that several million released Mallard make to UK population and harvest estimates (where and how many Mallard are released and shot in the UK, how they disperse after release, and how they interact/breed with wild Mallard)

These knowledge gaps could be addressed through focussed analyses of, for example, Poultry Register records (scale and location of releases), and/or through field studies of tagged Mallard released on shoots to illuminate harvest, movement and interaction patterns.

7. Policy and implementation options to fill knowledge gaps

7.1. Options for collecting accurate harvest data

Collecting accurate harvest data requires input from a large and representative sample of Guns and shoot owners. This can either be obtained voluntarily, (as is currently the case in the UK) through incentive, or through legislation. Suggestions for improving the collation of harvest data are not new and have been reviewed previously by Parrott *et al*. (2003), whose work is largely still valid. Some methods that they evaluated are not appropriate (e.g. the Game Licence has been abolished, game dealer licences do not cover many waterbird species). They made three recommendations: establishing hunter and shooting estate registers; Shotgun licence questionnaire survey and Game Conservancy Trust (GCT; now GWCT) estate database; utilisation and calibration of existing BASC and GCT surveys. The authors preferentially recommended the establishment of hunting registers, analogous to schemes in six of the nine European countries that they considered. This approach would also permit alignment with other European countries sharing AEWA obligations. However, none of these recommendations have been enacted in the past 20 years. Aebischer and Harradine (2007) investigated a potential tool to improve harvest data estimates, but in the intervening years the surveys underpinning their method have either ceased or changed making this approach challenging to implement.

Currently, harvest data of waterbirds in the UK are strongly skewed by the inclusion of records from released Mallard. We estimate that these may make up 910,000 of the approximately 1.4 million (65%) waterbirds shot annually (Section 3.1.1), and their inclusion may: a) bias overall estimates of harvest of other species if the crude classification “duck” is used with poor/absent separation of species during harvest reporting; b) distort harvest, population and life-history estimates of wild Mallard. Consequently, we suggest that effort be made to a) account for/exclude released Mallard from estimates and predictive modelling efforts by understanding their movements post-release, survival and contribution to wild populations; and/or b) identify and exempt released Mallards from harvest data by marking released birds so that their contribution can be ignored. This needs a method of low cost/high volume tagging of some kind – wing tags or plastic domestic fowl leg ring seem most suitable and safe for general use by those releasing birds, but may impose time, financial and welfare costs during the tagging process. Research into this option is desirable.

One option to allow a clearer picture of wild Mallard harvest could be to ban the release of reared Mallard. However, this disturbance (effectively a cessation) to an established recreational activity would have large-scale socio-economic consequences as well as possible effects on the wild population supported by the surviving released birds or their management, which should be carefully considered. We cannot recommend banning releases of Mallard without such a detailed consideration, which falls outside the scope of this review.

We consider five possible new methods of collecting waterbird harvest data below (summarised in Table 5) which are subsequently explore in detail.

Table 5. Summary of the five possible new methods for collecting harvest data in the UK, in decreasing levels of complexity and legislative/organisational requirement.

No.	Method	Coverage	Accuracy	Cost
1	Mandatory licence	Complete/Medium	High	High/Medium
2	Survey of shotgun certificate holders	Medium	Medium	Medium
3	Constant effort monitoring	Low	Medium	Medium
4	Voluntary return scheme	Very low	Medium-low	Low
5	Combine existing National Gamebag Census, Crown Estate schemes and other data	Low	Medium	Low

7.1.1. Mandatory Licence

A mandatory licence, with licence renewal linked to harvest returns. This could operate with shoot providers, wildfowling clubs, game bailiffs or the police able to inspect licences on demand. Mandatory reporting schemes for hunting and fishing exist globally. In the UK, anglers buying over 903,000 licences raised £20.3 million in 2022/23, and failure to provide catch returns for those holding migratory fish licences results in significant fines (Fisheries annual report 2022 to 2023). One potential variant of the use of licences to provide data is that they do not have to monitor all participants provided a suitably large and random selection of participants take part (e.g. North American system). The North American Waterfowl Harvest Plan conducts several surveys that operate this way to obtain harvest return information as well as information on responses on the views and experiences of hunters to help develop "biologically feasible" harvest management decisions that also provide the greatest benefits to multiple stakeholders (Patton 2018). This information is often supplemented by species specific bag data from bag checks by wardens, particularly when hunters visit public hunting areas (Gammonley & Runge 2022).

7.1.1.1. Pros

- A total (representative) coverage of Guns across geographic areas and hunting scenarios.
- Annual returns to feed into a more adaptive management process.
- Would improve social acceptance for hunting if harvest sustainability has greater transparency.
- Would improve social licence for hunting if hunting licence can generate income for habitat restoration and conservation initiatives.

7.1.1.2. Cons

- Additional costs and bureaucracy to administer (though could be offset by charge for licence and use of technology (e.g. Apps) may be possible, providing some automation of data collection and thus reduction in administration).
- Would require legislation to instate.

- Accuracy of the harvest return is still dependent on the identification skills, memory, record keeping and honesty of individual Guns (some of these may be addressed by the use of an integrated App that permits photo-records and prompts contemporaneous data recording).
- Likely to provoke objections from Guns, etc., due to cost and privacy issues.

One variant of this idea is that instead of collecting returns from all licence holders, a more detailed, but representative sample is taken, rather like in the USA. This may reduce data quantity, but it likely offers little reduction in the administrative costs of issuing and policing licences so seems to offer little benefit.

7.1.1.3. Costs

- Licencing system, including collation and storage of harvest returns. The nearest equivalent that we are aware of in the UK is the Salmon and Sea trout (Migratory Fish) rod licence which requires an annual catch return (including null catch). This currently costs the angler £90.40/year, administered by the Environment Agency and operates with a surplus which is reinvested in to maintaining fishing and waterways. A licence that is concerned only with recording information and covering costs of some compliance activity could be cheaper.
- Enforcement, either as part of general policing or through a series of bailiffs/warden as in fishing.
- Modelling. This would become relatively straightforward once pipelines have been developed to feed harvest returns available in a standard form into model code.

7.1.2. Survey of shotgun certificate holders

A survey linked to shotgun certificate holders could either occur randomly each year with a subset being drawn from the complete list or could be linked to renewals (currently 5 years) with the portion of certificate holders renewing each year serving as the sample. This could operate voluntarily, or certificate renewal could be contingent on submission.

7.1.2.1. Pros

- A representative coverage of Guns across geographic areas and hunting scenarios.
- Administrative system currently in place.
- Annual returns to feed into a more adaptive management process.
- Social acceptance as per option 1.

7.1.2.2. Cons

- May require legislation to instate.
- Imposes additional costs, likely not covered by current certificate prices.
- Potential General Data Protection Regulation issues with police sharing contact details.
- Accuracy of the harvest return is still dependent on the identification skills, memory, record keeping and honesty of individual Guns.
- Many shotgun certificate owners will not shoot game, but only clays. They will provide null data. (see evaluation of the extent of null data provided in Parrott *et al*. 2003).

- Reliance on records for a 5-year period of a certificate may be unreliable.
- Likely to provoke objections from Guns etc due to privacy issues (or claims to be clay shooters).
- If voluntary – sample size may be low and unrepresentative.

7.1.2.3. Costs

- Survey/sampling could be incorporated into the overall shotgun licence renewal process with costs also incorporated.
- Enforcement, as part of the overall shotgun licencing process.
- Modelling. This would become relatively straightforward once pipelines have been developed to feed harvest returns available in a standard form into model code.

7.1.3. Constant effort monitoring

A constant effort monitoring program at a representative set of sites. Wardens/bailiffs at these sites would collect accurate data on numbers of birds harvested throughout the season by direct counts, interviews with visiting Guns and inspection of bags at the end of the day. For example, Illinois, USA has a number of waterbird hunting sites where hunters are required to report harvest to check stations prior to leaving the site, such as the Mississippi River State Fish and Wildlife Area. In addition, Illinois has a series of sites where hunters' harvest is monitored by wardens and compared with harvest reported subsequently at the end of the season to assess the accuracy of and calibrate reported harvests. These are likely useful and complimentary to options 1 and 2 above and not a replacement.

7.1.3.1. Pros

- High quality data collected by experts (age/sex/species identification can be very precise).
- Constant effort makes comparisons between years robust.

7.1.3.2. Cons

- Given the small numbers of some species shot annually, and the site specificity of some species, there may need to be a large number of constant effort sites required, each demanding a trained monitor. This could be expensive to establish and maintain. While this may work for organised driven (wild or released) game shoots where dates are planned a year in advance, it may be less suitable for wildfowling or IWB where shooting parties are smaller, less organised and operate on a more ad hoc basis when conditions are right.
- Access by monitors to shoots on private land may be opposed by land/shoot owners. Learning how this works with fishing licences on private land would be a good way forward.

7.1.3.3. Costs

- Training and Employment of monitors. These jobs would only last for the shooting season, which depending on species would be August to February. One option could be to use river bailiffs whose fishing seasons typically occupy roughly March to September. Seasonal staff could be used. Local staff could be employed by particular shoots/clubs, perhaps as part of an accreditation or licencing scheme for shoots that release gamebirds.

- Modelling. The standardised returns from a fixed set of shoots would render this relatively straightforward once pipelines have been developed to feed harvest returns available in a standard form into model code.

7.1.4. Voluntary return scheme

A voluntary returns scheme could be advertised to Guns and shoots and promoted more vociferously than it currently is by the shooting industry/community to encourage uptake, with Guns being encouraged to participate via, for example, posters with QR links to online survey forms; well-advertised apps; peer encouragement (see e.g. shoot sweepstakes contributing to GWCT appeals); and/or incentives such as prizes.

7.1.4.1. Pros

- Potentially large sample size (however, email survey response rates down to around 2–10% (BASC, pers. comm.).
- Relatively low costs to establish and run.
- Applicable to all sorts of hunting scenarios and can operate in remote locations.
- Promotes engagement with Guns.
- Voluntary participation may reduce antagonism of Guns.

7.1.4.2. Cons

- Likely a strong bias in participants if the returns are “opt in”. Bias may be due to participation by more conservation-minded Guns, or under-representation from driven shoots if the Guns assume that shoot owners will provide data.
- Risk of a very low (and skewed) sample size if process is onerous or perceived as intrusive.
- Voluntary participation risks bias in data collection.
- Voluntary participation risks small sample sizes.
- Data accuracy depends on Guns’ identification skills.

7.1.4.3. Costs

- Administration and advertising of the survey scheme through shooting stakeholder networks (press, social media, events such as shooting shows/country fairs). This could partly be offset by collaboration with and thus promotion by shooting organisations.
- Initial establishment of app/survey software and ongoing maintenance.
- Prizes/incentives to promote participation.
- Some form of regular data-checking/validation to ensure data quality.
- Modelling. This would become relatively straightforward once mechanisms have been developed to feed harvest returns into model code.

7.1.5. Combine existing NGC, Crown Estate schemes and other data

The current, existing, survey methods (described in Sections 2.2 to 2.5) could be continued, strengthened, validated and combined in a hybrid approach to produce annual indices.

Additionally, a 5- or 10-yearly (or, 6-, 9- or 12-yearly to tie in with AEWA triennium) validation survey could be conducted to calibrate the indices. Such work is currently conducted on an ad hoc basis by a range of parties as interest arises rather than in any coordinated manner. Committed funding could be used to ensure that this work is conducted regularly and following the same protocols for consistency.

7.1.5.1. Pros

- Survey and monitoring schemes already in place and currently funded by shooting stakeholders.
- No need for new legislation.
- Relatively cheap to run.

7.1.5.2. Cons

- Further work needed on representativeness of samples.
- Low sample sizes, due to low voluntary participation, means low confidence for some species (but less commonly shot so potentially less concern).
- Mix of data likely requires more complex modelling approaches.
- For the NGC at least, data anonymity is promised, and specific uses are defined when shoots enter the scheme. Data use in a broader, monitoring, context may deter participation.

7.1.5.3. Costs

- Organisations currently running schemes (BASC/GWCT) may want/need financial support to make data public.
- Costs for a project to develop and validate the index against known data. Probably would require a one-year post-doctoral academic position, or similar.
- Regular (5–10 year) more detailed validation surveys would be continually required to ensure data reliability.
- Modelling may be more complicated each year due to the variety/range of data used/available.

7.2. Options for collecting accurate data on Gun behaviour and attitudes

Accurate data on how Guns behave in different scenarios (e.g. density/encounter rate dependent harvest decisions; quarry choice switching in different mixes of available quarry, financial choices in a fluctuating market; personal preferences and willingness to travel) and an understanding of their anticipated behaviour if there are changes in quarry populations or harvest opportunities are essential to model future scenarios. Data could be collected through surveys, relying on self-reporting. However, hunting self-report data may be unreliable (e.g. Chu *et al.* 1992; Beaman *et al.* 2005) and this may be exacerbated in a situation where the respondent/Gun knows or believes that their data may be used to support advice, policy or legislation that restricts an activity which they enjoy. Therefore, the work should also include more direct behavioural observations of how Guns act in different scenarios. Thus, the work might comprise two strands: 1) surveys of Guns; and 2) observation of Guns.

7.2.1. Surveys of Guns

Existing surveys of Gun behaviour could be used, for example Value of Shooting (conducted every 8–10 years; see Section 2.5), or the Game Shooting Census (conducted annually; see Section 2.7). This would offer a low-cost opportunity in collaboration with shooting organisations that already have connections with Guns. By agreement, specific questions about shooting behaviour, attitudes and harvest decisions, both actual and anticipated, could be asked.

7.2.1.1. Pros

- Low cost.
- Established connections to Guns.
- Basic structure and logistics already established.
- Likely good coverage of all waterbird harvest scenarios.
- Large sample size.

7.2.1.2. Cons

- The surveys are already quite detailed and long, so respondents may become fatigued and data quantity and quality may suffer.
- Voluntary participation may lead to low and biased return rates.
- Responses may be deliberately or unintentionally inaccurate as Guns struggle to imagine novel scenarios and how they may behave.

7.2.1.3. Costs

- Cost of survey – If the survey was integrated with surveys 1, 2, or 4 described in Section 7.1, then the costs of adding questions may be negligible.
- Data extraction and analysis – variable, depending on the number and complexity of the questions being asked.

7.2.2. Observation of Guns

Observers would follow Guns while hunting, either in proximity (which would permit data collection on shot decisions, crippling and retrieval rates, quarry identification) or at a distance on a site. This could involve monitors at constant effort sites or others, collecting data on harvest, shot numbers, quarry availability, weather conditions, etc. This could include deliberate local scale/single site experimental manipulations of shooting conditions (quarry restrictions, bag limits, time restrictions) so that novel scenarios can be simulated, and behavioural responses observed.

7.2.2.1. Pros

- High resolution, precise and accurate data about harvest behaviour.

7.2.2.2. Cons

- High cost to train and deploy observers. See Section 7.1.4 for similar concerns about costs and logistics of employing monitors and some potential solutions.

- Small sample size due to high investment in high quality data. This may make it difficult to ensure representative data across all shooting scenarios.
- While this may work for organised driven (wild or released) game shoots where dates are planned a year in advance, it may be less suitable for wildfowling or Inland Waterbird Shooting where shooting parties are smaller, less organised and operate on a more ad hoc basis when conditions are right.
- Access by monitors to shoots on private land may be opposed by land/shoot owners.

7.2.2.3. Costs

- Training and employment of observers. Managing observers, connecting them with representative Guns and collating their data.
- Producing models to predict population estimates under differing harvest scenarios.

8. Concluding summary and recommendations

Our review indicates that the *status quo* simply is not providing a sufficient quantity and quality of data to model accurately and precisely the current level of waterbird harvest in the UK. The absence of robust and representative harvest data, coupled with uncertainty over waterbird population size estimates, means that our models of the sustainability of waterbird harvest have broad confidence intervals. However, for most species the current level of harvest is small compared to the populations and so even in the presence of large uncertainties we can be confident that harvest is not likely to be unsustainable. For species where harvest represents a greater proportion of the population (including Mallard, Wigeon, Teal and Greylag Goose) the high levels of uncertainty around both population size and harvest levels mean we can be less certain that harvest is sustainable, but we can estimate the degree of uncertainty. For Mallard, the numbers being harvested are likely to be largely composed of reared and released rather than wild birds, meaning that the harvest of this species is also likely to be sustainable, but the releases may affect the wild population in other ways. However, we also conclude that the lack of “perfect” information on life history variables, population sizes or harvest bags is not a barrier to starting population models for waterbirds and their harvests, which can be refined as data quantity and quality are increased.

To improve our ability to reliably determine the sustainability of the harvest of AEWA waterbird species, **our key recommendation is that the quantity and quality of data relating to waterbird harvest and Gun behaviour is markedly improved**. How this is achieved is dependent on a balance of considerations of resources, legislation and compliance. Prescribing this balance is beyond the scope of this review but we present a range of options for consideration in Section 7.

Our preferred approach to achieve this is:

- **A national reporting scheme, supported by additional focussed surveys of Gun and shoot behaviour** to obtain behavioural information and validate the broader national surveys. This should separate out harvest records by species, age, sex, harvest type and, for Mallard, whether the birds are wild or released. We believe that a voluntary approach is likely to find greater acceptance, especially if backed by credible shooting and conservation organisations. However, current voluntary schemes have relatively low levels of engagement. If there is no marked increase after a period of concerted effort (perhaps five years) to increase engagement voluntarily, backed by shooting organisations, then we recommend that a (more costly and perhaps less accepted) mandatory approach is enacted.

We also recommend a series of lesser improvements that can be implemented rapidly and at relatively low cost that would increase data quantity and quality on which models can be based. These may contribute only incremental improvements to current models and this low cost ‘desk based’ exercise in data collection should be matched by a more rigorous, bespoke and representative approach to improving harvest data as described above. These lesser improvements include:

- **Analysis of archive BTO ringing data** that could provide a marked improvement in life history data for several key waterbird species, which would refine current models of harvest sustainability.
- **A reanalysis of the existing raw PACEC/Value of Shooting data** that might reveal key insights into Gun harvest behaviour and attitudes.

- **An exploration of the APHA Poultry Register** that records the locations and numbers of Mallard releases, to inform our understanding of their contribution to population estimates and harvest sustainability.

Our understanding of waterbird harvest is at present entirely retrospective and dependent on population numbers, species life-history and, crucially, current Gun behaviour. The behaviour and distributions of waterbirds are likely to alter with climate change and Gun behaviour is also likely to alter, perhaps because of opportunity or education. There is also likely to be a subtle interaction between quarry availability and Gun behaviour. Therefore, to produce better future estimates of waterbird harvest effects, perhaps by considering an "adaptive harvest management" approach to waterbird harvesting (Nichols *et al.* 2007), it would be helpful to construct prospective models and to have validation of their outputs. Therefore, to estimate and ensure robustness of future estimates under uncertain conditions, we recommend:

- **Prospective models be developed**, based on current modelling approaches, informed by the improved data on Gun attitudes (obtained in Recommendations 1 and 3) and predicted waterbird population and life history data (obtained in Recommendation 2 and the literature).
- **Validation of prospective models via (quasi) experimental approaches**. For example, short term changes to, for example, season length, bag limits or quarry lists could be implemented. Such manipulations could be applied regionally and voluntarily with the engagement and consent of local Guns and shoots. This would likely achieve greater acceptance. Alternatively, manipulations might be applied more widely, reliant on use of Sections 2.5 and 2.7 in the Wildlife and Countryside Act. We suspect that compulsory manipulations, nationally or locally might be poorly received by Guns and shoots. For example, an experimental shortening of season for Mallard could be applied by delaying the start of the season by two weeks and monitoring this for three years, to test the prediction of a reduction in total mortality and a change (increase) in the juvenile/adult ratio.

Changes in data collection and monitoring of harvest activity risk antagonising stakeholders if the benefits are not tangible to them. Accurate data is most efficiently provided by Guns and shooting organisations and novel methods deployed to increase data quantity and quality will be more effective with Gun assent and compliance. Therefore, we recommend (indeed suggest it is vital) that:

- **Any new data collection techniques be conducted in close collaboration with shooting organisations and Guns.**

We note that some of these recommendations have been made repeatedly over the past 20 years (Parrot *et al.* 2003; Aebischer & Harradine 2007) but have not yet been enacted. Continued poor data risks leading to inaccurate or sub-optimal decisions being made about waterbird harvests and this fails all stakeholders. If these key knowledge gaps can be addressed, then we believe that current modelling approaches can be used to better determine the sustainability of the harvest of AEWA waterbird species, satisfying the obligations of the UK to the agreement.

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Appendix 1: National Gamebag Census summary data

Table A1.1. The average number of sites reporting non-zero bags for each country in the National Gamebag Census from 1961 to 2004 (Aebischer & Harradine 2007).

Species	Average number of sites reporting non-zero bags in England	Average number of sites reporting non-zero bags in Wales	Average number of sites reporting non-zero bags in Scotland	Average number of sites reporting non-zero bags in Northern Ireland	Average number of sites reporting non-zero bags in the UK
Greylag Goose *Anser anser*	14.3	0.5	17.0	0.5	32.2
Pink-footed Goose *Anser brachyrhynchus*	1.1	0	7.8	0	8.9
Greater White-fronted Goose *Anser albifrons albifrons*	0.5	0	0	0	0.5
Common Goldeneye *Bucephala clangula*	2.8	0.4	4.8	0.1	8.1
Common Pochard *Aythya ferina*	15.6	0.6	1.2	0.2	17.6
Tufted Duck *Aythya fuligula*	34.7	0.7	7.9	0.1	43.5
Greater Scaup *Aythya marila*	0	0	0	0	0.0
Northern Shoveler *Spatula clyptea*	11.6	0.9	0.3	0	12.8
Gadwall *Mareca strepera*	16.2	0.5	0.2	0	17.0
Eurasian Wigeon *Mareca penelope*	25.2	1.9	10.3	0.9	38.2
Mallard *Anas platyrhynchos*	183.4	5.4	47.5	2.1	238.5
Northern Pintail *Anas acuta*	5.0	0.6	0.4	0	6.0
Common Teal *Anas crecca*	120.5	6.2	30.5	2.7	159.8
Eurasian Golden Plover *Pluvialis apricaria*	3.8	0.2	2.5	0	6.5

Species	**Average number of sites reporting non-zero bags in England**	**Average number of sites reporting non-zero bags in Wales**	**Average number of sites reporting non-zero bags in Scotland**	**Average number of sites reporting non-zero bags in Northern Ireland**	**Average number of sites reporting non-zero bags in the UK**
Eurasian Woodcock *Scolopax rusticola*	295.3	12.4	98.2	4.3	410.3
Common Snipe *Gallinago gallinago*	119.9	8.5	96.0	3.2	227.6
Jack Snipe *Lymnocryptes minimus*	0	0	0	0	0
Common Moorhen *Gallinula chloropus*	61.9	1.0	2.2	0.6	65.6
Common Coot *Fulica atra*	25.8	0.6	1.7	0	28.0

Appendix 2: Crown Estate wildfowling returns: summary data

Table A2.1. The average number of wildfowling clubs submitting non-zero bag data per species, per country and the average harvest per species reported by clubs shooting over the Crown Estate foreshore over the period 1994/95 to 2022/23.

Species	Number of clubs reporting non-zero bags in England	Number of clubs reporting non-zero bags in Wales	Number of clubs reporting non-zero bags in Scotland	Number of clubs reporting non-zero bags in Northern Ireland	Number of clubs reporting non-zero bags in the UK	Average annual harvest
Greylag Goose *Anser anser*	10.76	4.72	0	1.76	17.24	200.3
Pink-footed Goose *Anser brachyrhynchus*	4.07	0.38	0	0.93	5.38	31.1
Greater White-fronted Goose *Anser albifrons albifrons*	1.24	-	-	-	1.24	1.7
Common Goldeneye *Bucephala clangula*	2.34	1.10	0	1.34	4.78	5.1
Common Pochard *Aythya ferina*	2.66	0.66	0	0.66	3.98	5.5
Tufted Duck *Aythya fuligula*	4.03	0.97	0	0.76	5.76	11.4
Greater Scaup *Aythya marila*	-	-	-	0.45	0.45	0.06
Northern Shoveler *Spatula clyptea*	7.59	2.34	0	1.48	11.41	28.2
Gadwall *Mareca strepera*	8.24	1.59	0	1.38	11.21	67.6
Eurasian Wigeon *Mareca penelope*	22.34	9.28	0	4.31	35.93	1,602.0
Mallard *Anas platyrhynchos*	23.00	9.24	0	4.38	36.62	1,019.6

Species	**Number of clubs reporting non-zero bags in England**	**Number of clubs reporting non-zero bags in Wales**	**Number of clubs reporting non-zero bags in Scotland**	**Number of clubs reporting non-zero bags in Northern Ireland**	**Number of clubs reporting non-zero bags in the UK**	**Average annual harvest**
Northern Pintail *Anas acuta*	12.28	4.86	0	1.86	19.00	139.6
Common Teal *Anas crecca*	22.52	8.83	0	4.10	35.45	1,619.8
Eurasian Golden Plover *Pluvialis apricaria*	2.45	0.69	0	1.07	4.21	1.7
Eurasian Woodcock *Scolopax rusticola*	0.59	1.00	0	0.90	2.49	8.7
Common Snipe *Gallinago gallinago*	4.17	2.45	0	2.21	8.83	43.2
Jack Snipe *Lymnocryptes minimus*	-	-	-	0.72	0.72	1.7
Common Moorhen *Gallinula chloropus*	0.03	0	0	-	0.03	0.04
Common Coot *Fulica atra*	0.07	0	0	-	0.07	0.1

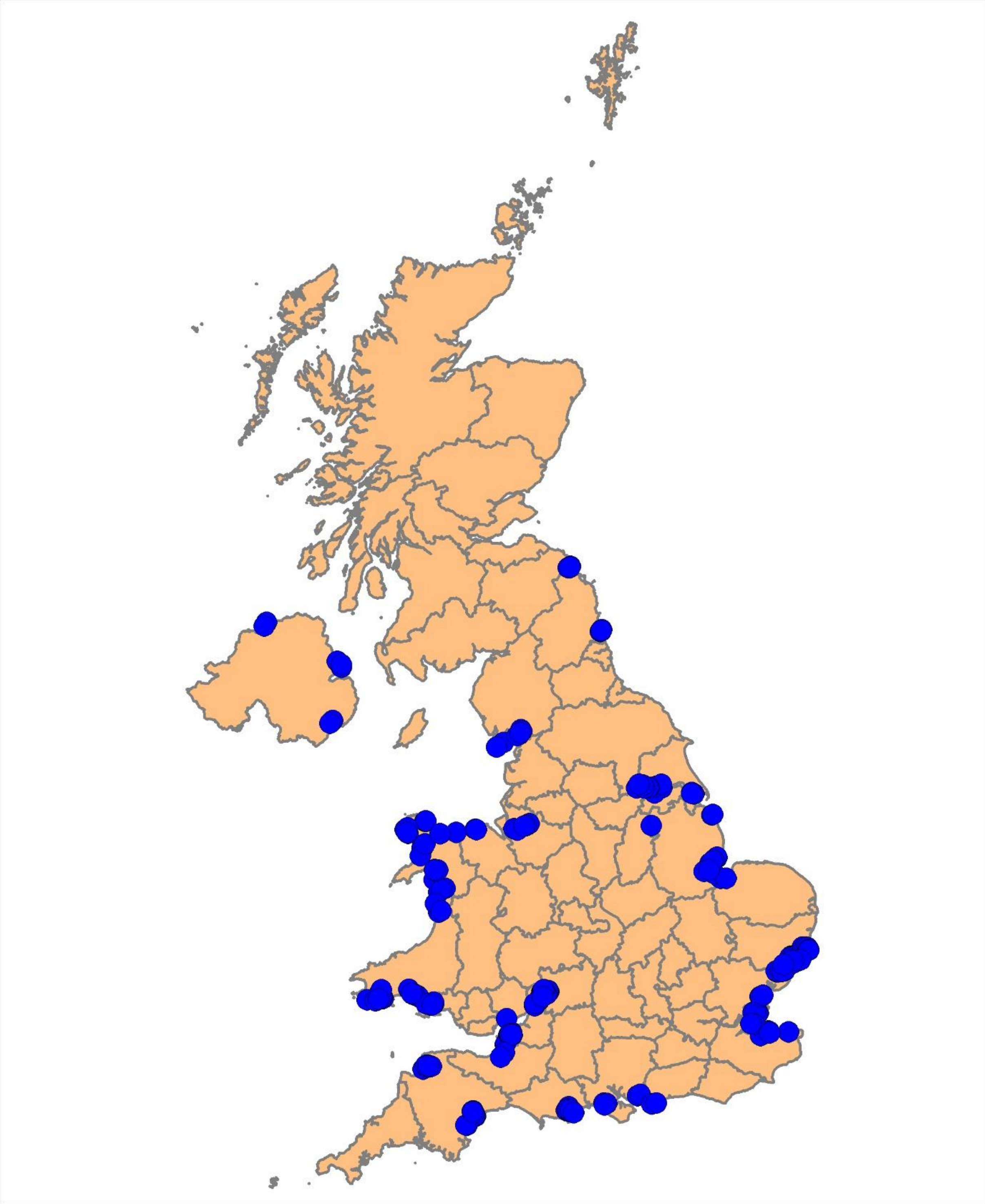


Figure A2.1. Locations of BASC affiliated wildfowling clubs which submit data to the Crown Estate wildfowling returns.

Appendix 3: BASC wing survey summary data

Table A3.1. Number of duck, goose and wader wings submitted to the BASC wing survey per country from 1986 to date.

Year	England	Wales	Scotland	Northern Ireland	Unknown	UK
1986/87	2,168	212	742	79	62	3,263
1987/88	478	35	336	31	14	894
1988/89	942	77	491	18	99	1,627
1989/90	255	12	43	0	0	310
1990/91	2,783	127	346	0	51	3,307
1991/92	58	0	155	0	0	213
1992/93	1,129	118	399	352	3	2,001
1993/94	3,571	562	825	435	2	5,395
1994/95	3,698	326	636	191	37	4,888
1995/96	5,077	390	558	384	138	6,547
1996/97	3,073	294	341	422	12	4,142
1997/98	1,843	209	144	219	5	2,420
1998/99	2,104	65	37	101	0	2,307
1999/00	1,361	0	49	0	0	1,410
2000/01	1,291	0	62	0	3	1,356
2001/02	1,291	0	62	0	3	1,356
2017/18	43	8	37	0	1	89
2018/19	757	332	163	188	49	1,489
2019/20	686	113	244	179	147	1,369
2020/21	468	53	132	30	240	923
2021/22	202	16	15	29	167	429
2022/23	233	20	0	17	247	517
2023/24 (to date)	45	12	0	8	45	110

Table A3.2. Wings submitted to the BASC Wing Survey, split by species.

Species	Average number of wings per year	Estimated percentage of annual harvest	Total wings 1986/87 – 2022/23*
Greylag Goose *Anser anser*	7.5	< 0.1	156
Pink-footed Goose *Anser brachyrhynchus*	10.0	0.1	210
Greater White-fronted Goose *Anser albifrons albifrons*	0.3	0.3	6
Common Goldeneye *Bucephala clangula*	1.6	0.4	33
Common Pochard *Aythya ferina*	32.4	8.8	680
Tufted Duck *Aythya fuligula*	55.1	1.1	1,157
Greater Scaup *Aythya marila*	0	0	0
Northern Shoveler *Spatula clyptea*	3.7	0.2	78
Gadwall *Mareca strepera*	4.9	0.1	103
Eurasian Wigeon *Mareca penelope*	1,100.0	2.6	23,101
Mallard *Anas platyrhynchos*	84.6	< 0.1	1,776
Northern Pintail *Anas acuta*	40.1	5.9	860
Common Teal *Anas crecca*	856.0	0.6	17,976
Eurasian Golden Plover *Pluvialis apricaria*	0.2	< 0.1	5
Eurasian Woodcock *Scolopax rusticola*	29.9	< 0.1	628
Common Snipe *Gallinago gallinago*	0.9	< 0.1	18
Jack Snipe *Lymnocryptes minimus*	0	0	0
Common Moorhen *Gallinula chloropus*	0.1	0	2

Species	Average number of wings per year	Estimated percentage of annual harvest	Total wings 1986/87 – 2022/23*
Common Coot *Fulica atra*	0	0	0

* Data from 2023/24 is excluded as the season has not finished and wing collection and identification is still ongoing.

Appendix 4: Relevant data from the PACEC 2014 report

Table A4.1. Estimates relating to waterbird shooting including participants, total harvest and area of land involved derived from the Public and Corporate Economic Consultants (PACEC) 2014 survey.

Metric	**Driven, predominantly game shooting (including duck)**	**Walked-up, predominantly game shooting (including duck)**	**Inland duck (e.g. flight ponds/ marshes) and goose shooting**	**Coastal wildfowling (duck/ goose/ wader shooting on foreshore)**
Estimates of the number of participants* (Table 21 PACEC 2014)	280,000	150,000	75,000	28,000
% of total respondents who participate in the shooting type (Table 24 PACEC 2014)	53%	30%	20%	8%
Number of "Gun days" per year (number of Guns x shooting days (Table 18 PACEC 2014)	120,000 days x 13 participants/ day 1,600,000 Gun days	100,000 days x 7 participants/ day 680,000 Gun days	38,000 days x 4 participants/ day 160,000 Gun days	28,000 days x 4 participants/ day 100,000 Gun days
Number of providers (Table 6 PACEC 2014)	23,000	21,000	16,000	4,000
Area of land affected by shooting per provider (ha) (Table 64 PACEC 2014)	850	1,350	1,400	2,600

* Estimates are not exclusive, so one Gun could participate in multiple shooting types.

Appendix 5: Prescribed Take Level (PTL) models

Comparison of PTL and PEG

In Ellis and Cameron (2022) an initial assessment of the sustainability of wildfowl harvest in the UK was made using the Potential Excess Growth (PEG) model of popharvest (Eraud *et al.* 2021). Recent literature (Johnson *et al.* 2024) suggests that the Prescribed Take Level (PTL) model in popharvest is a more desirable approach, and so we have rerun the models of Ellis and Cameron (2022) using the same data, but through a PTL model. The outputs from both the PEG and PTL models are a Sustainable Harvest Index (SHI) and a probability that the harvest is unsustainable.

Given the uncertainties in life history characteristics, we have produced “low” and “high” scenario estimates based on life history characteristics specified in the model which we have previously demonstrated (Ellis & Cameron 2022) produce the minimum and maximum estimates of SHI, but for which we are unable to accurately determine the correct formulation from existing data. It is important to note that in the PTL models an SHI less than 1 for any value of F_{obj} up to one is theoretically sustainable in an ecological sense but may not be in social sense if it prevents populations from increasing. An SHI = 1 indicates that harvest is equal to the management objective, and an SHI = 1 at F_{obj} = 1 indicates that harvest is equal to the maximum sustainable yield – such a harvest should result in a population at equilibrium at a fraction of its carrying capacity but may be unstable in the presence of other ecological pressures. Additionally, an SHI below 1 does not indicate that the harvest is definitely sustainable, rather that the level of take is below the management objective and that on its own and without any additional factors acting on the population, is unlikely to be unsustainable.

The two scenarios are generated with the following assumptions:

- Low scenario: short living rate with Rmax estimated from reported survival rates.
- High scenario: long living rate with Rmax estimated from adult mass and type.e and type.p set to determinist.

Estimates of SHI were generated based on 10,000 simulation runs per species, per scenario using the same data as Ellis and Cameron (2022).

Below, we present the outcomes of the low and high scenario estimates for SHI (Table A6.1) and the probability of unsustainable harvest (Table A6.2) using the PTL method and the current best estimates of harvest and population. This is provided to allow a direct comparison with Ellis and Cameron (2022), but it should be noted that it still only represents a snapshot and does not allow for any consideration of changes in harvest or population. Furthermore, the same assumptions and limitations discussed in Ellis and Cameron (2022) apply to these data (see also Section 5.1). This is important for Mallard, Woodcock and Greylag Goose where the proportion of reared versus wild Mallard, and the proportion of British or resident versus migratory Woodcock and Greylag Goose is uncertain.

Table A5.1. Average Sustainable Harvest Index (SHI; 95% CI) under the (A) low scenario or (B) high scenario for various desired management objectives (F_{obj}) for all species in AEWA Table 1 hunted in the UK, calculated using a Prescribed Take Level (PTL) model approach. Figures in brackets provide Confidence Intervals for estimates.

A)

Species	F_{ob} = 0.1	F_{ob} = 0.2	F_{ob} = 0.3	F_{ob} = 0.4	F_{ob} = 0.5	F_{ob} = 0.6	F_{ob} = 0.7	F_{ob} = 0.8	F_{ob} = 0.9	F_{ob} = 1
Mallard *Anas platyrhynchos*	9.37 (3.67–19.81)	4.70 (1.82–10.00)	3.12 (1.24–6.62)	2.36 (0.92–4.98)	1.89 (0.75–3.97)	1.58 (0.63–3.34)	1.34 (0.52–2.85)	1.18 (0.46–2.48)	1.05 (0.41–2.22)	0.95 (0.37–2.01)
Common Teal *Anas crecca crecca*	11.02 (4.39–23.22)	5.53 (2.18–11.8)	3.65 (1.43–7.70)	2.75 (1.07–5.77)	2.22 (0.87–4.66)	1.83 (0.71–3.89)	1.57 (0.62–3.31)	1.37 (0.54–2.88)	1.23 (0.49–2.61)	1.10 (0.43–2.34)
Eurasian Wigeon *Mareca penelope*	3.25 (1.29–6.80)	1.65 (0.65–3.51)	1.09 (0.43–2.29)	0.82 (0.32–1.74)	0.65 (0.25–1.40)	0.55 (0.22–1.15)	0.47 (0.18–0.98)	0.41 (0.16–0.86)	0.36 (0.14–0.76)	0.33 (0.13–0.68)
Gadwall *Mareca strepera strepera*	5.15 (2.02–10.87)	2.60 (1.02–5.44)	1.73 (0.68–3.60)	1.29 (0.51–2.73)	1.03 (0.41–2.19)	0.86 (0.34–1.83)	0.73 (0.29–1.53)	0.65 (0.25–1.38)	0.57 (0.22–1.21)	0.52 (0.20–1.09)
Northern Pintail *Anas acuta*	1.11 (0.43–2.36)	0.55 (0.22–1.17)	0.37 (0.14–0.79)	0.28 (0.11–0.59)	0.22 (0.09–0.47)	0.19 (0.07–0.39)	0.16 (0.06–0.33)	0.14 (0.06–0.29)	0.12 (0.05–0.26)	0.11 (0.04–0.23)
Northern Shoveler *Spatula clyptea*	3.25 (1.29–6.79)	1.64 (0.64–3.44)	1.08 (0.42–2.31)	0.82 (0.32–1.73)	0.65 (0.26–1.39)	0.55 (0.21–1.16)	0.46 (0.18–0.98)	0.41 (0.16–0.85)	0.36 (0.14–0.76)	0.33 (0.13–0.7)
Tufted Duck *Aythya fuligula*	1.13 (0.45–2.41)	0.57 (0.23–1.19)	0.38 (0.15–0.80)	0.29 (0.11–0.61)	0.23 (0.09–0.49)	0.19 (0.07–0.40)	0.16 (0.06–0.34)	0.14 (0.06–0.31)	0.13 (0.05–0.27)	0.11 (0.05–0.24)
Common Pochard *Aythya ferina*	0.42 (0.16–0.88)	0.21 (0.08–0.45)	0.14 (0.06–0.29)	0.11 (0.04–0.22)	0.08 (0.03–0.18)	0.07 (0.03–0.15)	0.06 (0.02–0.13)	0.05 (0.02–0.11)	0.05 (0.02–0.1)	0.04 (0.02–0.09)

Species	F_{ob} = 0.1	F_{ob} = 0.2	F_{ob} = 0.3	F_{ob} = 0.4	F_{ob} = 0.5	F_{ob} = 0.6	F_{ob} = 0.7	F_{ob} = 0.8	F_{ob} = 0.9	F_{ob} = 1
Common Goldeneye *Bucephala clangula cla**n**gula*	0.71 (0.28–1.50)	0.36 (0.14–0.75)	0.24 (0.09–0.50)	0.18 (0.07–0.37)	0.14 (0.06–0.30)	0.12 (0.05–0.25)	0.10 (0.04–0.21)	0.09 (0.04–0.19)	0.08 (0.03–0.17)	0.07 (0.03–0.15)
Pink-footed Goose *Anser brachyrhynchus*	1.49 (0.58–3.16)	0.74 (0.29–1.56)	0.50 (0.20–1.05)	0.37 (0.15–0.79)	0.30 (0.12–0.64)	0.25 (0.10–0.52)	0.21 (0.08–0.45)	0.19 (0.07–0.39)	0.17 (0.07–0.35)	0.15 (0.06–0.31)
Icelandic Greylag Goose *Anser anser anser*	18.7 (7.46–39.1)	9.50 (3.67–19.79)	6.31 (2.47–13.31)	4.70 (1.88–9.92)	3.76 (1.48–7.8)	3.11 (1.22–6.65)	2.67 (1.05–5.56)	2.34 (0.9–4.94)	2.09 (0.82–4.43)	1.88 (0.74–4.00)
British Greylag Goose *Anser anser anser*	19.2 (7.55–40.31)	9.66 (3.82–20.30)	6.40 (2.49–13.50)	4.84 (1.90–10.26)	3.89 (1.54–8.13)	3.22 (1.27–6.72)	2.76 (1.09–5.78)	2.41 (0.94–5.12)	2.15 (0.84–4.52)	1.92 (0.76–4.00)
Common Snipe *Gallinago gallinago gallinago*	3.09 (1.21–6.56)	1.54 (0.61–3.21)	1.02 (0.41–2.16)	0.77 (0.30–1.61)	0.61 (0.24–1.28)	0.51 (0.20–1.08)	0.44 (0.17–0.92)	0.38 (0.15–0.81)	0.34 (0.13–0.72)	0.31 (0.12–0.65)
Eurasian Golden Plover *Pluvialis apricaria altifrons*	0.07 (0.03–0.15)	0.04 (0.01-0.07)	0.02 (0.01–0.05)	0.02 (0.01–0.04)	0.01 (0.01–0.03)	0.01 (0.01–0.03)	0.01 (0–0.02)	0.01 (0–0.02)	0.01 (0–0.02)	0.01 (0–0.01)
Resident Eurasian Woodcock *Scolopax rusticola*	4.74 (1.87–9.92)	2.34 (0.92–4.92)	1.59 (0.62–3.38)	1.18 (0.47–2.45)	0.95 (0.37–2.01)	0.79 (0.31–1.66)	0.67 (0.26–1.43)	0.59 (0.24–1.25)	0.53 (0.21–1.13)	0.48 (0.19–1.01)
Migratory Eurasian Woodcock *Scolopax rusticola*	4.24 (1.67–8.98)	2.10 (0.84–4.48)	1.40 (0.55–2.97)	1.06 (0.42–2.22)	0.84 (0.33–1.8)	0.70 (0.28–1.48)	0.60 (0.24–1.27)	0.53 (0.21–1.09)	0.47 (0.18–0.99)	0.42 (0.17–0.89)

B)

Species	$F_{ob} = 0.1$	$F_{ob} = 0.2$	$F_{ob} = 0.3$	$F_{ob} = 0.4$	$F_{ob} = 0.5$	$F_{ob} = 0.6$	$F_{ob} = 0.7$	$F_{ob} = 0.8$	$F_{ob} = 0.9$	$F_{ob} = 1$
Mallard *Anas platyrhynchos*	14.2.0 (5.55–30.00)	7.10 (2.76–14.77)	4.75 (1.84–9.92)	3.54 (1.40–7.44)	2.82 (1.10–5.95)	2.36 (0.91–4.96)	2.04 (0.80–4.27)	1.77 (0.69–3.75)	1.59 (0.61–3.36)	1.41 (0.56–2.99)
Common Teal *Anas crecca crecca*	14.38 (5.64–30.14)	7.20 (2.77–15.1)	4.82 (1.88–10.15)	3.58 (1.42–7.54)	2.88 (1.12–6.04)	2.39 (0.95–5.04)	2.06 (0.80–4.36)	1.79 (0.70–3.76)	1.61 (0.62–3.42)	1.44 (0.57–3.02)
Eurasian Wigeon *Mareca penelope*	4.53 (1.77–9.66)	2.27 (0.90–4.74)	1.52 (0.60–3.21)	1.15 (0.45–2.39)	0.92 (0.36–1.94)	0.76 (0.30–1.60)	0.65 (0.25–1.37)	0.57 (0.23–1.21)	0.51 (0.20–1.07)	0.46 (0.18–0.95)
Gadwall *Mareca strepera strepera*	7.47 (2.93–15.85)	3.74 (1.49–7.81)	2.52 (0.99–5.31)	1.88 (0.75–3.99)	1.52 (0.60–3.17)	1.26 (0.49–2.62)	1.08 (0.43–2.28)	0.95 (0.38–2.01)	0.84 (0.34–1.77)	0.76 (0.30–1.60)
Northern Pintail *Anas acuta*	1.63 (0.64–3.43)	0.82 (0.32–1.71)	0.54 (0.21–1.15)	0.41 (0.16–0.86)	0.33 (0.13–0.70)	0.27 (0.11–0.57)	0.23 (0.09–0.50)	0.21 (0.08–0.43)	0.18 (0.07–0.39)	0.16 (0.07–0.35)
Northern Shoveler *Spatula clyptea*	4.60 (1.80–9.60)	2.32 (0.90–4.86)	1.53 (0.61–3.21)	1.14 (0.44–2.42)	0.92 (0.36–1.96)	0.76 (0.30–1.59)	0.66 (0.26–1.37)	0.57 (0.23–1.20)	0.51 (0.20–1.08)	0.46 (0.18–0.97)
Tufted Duck *Aythya fuligula*	1.68 (0.65–3.53)	0.84 (0.33–1.75)	0.56 (0.22–1.18)	0.42 (0.17–0.89)	0.33 (0.13–0.71)	0.28 (0.11–0.59)	0.24 (0.09–0.50)	0.21 (0.08–0.44)	0.19 (0.07–0.40)	0.17 (0.07–0.35)
Common Pochard *Aythya ferina*	0.62 (0.25–1.32)	0.31 (0.12–0.67)	0.21 (0.08–0.44)	0.16 (0.06–0.33)	0.13 (0.05–0.26)	0.10 (0.04–0.22)	0.09 (0.04–0.19)	0.08 (0.03–0.16)	0.07 (0.03–0.14)	0.06 (0.02–0.13)
Common Goldeneye *Bucephala clangula clangula*	1.04 (0.41–2.17)	0.52 (0.20–1.07)	0.34 (0.13–0.73)	0.26 (0.10–0.55)	0.21 (0.08–0.44)	0.17 (0.07–0.36)	0.15 (0.06–0.32)	0.13 (0.05–0.27)	0.12 (0.05–0.25)	0.10 (0.04–0.22)

Species	$F_{ob} = 0.1$	$F_{ob} = 0.2$	$F_{ob} = 0.3$	$F_{ob} = 0.4$	$F_{ob} = 0.5$	$F_{ob} = 0.6$	$F_{ob} = 0.7$	$F_{ob} = 0.8$	$F_{ob} = 0.9$	$F_{ob} = 1$
Pink-footed Goose *Anser brachyrhynchus*	1.92 (0.75–4.07)	0.96 (0.38–2.01)	0.64 (0.25–1.35)	0.48 (0.19–1.01)	0.38 (0.15–0.80)	0.32 (0.13–0.67)	0.27 (0.11–0.58)	0.24 (0.10–0.50)	0.21 (0.08–0.45)	0.19 (0.07–0.40)
Icelandic Greylag Goose *Anser anser anser*	24.98 (9.77–53.15)	12.31 (4.84–25.84)	8.22 (3.27–17.12)	6.19 (2.43–13.21)	5.01 (1.94–10.62)	4.16 (1.64–8.69)	3.54 (1.40–7.47)	3.13 (1.24–6.57)	2.75 (1.07–5.83)	2.48 (0.97–5.25)
British Greylag Goose *Anser anser anser*	25.43 (9.97–53.53)	12.69 (4.93–26.81)	8.48 (3.30–17.98)	6.34 (2.50–13.58)	5.12 (1.99–10.80)	4.22 (1.68–8.80)	3.61 (1.41–7.58)	3.18 (1.25–6.67)	2.83 (1.11–5.93)	2.53 (1–5.32)
Common Snipe *Gallinago gallinago gallinago*	4.54 (1.80–9.57)	2.27 (0.89–4.83)	1.51 (0.59–3.22)	1.14 (0.44–2.43)	0.91 (0.36–1.93)	0.76 (0.30–1.60)	0.64 (0.25–1.35)	0.56 (0.22–1.19)	0.51 (0.20–1.08)	0.46 (0.18–0.96)
Eurasian Golden Plover *Pluvialis apricaria altifrons*	0.09 (0.04–0.19)	0.05 (0.02–0.10)	0.03 (0.01–0.06)	0.02 (0.01–0.05)	0.02 (0.01–0.04)	0.02 (0.01–0.03)	0.01 (0.01–0.03)	0.01 (0.01–0.02)	0.01 (0–0.02)	0.01 (0–0.02)
Resident Eurasian Woodcock *Scolopax rusticola*	6.92 (2.73–14.52)	3.49 (1.37–7.43)	2.33 (0.92–4.9)	1.75 (0.69–3.68)	1.4 (0.55–2.94)	1.17 (0.46–2.47)	0.99 (0.4–2.08)	0.88 (0.35–1.84)	0.78 (0.31–1.64)	0.7 (0.27–1.47)
Migratory Eurasian Woodcock *Scolopax rusticola*	6.22 (2.44–13.3)	3.14 (1.22–6.66)	2.07 (0.81–4.4)	1.55 (0.61–3.27)	1.25 (0.49–2.64)	1.04 (0.42–2.2)	0.89 (0.35–1.89)	0.78 (0.31–1.64)	0.7 (0.27–1.46)	0.62 (0.25–1.31)

Table A5.2. Probability of unsustainable harvest under the (A) low scenario or (B) high scenario for various desired management objectives (F_{obj}) for all species in AEWA Table 1 hunted in the UK.

A)

Species	F_{obj} = 0.1	F_{obj} = 0.2	F_{obj} = 0.3	F_{obj} = 0.4	F_{obj} = 0.5	F_{obj} = 0.6	F_{obj} = 0.7	F_{obj} = 0.8	F_{obj} = 0.9	F_{obj} = 1.0
Mallard *Anas platyrhynchos*	1	1	0.99	0.96	0.88	0.79	0.68	0.57	0.46	0.36
Common Teal *Anas crecca crecca*	1	1	1	0.99	0.94	0.87	0.79	0.69	0.60	0.50
Eurasian Wigeon *Mareca penelope*	0.99	0.81	0.49	0.26	0.13	0.06	0.02	0.01	0	0
Gadwall *Mareca strepera strepera*	1	0.98	0.84	0.64	0.44	0.29	0.19	0.13	0.08	0.05
Northern Pintail *Anas acuta*	0.51	0.06	0	0	0	0	0	0	0	0
Northern Shoveler *Spatula clyptea*	0.99	0.81	0.48	0.26	0.13	0.06	0.02	0.01	0	0
Tufted Duck *Aythya fuligula*	0.53	0.08	0.01	0	0	0	0	0	0	0
Common Pochard *Aythya ferina*	0.01	0	0	0	0	0	0	0	0	0
Common Goldeneye *Bucephala clangula clangula*	0.17	0	0	0	0	0	0	0	0	0
Pink-footed Goose *Anser brachyrhynchus*	0.75	0.20	0.04	0.01	0	0	0	0	0	0
Icelandic Greylag Goose *Anser anser anser*	1	1	1	1	1	0.99	0.98	0.95	0.92	0.88

Species	**F_{obj} = 0.1**	**F_{obj} = 0.2**	**F_{obj} = 0.3**	**F_{obj} = 0.4**	**F_{obj} = 0.5**	**F_{obj} = 0.6**	**F_{obj} = 0.7**	**F_{obj} = 0.8**	**F_{obj} = 0.9**	**F_{obj} = 1.0**
British Greylag Goose *Anser anser anser*	1	1	1	1	1	0.99	0.98	0.96	0.93	0.89
Common Snipe *Gallinago gallinago gallinago*	0.99	0.77	0.43	0.22	0.10	0.04	0.01	0.01	0	0
Eurasian Golden Plover *Pluvialis apricaria altifrons*	0	0	0	0	0	0	0	0	0	0
Resident Eurasian Woodcock *Scolopax rusticola*	1	0.96	0.80	0.57	0.37	0.23	0.14	0.09	0.05	0.03
Migratory Eurasian Woodcock *Scolopax rusticola*	1	0.93	0.71	0.46	0.28	0.16	0.09	0.05	0.02	0.01

B)

Species	**F_{obj} = 0.1**	**F_{obj} = 0.2**	**F_{obj} = 0.3**	**F_{obj} = 0.4**	**F_{obj} = 0.5**	**F_{obj} = 0.6**	**F_{obj} = 0.7**	**F_{obj} = 0.8**	**F_{obj} = 0.9**	**F_{obj} = 1.0**
Mallard *Anas platyrhynchos*	1	1	1	1	0.99	0.96	0.91	0.85	0.79	0.72
Common Teal *Anas crecca crecca*	1	1	1	1	0.99	0.96	0.91	0.86	0.79	0.73
Eurasian Wigeon *Mareca penelope*	1	0.95	0.77	0.54	0.34	0.21	0.13	0.08	0.04	0.02
Gadwall *Mareca strepera strepera*	1	1	0.97	0.88	0.76	0.63	0.48	0.37	0.27	0.21
Northern Pintail *Anas acuta*	0.81	0.25	0.06	0.01	0	0	0	0	0	0

Species	F_{obj} = 0.1	F_{obj} = 0.2	F_{obj} = 0.3	F_{obj} = 0.4	F_{obj} = 0.5	F_{obj} = 0.6	F_{obj} = 0.7	F_{obj} = 0.8	F_{obj} = 0.9	F_{obj} = 1.0
Northern Shoveler *Spatula clyptea*	1	0.95	0.77	0.54	0.34	0.21	0.13	0.08	0.04	0.02
Tufted Duck *Aythya fuligula*	0.83	0.27	0.07	0.01	0	0	0	0	0	0
Common Pochard *Aythya ferina*	0.11	0	0	0	0	0	0	0	0	0
Common Goldeneye *Bucephala clangula clangula*	0.44	0.04	0	0	0	0	0	0	0	0
Pink-footed Goose *Anser brachyrhynchus*	0.89	0.38	0.12	0.03	0	0	0	0	0	0
Icelandic Greylag Goose *Anser anser anser*	1	1	1	1	1	1	1	1	0.99	0.97
British Greylag Goose *Anser anser anser*	1	1	1	1	1	1	1	1	0.99	0.98
Common Snipe *Gallinago gallinago gallinago*	1	0.95	0.75	0.53	0.33	0.21	0.12	0.07	0.04	0.02
Eurasian Golden Plover *Pluvialis apricaria altifrons*	0	0	0	0	0	0	0	0	0	0
Resident Eurasian Woodcock *Scolopax rusticola*	1	1	0.96	0.85	0.71	0.55	0.41	0.31	0.23	0.17
Migratory Eurasian Woodcock *Scolopax rusticola*	1	0.99	0.92	0.78	0.62	0.45	0.32	0.23	0.16	0.11

Accounting for harvest and population uncertainty

The models (Tables A5.1 and A5.2) provide a snapshot assessment of the sustainability of harvest given our best estimates of harvest and population size. However, these estimates are imprecise and will change annually. To account for this, we also look at how SHI is predicted to change across a range of harvest and population estimates.

Current population estimates for waterbirds are based on five-year mean peaks from 2012/13 to 2016/17 (Frost *et al.* 2019). This means that for species that exhibit year-to-year variations in size, the reported population estimate inherently also contains substantial variation. In addition, for some species it is common to use a multiplier to scale up counts to estimate birds which may be missed by normal count methods. There can be substantial variation in these multipliers when assessed in the field, and combination with the variation in counts over five years produces a wide range within which the "true" population estimate likely falls. The methodology for scaling up site counts to population estimates is well explained and robust (Frost *et al.* 2019; Musgrove *et al.* 2011), but the potential for uncertainty in the counts is not always very clear. Although it is a simplification of the methods, we attempt to explore that uncertainty by looking at the estimated percentage difference between (Table A5.3):

- the mean WeBS index for a species, multiplied by the average multiplication factor (where used)
- the maximum WeBS index for a species over the same period, multiplied by the maximum multiplication factor (where used)

Common Snipe and Woodcock population estimates are not made based on the same methodology but are included for comparison.

Table A5.3. An exploration of the uncertainty in population estimates according to how corrections are made to count data based on multiplication factors and their range.

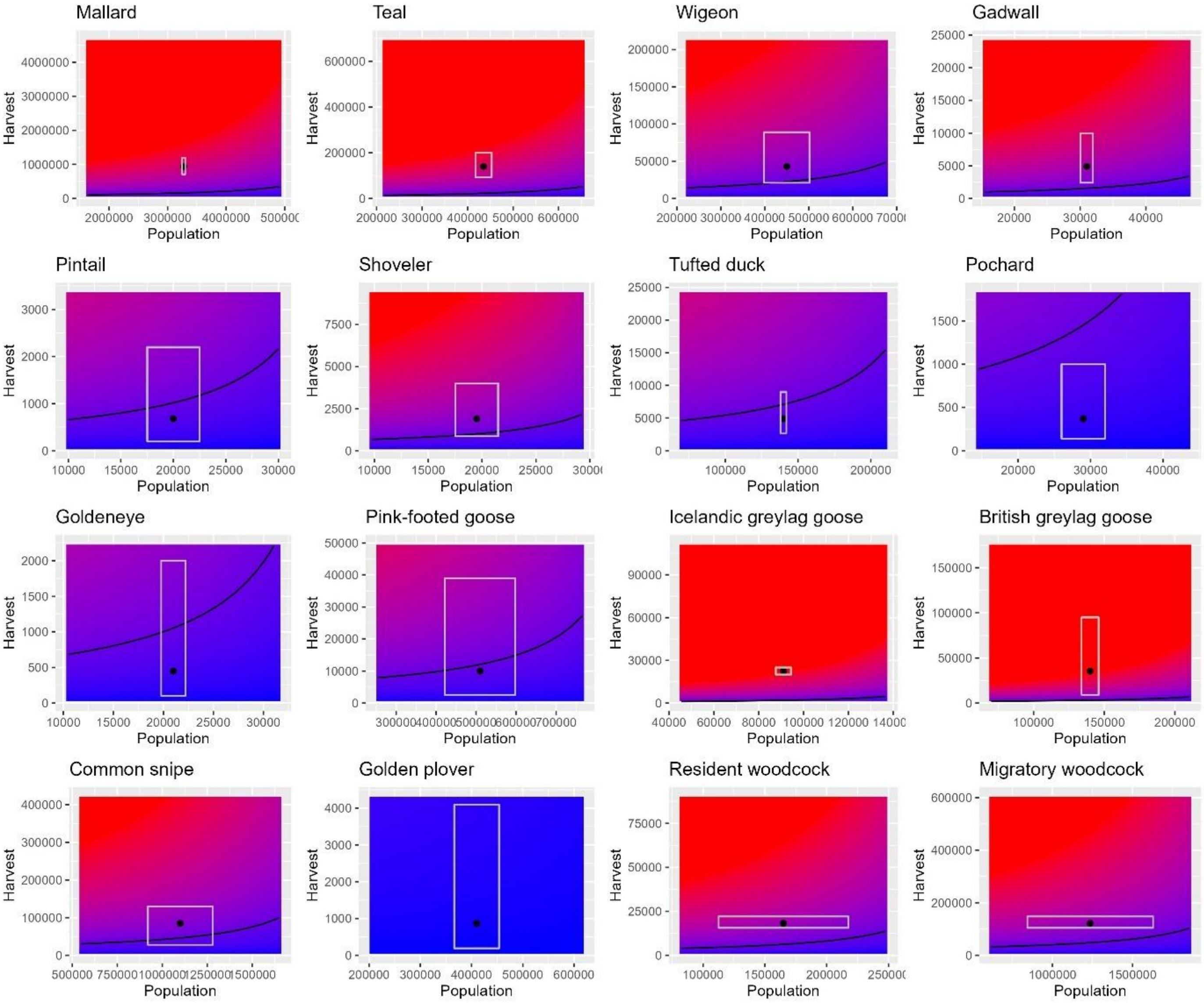
Species	Multiplication factor, mean	Multiplication factor, range	WeBS Index (2012/13-2016/17), mean	WeBS Index (2012/13-2016/17), range	Estimated percentage variation	Reliability score*
Mallard *Anas platyrhynchos*	4.00	1.43–4.00	119.0	115–128	7.6	3
Common Teal *Anas crecca crecca*	1.21	1.03–1.40	120.2	115–128	6.5	2
Eurasian Wigeon *Mareca penelope*	1.05	1.00–1.15	116.0	101–141	21.6	1
Gadwall *Mareca strepera strepera*	1.18	1.06–1.37	103.6	98–108	4.2	1
Northern Pintail *Anas acuta*	1.00	1.00–1.01	95.0	84–117	27.3	1
Northern Shoveler *Spatula clyptea*	1.16	1.01–1.27	85.2	73–99	16.2	1
Tufted Duck *Aythya fuligula*	1.48	1.17–2.02	116.0	112–118	1.7	1
Common Pochard *Aythya ferina*	1.22	1.10–1.38	143.4	127–169	17.9	1
Common Goldeneye *Bucephala clangula clangula*	1.26	1.02–1.60	122.4	114–138	12.7	2
Pink-footed Goose *Anser brachyrhynchus*	-	-	92.2	76–117	53.6	1
Greylag Goose *Anser anser anser*	1.96	1.06–3.10	93.8	91–98	4.5	2
Common Snipe *Gallinago gallinago gallinago*	-	-	119.4	96–151	40.8	3
Eurasian Golden Plover *Pluvialis apricaria altifrons*	-	-	123.0	103–136	18.2	2
Eurasian Woodcock *Scolopax rusticola*	-	-	92.0	64–151	204.8	3

* The Reliability score for each multiplication factor is reported from Woodward (2020). 1 – an estimate based on good-quality counts of a large proportion of the individuals involved; 2 – an estimate which is heavily based on count data but for which these data have had to be extrapolated to a large degree; 3 – an estimate which is not strongly based on actual count data and/or for which large assumptions have had to be made.

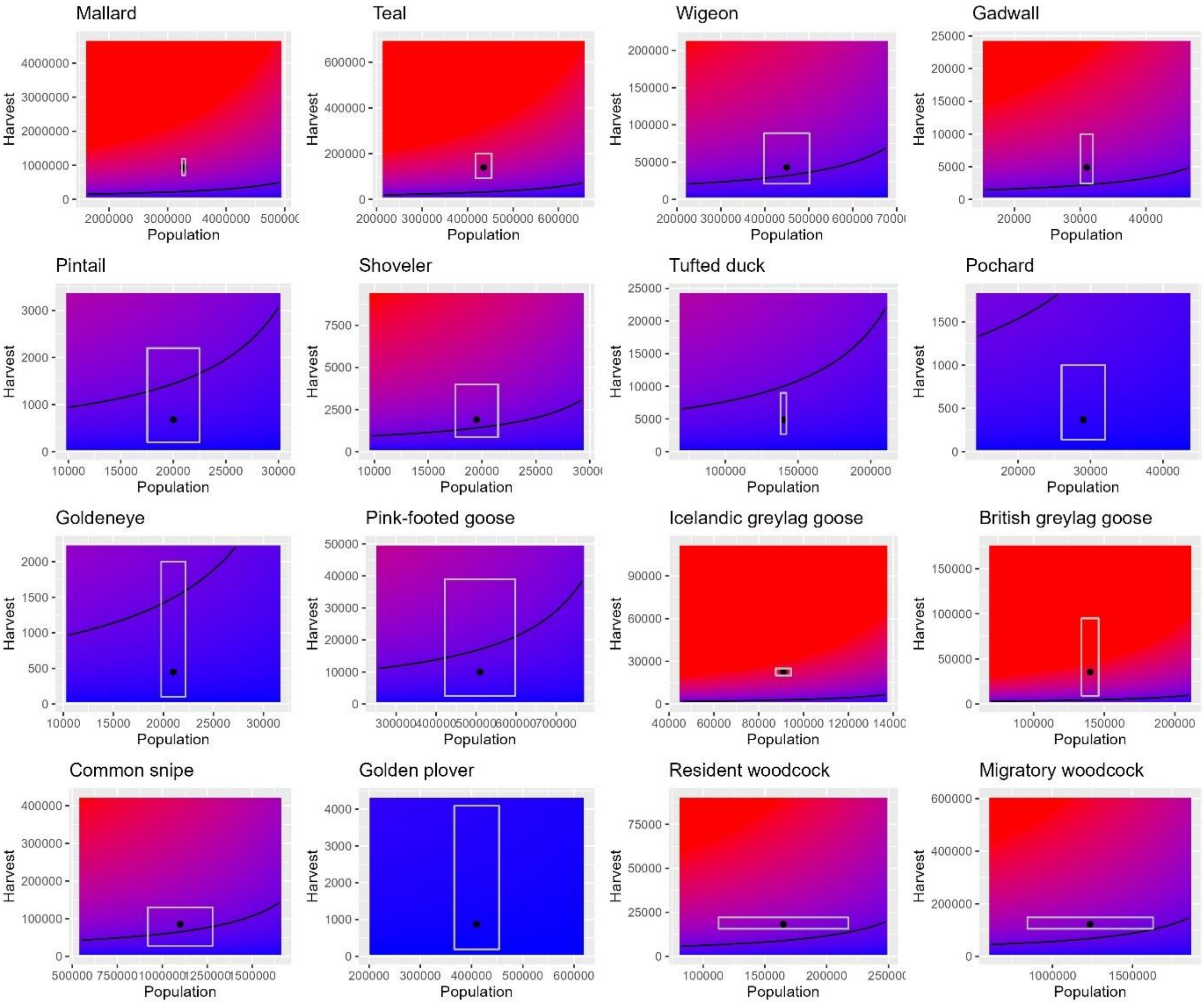
The maximum estimated percentage difference over the five-year period (excluding Woodcock for which survey methodology changed) was 53.6% for Pink-footed Goose (Table A5.3). We therefore will assume that ± 50% for all species population estimates is a reasonable degree of variation over which to model the sustainability of harvests. For harvest we model the effect from zero to five times the current best estimate of harvest levels. This likely largely overestimates the variability in harvest but provides a good overview of the “head room” available above current best estimates of annual harvests. For both harvest and population estimates we present confidence intervals for the current best estimates as described in Ellis and Cameron (2022).

The range in population and harvest is achieved by setting the population and harvest estimates equal to the stated range and allowing popharvest to draw from a uniform distribution across that range. The current best estimates for population and harvest are marked on each figure with a dot (Figure A5.1), along with a box marking the confidence intervals for harvest and population estimates. We present graphs for values of F_{obj} equal to 0.25, 0.5, 0.75 and 1.0.

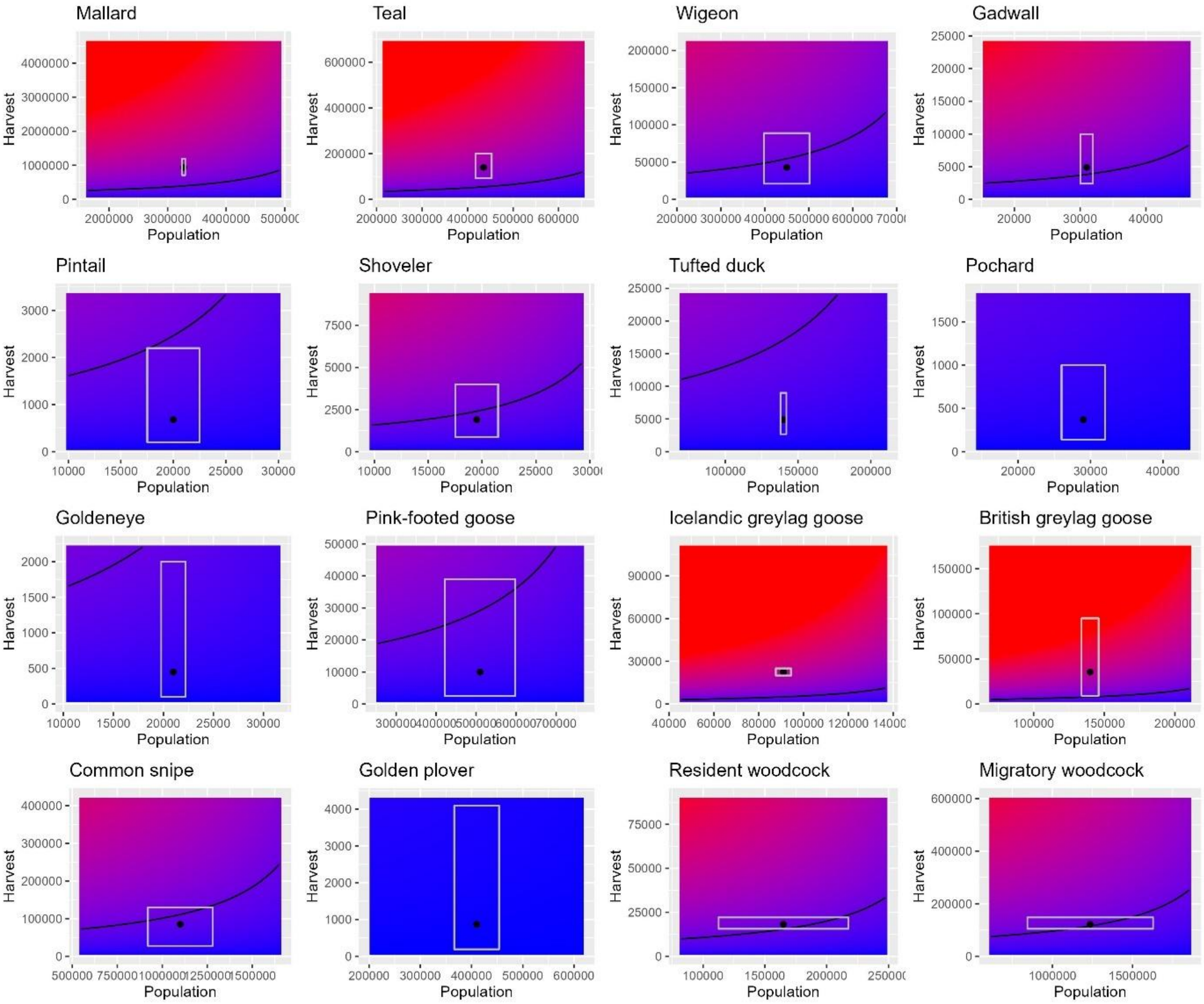
A) F_{obj} = 0.25



B) F_{obj} = 0.5


Mallard
Harvest
Population
0
1000000
2000000
3000000
4000000
2000000 3000000 4000000 5000000
Teal
Harvest
Population
0
200000
400000
600000
200000 300000 400000 500000 600000
Wigeon
Harvest
Population
0
50000
100000
150000
200000
200000 300000 400000 500000 600000 700000
Gadwall
Harvest
Population
0
5000
10000
15000
20000
25000
20000 30000 40000
Pintail
Harvest
Population
0
1000
2000
3000
10000 15000 20000 25000 30000
Shoveler
Harvest
Population
0
2500
5000
7500
10000 15000 20000 25000 30000
Tufted duck
Harvest
Population
0
5000
10000
15000
20000
25000
100000 150000 200000
Pochard
Harvest
Population
0
500
1000
1500
20000 30000 40000
Goldeneye
Harvest
Population
0
500
1000
1500
2000
10000 15000 20000 25000 30000
Pink-footed goose
Harvest
Population
0
10000
20000
30000
40000
50000
300000 400000 500000 600000 700000
Icelandic greylag goose
Harvest
Population
0
30000
60000
90000
40000 60000 80000 100000 120000 140000
British greylag goose
Harvest
Population
0
50000
100000
150000
100000 150000 200000
Common snipe
Harvest
Population
0
100000
200000
300000
400000
500000 750000 1000000 1250000 1500000
Golden plover
Harvest
Population
0
1000
2000
3000
4000
200000 300000 400000 500000 600000
Resident woodcock
Harvest
Population
0
25000
50000
75000
100000 150000 200000 250000
Migratory woodcock
Harvest
Population
0
200000
400000
600000
1000000 1500000

C) F_{obj} = 0.75


Mallard
Teal
Wigeon
Gadwall
Pintail
Shoveler
Tufted duck
Pochard
Goldeneye
Pink-footed goose
Icelandic greylag goose
British greylag goose
Common snipe
Golden plover
Resident woodcock
Migratory woodcock
Harvest
Population

D) F_{obj} = 1

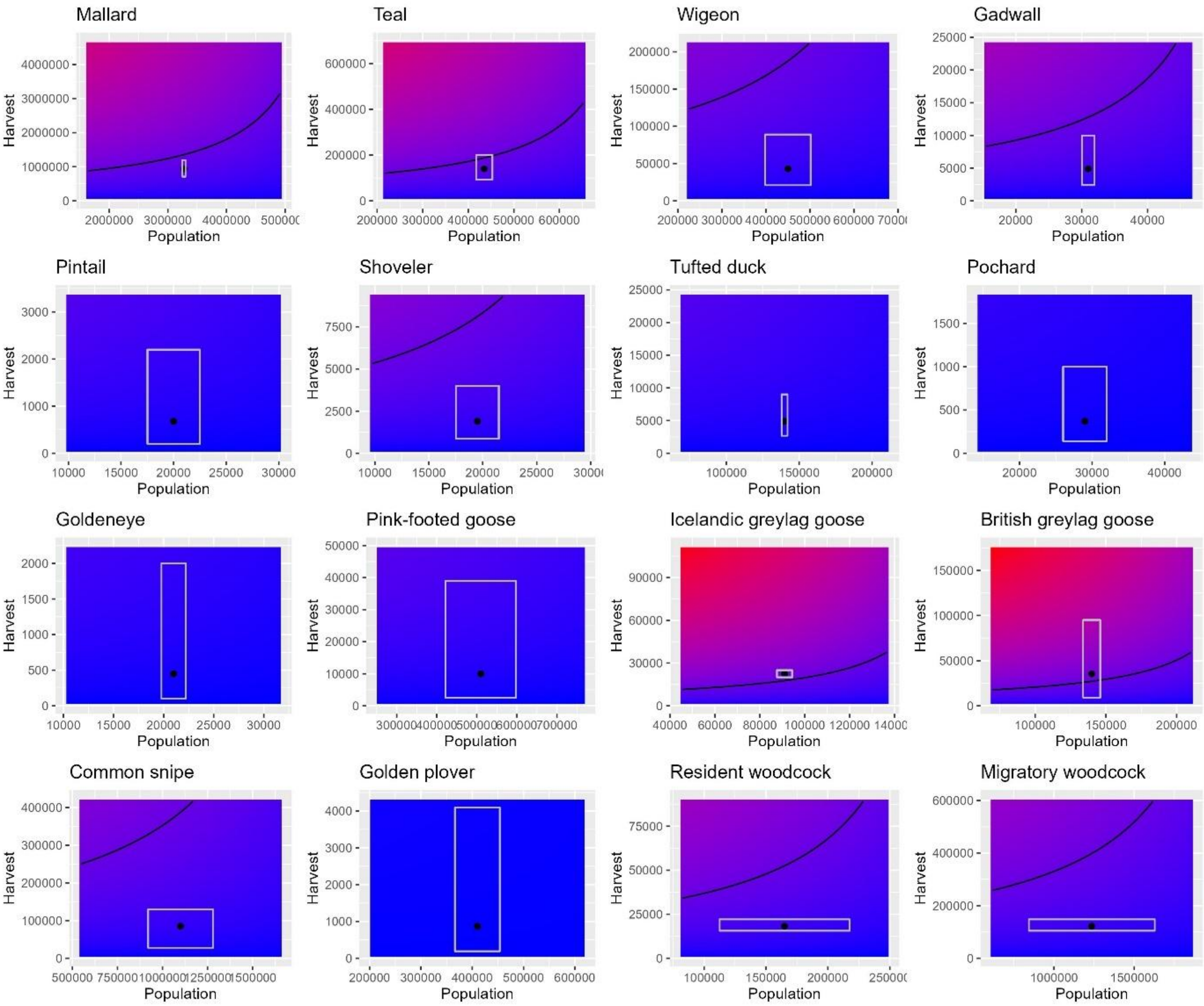


Figure A5.1. Estimated Sustainable Harvest Index (SHI) for Table 1 species across a range of harvest and population estimates, averaged across high and low scenarios and A) F_{obj}=0.25, B) F_{obj} = 0.5, C) F_{Obj} = 0.75, D) F_{obj} = 1 using the Prescribed Take Level (PTL) approach. SHI < 1 is in blue, SHI > 1 is in red and SHI = 1 is marked with a black line. The current best estimates for harvest and population sizes are marked with a black dot, and their confidence intervals with a grey box. For species with a dot and box lying above the black line and in a red region, current harvests exceed the management objective and may not be sustainable.

Appendix 6: Matrix-based population models

We were asked to consider other modelling approaches to Ellis and Cameron (2022) to examine the sustainability of waterbird hunting. There are many alternative modelling approaches that could be taken, but commonly stage-based discrete models or matrix models have been well utilised to model exploited game birds (Mills *et al.* 1999) and waterfowl (Flint *et al.* 1998). In Ellis and Cameron (2022), the sustainability is assessed by asking whether the harvest (mean±CI) exceeds half the maximum annual productivity as a snapshot in time. In a matrix model we model the population growth rate over time and ask if the population growth trajectory is affected by the harvest such that it declines or results in population extinction. One can then ask what the likelihood of population decline, or extinction is as a result of the harvesting.

Otherwise, the two modelling approaches, demographic invariant models in ‘popharvest’ (DIM; Ellis & Cameron 2022) and matrix models are actually very similar as they are dependent on the same data for parametrisation and ask the same questions. Before we look at a matrix model let us examine our expectations: to run a matrix model of a waterbird we require information on breeding productivity, juvenile survival, adult survival and harvest rates or harvest bag size – similar but certainly more data that what is required to run a DIM model. For a matrix model we can simplify it by running a female only model, ignoring males, and ask what will happen if we start the population at the current estimated size and apply the current estimated harvest? If the DIM models in Ellis and Cameron (2022) are approximately correct, then we would expect some congruence between the two approaches. If we take Teal as an example where for a species with known short life histories, breeding at 1 year and average lifespan of three years with mass estimated adult survival the DIM model is 50:50 on whether the Sustainability Harvest Index is less than 1 (Figure A6.1; Ellis & Cameron 2022). Our expectation of a matrix model approach then would be that the modelled population should be relatively stable with as many stochastic models runs resulting in increased population growth as decreasing population growth.

In a first model this is exactly what we asked. We created a two stage-matrix model of Teal. We built productivity into the model based on 0.94 hatched broods per female and 3.74 ducklings per hatched brood. This halves to 1.87 female ducklings per hatched brood (Čehovská *et al.* 2022). This is high nest success but is close to the PEG approach that examines “maximum productivity” (Ellis & Cameron 2022). For adult female annual survival, we reviewed four studies, and we selected 0.445 based on studies which estimated Teal survival in Europe and on female Green-winged Teal in North America where 0.445 was within a wider range of observed values (0.44–0.49). Of importance, these ringing studies-based survival estimates include hunting mortality already – so in the first instance we do not need to explore the addition of hunting mortality as it is already included in the survival estimate. Given that mass estimated survival of Teal is much higher, over 0.6 – this represents a significant decline in annual survival from hunting. Juvenile survival was taken from a range of estimates based on Thompson *et al.* (2022). As we can see in Figure A7.1 the median population growth prediction is for moderate positive population growth with slightly more stochastic population runs predicting population growth than population decline. Despite the more realistic representation of Teal life history, and much more parameterisation, the PVA model returns a similar result to the PEG model but with slightly more probability than not that harvest the population will grow or stabilise under survival rates that include hunting mortality. A fifty-year extinction risk – a population size of zero – is within the 95% confidence intervals of this population model – but is highly unlikely.

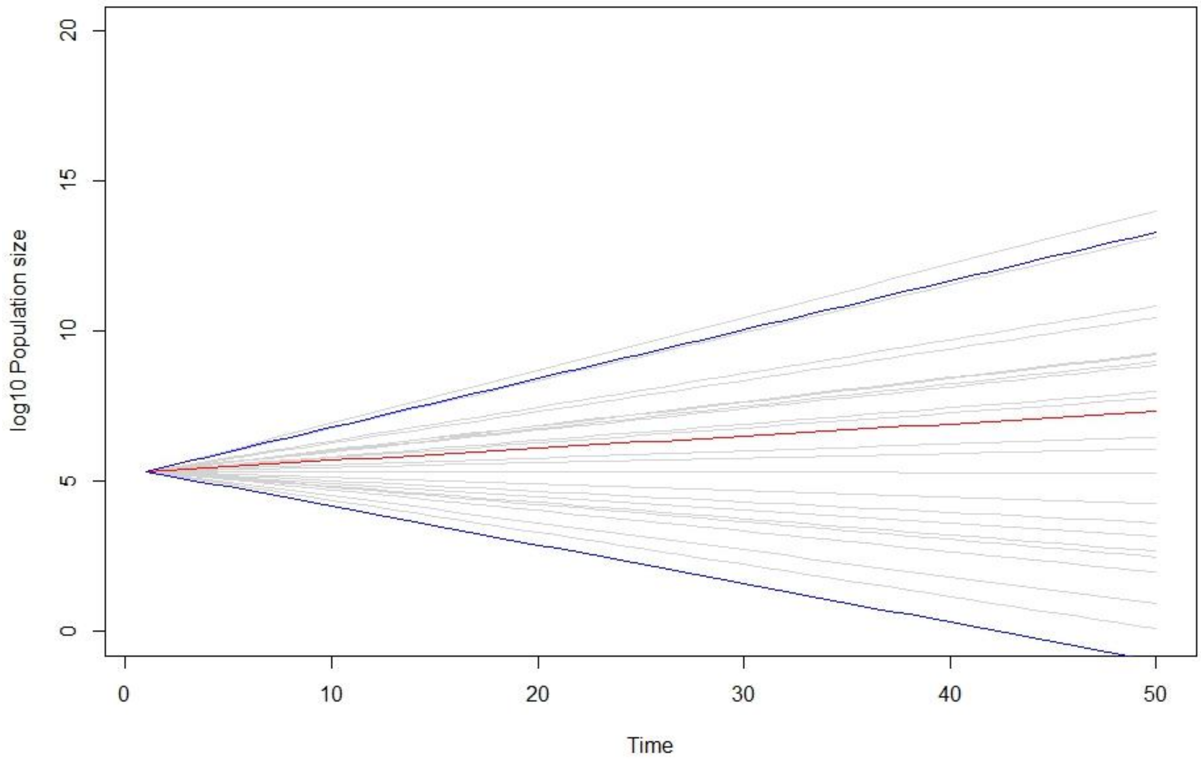


Figure A6.1. Median (female) Common Teal population over time (red) and 5th and 95th percentiles (blue) from 10,000 stochastic simulations, with some example trajectories (grey). Model uses transition probabilities between life stages (egg and adult) multiplied by the current population to calculate the next years population. Adult (female) survival probability includes hunting mortality, based on estimate from Devineau *et al.* (2010) (chosen from a uniform distribution between 0.26–0.69). Survival of egg to adult based on the probability of egg to juvenile survival and juvenile to adult survival (including hunting mortality, chosen from a uniform distribution between 0.18–0.56 from Thompson *et al.* 2022). Egg to juvenile survival (0.37) calculated as probability of nest survival (0.94 from Čehovská *et al.* 2022) multiplied by number of female ducklings per hatched brood (3.7/2 from Čehovská *et al.* 2022) and divided by the expected number of (female) eggs laid (BTO website; clutch size 8–11, 9.5 used, assume half would be female).

While this is already a satisfying result there are several limitations of this model as it does not explicitly include estimates of harvest mortality in the United Kingdom (Aebischer 2019), and it does not include other realities such as the increased vulnerability of juvenile birds to hunting mortality. In the next model these are integrated by re-estimating adult and juvenile female annual survival, by setting the proportion (p) of shot birds that are juveniles to 0.6 and allowing each annual harvest rate to draw from the range estimated by Aebischer (2019) for 2016 of 45,000–100,000 females per year.

To estimate the annual survival of adult females we looked to the DIM approach taken in Ellis and Cameron (2022), where a body mass of 340 g for a species with a short life history result in an adult female survival estimate from 'popharvest' of 0.607. The estimate of annual adult female survival rate of Teal at 0.607 is reasonable when considered alongside the maximum recorded longevity of the species of 18 years, falling within the distribution of other birds of similar body size and life history, depicted in a plot of longevity against annual adult survival for a range of UK bird species in (What makes for a long life?).

Switching to a model with annually variable fixed hunting mortality based on the estimated bag records for Teal in the United Kingdom, as opposed to have a proportional effect on survival as we used in model 1, results in an entirely different outcome (Figure A7.3). Despite realistic hunting mortalities, including increased vulnerability to hunting of juveniles over adults, the second model suggests that Teal populations should be increasing. The fifty-year extinction risk of the second model is zero, such that extinction is not within the 95% confidence intervals of the model runs. This result suggests that the PEG/PTL modelling approach of Ellis and Cameron (2022) is much more conservative that a more realistic PVA modelling approach. A more conservative approach is warranted when modelled systems are data poor and the biological response of the population to harvest is poorly understood. In both the first and second PVA model – we have taken a maximum productivity approach as was used in the PEG/PTL models, but PVA models also allow us to examine the role of productivity in the sensitivity of waterbird population growth to harvest. In a final model we explore the role of nesting success, specifically a range of studies that find that nest survival varies between studies and years between zero and 94%, where here we have chosen to explore the role of nest survival values between 40 and 70% (Figure A7.4).

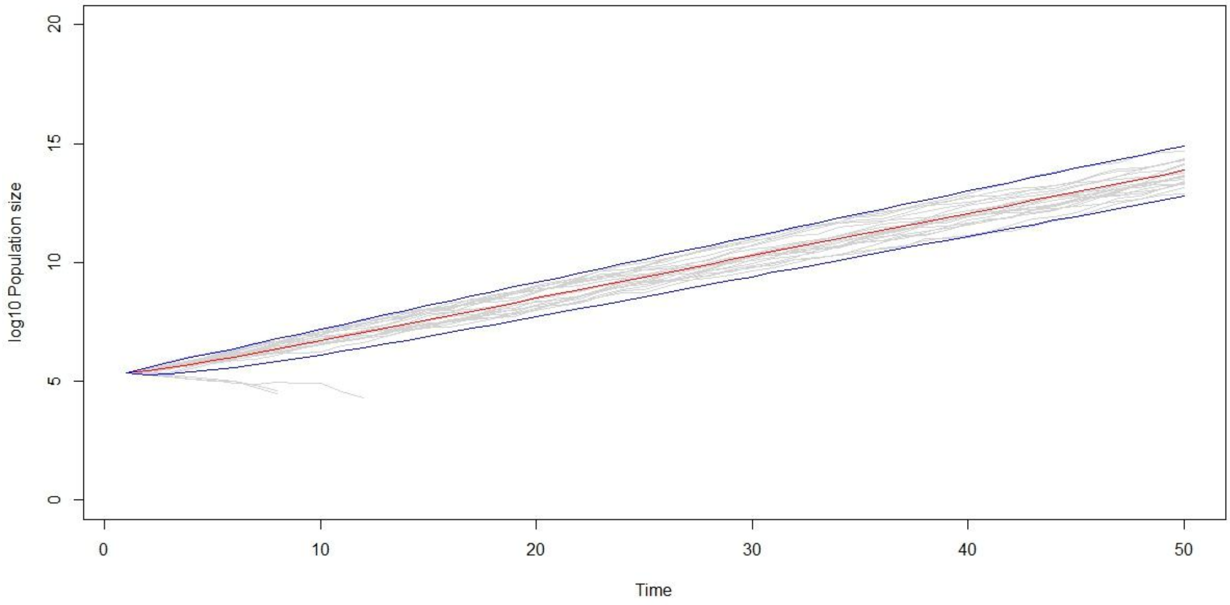


Figure A6.2. Median (female) Common Teal population over time (red) and 5th and 95th percentiles (blue) from 10,000 stochastic simulations, with some example trajectories (grey). Some trajectories specifically chosen to show examples of crashing populations.The Model uses transition probabilities between life stages (egg and adult) multiplied by the current population to calculate the next years population with a stochastically chosen harvest number subtracted each year. Harvest is chosen each year from a uniform distribution on the interval [46,500, 100,000] females, with the proportion of shot birds that are juveniles of 0.6. Model subtracts the “lost” expected egg production of killed birds and number of expected adults that were killed at the wintering grounds that would have survived the return to the breeding ground from the non-harvested expected numbers. To calculate this, it is assumed that surviving the winter and subsequent spring migration is equally as likely as surviving the summer and subsequent autumn migration, for adults. For young birds (from the moment of being laid), it is assumed that the first half of the yearly life cycle is twice as hazardous as the second half. Underlying adult (female) survival probability does not include hunting mortality, and is based on an estimate of best possible survival from (Ellis & Cameron 2022; BTO blog) (chosen from a uniform distribution between 0.425–0.791). Survival of egg to adult based on the probability of egg to juvenile survival and juvenile to adult survival (excluding hunting mortality, chosen from a uniform distribution between 0.342 and 0.722 (i.e.) survival increased by the same amount as for adults from exploited populations, with the original range from Thompson *et al.* 2022). Juvenile survival and adult survival are linked by splitting the respective ranges into very bad, bad, average, good and very good subranges and choosing from within them each year (i.e. they are auto-correlated such that good years are good for both adult and juveniles). Egg to juvenile survival (0.37) calculated as probability of nest survival (0.94 from Čehovská *et al.* 2022) multiplied by number of female ducklings per hatched brood (3.7/2 from Čehovská *et al.* 2022) and divided by the expected number of (female) eggs laid (BTO website; clutch size 8–11, 9.5 used, assume half would be female).

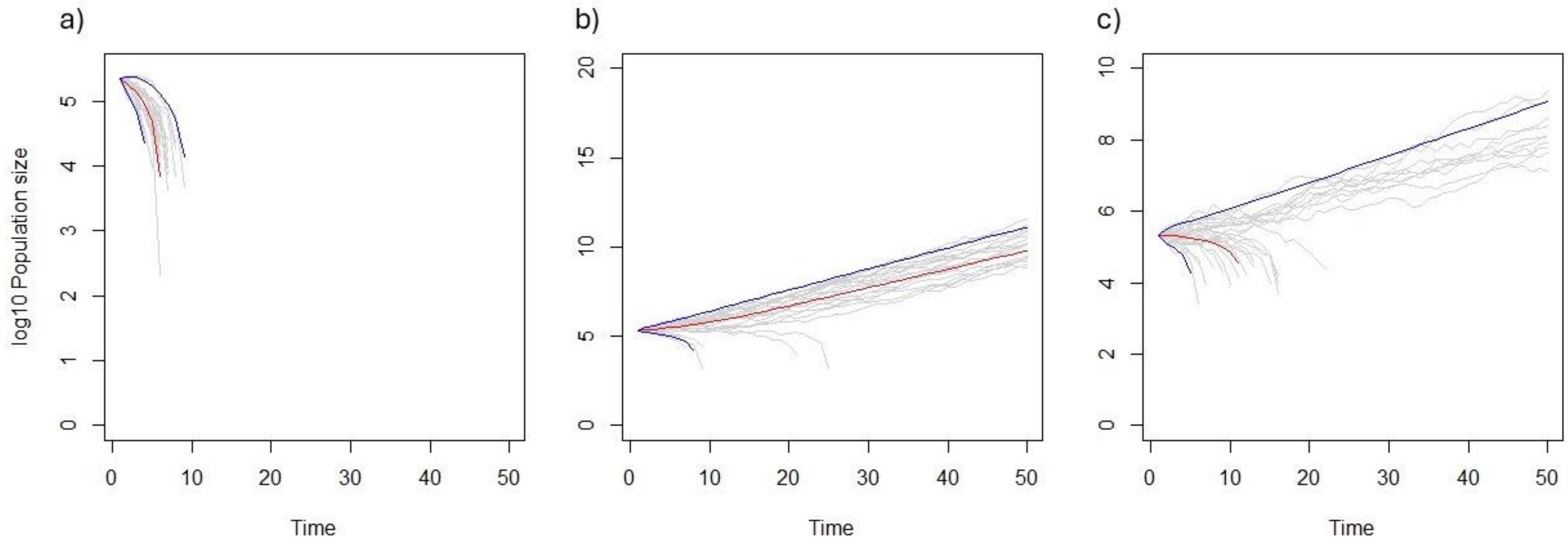


Figure A6.3. As Figure A6.2 but with probability that a nest survives (i.e. any chicks are hatched from a clutch) (a) fixed at 0.4, (b) fixed at 0.7, (c) drawn each year from a uniform distribution between 0.3 and 0.9.

Compared to earlier models, the third model demonstrates how dependent modelled population growth, and the effects of harvest, are on waterbird productivity. This is a classic result across waterbird species and studies, that the population growth is most sensitive to recruitment relative to adult survival. With constant low rates of nest survival such as 40% combined with realistic brood sizes and estimated UK harvest mortality, Teal populations are predicted to decline (Figure A6.1). Constant but higher nest survival (e.g. 70%) results in greater probabilities of stable or increasing population growth despite the UK harvest (Figure A6.3b). A stochastic approach to nest success, where we allow the model to select from a uniform distribution of nest survival values ranging from 40–70% each year results average population trajectories that decline (Figure A6.3c). While declining productivity is not currently of concern for Teal, it is of concern for other species (i.e. Pintail) and it is clear the sustainability of any harvest is strongly coupled to population productivity.

Like the PEG/PTL approaches introduced by Ellis and Cameron (2022) and presented in this report, PVA models are relatively straightforward tools to explore the sensitivity of wildlife populations to change, whether that be changes to productivity or survival. Both approaches, PEG/PTL or PVA, are accessible and run on basic computers using simple code. For example, the models presented here were run on a standard laptop using base R code. What is different between the two modelling approaches is the data required to parameterise them, with the more realistic life history and ecological scenarios that can be represented by matrix models in a PVA approach requiring suitable field-based data for the populations they are trying to represent. Even for Teal in this example we have had to rely on data from North America to gather enough data. In terms of costs and constraints, it is in investing in enough data from breeding ranges of UK wintering waterbird that makes a PVA approach more challenging. This is where the PEG/PTL approach is more attractive, and specifically why it is being promoted, to open numerical approaches to assessing harvest sustainability in data poor systems.