Ecological Consequences of Gamebird Releasing and Management on Lowland Shoots in England (NEER016)

A Review by Rapid Evidence Assessment for Natural England and the British Association of Shooting and Conservation

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Executive Summary

This review was commissioned and jointly funded by Natural England and the British Association of Shooting and Conservation (BASC).
The format of the work is that of a Rapid Evidence Assessment (REA): collating the available evidence, setting it within a comprehensible framework; describing material in terms of its relevance to the central question about the effects that releasing gamebirds has on the habitats and wildlife of England; and highlighting knowledge gaps.

The Issue
The recreational hunting of gamebirds (shooting) is widespread and long established in lowland England. Over the past century there has been a marked shift in the species and origins of the gamebirds shot such that in the 21st century, the majority of the quarry are made up of three species (pheasants *Phasianus colchicus,* red-legged partridges *Alectoris rufa* and Mallard A*nas platyrhynchos*) that have most commonly been reared under artificial conditions before being released in woodland and on farmland. Releasing has been undertaken since 1900, albeit originally at relatively few sites and in relatively small numbers, but the practice took off in the 1960s when wild bird populations could no longer support shooting demand. It has been increasing ever since and it is currently estimated that between 39 and 57 million pheasants and 8.1 and 13 million partridges are released in the UK, with 85% of these in England.

Released gamebirds themselves have effects on the fauna and flora of the habitats into which they are released, and their release is accompanied by habitat and other management activities by shoot owners, game keepers or shoot members which also have a range of effects on habitats and wildlife. Once released, a proportion of the gamebirds are shot by recreational hunters, assisted by non-shooting participants such as beaters. The increasing numerical scale and spatial extent of releases and their associated management means that effects on the habitats and wildlife of England have stimulated a small but growing body of research. Several authors have produced reviews that encompassed elements of the literature from a range of different perspectives. Our review aims to identify and appraise the relevant peer reviewed and grey literature systematically and comprehensively and to bring things up to date in order to gain a most complete and holistic understanding of the effects of releasing gamebirds on the habitats and wildlife of England. In agreement with the commissioning partners, we have deliberately excluded a consideration of the ethical, social and economic dimensions of shooting released gamebirds. We also did not consider the effects of lead shot used in shooting released gamebirds on the health of either humans or other wildlife, but instead direct readers to a recent comprehensive review.

The focus of the current Review is on ecological effects of releases of pheasant, red-legged partridge and mallard, and their associated management, on habitats and wildlife. Being a perturbation of a natural ecosystem, these effects are unlikely to be

ecologically simple. Such complexity is exacerbated by the fact that these releases are accompanied by deliberate human actions (game, predator and habitat management; organised shooting). Understanding such socio-ecological systems requires a simultaneous consideration of the actions of the released birds themselves and the human actions that accompany these releases that affect the habitats and wildlife in and around the release area, and the feedback loops between them. Therefore, we have outlined a conceptual model that integrates these diverse influences and interactions necessary to comprehend the net environmental effects of gamebird release (Fig 1). It is within such a Conceptual Model that the evidence presented in this Review should be assessed.

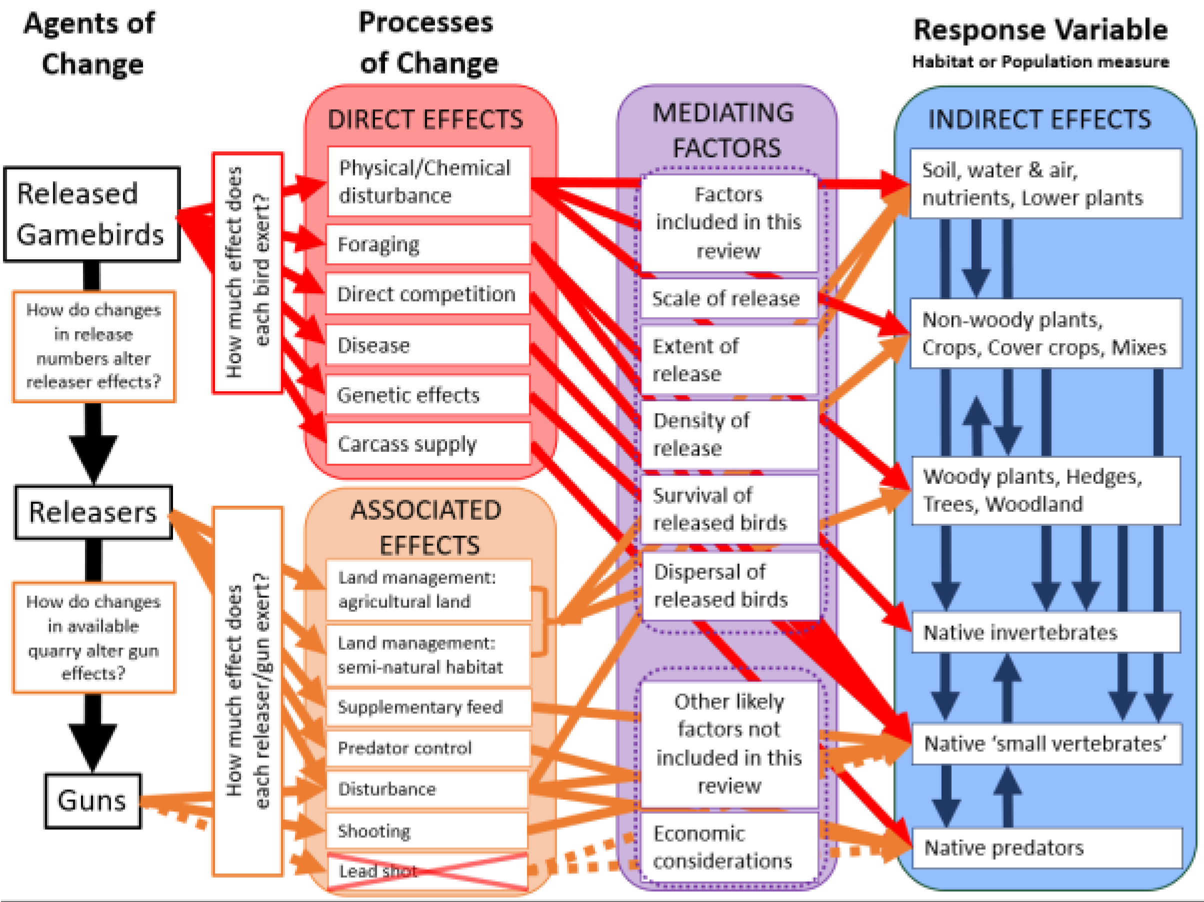

Figure 1. A conceptual model of the environmental effects on habitats and wildlife caused by the release of gamebirds and their associated management.

Method

The REA protocol involved a series of literature searches of a suite of public and private databases and approaches to other active researchers in the field, considering material pertinent to the release of gamebirds in the UK and Europe between 1961 and 2020. We conducted formal literature searches of four databases known to include both peer-reviewed and grey literature using defined search strings. We approached relevant organisations and individuals that had recently been working on released gamebirds in the UK for any relevant literature based on unpublished work or other material accumulated by them while addressing similar questions. We followed up additional references in this material that had not been identified using the first two methods. The resulting material, (collectively 'papers') comprises peer-reviewed papers, books, book chapters, academic theses, published and unpublished reports, and unpublished manuscripts. Each paper was scored according to its relevance to the central question of what effect the release of gamebirds has on the habitats and wildlife of England and was critically appraised. This process left us with a total of 229 papers that we included in our Review.

Summary and Conclusions

We found a set of 58 highly relevant papers that permitted us to formally compare measures of interest relevant to releasing and management between a treatment group and a control group, or to explore relationships between such measures and variation in release size or across spatial scales. Most of these papers made use of 'natural experiments' in which some areas hosted game shoots while others did not. We found no studies that experimentally manipulated gamebird release patterns and monitored associated change. We found an additional 60 papers that we described as moderately relevant. These included material that quantitatively described environmental variables of interest or the actions of game managers and/or guns at release sites or on game shoots or it described the behaviour of released gamebirds such as their diet composition, activity budgets or habitat preferences. Even though the study design of these papers precluded a formal comparison between these measures at sites with and without releasing, they provided useful background information with which cruder comparisons could be made. The remaining 101 papers, that we describe as weakly relevant, describes variables of interest, human actions or the behaviour or natural history of released gamebirds in a more qualitative manner. Even though such material cannot be formally evaluated or used to conduct quantitative comparisons, it may provide indications of where future work might focus efforts

The findings we considered were obtained from work undertaken at many hundreds of different release-based shoots over several decades. Such field studies are often dependent on the (voluntary) participation of the land owner or game manager and as such may not be an unbiased representation of the size or structure of English shoots or of management practices conducted nationally, but we have no evidence to support this concern. Therefore, we believe that until further data are forthcoming, the findings of the Review should be interpreted as representing a median type of shoot in terms of size and adherence to good practice over that period. During the period covered by our Review (and especially since the mid 1990's), releasing numbers have steadily increased. The pace of change in gamebird release, game management and shooting is such that the relevance of the (relatively small amount of) earlier work is less certain than that of more recent work. However, we have not attempted to weight studies by their date of completion but advise that older studies are examined carefully for their current relevance. The fact that several studies reported spatial variation in their results indicates that findings from one area may not simply transfer to other areas. There are currently insufficient studies to permit us to account for these spatial variations in a robust and formal manner. We have not attempted to weight or account for spatial variation but we advise that drawing national conclusions based on studies from single sites is risky.

Some previous Reviews related to this subject have defined the various effects of a released gamebird or the game manager as either positive or negative. We believe that whether a Direct, Associated or Indirect Effect is classed as positive or negative may sometimes be subjective and/or context dependent. For example, the availability of gamebird carcasses may support a higher population of predators. If the predator species is rare or endangered, then an increase in their numbers may be desirable and seen as a positive outcome of gamebird release. Alternatively, if the predator is a generalist and its increase leads to depletion of non-gamebird species then this may be undesirable and considered a negative outcome of gamebird release. Another example: supplementary feeding may support larger populations of overwintering passerines but may simultaneously support increased rodent numbers. Such rodents may be pests but may also be desirable species of conservation concern or provide prey for predators of conservation concern. Should supplementary feeding be considered to exert a positive or negative effect? Therefore, we have avoided defining the direction of each individual effect on the Response Variables. We advise that future work needs to clearly determine the specific ecological outcomes that are of interest and carefully consider and assign the direction of each effect in order to arrive at meaningful net outcomes. The net direction (positive or negative) and value for any one Response Variable may only become apparent when the cumulative effects of multiple Processes of Change and Mediating Factors are considered using our Conceptual Model.

In general, effects that we might consider to be subjectively positive are usually a consequence of gamebird management activities (Associated Effects) and most effects that we might consider negative are caused by the released birds themselves (Direct Effects). We found reasonable evidence for physical disturbance of soil, nutrient enrichment of water and soil, reductions in non-woody plants (especially those of conservation interest) due to damage or enrichment and reductions in abundance and/or diversity of at least some invertebrate species at or close to release sites. We found weaker, less or more ambiguous evidence that the released birds predated small vertebrates, posed a direct competition to non-game species, spread disease to non-game species, influenced the genotypes of wild conspecifics (in England) or that their carcasses supported increases in generalist predators. Such Direct Effects may be moderated by changing the scale and location of releases. Some negative effects such as effects on reptiles or sensitive lichen communities involve very specific conflicts with nature conservation interests which can likely be reduced or eliminated if sensitive sites are identified and avoided when releasing gamebirds. Several studies revealed that the strength of these (negative) effects grew stronger as the density of released birds in the pen and surrounding area increased. Working within the normal range of releases described in the papers we reviewed (a few hundred birds to a few thousand gamebirds in any one pen) it was a consistent result across studies that smaller releases had a reduced effect.

The actions of game managers had a range of effects associated with, or motivated by, gamebird release. We found reasonable to good evidence that they engaged in land management of agricultural and semi-natural habitats at levels higher than other land owners. This land management was typically accompanied by increases in numbers or diversity of plants, invertebrates and non-game vertebrates in those areas of the game shoot. It is important to better understand whether and to what extent these benefits are spread more widely to areas of the game shoot that are not so intensively managed and further out to neighbouring areas where release and shooting does not take place. While it is usually clear that releasing provides the motivation and economic underpinning of the management of these habitats, it would be useful to better understand this link and the shape of the relationship between numbers of gamebirds released and the extent of management actions. This would permit more accurate predictions of management changes and accompanying ecological effects in response to any future alterations in gamebird releases.

The legal control of generalist predators led to lower than expected numbers of such predators locally, although the evidence for this was weak and this may reflect the mobility of predators or the inefficiency of much predator control. We found little evidence that predator control specifically associated with gamebird release itself led to increases in populations of non-game small vertebrates. Predator control can help non-game species of for example birds, especially when breeding but the effectiveness of predator control in association with releasing, which mainly occurs after the end of the breeding season isn’t so clear and will vary from site to site. We found no evidence to either support or refute the hypothesis that generalist predators thrive on abundant gamebird carcasses and that this leads to overall decreases in non-game species. However, we advise that efforts be made to explore, experimentally and in the context of other contributing factors, such as farming activities, whether gamebird release can drive an increase in abundance of generalist predators, and whether this has consequent effects on the abundance or diversity of non-game species. We found weak evidence to suggest that illegal killing of predators was directly related to gamebird release (as opposed to wild game management), although we acknowledge that data about individual crimes is hard to obtain and the aim of prosecution is usually to determine guilt rather than motivation. We found moderate evidence that the provision of supplementary feed was accompanied by increases in a range of non-game small vertebrates and while many birds and mammals benefit, it is sometimes a matter of perspective as to whether this is ecologically positive or not, depending on the species considered. We found weak evidence that the actions of the guns and beaters during shooting led to disturbance of and potentially unintentional killing of non-game species. The effects of guns and beaters (disturbance, killing of wild species, use of lead shot) is not inevitably linked to gamebird release but may occur when shooting other wild game species.

In order to calculate the net effects of gamebird release either at a local or national scale, it is necessary to have reliable fine scale data about the size, location and past history of releases and the size of area over which released birds disperse and associated game management occurs (some of what we describe as the mediating factors). At present, none of these data are easily available. While individual shoots may be willing to report this data and provide information about where they practice land management or the extent to which they engage in supplementary feeding or predator control, it is unclear how representative or reliable these critical measures are. From a research perspective, such data would be desirable, but reliable mechanisms to collect these data do not currently exist. These data are vital in order to robustly evaluate net effects because the spatial scale of the different effects varies and may co-vary with release numbers or density.

Our Review has identified a series of Processes of Change that can lead to damaging Direct and Associated Effects on habitats and wildlife of England related to gamebird release. The evidence suggests that at least some of these effects can be ameliorated by following best practice relating to release sizes and densities and by consideration of release site locations and the rearing conditions of gamebirds for release. Equally, our Review has also identified a series of Processes of Change that can lead to beneficial Direct and Associated Effects on habitats and wildlife of England related to, and motivated by, gamebird release. These actions such as habitat management of woodlands and field edges, careful and timely supplementary feeding and appropriate predator control, can be achieved and enhanced through the deployment of best practice. Our Review indicates that ultimately, either an increase in damaging effects arising from poor or excessive game management, or a decrease in beneficial effects arising from the reduction or cessation of gamebird release could contribute to net negative ecological outcomes. To achieve net positive ecological outcomes for the habitats and wildlife of England, it is necessary to carefully consider and act to simultaneously reduce the negative and enhance the positive effects of gamebird release, both today and in the future.

Contents

Foreword

This review was commissioned and jointly funded by Natural England and part funded by the British Association of Shooting and Conservation (BASC). The authors met with David Stone (Natural England) and Matthew Ellis (BASC) on 8 Jan 2020 to discuss the scope of the review and appropriate methodologies before commencing work. The draft of the review report was submitted to Natural England and BASC on 18 March 2020.

The format of the work is that of a Rapid Evidence Assessment. Therefore, we restrict ourselves to collating the available evidence, setting it within a comprehensible framework, describing material in terms of its relevance to the central question about the effects that releasing gamebirds has on the habitats and wildlife of England, and highlighting knowledge gaps with recommendations for what material might be required to fill these gaps. We do not make formal recommendations about policy or best practice.

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We are very grateful to all respondents who suggested additional material for us to consider and who let us access their unpublished data. David Welchman (APHA) commented on material relating to disease and pathogens. Dr Nicholas Aebischer, Edward Baxter, Prof Nick Sotherton and Dr Mark Whiteside read and commented on drafts of the Review. Both authors contributed equally to the writing of this Review and authorship is given alphabetically.

18 March 2020

NATURAL ENGLAND   BASC   UNIVERSITY OF EXETER   Game & Wildlife CONSERVATION TRUST

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Introduction

The recreational hunting of gamebirds (shooting) is widespread and long established in lowland England (Martin 2011, 2012). Over the past century there has been a marked shift in the species and origins of the gamebirds shot such that in the 21st century, the majority of the quarry are made up of three species (pheasants *Phasianus colchicus*, red-legged partridges *Alectoris rufa* and Mallard *Anas platyrhynchos*) that have most commonly been reared under artificial conditions before being released in woodland and on farmland (Aebischer 2019). Releasing has been undertaken since 1900, albeit originally at relatively few sites and in relatively small numbers, but the practice took off in the 1960s (Robertson et al. 2017) when wild bird (mainly grey partridge *Perdix perdix*) populations could no longer support shooting demand. It has been increasing ever since and it is currently estimated that between 39 and 57 million pheasants and 8.1 and 13 million partridges are released in the UK, 85% of these in England (Aebischer 2019).

Released gamebirds themselves have effects on the fauna and flora of the habitats into which they are released (hereafter shooting estates), and their release is accompanied by habitat and other management activities by shoot owners, game keepers or shoot members (hereafter game managers) which also have a range of effects on lowland habitats and wildlife (described below in Critical Appraisal of Evidence for Processes of Change). Once released, a proportion of the gamebirds are shot by recreational hunters (hereafter guns), assisted by non-shooting participants such as beaters, pickers up etc (hereafter beaters). The increasing numerical scale and spatial extent of releases and their associated management (described below in the Some Evidence for Mediating Factors) means that these effects on the habitats and wildlife of England have stimulated a growing body of research. Several authors have produced reviews that encompassed elements of the literature from a range of different perspectives. We briefly summarise those written over the last 25 years below detailing their methodologies (if any) and specific areas that they cover.

Callaghan (1996) conducted an international review of the biodiversity and sustainability effects of release of waterfowl for shooting. This is the only review that we are aware of describing the effects of releasing mallard (or other waterfowl) for shooting. Callaghan's Review contained no description of methods as to what or why material was included. It considered effects including: hybridization, non-sexual inter-specific interactions, nutrient dynamics, disease, shooting pressure, distortion of wild population monitoring, depletion of conservation resources, restriction of more damaging activities, and re-establishment of species. It concluded that there were six negative effects and two positive effects arising from release and suggested that release of non-native wildfowl should be banned and that the release of native waterfowl for shooting should be discouraged and codes of practice developed. Not all of the findings or conclusions are relevant to the release of mallard in England.

Jones (2009) considered the effects of 'bloodsports' (the author's preferred term for hunting, game shooting and angling) on the habitats and wildlife in the UK. This Review takes a social science approach and expressly does not follow a methodical search strategy arguing that it comprises a "synoptic account that does not engage in heaping Ossa upon Pelion [sic] by way of intensive documentation, much less original sources" (p52). It considered effects including: landscape effects (planting and tending woodlands, installing hedges, stream clearing), wildlife effects (predator control) and agricultural effects (disturbance to agricultural land, and withdrawal of land from cultivation). It did not formally compare positive and negative effects. It is hard to comprehend what final conclusions about environmental effects, if any, were drawn. The Review included, and sometimes failed to distinguish, effects due to a variety of different 'bloodsports' making it hard to determine how relevant it may be to the releases of gamebirds in England.

Bicknell *et al*. (2010) specifically reviewed the impacts of non-native gamebird release in the UK. This Review briefly described how literature was selected, reporting that it "involved literature searches, particularly scientific publications, and contacting the various organisations of interest to this review [GWCT, RSPB, BTO, Avon Wildlife Trust, Buglife, Butterfly Conservation]"(pI). However, no search terms or selection criteria were reported. It was noted that most work related to the effects of pheasant release, and that although red-legged partridge were ecologically similar, their smaller body size and release scale meant that the absence of data specific to them was not especially important. The geographic scale at which any effect was likely to operate, and the expected direction and extent of effect was reported. The review considered the 75 studies shown in their Table 1. It considered effects including: Woodland creation and retention, woodland management, farmland (including hedgerow) management, legal predator control, supplementary feeding, browsing by gamebirds, predation of invertebrates, direct competition with breeding native wildlife, soil enrichment, disease spread, consumption of lead-poisoned prey, lead ingestion by native wildlife, lead concentration in the environment, grey partridge 'by catch', increased predator abundance, illegal predator control, economic input, employment, traffic accidents caused by gamebirds, zoonotic disease, and lead consumption by humans eating game. It concluded that 'the management of land and wildlife for [lowland game] benefits certain habitats and species', that 'current nationwide average release densities appear to be below the levels where known negative impacts occur' (although data are lacking and this may be spatially heterogeneous) and that 'there is also a significant and growing body of evidence that the negative impacts of gamebird release and related activities are considerable.' All work was directly relevant to the issue of releasing gamebirds in England.

Gallo & Pejchar (2016) reviewed an international set of literature reporting the effects of game management on non-targeted animals. They provided a detailed search criteria and process. Their approach revealed only 26 papers, most from outside the UK. It broadly described positive, negative and neutral effects deriving from: artificial water catchment management, increased abundance of game animals, mechanical reduction of woody vegetation, planting wildlife crops and prescribed burning. They examined 43 relationships (shown in the Supplementary Material) and concluded that 40% were positive for non-targeted species; 37% were negative and 23% had no effect. Four papers related to released gamebirds with another paper relating to wild pheasant management in the UK.

Mustin *et al.* (2018) conducted a pan-European review of the effects on non-game species of managing gamebirds for shooting. This Review described search, inclusion and extraction criteria in detail, based on records in Web of Science and Google Scholar as well as a non-systematic supplementation. 1735 initial studies were refined to 35 which contained 122 significant effects. It considered effects including: habitat management, predator control, parasite control, supplementary feeding, rear and release. It concluded that '63% of the 122 effects on non-game species were positive' and 'effects of rear and release were mixed (8 positive and 7 negative)'. The work related to the shooting of both reared and wild-born gamebirds and little effort was made to separate these two bases of shooting. Because it drew on material from across Europe (although most work was from the UK), the different landscapes, wildlife and shooting practices mean that the conclusions will not all be relevant to releases of gamebirds in England.

Avery (2019) presented a summary of the natural history of the pheasant in the UK and a description of their release and shooting with some mention of the effects that this release may have on habitats and native wildlife. This Review contained no description of methods as to what or why material was included. It considered effects including: habitat quality on shooting estates, impacts on reptiles, effects on invertebrates, impacts on seed-eating birds, effects on predator numbers, conflicts between pheasant shooting and raptor conservation, lead in the environment, road traffic accidents, and wildlife and zoonotic disease. It did not formally compare positive and negative effects, but concludes that "...pheasant shooting can coexist with nature conservation objectives – but I'm guessing really"(p386). All work was directly relevant to the issue of releasing gamebirds, specifically pheasants, in England.

Chapman (2019) reviewed the direct ecological effects of non-native gamebird releases in lowland UK. This Review contained no description of methods as to what or why material was included. The literature considered included studies from the UK and Western Europe and explicitly excluded indirect effects such as associated habitat management (including management behaviour such as predator control and supplementary feeding). It considered effects including: spread of pathogens and parasites, competitive interactions with other birds, and disruption of predator-prey interactions. It did not formally compare positive and negative effects. There are no overall conclusions drawn although there are knowledge gaps highlighted.

There are also a number of PhD theses completed in the last 20 years that summarise aspects of this literature in their Introductory Chapters (Callegari 2006b; Greenall 2007; Davey 2009; Pressland 2009; Whiteside 2015; Swan 2017; Gethings 2018; Hall in prep).

A common thread in these reviews is that more work is needed. While there may be relatively little material published in the peer-reviewed scientific literature of direct relevance to this issue, for many areas there is a wider knowledge base than a standard literature search deployed by previous Reviewers might suggest. Our review aims to identify and appraise the relevant peer reviewed and grey literature systematically and comprehensively and to bring things up to date in order to gain a most complete and holistic understanding of the effects of releasing gamebirds on the habitats and wildlife of England.

Methods

Review Scope

Our review method (see below) follows the protocol for a Rapid Evidence Assessment (REA) as recommended by Natural England and described in Collins et al. (2015) and incorporating details agreed with the Steering Group. The focus of the Review is on ecological effects of releases of pheasant, red-legged partridge and mallard (An estimated 0.2 – 0.3 million grey partridges are also released annually but the effects of these relatively small releases are not included in this review in agreement with Natural England), and their associated management, on habitats and wildlife.

Recreational shooting involves a wide and diverse range of social factors. It has an economic value with various estimates of its contribution to the English economy (e.g. PACEC 2006, 2014). However, we exclude material relating only to the socio-economic value of, for example, shooting or supply of game meat, mental and physical health benefits from participating in shooting (Anon 2015), or the associated socio-economics of traffic accidents (Madden & Perkins 2017). The effects on zoonotic disease (Craine et al. 1997, Kurtenbach et al. 1998) or damage to crops (Rice 2016) linked to gamebird releases are also excluded, in agreement with Natural England, as is any consideration of the welfare of gamebirds during rearing and release (Madden et al. 2020). There is a growing appreciation that future ecosystems may differ substantially from current ones because of changes to the climate. While we acknowledge this likelihood, we have not tried to account for climate change scenarios in our evaluation of material. Finally, we do not consider the ethical or moral associations with gamebird shooting – for a recent discussion of these see e.g. Feber et al. (in press).

One further factor related to shooting of released gamebirds that is considered to have ecological effects is the use of lead ammunition. Lead shot, commonly used as ammunition for shooting gamebirds, whether wild or released, has multiple effects on the quarry species themselves and other fauna inhabiting the shoot that may consume spent ammunition either when predating injured or unretrieved gamebirds, or when ingesting grit to aid digestion. With the agreement of Natural England, we do not re-consider this extensive literature in our review but rather we direct readers to the comprehensive and recent review devoted entirely to this issue by Pain et al. (2019). Furthermore, we note that a consortium of interest groups has announced the intention to oversee the phasing out of lead shot over the next five years (https://basc.org.uk/lead/).

Constructing our Database

The REA protocol involves a series of literature searches of a suite of public and private databases and approaches to other active researchers in the field. The initial scoping document underlying this Review suggested that applicable peer-reviewed evidence may be limited and that there was thought to be a body of high-quality grey literature that may be of use. Consequently, we deployed three complementary strategies to obtain the set of studies to be considered. First, we conducted a formal literature search of four databases known to include both peer-reviewed and grey literature using defined search strings. Second, we approached likely relevant organisations and individuals that we knew were or had recently been working on released gamebirds in the UK to ask if they had any relevant literature based on their own unpublished work or other material accumulated by them while addressing similar questions. This included a GWCT database of relevant literature. Finally, after we had refined the material from our first two searches, we noted and followed up any relevant citations in that material which we had missed. This could include material from pre-1961. This material comprises peer-reviewed papers, books, book chapters, academic theses, published and unpublished reports, and unpublished manuscripts. We collectively term these as papers from hereon for simplicity.

1) Formal Literature Searches

We adopted a broad-term approach and conducted searches of four databases between 13 and 15 Jan 2020 (Table 1). We only considered research published post 1961 to present for two reasons. First, this period matches the duration of the National Gamebird Census from which it is possible to start to reliably extract patterns of change in gamebird release (Robertson et al. 2017, Aebischer 2019). Second, prior to this period, the style of shooting, land management and release practices in the UK exhibit clear differences from those practices seen in the 2020s.

Database	Date of Search	Search Term	Search Limits	Number of Records Retrieved
Google Scholar	13 Jan 2020	Phasianus colchicus shoot* [with the exact phrase Phasianus colchicus] Alectoris rufa shoot* [with the exact phrase Alectoris rufa] Anas platyrhynchos shoot* [with the exact phrase Anas platyrhynchos]	Words anywhere in the article, 1961-2020	2250 708 3930
Web of Science	13 Jan 2020	Phasianus colchicus AND shoot* Alectoris rufa AND shoot* Anas platyrhynchos AND shoot*	Timespan 1961-2020	32 24 14
Web of Science	14 Jan 2020	Phasianus AND colchicus Alectoris AND rufa Anas AND platyrhynchos	Species name in keywords and papers in categories: zoology, ecology, biodiversity conservation, environmental science, biology, agriculture multidisciplinary, entomology, forestry, environmental studies, soil science, agronomy Timespan 1961-2020	727 419 3041
ETHOS (Library of theses published by UK Universities)	15 Jan 2020	Phasianus AND colchicus Alectoris AND rufa Anas AND platyrhynchos	None	15 0 7
NDLTD (Library of theses published by International Universities)	15 Jan 2020	Phasianus AND colchicus Alectoris AND rufa Anas AND platyrhynchos	Published in English	16 2 92

Table 1. List of databases searched in our Formal Literature Search

The literature returned from these searches was then rapidly refined by JRM using the following exclusion criteria in this order: We removed any duplicates; We excluded literature not written in English, due to time and resource constraints on obtaining any relevant translations; We read the titles and abstracts of remaining papers and we excluded literature based on ecological research outside Europe. Although gamebird release does occur in the USA and New Zealand, the management, shooting practices and general ecology in those regions differ substantially from that seen in England, making it hard to interpret any findings with respect to the central question we were addressing. However, we retained a few studies relating to gamebird physiology and behaviour likely to be conserved across locations independent of ecology. We have clearly highlighted such studies as having been conducted outside Europe. We then excluded literature that included: A focus on lead or other heavy metals or toxins on gamebird health; Evidence of non-European location of study in title or abstract, OR an ecological relationship with a species not commonly found in UK; Any without the species name (in Latin or

English) in title that were not ecological (e.g. behaviour/parasitology/development); Any with the species name (in Latin or English) in title that only involved birds in captivity or solely related to social behaviour, mate choice/sexual selection, genetic structure.

2) Accessing Existing Organisational Datasets

We wrote to 22 individuals who are contributing to a meeting about the Biology and Ecology of released Gamebirds on 11 March 2020 (The email text is in Appendix 1). These people have all published on aspects of released gamebird ecology in the UK in the past 10 years and/or are currently working in this area. They include academics, independent ecologists, government scientists and researchers in relevant NGOs including BASC, BTO, GWCT and RSPB. We asked them for suggestions of any relevant grey literature they were aware of as well as any unpublished data or draft manuscripts that they would like to be considered for inclusion. The formal search and grey literature databases were combined.

Screening our Database

The contents of the database were then reviewed in detail by RS. He read the remaining Titles, Abstracts and Text and again excluded literature that included: Any mention of lead or other heavy metals or toxins on gamebird health; Any indication of non-European location of study, OR an ecological relationship with a species not commonly found in UK; Any studies for which no data were presented. He then scored each paper as directly relevant or indirectly relevant to the central question of what effect the release of gamebirds has on the habitats and wildlife of England (see Appraisal of Evidence below). We (RS and JRM) then read those papers and followed up any relevant citations within them that we had not encountered during our previous searches. There are also references in the review which provide background only that we specifically searched for to support general assertions about broader ecological processes or taxon specific natural history. Note that most studies indicate whether releases occur or not, for those that don’t we make our own assessment of this and where it is likely the study concerns only wild bird management it is excluded from the review. Papers were not differentially scored on scientific approach (experimental or otherwise) or whether peer reviewed or grey.

This process left us with a total of 229 papers that we included in our Review.

Appraisal of Evidence

The material most pertinent to the question that we are asking is that which makes a direct comparison in environmental variables of interest (e.g. wildlife populations or habitat coverage or quality) between sites where gamebirds are released and control sites where they are not, or material that considered correlated changes in environmental variables of interest with variation in the size of gamebird releases or variation across different areas hosting different amounts of gamebird releases, or material that compares the behaviour of released gamebirds with that of their wild-born conspecifics. We class such material as directly or highly relevant. We have summarised this literature (58 papers) in Appendix 2. In the following Sections, we denote this literature with ***.

Material that quantitatively describes environmental variables of interest or the actions of game managers and/or guns at release sites or on game shoots is also informative even though there may not be a direct comparison with control sites. Equally, material that describes the behaviour of released gamebirds such as their diet composition, activity budgets or habitat preferences may be informative. Even though the study design precludes a formal comparison between these measures at sites with and without releasing (because data were not collected at control sites within the study), it may be possible to obtain control values from other studies and make such comparisons. We have not conducted such comparisons in this Review due to time constraints. Such material also provides baseline values that might permit us or others to quantify the regional or national scales of releases and/or their effects. We class such material as moderately relevant (60 papers). We have included descriptions of such material in the sections below. In the following Sections, we denote this literature with **.

Finally, there is a body of material that describes variables of interest, human actions or the behaviour or natural history of released gamebirds in a more qualitative manner. Even though such material cannot be formally evaluated or used to conduct quantitative comparisons, it may provide indications of where future work might focus efforts or indicates whether particular effects do or do not occur. This material is especially important for understanding those effects that we suspect are likely to occur but which have not yet been formally investigated. We class such material (101 papers) as weakly/somewhat relevant and include descriptions of this material in the sections below. In the following Sections, we denote this literature with *.

The same reference may be given a different relevance class depending on the data set and analysis being cited from it. This is especially likely for theses and larger pieces of work. Other supporting references relating to general information relating to gamebird or non-gamebird more general ecology and biology, but of little direct relevance to the central question are represented without denotion.

Many studies reported comparisons between sites where shooting occurs with those where it does not occur. In most cases, it is mentioned or explicitly stated that releasing occurs in and around those sites. However, for some studies this is implicit rather than explicit. We have assumed that this shooting relies on gamebird release. The PACEC (2006) report states that 83% of shooting providers surveyed released gamebirds, with only 9% saying that they did not release birds. These shoot providers included grouse and wildfowl shoots where releases are never practiced and the report states that an absence of release was most common in Scotland (where grouse and wildfowl shooting may be more common). Therefore, although it is not inevitable that sites where shooting occur also release birds, it is highly probable. Where shooting of wild-bird populations occurs, we mention this in our text.

Collecting Novel Data on Scale and Extent of Release

The effects of released gamebirds and their associated management on habitats and wildlife is unlikely to be homogenous and therefore to understand the intensity and spatial extent of any effects of release, it is essential to consider when, where and how many gamebirds are released, how long they remain at the release site or on the game shoot and how long they survive, and where they disperse to. Gamebird release in England is not formally documented. There is no single, national level, reliable record of how many gamebirds are released annually, nor any recording of the exact sites where they are released, nor records of the ground that is managed or shot over, nor description of land where gamebirds may inhabit either during or after the shooting season. Therefore, we have drawn on a range of sources from which to extrapolate the scale and extent of releases. Each of these is imperfect and subject to a number of biases and limitations. JRM undertook this data collection for the section on Mediating Factors.

1) APHA Poultry Register:

Compulsory registration is required for individuals or organisations that breed, rear or release >50 gamebirds. Voluntary registration is available to those releasing <50 birds (Anon 2019). During the registration process, registees are asked to report: Species (pheasant, partridge (no separation of red-legged and grey) and duck (no distinction by species); Livestock Unit Animal Production Usage (Shooting, Other); Livestock Unit Animal Purpose (breeding for shooting, rearing for shooting, release for shooting); and Usual Stock Numbers. We made a FOI request for this information on 29 Jan 2020 and received a response on 13 Feb 2020. There were 7902 records but this does not correspond to 7902 separate locations because a single location may include all three species (three records) and/or up to three Animal Purposes per species. When filtered by Species and Animal Purpose, we can be more certain that a single record relates to a single location. Due to data protection issues, locations were provided at the postcode district level and for those districts where less than five records were present, no location data were provided. This included 1162 records. There were also 882 records for which no postcode was recorded. These records cover Great Britain, so we converted postcode district to County using (https://www.doogal.co.uk/PostcodeDownloads.php) and only considered those from English counties. These data provide raw numbers of birds held in captivity for releasing but were available only for a single year. More detailed and precise datasets would be available via a Data Sharing Agreement. We have not pursued this opportunity due to time constraints.

2) Import Figures:

Kerry McCarthy MP asked DEFRA how many partridge (a) eggs for hatching and (b) live birds were imported into England from each (a) EU country and (b) third country in the last 12 months for which figures are available. They responded on 13 May 2019 (Rutley 2019) with details of import figures for both partridges (presumed red-legged partridge) and pheasants over the previous 12 months. The question is asked annually so a time series could be explored. There is no detail provided as to where these eggs/birds were delivered or whether or where they were released.

3) National Gamebag Census:

The National Gamebag Census (NGC https://www.gwct.org.uk/research/long-term-monitoring/national-gamebag-census/) was established by the GWCT in 1961 to provide a central repository of records from shooting estates in England, Wales, Scotland and Northern Ireland. Participation is voluntary and several hundred shoots contribute annually. However, shoot participation may fluctuate as may the size of areas shot over and number of birds released. Therefore, data are interpreted as an index rather than raw numbers. This permits temporal analysis but make fine-scale spatial analysis difficult. The dataset is likely to be non-random with a bias towards shoots with links to the GWCT.

4) Bird Surveys by the British Trust for Ornithology:

The BTO conduct a series of surveys and from these, compile Atlas records of occurrence and abundance available for both the winter and breeding season (Balmer et al. 2013). Replicated methods allow changes in these measures over decades to be detected. We have not analysed these raw data (given time constraints) but rather have studied the summary maps and data presented in Balmer et al. (2013). Although these surveys offer a national coverage, they seldom coincide with periods of gamebird release (August/September) and typically extrapolate data for 10km^2 tetrads from a smaller number of transects or surveys. Depending on whether these surveys intersect release areas, local densities may be missed or under/over-represented.

5) Survey of advertising game shoots via Guns on Pegs:

Guns on Pegs (https://www.gunsonpegs.com/) is a commercial advertising site where shoots looking to let days or attract syndicate members can advertise. They can enter free-text descriptions of their shoot. Entry dates are not recorded but it has been operating for 7-8 years. We manually read through descriptions of 697 lowland shoots in England that offered shooting of pheasant, partridge (no attempt is made to distinguish red-legged from grey partridge) and 'duck' (not specified as mallard but often contrasted with 'wildfowl'). We extracted data on the quarry species and the bag sizes offered (with all quarry species combined) and the area over which each shoot operates. Shoot location is provided at county level. This database is likely to be biased towards larger, more commercially orientated shoots.

A conceptual model

Being a perturbation of a natural ecosystem, the release of gamebirds and their effects on the habitats and wildlife of England are unlikely to be simple. Such complexity is exacerbated by the fact that these releases are accompanied by deliberate management of the habitats and wildlife by game managers and organised shooting of the birds by guns and beaters and this adds human motivation and behaviour to a consideration of such effects. When attempting to consider the ecological effects of gamebird release it is vital to simultaneously account for the actions of the released birds themselves and the human actions that accompany these releases, especially the actions of the people managing the landscape into which the birds are being released.

In order to account for this complexity and to allow consideration of both anthropogenic and natural effects, we believe that it is helpful to consider the specific effects examined in individual published studies within a holistic framework. Therefore, we outline a Conceptual Model, based on our reading of the literature and prior understanding of gamebird ecology and game management in England (Fig. 1). It is within such a Conceptual Model that the evidence presented in this Review should be assessed.

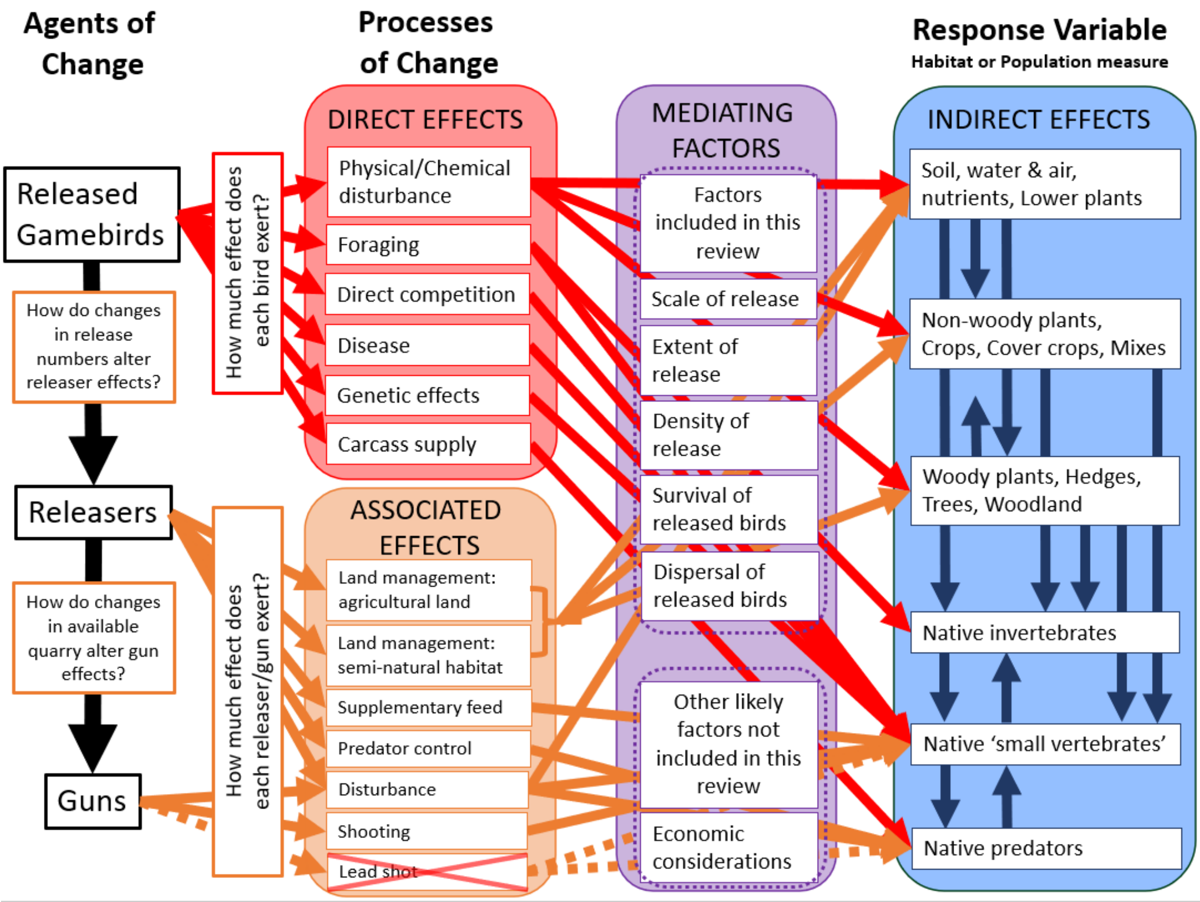


Figure 1. A conceptual model of the environmental effects on habitats and wildlife caused by the release of gamebirds and their associated management. While the effects of lead shot should be considered in such a model, we do not include evidence relating to this in our review and direct readers to Pain et al. (2019)

The primary focus of this review is the effect that released gamebirds and associated management activities have on the habitats and wildlife of England. Broadly, we assume that an **AGENT OF CHANGE** exerts an effect via a **PROCESS OF CHANGE** on some measure that captures aspects of habitat or wildlife worthy of interest or conservation (the **RESPONSE VARIABLE**). Such Response Variables might comprise a population measure of a particular taxa (e.g. numbers of songbirds or measures of biodiversity/richness in an area) or a quantitative descriptor of the health or extent of a particular habitat (e.g. quantity or quality of native woodland). In our Conceptual Model, we very crudely present six, illustrative, Response Variables: Soil, air and water (their structure and quality); Non-woody plants (e.g. game crops, annual flowers); Woody plants (e.g. perennial plants offering structure, woodland and hedgerows); Invertebrate populations (e.g. small herbivorous/detritovorous animals that could be eaten by gamebirds); Small vertebrate populations (e.g. songbirds, herptiles, rodents that may eat food provided for gamebirds, utilise habitats managed for gamebirds or be eaten by the gamebirds); and Vertebrate predators (e.g. foxes *Vulpes vulpes*, corvids, raptors that may predate released gamebirds and/or small vertebrates). We expect that future more detailed, formal analyses would use different, targeted Response Variables relevant to specific questions of interest. For example, a single species of conservation concern may comprise a Response Variable on its own.

We can conceive of three distinct classes of Agents of Change. First, the birds themselves may exert a **DIRECT EFFECT** by a series of Processes of Change on one or more Response Variable. For example, an individual released bird may both eat invertebrates and alter soil nutrient concentrations via defecation. Second, the release of these birds may be accompanied by management practices aimed to make the location into which the birds are released more clement for those released birds (e.g. woodland & hedgerow management, provision of game and wild bird cover crops, provision of supplementary food, predator control). We describe such practices conducted by humans motivated by the release of gamebirds as **ASSOCIATED EFFECTS**. As with direct effects, these act via a process of change on one or more Response Variables. For example, game managers may both increase perennial plant coverage and reduce numbers of predators. Third, the release of gamebirds is inevitably accompanied by shooting activities during which the birds are harvested. These are also considered to be Associated Effects. For example, guns and beaters may create physical and aural disturbance while harvesting gamebirds, affecting non-target wildlife.

The separation of the Agents of Change and the Direct and Associated Effects within our model is helpful when considering the ecological effects of released gamebirds for two reasons. First, the Associated Effects may operate in the absence of, or unrelated to the extent of, gamebird release. For example, game managers may choose to conduct game management actions even if they are not releasing gamebirds, perhaps because they wish to support wild-breeding populations of game species to hunt, or more generally because they believe them to enhance populations of non-game species on their land. Or guns may continue to harvest wild game in the absence of artificial releases. Second, even when gamebirds are released, the scope and extent of these Associated Effects may not relate in a linear or otherwise predictable manner to the number of gamebirds being released. For example, a game manager may only engage in planting cover crops if they release >1000 gamebirds, but go on to plant the same area if they release 2000, 3000 or 4000 birds, only planting additional areas if they release >5000 birds. Or guns may create the same level of disturbance during a day of shooting regardless of whether they are harvesting 50 or 500 gamebirds. Therefore, in order to understand how differences or changes in patterns of gamebird release may exert effects on habitats or wildlife, it necessary not only to simply count the numbers of birds being released, but also to consider how such numbers motivate associated actions by game managers, guns and beaters which may themselves exert effects on habitats or wildlife.

Agents of Change exert effect on habitats and wildlife via a series of Processes of Change. One agent may conduct a number of processes of change. For example, an individual released bird may scratch up vegetation, depredate native fauna and flora and eventually die and provide a food source for native predators. Or an individual game manager may plant woodland, control predators and provide supplementary feed. Because these processes exert very differential effects on different Response Variables of habitats and wildlife, it is helpful to consider them individually and account for the mechanisms by which they act. Simplistically, one might hypothesise that the release of a pheasant corresponds to a reduction in the number of invertebrates at their site of release. However, such reductions may arise because of direct predation or disturbance to the habitat or because the nutrient enrichment at the site facilitates a competitor species that themselves reduce the focal invertebrate population. If specific processes of change are identified and understood, then interventions via management may be able to ameliorate some of these effects associated with release processes. For example, if invertebrate populations at release sites decline because they are being predated by the gamebirds, then increased food provision may reduce such foraging and thus reduce the effect of release. Conversely, if invertebrate population declines occur because their habitat is being physically damaged, then attempts to halt declines by increased food provision will be ineffective.

The Response Variables are ecological units, such as a population of invertebrates or an area of grassland, and thus are likely to be connected with one another via a web of trophic interactions. Therefore, if one Response Variable is affected by gamebird release, resulting changes will exert effects on other connected Response Variables. We term such effects of gamebird release **INDIRECT EFFECTS**. For example, carcasses of released gamebirds may support populations of one Response Variable (Vertebrate predators) that also prey on a second Response Variable (small vertebrates). Or, if a larger area of annual plants were provided by game managers (because gamebird releases were increased), this may also increase numbers of Small vertebrates or Invertebrates that feed on such plants. In order to understand such indirect effects, it is necessary to comprehend the web of trophic interactions involving the habitats and wildlife of non-game species at and beyond the sites of gamebird releases. This is not trivial and reviewing the literature on the forms of trophic interactions among the habitats and wildlife of England is beyond the scope of this review. However, future work that attempts to assess or model changes in patterns of gamebird release should fully and impartially account for such Indirect Effects.

It is unlikely that a Process of Change will impact on a Response Variable in a uniform manner under all circumstances. Therefore, it is helpful to incorporate **MEDIATING FACTORS** into the model. For example, the effect of predation of invertebrates by released gamebirds may vary across the year as the natural dietary requirements of the gamebirds change such as during their breeding period, or as the availability of invertebrates declines over winter. Or, the dispersal of gamebirds from their point of release, and hence the area that they effect, may vary according to the density at which they are released. Consequently, in order to establish the effects that released gamebirds have on habitats and wildlife, it is necessary not just to understand the specific processes that occur (e.g. the number of invertebrates eaten by a single bird or the amount of woodland maintained by a single game manager), but to understand how these processes are distributed in time and space, depending on when and where gamebirds are released. The effects are not restricted to the point of release (in either time or space), but the gamebirds disperse over time, spreading the extent of effect but likely reducing the intensity of the effect as gamebird density declines with a fixed (and declining - released gamebirds die at high rates either due to shooting or natural mortality) number of birds distributing over an increasing area of country. The Associated Effects will also be spatially heterogeneous, coinciding with areas where gamebirds are released and perhaps scaling with the size of releases in those particular areas. Therefore, it is critical to consider the scale and extent of releasing and the subsequent dispersal and mortality of the released gamebirds and the game management that accompanies this.

First, this requires an understanding of the ***Scale of Release***, revealing how many gamebirds are released. Second, gamebirds are not released uniformly across the country but, because of history or geography, some parts of the country host especially high levels of release with more birds/km^2. Within these areas, releases are again clumped, with concentrations of gamebirds being placed in release pens (pheasants & partridges) or on particular ponds (mallard). Therefore, it is important to understand the ***Extent of Release*** i.e. the distribution of areas where releases occur. Third, in order to describe how the Direct Effects change in time and space as released birds leave the point of release and die, and to predict where Direct Effects may occur beyond the immediate release points, it is necessary to understand the post-release movement of gamebirds and their mortality patterns, providing information about the ***Density, Dispersal and Survival*** of each species.

Using this Conceptual Model assists our Review by allowing us to integrate a disparate literature drawn from field ecology, population biology, wildlife management, rural policy and human geography within a single framework. Although we do not convert all such studies to a single currency in this Review, future work could attempt this such that values from each study could be included within a single analysis. It also clearly indicates where gaps in our knowledge lie. These gaps are the links in our model between agents of Change, Processes of Change and Response Variables for which we currently have no data. At the end of our Review we will highlight these knowledge gaps.

More generally, this Conceptual Model offers two advantages when considering the effects of releasing gamebirds on the habitats and wildlife of England. First, it incorporates, without subjectivity, the three distinct forms of effect that gamebird release may have on habitats and wildlife, linking the Direct Effects of the birds themselves to the Associated Effects of humans involved in their release, be they game managers, guns or beaters, to the Indirect Effects that arise through the large-scale perturbations of natural systems likely to accompany release programmes. This makes accounting for net effects of gamebird releases or particular Response Variables feasible. Second, it permits future researchers considering recommending changes to patterns of release to construct predictive models that could more robustly indicate consequences of such changes for English habitats and landscapes. Some of these consequences may be unexpected if a holistic view is not taken. We will not, in this Review, attempt to parameterise and run such models or use them to draw conclusions about the quantitative effects of gamebird release, but we recommend that future work in this field adopts such a holistic approach.

Critical appraisal of evidence for processes of change: direct, associated and indirect effects of gamebird release

The release of gamebirds may have three types of effects on the habitats and wildlife of England as described above in A Conceptual Model. Our use of the terminology of Direct, Associated and Indirect does not indicate that we believe or conclude that any one set of effects is more or less influential than any other. They need to be considered in conjunction with one another (we suggest within our Conceptual Model) in order to calculate the combined effects that gamebird release and associated land management activities for those releases may have on a particular Response Variable of interest. In this section, we have organised the available evidence under the three broad types of effects, subdividing these by the Response Variables upon which each effect acts.

1) Direct effects

A) Direct effects of bird actions on soil, water and air

Direct effects of physical damage by gamebirds or nutrient enrichment through dunging are most evident at the particular sites where the birds are released. In a sample of five pheasant release pens from ***Sage et al. (2005a), soil potassium was higher in pens (47 mg/l compared to 14 mg/l) and phosphate was higher in pens (337 mg/l compared to 204 mg/l) than in control areas nearby. pH and magnesium levels were not detectably different in this small sample.

Effects of enriched soil chemistry may persist even after gamebird release has ceased and alter floral populations in following years. A study of the possible recovery of ground floras and soils in abandoned or disused pheasant release pens was undertaken over three years. ***Capstick et al. (2019a) compared the soil chemistry, ground flora structure and community composition of abandoned (including some 14+ years) release pen sites in 65 ancient semi-natural woodlands (ASNW) with nearby paired control areas in the same woodland with no history of being inside a pen. Soil fertility remained higher in abandoned pens than in control plots (phosphate 4.4 mg/l compared with 2.5 mg/l and potassium 7 mg/l compared with 5.2 mg/l). Nitrate levels, pH and soil organic matter, however, were not different. Note that these numbers (mg/l) for soil nutrients are not comparable with those described at the start of this section (Sage et al. 2005a) as different extraction methods were used in the two studies. There were more species of highly fertile soils in the abandoned pens than in the controls (14.3 per quadrat compared to 12.5) and fewer winter green perennials (14 compared to 22), which were the group of plants most affected in Sage et al. (2005a). Overall vegetative percentage cover had recovered, however, and there were no longer differences in the proportion of grasses and annual herbaceous or species of disturbed ground. The sensitive ground flora community and soil chemistry showed signs of recovery in the oldest group of pens in the study, those that had been abandoned for at least 14 years.

However, this long-term recovery was less marked at sites where a higher density of pheasants (>1000 per hectare) had been released.

Nutrient enrichment of water bodies may arise from the release of mallard. Estimations of nutrient enrichment by mallard in North America reveals that each bird introduces 0.72g N and 0.23g P per day into their water body (*Manny *et al*. 1994). There is the potential of eutrophication arising from large numbers of released mallard on a small pond which could degrade local biodiversity. However, such eutrophication may already be accounted for by depositions by wild birds. A population of wild birds, including mallard, were reported to contribute 73% of the external P and 17% of external N entering a pond in a UK SSSI (*Chaichana *et al*. 2010).

There are other, mainly anecdotal, reports of changes to soils where pheasant congregate. **Alsop and Goldberg (2018) is a recent example of a Natural England investigation at a designated site which included some systematic assessments. A National Nature Reserve NNR in the Derbyshire Dales contained a wooded ravine with a small but overstocked pheasant release pen and associated feeders. Over several years, next to release and/or feed sites, they documented soil erosion and soil enrichment and associated concentrations of droppings, a reduction in natural regeneration of tree and shrub species, more bare ground and a coarse and rank ground flora. The report concluded that the licence and consent to feeding and driving pheasants at the NNR should be withdrawn.

Lower order plants such as bryophytes (mosses and liverworts) and lichens are particularly sensitive to damage through enrichment of the soil or atmosphere and typically only remain common in woodlands that are in relatively clean-air regions of the country (e.g. Mitchell et al. 2004). Some preliminary calculations from the Centre for Ecology and Hydrology on atmospheric nitrogen caused by pheasant excrement suggested that some woodlands with large pheasant releases could have atmospheric nitrogen levels that are elevated to a level that will cause these sensitive plants to decline (unpublished data, Centre for Ecology and Hydrology, 2020).

A detailed investigation of the possible effect of pheasant releasing next to an area with feed hoppers and a rich lichen flora was conducted at a woodland SSSI in Leicestershire (*Smith 2014). The resulting report describes a complicated situation that inevitably leads to a lack of clarity on the central issue but provides some guidance plus various suggestions for further work. Briefly, the assessment work indicated that where the birds congregated away from the release site N deposition rates and atmospheric NH_3 levels were high enough to ‘merit lichenological concern’. The report provides an overview of atmospheric pollution in Britain which suggests that 20% of UK SAC’s (and Croxton Park SSSI) have background N and NH_3 concentrations which can damage lichens and bryophytes. Another principle source of atmospheric pollution at the site is identified as local livestock farming. Applying a precautionary principle i.e. given that the site conservation interest is under pressure anyway, the report indicates that potential pollution sources that contribute additional N will be contrary to good conservation management. Waste feed provided for the pheasants and pheasant faeces at the site could represent such an additional source.

Another SSSI wood with released pheasants, notable for a rich lichen flora was studied by **Bosanquet (2018). Surveys of lower order plants on trees and twigs were undertaken, which reinforced the conservation interest but also identified relative degradation of the flora in part of the woodland which was enclosed by a pheasant release pen. This took the form of the appearance of 'free' algae and N tolerant lichens which can lead to species loss inside the pen. Survey methods did not include randomly selected assessment points. While it is also plausible that the North East facing side of the wood (which contained the pen) was vulnerable to some other source of enrichment, the report described this as unlikely.

In 2015 the GWCT undertook a survey of bryophytes and lichens on the ground and on tree trunks in pheasant releasing woods at seven estates in Devon, SW England, comparing them with control areas in the same woods and in woods without release pens on the same estates (***Sage 2018a,b). They did not measure the size of pens or density of pheasants but at all of the study sites the releases would be categorised as large in terms of overall numbers. The abundance and diversity of bryophytes and lichens on trees overall was not different between release pen plots and controls. Moss diversity was about 25% lower on trees in both release pen plots, and in the plots outside of the release pens but in the same woods, compared to the estate woods without release pens. There was no difference in lichen species diversity between plot types. Liverwort species diversity was between ~30 and 50% lower in the release pens and release pen wood controls, compared to the non-pen estate woods. For moss and lichen abundance overall, and abundance of common species such as *Isothecium myosuroides*, and *Lecanactis abientina,* there was no difference between estate plot types, or between estate controls and other non-estate woods. Liverwort species as a group, and the common species *Frullania tamarisci*, were about 50% less common in pheasant release pens and in the control plots in the same woods, compared to the non-pen estate controls. *I. myosuroides* and *F. tamarisci* are relatively common species that are considered to be sensitive to atmospheric nitrogen enrichment and the presence/absence or abundance of these species is sometimes used as indicator of this in woodlands (Mitchell et al. 2004). It is possible that the differences between woods with and without release pens arise from increased nitrogen in the air, but other factors may also be involved. For example, management undertaken to create sunny areas in and around pens may reduce the suitability of the microclimate in those areas. **Rothero (2006) reported changes to the soil composition and effects (reductions) on bryophytes at a SSSI In Scotland when the estate started to release red-legged partridges nearby.

B) Direct effects of bird action on woody/non-woody plants

The direct effects of disturbance to plant life by released gamebirds are most evident within and around woodland-based, release pens. There are several mechanisms by which woodland ground floras might be changed where gamebirds are released. Changes to soil chemistry is one (see above) and plants that are still present in late summer and autumn can be damaged directly by pecking and trampling following the release. Perhaps less obviously, the woodland ground flora will also be affected by management of shrubs and trees in and around release pens. For example where the tree canopy is thinned in a woodland pen, while certain plants and animals can benefit (see 3.A,B,D), a shade tolerant flora may be reduced. Released gamebirds may encourage and stimulate growth by dispersing seeds from one area to another. We are not aware that this has been studied in pheasants or red-legged partridge, but Ducks, including mallard, are important dispersers of seeds of terrestrial and wetland plants, especially those with small seeds (*Kleyheeg *et al*. 2016). Mallard (and perhaps other gamebirds) may serve as vectors for both native and invasive species of fauna and flora (Green 2016).

Sage et al. (2005a) (see also **Ludolf *et al.* 1989b) reported the effects of pheasants on ground flora and soil at 43 woodland-based pheasant releasing sites by comparing quadrats inside and alongside the release pens with quadrats in the same woodland but away from the release areas. The sites were ASNW in southern England (Tapper 1992). ASNWs are considered to be valuable in terms of wildlife and cultural heritage and as a consequence, have a high sensitivity to damaging activities (Rackham 2003; Peterken & Game 1984). On average, release pens had more bare ground than control areas (18% compared to 10%) and reduced vegetation cover below 50 cm (42% compared to 55%) (Sage et al. 2005a). The release pens had lower average species diversity (3.4 species per quadrat compared to 4.1) and lower percentage cover of shade tolerant perennials, in particular winter-green perennials (6.4% compared to 25%) than the control areas. These perennials such as wood avens *Geum urbanum*, dog violet *Viola riviniana* and wood speedwell *Veronica montana* were present in relatively few release pens (5,6 and 8 respectively) compared to the control areas surveyed (8, 14 and 22). Annual species and some perennials preferring fertile or disturbed soil such as annual meadow grass *Poa annua* or chickweed *Stellaria media* were present in more pens (14 and 15) than in control areas (3 and 7) and increased in percentage cover as stocking density increased over about 1000 pheasants per hectare of pen. The proportion of bare ground was greater in smaller woods. Perennials characteristic of shady habitats like wood millet *Milium effusum* or wood anemone *Anemone nemorosa* decreased as stocking densities went up over about 1000 birds per ha of pen. The reduction of winter-green perennials was greatest in smaller, older pens. For these characteristic woodland plants there was no threshold stocking density at which degradation began to occur.

In another study (***Sage 2018a), the ground flora communities were measured in 12 quadrats inside release pens at seven shooting estates in the Exmoor region and in 12 quadrats in the same woodland but outside the pen. At four of these estates similar surveys were undertaken in woodlands without a release pen. The study did not look at pen stocking densities but the sample included several very large shoots (in terms of numbers released) with unusually large pens (up to around 10 ha). There was no difference in overall plant diversity between plot types. In terms of plant abundance, there was more bare ground (40% compared with 10%) and the proportion of ground covered by woodland herbaceous plants (15% compared with 30%) inside release pens compared to the areas outside of the release pen. The proportion of ground covered by ferns was over 20% in plots outside of the release pens and less than 10% in the release pens. Fern coverage was also less than 10% in the non-release control woods. Fern diversity was twice as high outside as inside the pens, and higher in comparison to the non-release control woods. The effects that were measured were confined to the pen and did not extend to other parts of the release woods.

Released gamebirds may have effects on flora further away from the release pen and these effects may also be density dependent. Pheasants are often encouraged to make daily movements along hedgerows between releasing woods and holding cover and partridges will use hedges close to their release sites. ***Sage et al. (2009) measured hedge and hedgebank structure, ground flora species composition and songbirds using hedgerows along transects leading away from release points at over 100 shooting estates in southern and eastern England in 2002 and 2003. They compared these data with numbers released and distance from hedge to the release pen. They found around twice as much bare ground on hedge-banks and inside hedges within 100m of release pens than in hedges further away from the release sites. The ground flora structure within hedges (but not on hedge-banks) at around knee height or below near to release sites was reduced by a similar amount. The hedge shrub structure between knee and waist height was reduced within around 100 m of release pens in areas that release more than 1500 birds compared to distant hedgerows. In arable areas, the diversity of perennial weed species inside hedgerows was about 25% greater where releases of more than 1500 birds occurred in the nearby pen. In grassland areas the diversity of desirable perennial plants was greater inside hedges where fewer than 1500 birds were released into the nearby pen. Alongside gamecrops, the study found fewer annual plants within hedgerows. In areas that released more birds there were around a third fewer perennial plants as well. They found no effects on the plant community of hedge-banks (ie alongside hedges) near release sites or gamecrops.

In their study using satellite imagery of hedgerow abundance and structure on 97 pheasant shoots where releasing was practiced and on 53 farms with no releasing ***Draycott et al. (2012) reported that hedgerow structure was similar on both types of sites and that woody species richness and woody cover was generally not depleted in hedges adjoining woodlands with pheasant release pens. Unlike Sage et al. (2009) this study used mean values and did not take account of the distance to release pens along hedges.

C) Direct effects of competition on small vertebrates

Released gamebirds have the potential to compete directly with other wildlife by disturbing them or displacing them from foraging sites, or by consuming food items that the other wildlife would otherwise eat.

A study in France reported that there was no effect of increasing (natural) mallard density on the foraging behaviour of other dabbling ducks, suggesting that released mallard are unlikely to directly compete with or interfere with the foraging of other granivorous ducks (*Guillemain & Fritz 2002).

While invertebrates are important to pheasant and partridge chicks, they are not a key food item for adult pheasants or partridges (Beer 1988). They can sometimes form a small part of their diet (although Holland et al. 2005 suggests not at all for partridge) and there is evidence that adult pheasants may reduce some invertebrate populations, particularly at release sites (see section F) below). It is therefore possible released pheasants may reduce food supplies for sympatric non-gamebird species at or near to those sites but there is little direct evidence of this. In their review of gamebird release, ***Bicknell et al. (2010) reported the chick dietary composition overlap between pheasant chicks and three farmland birds of conservation concern (grey partridge, yellowhammer *Emberiza citrinella*, corn bunting *E. calandra*), calculated a similarity coefficient for the invertebrate component of their diets as being 0.83, 0.79 and 0.69 respectively (with a score of 1 indicating a perfectly matched diet) and suggested possible resource competition 'where gamebirds breed in abundance'. Pheasants and partridges however do not breed well at the vast majority of release-based shoots in England (e.g. *Draycott *et al*. 2008b; *Sage *et al*. 2018c). There may be dietary competition between breeding pheasants and non-game species at shoots that successfully manage for wild game.

D) Direct effects of disease on small vertebrates

Released gamebirds may introduce pathogens into the wild that they acquired while being reared at high densities. Their high densities at release may facilitate or enhance their susceptibility to, or spread of, pathogens in local areas. Such pathogens may go on to infect other non-game species.

i) Endoparasites

In Britain released pheasants and red-legged partridges are prone to infection by a range of endo-parasitic worm species the commonest being *Heterakis gallinarum*, *Capillaria* spp. and *Syngamus trachea* (**Clapham 1961; **Draycott *et al*. 2000; **Gethings *et al*. 2015). The process of releasing in large groups onto the same area each year combined with the lifecycle and survivability of the parasites in the environment means that infections are maintained from one year to the next. While parasites are often specific to certain host species or closely related groups (Anderson 2000) some of the worms found in released gamebirds are also found in other birds (**Clapham 1957; ***Gethings et al. 2016b, *Bandelj *et al*. 2015; **Tompkins *et al*. 2000).

H. gallinarum is very common in pheasants and has been recorded in high numbers in apparently healthy birds (**Draycott *et al.* 2002; **Woodburn 1999). Between 2000 and 2003 a field experiment was conducted on two plots on each of nine estates in eastern England to determine the effect of *H. gallinarum* on pheasant breeding success (**Draycott *et al*. 2006). On one plot pheasants were treated for worms using treated grain in feed hoppers. These birds had reduced burdens of *H. gallinarum*, similar adult survival but better breeding success in the spring (on average 25% more young observed) than untreated birds on the other plot. **Woodburn (1999) found that catching and treating individual hen pheasants with an anthelmintic reduced subsequent *H. gallinarum* infections and improved the survival of hen pheasants during nesting. She hypothesized that this may be due to reduced scent emission by birds with reduced parasite burdens. This may have been the mechanism for Draycott et al.'s (2006) results.

Released pheasants may act as a reservoir for *H. gallinarum* by maintaining significant infections which could be picked up by other birds through the usual transmission route of parasite eggs excreted onto the soil. **Tompkins *et al.* (2000; 2001) experimentally infected eight grey partridge with *H. gallinarum* and found that at relatively low infections rates (i.e. at which pheasants would be unaffected) after 50 days the partridge showed on average a decrease in caecal activity of 32%, in food consumption of 19% and in body mass of 11% (Tompkins et al. 2001). The findings from this small sample of birds were used to support a model that suggested that pheasants carrying *H. gallinarum* could compete and ultimately exclude grey partridges via the parasite alone (Tompkins et al. (2000). Red-legged partridges were not affected by *H. gallinarum* but unlike the pheasant the parasite does not establish well in the red-legged partridge (Tompkins et al. 2002). This suggests that red-legged partridges would not play a role in any potential *Heterakis* mediated competition as hypothesised in his other papers.

The results of an infection of *H. gallinarum* on grey partridge were not repeatable in a study by **Sage *et al.* (2002) using 26 experimentally infected partridges and 26 uninfected ones. They found no reduction in food eaten or caecal production in the infected birds. Infected female grey partridge (but not males) lost 2% body mass and this was related to the number of worms in the caeca. Whether *H. gallinarum* can have a serious deleterious effect on grey partridge (or any other birds that may be exposed to infected pheasant faeces) in the field (as suggested by Tompkins work) remains unclear. The grey partridges may face greater exposure when closer to release sites. In a study on two large estates in southern England ***Ewald & Touyeras (2002) looked for associations between grey partridge productivity and proximity to pheasant release pens. At one site over 18 years, grey partridge young-to-old ratio was related to pen proximity but so were habitat quality factors and these were thought to be causal. At the other site over 33 years, no associations were found. Moreover, despite increasing numbers of released pheasants since the 1960s (Aebischer 2019), an analysis of 12,056 post-mortem reports found that the rate of infection of wild grey partridges by *H. gallinarum* fell by over 90% since 1951. The authors suggest that this indicates that the source of the parasite was free-ranging domestic fowls *Gallus gallus*, now vanished from the British countryside (**Potts 2009, 2010).

Syngamus trachea or gapeworm, which also infect non-gamebird species, is a common problem for pheasant and partridge releases and many game managers treat gamebirds for infections via their food or drink when released. Following treatment, birds can often re-infect themselves because the parasite eggs can survive on/in the soil from one season to the next. Gethings et al. (2015) showed that the parasite eggs were highly aggregated around feed points in and around released pens. A strong negative association between worm number and body condition in pheasants and carrion crows (found on the same shooting estates) has been shown (Gethings et al. 2016 **a, ***b) even at low infection levels where just one pair of worms is associated with an apparent deleterious effect. It is not known whether birds in poor condition are more likely to acquire these worms or whether the worms reduce condition.

The relationship between *Syngamus* in gamebirds and wild birds is unclear. Wild birds are thought to cause *Syngamus* infections in poultry or in released gamebirds and *vice versa* (Anderson 2000). Bandelj et al. (2015) looked for parasites in 385 passerines including 43 species and found *Syngamus* spp. in 2.5% of the non-migratory birds. Most were omnivorous species indicating that insectivorous or granivorous species were rarely affected. Holand et al. (2015) found that house sparrows *Passer domesticus* infected with *S. trachea* showed reduced reproductive success compared with uninfected birds. ***Millan et al. (2004) suggests that the 4 million red-legged partridges released in Spain might introduce parasites to wild red-legged partridges. They sampled red-legged partridges from 9 areas with released birds and 7 areas with only wild ones and between them they had 16 different helminth species, mainly nematodes. However only one of these occurred in both the reared and wild sample. This suggests at least that wild and released red-legged partridges do not necessarily share the same endoparasite.

Other animals especially birds are known to use pheasant and partridge feeders (see 2.D) making them potentially susceptible to infection. However, there is little information on whether parasite control treatment for releases, which is frequently undertaken using anthelmintic-treated grain or water in pheasant feeders inside release pens, has any positive or negative effect on other wildlife (Mustin et al 2018).

ii) Ectoparasites and associated pathogens

Ticks are commonly found on pheasants (**Hoodless *et al*. 1998). Ticks, by themselves, may infect non-game species and cause adverse effects via blood loss and physical damage (Proctor & Owens 2000). Ticks may also spread other pathogens. For example, Lyme disease in humans, caused by the bacteria *Borrelia* spp. is acquired through tick bites, predominantly from sheep ticks *Ixodes ricinus*. *Borrelia* bacteria are maintained in an enzootic tick-wildlife cycle, infecting rodents and certain other mammals and ground-feeding birds. The relative importance of the different factors on the incidence of *Borrelia*-infected ticks and the effect they have on their wildlife carriers is unknown and complex interactions can be expected (Ostfeld et al. 2018). Small mammals have been identified as the likely key vectors (i.e. a host that can get the bacteria from one tick and then pass it to another) of *Borrelia* in woodlands (Perez et al. 2016). While a review of releasing and zoonotic disease is not included in this review (see Methods) interactions between released game and wildlife via ticks, and hence the literature on *Borellia*, is relevant. Pheasants have been identified as a potential vector of *Borrelia* (**Kurtenbach *et al.* 1998). Whether that potential is realized and whether pheasant release sites play a role is unknown. Numbers of released pheasants in a landscape are at their lowest during the spring and early summer when questing ticks are most active. Nevertheless, whether there are particular tick-host community/habitat scenarios where pheasant releases might maintain or increase the prevalence of *Borrelia* and Lyme disease requires investigation. Millins et al. (2017) concluded that the presence and management of non-native species (they were using grey squirrel as a model) is likely to have a limited effect on tick abundance in areas that have [other] tick reproduction hosts, in particular deer.

Ehrmann et al. (2018) identified a range of habitat properties that promote tick-hosts and ticks in woods. The paper discusses how the structure and microclimate of woodland understoreys might influence the survival of ticks when not on hosts, and also the interactions between ticks and hosts. On average woods managed for released pheasants tend to have more shrubs and greater ground cover than other woods (see 3.A). This and other woodland conservation practices may promote ticks and tick-host interactions.

iii) Disease and Pathogens

Reared gamebirds are susceptible to a range of diseases in addition to the internal and external parasites referred to above. These diseases may infect local wildlife with negative individual health and population level effects. The overabundance of game species in an area, resulting from excessively large releases (or under-shooting) can support increased rates of disease transmission both within the game species and to populations of other wild species (**Gortázar *et al*. 2006). The occurrence of these diseases on a particular site depends on factors such as the sources of the gamebirds, contact with other wildlife, stocking density, management of the birds during rearing and prior to release and also on external factors such as weather conditions.

In recent years respiratory disease has become increasingly important in reared gamebirds before and after release. The principal pathogen is *Mycoplasma gallisepticum* (MG) (*Welchman *et al*. 2002) but the severity of disease can be exacerbated by other agents such as the bacterium *Ornithobacterium rhinotracheale* (ORT) in pheasants (*Welchman *et al.* 2013). MG has been detected in rooks (Pennycott 2005) and is widely recognized in songbirds in North America, although this finding has not been replicated in the UK. There is opportunity for transmission of MG from released gamebirds to wild birds in the UK and *vice versa* if, for example, gamebirds and corvids come into close contact when feeding. Antibodies to another respiratory agent, avian (meta) pneumovirus were detected in both reared and free-living pheasants in Italy in the 1990s (*Catelli *et al.* 2001).

An important respiratory disease in poultry is infectious bronchitis (IB), which is caused by various strains of coronavirus, and some strains have been detected in gamebirds (*Welchman *et al*. 2002, *Cavanagh *et al.* 2002)[1]. Coronaviruses have also been associated with kidney disease (nephritis) in released pheasants (*Lister *et al*. 1985), and there is the potential for coronavirus to be transmitted to wild gamebirds and cause both types of disease. Coronavirus was detected in 16.8% of birds in a survey of diseases of free-living pheasants in northwestern Germany (**Curland *et al*. 2018) and around a quarter of a wild pheasant population in East Anglia died of related kidney failure (**Draycott 2013).

Intestinal disease is common in young reared gamebirds, and is often attributed to protozoal infections, particularly coccidiosis and spironucleosis (formerly known as hexamitosis). However, neither of these diseases appear to have been recorded as causing clinical disease in wild gamebirds; coccidia (*Eimeria* species) are host specific, therefore pheasant *Eimeria* are not known to parasitise birds other than pheasants, and the same applies to partridge *Eimeria* species.

Intestinal disease is also commonly associated with bacterial infections, such as with *Salmonella* species and particular strains of *Escherichia coli.* Both of these bacteria are associated with disease in younger birds during captive rearing and therefore unlikely to spread to wild birds as a result of release. ***Díaz-Sánchez et al. (2012a,b) showed a significantly higher prevalence of *E. coli* and avian pathogenic *E. coli* (APEC) in released red-legged partridges (45-60%) in Spain compared to wild populations (6%). The prevalence of *Campylobacter* sp. (23%) did not differ

[1] Added 21 May 2019: Note that none of these strains of coronavirus are SARS-CoV-2, currently causing COVID-19, and that chickens and ducks (and thus likely gamebirds) are not susceptible to SARS-CoV-2 (Shi *et al*. 2020. Susceptibility of ferrets, cats, dogs, and other domesticated animals to SARS–coronavirus 2. *Science*, DOI: 10.1126/science.abb7015)

significantly between these husbandry groups, and *Salmonella* sp. was only detected one of the farms studied (0.9%, 5 out of 544). These results suggest that released partridges can act as carriers of these enteropathogens and highlight a potential but as yet unidentified risk of transmission to natural populations via the releases of farm-reared partridges.

As with farmed poultry, gamebirds are susceptible to the notifiable diseases, avian influenza and Newcastle disease which may infect non-game species, although pheasants and partridges are less susceptible to clinical effects than are turkeys and chickens. The actions taken in the event of notifiable disease are covered by UK legislation and gamebirds that are reared or kept in captivity fall within the legal definition of poultry. Once released however they are categorized as 'wild birds'. There is a legal obligation to report suspicion of notifiable disease immediately. Highly pathogenic avian influenza (HPAI) was confirmed in pheasants in England in 2017, and Newcastle disease was confirmed in pheasants in England in 1996 and 2005 (*Aldous & Alexander 2008). These diseases are subject to a stamping out policy and, if confirmed, restrictions are imposed on the site, the birds are slaughtered and tracings and other measures implemented to prevent any risk of spread, and action is taken if further cases are detected. Although there is the potential for gamebirds to spread these diseases to native fauna (*Bertran *et al*. 2014), including after release, in practical terms the likelihood of spread from infected gamebirds in the UK is low once an outbreak has been confirmed and game managers take appropriate remedial actions.

Disease or other pathogens may accumulate in mallard flocks under dense, stressful rearing conditions and these may leak into the wider landscape [e.g. botulism (*Otter *et al.* 2018). No effects on other wild fauna were reported in that case. Reared mallard may be susceptible to infection from wild conspecifics that share food or water supplies and this was seen in one case of avian influenza (H7N1) where a strain of the virus was detected in both a reared flock and local wild birds (*Therkildsen *et al.* 2011). In that case, the infected reared flock was destroyed but there is the potential that if undetected, infected released mallard may spread such infections more widely and in the case of avian flu, this could affect wild bird populations (Stallknect & Brown 2007).

iv) Effects of medication of gamebirds on disease susceptibility of wildlife

Medication of gamebirds during rearing and soon after release may also affect the disease susceptibility of non-game species. Antibiotics have been widely used in gamebird rearing to control a variety of disease conditions (although the use of antibiotics in the gamebird sector declined by 52% in 2018 compared with 2016 (*UK-VARRS 2019) and some bacteria may have developed resistance. Resistance to common antibiotics (anti-microbial resistance – AMR) by a range of commensal microorganisms is detected in samples from pheasants long after medication. AMR of various types has been reported from pheasant samples (vanA-mediated glycopeptide resistance in *Enterococcus gallinarum* (*Devriese *et al*. 1996); Erythromycin, tetracycline, ciprofloxacin, nitrofurantoin, rifampicin & quinupristin-dalfopristin (all 100% resistance) (*Guerrero-Ramos *et al*. 2016); and a range of antibiotics in pheasant and partridge in *E coli* (*Abbasi *et al.* 2012). A case of LA-MRSA was reported in a pheasant (UK-VARRS 2019). *E. coli* isolated from a small percentage of wild partridges in Spain *Díaz-Sánchez *et al*. (2012) showed resistance to three selected antibiotics. The authors suggested that releasing treated birds was a potential means of disseminating antibiotic resistant bacterial strains among wild birds. A similar (unsubstantiated) explanation has been proposed for the high levels of AMR found in wild rodents that may utilise gamebird feeders (Gilliver et al. 1999). Gamebird carcasses that are consumed may contain veterinary antibiotics which can then enter the ecosystem. This may explain why AMR is detected in microorganisms in buzzard and other raptor faeces (e.g. Radhouani et al. 2010) and foxes (e.g. Carson et al. 2012), both common gamebird predators. However resistant bacterial strains are also likely to spread to wild birds from all farmed livestock, and the reverse can occur as well.

v) Direct effects of foraging on non-woody plants

Pheasant, red-legged partridges and mallard are all omnivorous (*Hill & Robertson 1988, *Potts 2012, *Dessborn *et al*. 2011a). Their adult diet is predominantly vegetarian (see 1.F) and may result in released birds consuming native flora.

Mallard commonly eat aquatic plants. The total seed biomass in digestive tracts of shot birds did not differ between captive-reared and wild–born mallard in France although there were some differences in the frequency of some species in the diet (more *Ludwigia peploides, T. aestivum, Polygonum lapathifolium* in captive bred birds; more *Potamogeton pusillus, P. nodosus, Echinocloa* sp. In wild-born birds) and generally, released mallard exhibited dietary preference for anthropogenic food (waste grain or bait) (***Champagnon et al. 2012).

Released pheasants that had been fed on a more diverse diet for their first 6 weeks, and so were considered to have experienced an early life more similar to wild born conspecifics, had higher numbers of natural food items (i.e. not grains that constituted supplementary feed) than did birds that had been fed on commercial chick crumbs and pellets (**Whiteside *et al.* 2015)

E) Direct effects of foraging on invertebrates

Pheasants and red-legged partridges in the wild consume animals, generally insects, particularly when they are chicks or laying hens (*Beer 1988). In the captive rearing system, the diet of young birds pre-release is usually pellet-based containing added protein. Adult birds, even wild ones, do not need a high protein diet but it is thought that released birds probably retain an instinctive interest in insects (and perhaps small vertebrates – See 1.G) and will eat or peck at them if they are easily available. A study of annual diet composition based on faeces from wild-living pheasants on Brownsea Island showed that insects and other animals comprised ~5-15% of their diet between July and September with much lower proportions in the remaining months (*Lachlan & Bray 1973 in *Hill & Robertson 1988). A longitudinal study in the USA of pheasant crop/gizzard samples reported a similar annual pattern but with a peak in insect and other animal consumption between June and July with levels of <5% for the rest of the year (*Dalke 1937 in *Hill & Robertson 1988). A meta-analysis of 15 pheasant diet studies based on crop contents of 1663 birds collected during the spring reported that animal matter (no separation of vertebrates and invertebrates) varied from 0.9-26.1% with a weighted average of 7.2% (Stromborg 1979). In all three studies, the great majority of the reported diet was of cultivated or wild grains, fruits and seeds. Because of the appearance of invertebrates in the diet of gamebirds at certain times of year, several studies have investigated whether their releases deplete local populations of invertebrates, especially close to release sites where high densities of birds occur. Studies of the effect of likely predation by released gamebirds on invertebrate populations have used several different approaches and have produced mixed results

A correlative study using distribution maps suggested that UK 10 km^2 tetrads where pheasants were reported (according to the BTO bird atlas) were less likely to have certain butterflies present. ***Corke (1989) suggested that predation of caterpillars by pheasants may be causing this effect. In a scientific critique of this work (Warren 1989) described how the size, timings and behaviour of these butterfly larvae meant that they were at a low risk of predation, and that Corke's correlations were probably not causal.

An experimental predation study was conducted at a field study in a Dorset woodland in 1990. Colonies of third instar larvae of two butterfly species identified by Corke as high risk (pearl bordered and small pearl bordered fritillary *Boloria Euphrosyne* and *B. selene*) larvae were established in the wood in September at varying distances from a pheasant release pen (***Clarke & Robertson 1993). Each colony consisted of four larvae on a potted pansy sunk into the ground with tree grease around the pot rim to prevent escapes. 20 colonies were monitored for between four and nine days. Overall 95% of larvae were recovered and there were no relationships with distance to pen.

The study by Clarke & Robertson also undertook surveys of 50 woods in central southern England that had fritillary colonies recorded in an historical survey undertaken in 1970. The surveys showed that the proportion of those 50 woods still with colonies had declined by around 35%, but that the decline was the same in woods with pheasant releasing and those without. Although the survey found no evidence, Clarke & Robertson (1993) discussed the possibility that pheasants may have an adverse indirect effect on these specialist butterflies if the violet host plants in or very close to pheasant release pens are themselves affected. Sage et al. (2005) showed that these types of plants can be reduced inside release pens (See 1.B). In an earlier study, *Porter (1981) looked at 150 pheasant droppings collected from a site with pheasants and a high density of butterflies including marsh fritillary *Eurodryas aurinia* which revealed that only two samples contained caterpillar remains.

***Neumann et al. (2015) looked at ground-active invertebrates inside pheasant release pens at 37 woodland sites in southern England over two years. At each site they compared samples from 10 pitfall traps in the central area of the release pen with 10 pitfalls in an area of the same woodland around 300 m away. They also measured aspects of the ground vegetation. The mean release density in the study pens was 1500 birds per hectare, i.e. more than recommended (*Sage 2007). They found that conditions for invertebrates inside the pens were altered in terms of leaf litter and plant species composition with more disturbance-tolerant annuals and perennials than outside the pens. They found no difference in overall invertebrate abundance between areas inside and outside the pens. Carabid and Staphylinid species richness was also the same. However, the release pens had a different community of ground beetles with fewer large woodland carabid beetles and more beetles characteristic of arable fields and grasslands. This is probably due to a combination of altered conditions in pens and predation of beetles. Release pens commonly have reduced shade due to tree canopy management which may favour the ground beetles recorded. There were also more detritivores such as snails in the release pens that released more than 1000 birds per ha.

***Hall (In prep) carried out pitfall trapping transects on invertebrate populations inside 65 release pens and comparing them to transects outside of those pens 2 weeks prior to release, 4 weeks post-release, and 9 weeks post-release. Prior to release, pen interior invertebrate biomass and detritivores counts both inside and outside but close to pens were lower, while slug counts both inside and outside of pens were higher. When invertebrates were overall more abundant, total invertebrate biomass and slug and detritivore counts were lower inside the pens 4 weeks post-release. When invertebrates were overall less abundant, the main differences between pen interior and exterior transects were only seen 9 weeks post-release, with higher total invertebrate biomass and higher total invertebrate, slug, and beetle counts within pens. Pen interiors prior to release had more bare ground. This increased further both 4 and 9 weeks after pheasant release. The effects of pheasants on invertebrates within and near to the release pen would probably be a consequence of predation and/or micro habitat changes.

Effects of gamebird predation on invertebrates may not be restricted to release pen areas or feed sites. ***Pressland (2009) studied 17 matched woodland pairs in SW England, one with pheasant releasing and one without. Ground invertebrates were sampled using pitfall traps both inside and outside (in grass field) the wood-edge both before (May/Jun) and after (Sept) pheasant release. There were fewer insects overall caught in grass fields outside of the releasing woods compared to inside pens before releasing occurred (May/June sampling). There was no detectable difference in insect numbers in wood-edge plots with or without releasing and before or after releasing, and between any plot type after release. Some insect groups were caught more frequently within woods where pheasants were released and some were more common in woods without releases, although these variations were not easily explained. Faecal analysis indicated that pheasants sometimes ate invertebrates and that the proportion in their diet increased in spring when more insects were available. The pheasants themselves were, of course, much less common in the spring (typically under 10% of the release) than in the early winter. Caterpillar biomass along woodland tracks was not strongly linked to pheasant density but to plant species richness and temperature (which correlated with pheasant density).

***Devlin (2019) set up one 25m pheasant exclosure and defined one control plot at each of three sites (woodland or grassland) in an area of mid Wales several km from several pheasant release pens. The open topped exclosure consisted of 1.5 m high chicken wire with an unknown number of 5cm holes at the base. He monitored the sites between March and July when dispersing pheasants might be expected to encounter them. Pheasants were not detected at one site. He counted pheasants and collected insects inside and outside the exclosure at each site and found fewer Orthoptera at one site in the exclosure compared to the control plot. He suggests this provides clear evidence that pheasants modify invertebrate communities from both a diversity and abundance perspective. There was a negative relationship at one site (over the duration of the study) between the density of free-living pheasants and both the abundance and diversity of invertebrates detected there at each sampling point. The presence of pheasants did not explain differences in abundance of Coleoptera or Lepidoptera within sites, although at one site (termed the upland grazing site), the density of pheasants observed each week was negatively related to abundance of Orthoptera at those times. The exclosures would have excluded other animals and there was no comparison between the control plot and exclosure to see if the exclusion of rabbits, hare or deer had resulted in altered vegetation structure and hence invertebrates. However, because there was only one fenced and one control at three different sites, there was no replication (the same plot was sampled weekly) and therefore it is difficult to attribute any observed difference between the two plots at each site to any measured factor.

Another study looked at the possible effects on invertebrates of high density releasing onto arable ground alongside sensitive chalk grassland habitats at six sites in central southern England, three with releasing and three without (***Callegari 2006a,b). Released red-legged partridges were found in their highest densities on the chalk grassland study sites in September and October. These birds were still only a small proportion of the total released in the vicinity of the grassland. Observational work established that the partridges (and pheasants) spend a considerable amount of time in feeding-related activity on the grassland in September following release and initial dispersal, which then declined into the winter. Gamebird exclosures were set up at each of the six sites before gamebird release and invertebrates sampled within them and in control plots the following spring. Of the nine invertebrate groups considered, the only difference recorded was a reduced number of Diptera species (but not abundance) emerging form the control plots. Analysis of faecal samples collected from two releasing sites found that 54% of pheasant and 44% of red-legged partridge samples had invertebrate fragments in September and then reducing to very small percentages by January as most of the invertebrate community hibernates. Flies, ants and weevils were common on the sites and common in the pheasant and partridge faecal samples. Other insect groups were eaten and in general the diet of the pheasants was more wide-ranging than that of the partridges. The diet of both species was determined by availability i.e. they were not eating particular insect groups preferentially. Although the gamebirds were eating invertebrates on the grassland following release in autumn, they did not appear to affect spring invertebrate densities. The study also used control exclosures over horseshoe vetch *Hippocrepis comosa*, the food plant of the Adonis blue *Polyommatus bellargus* butterfly. No difference was found in the numbers of emerging butterflies between the plot types. However low productivity in the Adonis Blue at the site may have compromised the study's ability to detect an effect if one had been present (***Callegari et al. 2014).

*Holland et al. (2005) reviewed the invertebrate diet of a suite of farmland birds including red-legged partridge but not pheasant. They found that the proportion of invertebrate food in the diet of adult red-legged partridge was 0% while for chicks the proportion was 35%. The key chick food item was Hemiptera.

***Jensen et al. (2012) investigated diet via the isotope signatures in feathers from wild and released pheasant populations and compared the levels and variation in isotope signature amongst the populations. Three feathers from each of five birds from each of three populations (two reared and one wild) were analysed using mass spectrometry. Clear differences were found despite the small sample size. Released pheasants had higher for $\delta^{13}C$ and this was attributed to maize in the diet of those birds, which wild birds did not eat. The released pheasants also had higher $\delta^{15}N$. This indicated that the wild pheasant had significantly greater amounts of invertebrates in their diet compared to the wild ones. It was not possible to tell if the released birds ate no invertebrates, or just relatively small amounts.

No such similar studies have been conducted on the effects of mallard diet on invertebrate populations. The diet of wild mallard may indicate what released mallard are likely to eat and hence what fauna and flora may be susceptible to predation by released mallard. Wild adult females and ducklings largely eat high-protein invertebrates in spring/breeding season with a higher proportion of animal matter in the diet during rather than outside the breeding season. Mallard ducklings aged 0-3 days old ate Diptera and Coleoptera, but by 12-24 days they had switched to feed on Mollusca and *Daphnia*. Post-fledging, mallard diet is predominantly of seeds including cereals (*Dessborn *et al*. 2011a).

Circumstantial evidence that high levels of waterfowl, presumably including mallard, affects invertebrate abundance is provided by work in Germany in which invertebrate abundance was observed to be higher in marshes where shooting was permitted compared to marshes that were protected, with the assumption being that disturbance and/or mortality reduced waterfowl numbers in hunted areas and so reduced predation on invertebrates (*Reicholf 1973 in *Callaghan 1996).

F) Direct effects of foraging on small vertebrates

The animal component of gamebird diet (described above 1.E,F), may include limited quantities of small vertebrates. The Amphibian and Reptile Conservation Trust (ARC) suggests that all six British reptile species could be vulnerable to predation by pheasants (although partridges are not excluded) and that this could affect their conservation status locally. According to ARC there are anecdotal observations of reptile predation by pheasants. The reptile management handbook (*Edgar *et al.* 2010) mentions possible pheasant predation and refers to general GWCT advice on releasing good practice. *Blanke & Fearnley (2015) cite earlier work, often anecdotal, that suggests a range of predators of sand lizards *Lacerta agilis* (38 bird species, 12 mammals, four reptiles and three amphibians) including pheasant. Pheasants are released into woodlands while reptiles tend to occupy more open habitats. However, pheasants are encouraged onto open ground to facilitate driven shooting and certain reptiles, common lizard *Zootoca vivipara*, slow-worm *Anguis fragilis* and adder *Vipera berus* in particular, will use open habitats at woodland edges, in woodland clearings and along woodland rides (e.g. Edgar et al. 2010) so there is scope for spatial overlap. The ecology of the six reptiles suggests that adults and juveniles can be exposed to pheasants in autumn after birds have dispersed from release pens and before reptile hibernation begins. Both adult and juvenile reptiles continue to be active in good weather throughout October (Beebee & Griffiths 2000, Edgar et al. 2010) but reducing temperatures will affect basking behaviour. Reptiles have responses that minimize risk to predators (Blanke & Fearnley 2015), but they are more sluggish on colder days. In the spring, numbers of remaining released gamebirds are typically 0-10% of the initial release size so the risk of conflict is lower. However, emerging reptiles post-hibernation are sluggish and less able to avoid predation and concentrations of adders emerging from hibernation in spring are thought to be particularly vulnerable. Pheasants are suggested to be attracted to the sinuous wriggling movements of snakes (Hand 2020).

There was no mention of reptile (or other vertebrate prey remains) in dietary studies including samples from > 2500 individuals surveyed in the UK and USA (*Dalke 1935, *Fried 1940, *Wright 1941, *Stromborg 1979, *Whiteside *et al.* 2015).

In 2012, using DNA Identification techniques, no reptile fragments were found in a sample of 50 pheasant droppings collected from a grassland / heathland area that contained released pheasants and reptiles (**Dimond *et al*. 2013).

There are observational accounts of pheasant adder interactions recorded by *Hand (2020). At a well-known site the introduction of pheasants coincided with the decline of adders. Individual adders have been found with injuries reported to be by pheasants, a pheasant was observed pecking an adder and there are photographs of a pheasant eating an adder.

***Berthon (2014) found that juvenile penned pheasants preferentially pecked at reptile shaped plastic objects compared to similar plastic objects in non-reptile shapes. Adult pheasants did not show a preference. Berthon also recorded no reptiles under refugia set out in a sample of pheasant releasing woods at three sites in the New Forest area but did record a small number of grass snakes *Natrix natrix* and slow worms in refugia in three non-release woods.

G) Direct effect of carcass availability on predators

It is likely that the potential food resource of released gamebirds will attract or sustain certain predators like foxes, corvids and raptors. In theory, generalist predators like these will respond numerically (i.e. increase in number) and/or functionally (switch to eating more pheasants) to an increase in abundance of a prey species such as released pheasants (Solomon 1949; Robertson and Dowell 1990). In early autumn each year the total biomass of released pheasants and partridges in the UK will be around 40,000 and 4,000 tonnes respectively. Using estimates of breeding birds in 2013, the spring pheasant biomass was calculated to be 3,740 tonnes (including wild and released birds), exceeded only by wood pigeon (**Blackburn and Gaston 2018). On average around 60% of pheasants released for shooting in the UK die of causes other than being shot. Most of these are predated but the corpses of others will also be available to scavenging predators (*Sage *et al*. 2018c). Most (~ 70%) recorded predation of reared pheasants in the UK is attributed to foxes (**Robertson 1988; **Sage *et al*. 2001; **Woodburn 1999). Raptors are implicated in < 1% of deaths of newly released pheasants, but on some sites, they are responsible for > 10% of deaths (**Parrott 2015). Predation is especially common immediately after release. One release pen in Ireland suffered the highest rate of loss (48%) in the first 10 days after the birds left the release pen (**Robertson 1988). Therefore, we may expect to see relationships between the scale of releases and the populations of these gamebird predators.

In their review of predation of birds, *Roos *et al.* (2018) found that the overall density of foxes in England / UK was higher than in eight other European countries (but not Italy and Spain). They speculated that this was because of habitat suitability factors, high farming production, lack of apex predators and the release of pheasants and partridges providing a food source throughout the winter. They also found that crow *Corvus corone* density was higher in the UK than in other European countries and suggested the same set of reasons.

There are five studies looking at relationships between gamebirds and their main predator, foxes.

***Robertson (1986) intensively studied a large pheasant release in Ireland. He searched for fox droppings fortnightly throughout 1984 and found four times as many within 200 m of the release pen in August and September (poults were placed in pens on 4th August) than before, and that these droppings contained more pheasant fragments.

***Madden et al. (in prep) looked at relationships in the public reporting of roadkill data (https://projectsplatter.co.uk) for foxes and pheasants, in relation to the presence or absence of commercial game shoots in 10kmx10km tetrads. The game shoot data were acquired from the main advertising portal for shooting in the UK 'Guns on Pegs' and 930 shoots, estimated to be around a third of the total, were assigned to tetrads. Around 7,000 records of pheasants and foxes were also assigned as proportions and associations with each other and with tetrad shoot classification (commercial shoot or not) were investigated. There was no overall difference in the proportion of fox roadkill with and without a commercial shoot. The overall proportion of foxes reported as roadkill decreased as the proportion of pheasant roadkill increased, so areas with more pheasants had fewer foxes. With no positive relationships Madden et al. suggest that these results show that these predators were not more common in tetrads with commercial shoots and were not attracted to roads to scavenge. The authors suggest that fox control at release sites has at least some effect at most sites which leads to a net decrease in generalist predators (see 2.E). Looking at temporal trends they also found that foxes were more frequently killed on roads in July and August, when pheasants are first released and largely confined to pens. They suggest that this might be because the releasing encourages greater movement of foxes in the area away from release points by gamekeeping activities. Robertson (1986) thought that the foxes were attracted to the recently released birds even though they were initially inaccessible. Young foxes also naturally disperse from territories at this time.

The GWCT's National Gamebird Census (NGC) indicated crudely that in each of 11 UK regions, as pheasant releasing has increased so has the fox bag i.e. the numbers of foxes killed per 100 ha (***Reynolds 1994). The article describes this as a striking correlation but explains that it does not necessarily follow that fox density or breeding success increases with increased releasing. For example, it may be that dispersing foxes remain longer on estates with released birds, and then are more likely to be culled.

***Porteus (2015) estimated immigration rates (foxes per km^2 per year) into managed NGC estates from annual fox culling rates (see also Porteus et al. 2018).

These immigration rate estimates varied significantly between seven landscape types (e.g. the estimate for arable was four times greater than the one for upland). As well as being related to fox density at the landscape scale he found that estimated immigration rates for each landscape type correlated with the mean density of pheasants released in each landscape. So, for example, there were 15 times as many released pheasants and four times as many foxes on arable than upland. One of several hypothetical causes of this correlation was that foxes were responding directly to the releases, but landscapes suitable for foxes are also suitable for releasing and vice versa and there may be other correlating landscape drivers involved. The main aspect of Porteus (2015) was to understand the effect of fox culling on another dataset of 22 sites that contributed to the GWCT Fox Monitoring Scheme where lamping / shooting effort was recorded daily over the five years 1996 – 2000. Five of these sites also provided data on gamebird releases via the NGC. At each site lamping effort was used to model fox density over time and estimate demographic parameters. Fox immigration rates (foxes per km^2 per week), and the carrying capacities of foxes appeared positively related to the NGC estimates of the number of gamebirds released, the gamebird bag and the number of gamebirds not shot. With only five sites these relationships were not statistically significant. These aspects of Porteus (2015) are currently in the process of being written up for peer review.

***Beja et al. (2009) recorded foxes, amongst other taxa, on 12 shooting estates and 12 similar areas in Portugal. Most of the shooting estates released red-legged partridges. The survey method involved only one diurnal transect survey at each site in Spring but they reported that foxes were more common on the shooting estates even though many of the sites targeted them for culling.

Similar questions have been asked of predators likely responsible for lower levels of released gamebird predation, specifically raptors. The buzzard *Buteo buteo* has increased substantially in population and range since the 1970s, in parallel with the widespread increase in gamebird releasing and some buzzards are sometimes interested in pheasant release pens (**Kenward *et al*. 2001; Parrott 2015; Swan 2017). Reductions in persecution, the banning of certain pesticides and the rabbit *Oryctolagus cuniculus* population have been suggested as the most likely drivers of this (Parkin & Knox 2010) but it is possible that gamebird releasing has also contributed. By studying home ranges, Kenward et al. (2018) looked at habitat selection by buzzards which indicated that rough ground, meadow and suburban land were most important. Coniferous and broadleaf woodland were not important which by inference suggests that pheasant release sites (usually situated in such woodland) weren't either. ***Swan (2017) found that buzzards nested at greater density in areas with more pheasants and rabbits. However, only rabbits were caught in proportion to their abundance and only rabbit provisioning rate was associated with buzzard productivity.

***Pringle et al. (2019) identified a series of positive (and some negative) correlations or associations between pheasants/partridges and buzzards/some corvids using spatial datasets from the Bird Atlas 2007-2011 (Balmer et al. 2013) and from APHA poultry register data, which identifies gamebird holding facilities in Britain including both rearing sites and release sites (although Pringle et al. assumed it was just rearing sites). Some of the model relationships suggest straightforward responses by the predators to the presence of gamebirds. For example, the abundance of pheasants in the breeding season was spatially correlated with buzzard and crow abundance, and temporal changes in red-legged partridge numbers were positively associated with buzzard numbers. Other relationships were harder to explain, such as the negative spatial correlation between magpies *Pica pica* and pheasant abundance or positive temporal relationships between jay *Garrulus glandarius* and red-legged partridge numbers. These suggest different factors may be involved in the correlations such as particular habitat factors not fully accounted for in the analysis. Reported relationships were frequently not linear and tended to have very low R-squared values and small effect sizes, with raw avian predator abundances being plotted against logged gamebird abundances suggesting that any actual changes in predator populations are small compared to changes in gamebird numbers. The paper suggests further work to explore these relationships.

***Kenward (1981) radio tagged 43 goshawks *Accipiter gentilis* in the vicinity of a large released pheasant shooting estate in Sweden. He found that goshawks were the main cause of over-winter mortality of pheasants. Goshawks on the release site were at a higher density, had smaller ranges and were heavier than goshawks elsewhere.

Released mallard may be predated by foxes but mallard are also predated by a range of different fauna depending on the life stage, with nests being predated by corvids (Mallard eggs constitute 13% of shell fragments found in carrion crow diet in Poland) (*Opermanis *et al*. 2001) and a suite of mammalian predators ([*]Padyšáková *et al*. 2011); adults and ducklings are killed on the water by pike *Esox lucius* ([*]Dessborn *et al*. 2011b) and Marsh Harriers *Circus aeruginosus* and American mink *Neovison vison* (Opermanis et al. 2001). Experimental increases in densities of (simulated) duck nests in the Czech Republic were not accompanied by increased rates of predation on them in that season, suggesting that short-term, local increases in density perhaps resulting from the survival of released mallard does not immediately increase risk of predation of the nests of similar resident ducks or a local, short-term increase in predator activity (*Padyšáková *et al*. 2011).

H) Direct effects of genetic disruption of wild populations

In situations where gamebirds are released into areas containing wild conspecifics or close relatives with which they can hybridise, there exists the possibility that the introduced birds may mate with the residents and disrupt local genotypes. The red-legged partridge from the Iberian Peninsula, France and Italy was introduced into the UK in the 17th century for shooting purposes (Lever 1977) and has bred in the wild since then. In the 1960s game farmers rearing the red-legged partridge in the UK and elsewhere in Europe began importing the chukar partridge *Alectoris chukar* (from further East), which they crossed with *A. rufa* to produce a more productive bird for rear and release. However, it was soon recognized that after release into the wild, these hybrid birds were breeding with genetically pure wild *A. rufa* population throughout its natural range and where it had been introduced (**Blanco-Aguiar *et al*. 2008; **Casas *et al*. 2012). This has resulted in the virtual loss of the native *A. rufa* genome (or at least one of its three subspecies *A. r. rufa* see Madge & McGowan 2002). Recent work looked at specimens of wild *A. rufa* in Norfolk, where historically many estates have not released and which is regarded as a stronghold of wild *A.*

rufa and grey partridge. The aim was to establish if there was a remnant pure population that could be used for conservation management of the species (Barbanera et al. 2015). Preserved samples from 19th century Norfolk were genetically pure but 94% of a sample of modern Norfolk birds had chukar ancestry. While there remains at least one genetically well-preserved and self-sustaining population of *A. ruf*a (in Italy), this rear-and-release practice has resulted in genomic biodiversity loss, most notably in *A. rufa's* natural range (**Barbenera *et al*. 2010) but also throughout the UK where the bird was introduced (**Barbenera *et al*. 2015). The release of *A. chukar* or its hybrids was banned in 1992.

In locations where mallard are not native, their release is commonly accompanied by hybridisation with native *Anas* species, resulting in perhaps as little as 15% of the duck population comprising pure native species genotypes (refs in Callaghan 1996). In the UK, where mallard are native, interbreeding with wild conspecifics is a more likely outcome. In the Camargue in France, where large scale release of mallard has occurred for around 40 years, the genetic signature of birds reared from mallard farms were still clearly distinct from those of presumed wild-born mallard caught in no-shooting areas, despite significant rates of hybridization between reared and wild-born birds (**Champagnon *et al*. 2013). This suggests that admixture and introgression may be constrained because of poor survival of released birds in the wild, being reported in central France as being about half that of the first year survival of local wild mallard (***Champagnon et al. 2016), or because captive-bred mallard preferentially mate with a partner of the same strain (***Cheng et al. 1978). A broader survey of mallard populations in northern mainland Europe also revealed clear genetic differentiation between farmed and presumed wild-born mallards, but with admixed individuals within the wild-born population suggesting introgression (**Söderquist *et al*. 2017).

2. Associated effects

A) Associated effects of land management on soil, water and air

There are an estimated 9,900ha of flight ponds managed for shooting in the UK (PACEC 2006) but this does not, and probably cannot, differentiate ponds used for release and those used to attract wild duck. When considering individual shoots, there is a reported increase in the percentage of shoots that engaged in land management of water/marshland (presumed motivated at least partially by the desire to shoot duck, although again it is not possible to distinguish released from wild bird shooting) from 18% of responding shoots in 1977-78 to 49% in 1991/1992 (***Cox *et al.* 1996). This almost tripling of the extent of management of wetlands crudely matches the magnitude of increase in releasing and harvest of ducks over that time. It also supports the inference that a substantial minority of UK shoots are actively managed to provide duck shooting although it cannot distinguish between shooting or released or wild mallards, or indeed between the quarry species being shot. The construction and management of open-canopy ponds (the type most usual for release and holding mallard) resulted in higher bird diversity (64 vs 37 species), abundance and activity compared to un-managed ponds. This effect was predominantly driven by management action rather than the surrounding landscape features (Lewis-Phillips *et al*. 2019).

B) Associated effects of land management on non-woody plants

Game managers may plant areas of annual or perennial plants in association with their gamebird releases. Winter game crops are widely planted on released shooting estates to provide feed areas and to hold pheasants and red-legged partridge as part of a shoot (GCT 1994). In addition, there are a range of field edge habitat management practices undertaken, perhaps by shoots releasing gamebirds, but especially by wild pheasant and partridge shoots in some, usually drier, parts of the country (Tapper 1999). These shoots will plant brood-rearing crops (supplying invertebrates and cover for young chicks) or employ conservation headlands in the crop edge zone, as well as maintaining plenty of nesting cover along hedge-banks and field corners and creating in-field beetle banks.

Of those shoots responding to surveys, many report planting cover crops or mixes. In the Savills Shoot benchmark survey 2018-19 (**Teanby *et al*. 2019) of 88 UK shoots, 81% have established buffer strips and/or field corners; 86% have established wild bird seed mix; 50% have established pollen and nectar mix; 47% have established conservation headlands. A survey of 65 farms in Essex (***Howard & Carroll 2001) reported that 86% of shooting farms planted game crops compared to 0% of non-shooting farms. More shooting farms were also selective in their use of chemicals (58% vs. 30%) and left edges unsprayed (66% vs. 33%) than non-shooting farms.

The areas that are covered by this planting may be reasonably large. Based on a questionnaire survey of professional gamekeepers, most self-selecting respondents managed larger shoots (average area 1900 ha) (**Ewald & Gibbs 2020) reported a mean area of 17ha of game cover crop/shoot not in a stewardship agreement i.e. about 1 % of the total area of the sample. This included a mean area of 7.7ha of maize cover/shoot not in a stewardship agreement and a mean area of 11ha of wild bird cover crop/shoot not in a stewardship agreement. All these areas may be greater if areas in stewardship agreements were included. Respondents to the PACEC (2006) survey reported that the average lowland pheasant shoot contained 8.9 ha of game crops.

Recent mapping work at three very large shooting estates in the Exmoor region that covered around 60 km^2 indicated 4.4%, or 256 ha of the estates were winter game crops (***Sage 2018a,b). The average size of 143 game plots was just under 1.8 ha, much larger than plots on shoots in other regions. One quarter (65 ha) was maize while three quarters contained 15 other crop types the commonest being kale (28 ha), miscanthus (24 ha), root crops 20 (ha) and wild bird mix (18 ha). Most of the farmed land in the region is grassland for livestock grazing. Comparisons with cropping maps from 1967/8 showed that the game interest on the Exmoor estates in 2017 contributed to a cropping pattern in the landscape that was more like that of the 1960s than of modern farmland without a game interest.

C) Associated effects of land management on woody plants

Woodland is commonly used both as the site of pheasant and partridge releases, as preferred habitat for them to inhabit before and during the shooting season, and to force the birds to fly high over the guns when flushed. This motivates its retention, management and creation. At least 14% of all woodland in the UK is to some extent managed for gamebirds, primarily pheasants (*Gilbert 2007). In Scotland and Wales it is around 5% each, while in England it is 28%, more than is managed for wildlife conservation (*Gilbert 2007; see also ***Firbank 1999). Some have commented that shooting estates are more likely to retain and manage existing woodlands and plant new ones (Rackham 2003; ***Oldfield *et al.* 2003).

The kind of management practiced by gamebird releasers is summarised in the early *Game Conservancy Green Guide* booklet series, e.g. GCT (1988), which provided advice on how to plant or manage neglected woodlands for releasing and shooting and provide a context for what has motivated game managers to do in their woods in recent decades. This included the design of sloping woodland edges, providing shrubby cover and the creation of flushing and rising points. Robertson et al. (1993a; 1993b) and *The Game Conservancy Annual Reviews* (1987 to 1993) reported on a programme of work that studied the characteristics of woodlands that made them good for pheasants (see also Robertson 1992 for a summary). Radio-telemetry showed that pheasants spend the majority of their time within 20 m of the woodland edge, so small deciduous woodlands of around 2 ha or less held the highest densities of pheasants in winter and summer. Woodland rides needed to be at least 30 m wide to create a woodland edge (Robertson et al. 1991). Using counts of flushed birds on drives, the work showed that woodlands with abundant shrubby cover between 0.3 – 2 m in height were favoured by pheasants during the winter (Robertson et al. 1993a). Edge length may be increased by the creation of woodland rides, strips of land that are carved into the area and which may be preferentially used by pheasants, permit game managers access to the woods and can be placements for guns.

Some indication of the woodland planting behaviour of game managers, and whether they are following the advice summarised above, can be obtained by surveying them. Using questionnaire surveys of land-owners, a survey by Reading University reported that 67% and 56% of landowners retained or planted woodlands of less than 10 ha with pheasants in mind (**Cobham Resource Consultants 1983). Savills Shoot benchmark survey 2018-19 (**Teanby *et al*. 2019) reported that of 88 UK shoots that between them put down > 1,000,000 birds and shot >1,800 days, 86% have managed woodland in the last 10 years averaging 308 acres each and 52% have planted woodland in the last 10 years averaging 24 acres each. Respondents to the **PACEC (2006) survey reported that the average lowland pheasant shoot contained 61 ha of woodland.

The actions of game managers should be compared with the woodland management of other landowners. ***Firbank (1999) classified 320 1-km^2 grid squares in England as game or non-game based on site visits. They found that game squares had more and larger woods than non-game squares, especially in the East, although the differences were small. In a sub-sample of 12 game squares there had been a significant increase in woodland since the 1960s (none had less), whereas there had been no increase in 12 non-game squares (3 had less). ***Duckworth et al. (2003) reported a subset of these data and concluded that game shooting encouraged landowners to retain existing woodlands, and to plant new ones, during the period 1960-1990. A survey of 965 responding keepers (including fulltime 52%, part time

18% and amateur 30%) (**Ewald & Gibbs 2020) reported managing over 1,625,216 ha land area (but this includes England & Scotland where moorland may predominate) including 192,051 ha woodland. This equated to a mean area of woodland under their management of 244.5ha per shoot or 11.8% of shot ground, similar to the national average coverage of 13%. Of the surveyed keepers, 71% reported planting trees, covering 47.3 ±11.4ha and 45% of these respondents cited shooting as prime reason for planting. ***Oldfield *et al.* 2003 surveyed 65 landowners in three study sites in England and reported that those that maintained gamebird shoots (and participated in fox-hunting) had a higher proportion of woodland on their farms (~6-7%) than neighbours who did not participate in field sports (<1%). Those landowners also were more likely to have planted woodland, but this effect was not seen for hedge planting.

The strongest evidence of the motivation that release and shooting has for game managers comes from closely paired comparisons between farms or estates that release gamebirds compared to those that do not. ***Cox et al. (1996) conducted a survey of 356 farms with shooting in Buckinghamshire, Cumbria, Devon, Gloucestershire and Nottinghamshire and a matching sample from the same area of 296 farms without shoots. They reported that 31% of farms with shoots had >5% woodland compared to 14% of farms without shoots. However, 29% of the shooting holdings did not manage their woodland for any game conservation purposes, although around 80% of woodland in the sample was shot over. The total acreage of woodland planted over the previous 10 years comprised 1,227 ha (90%) on 104 farms with shooting compared to 128 ha (10%) on 34 non-shooting farms. They then refined their survey to specifically compare farms where pheasants were released with farms where shooting occurred but pheasants were not released. On those farms where pheasants were released, 61% of respondents reported planting new woodland in the previous 10 years compared to only 21% of those not releasing pheasants. Releasing farms were also reported to have more conservation headlands, hedges, ponds and marshland than non-releasing shooting farms, but data on areas were not provided. In the 1990s, in a survey of 261 farms, 61% of holdings that released pheasants planted new small woodlands (1 – 5 ha) compared to 21% of non-release sites (***Short et al. 1994). 31% of the release sites had more than 5% woodland cover compared to 14% of the non-release sites. A survey of shooting (n = 38) and non-shooting farms (n = 27) in Essex by ***Howard & Carroll (2001) reported that woodland occupied 8.0% of land on shoot farms and 1.2% on non-shoots farms, such that woodland on shoot farms accounted for 95% of woodland in the survey. Woodland planting was also more common on shooting farms.

Gamebird release and shooting also stimulates the management of woodland as well as its creation. In the past, areas were commonly established along rides within woodlands to promote natural forage areas for pheasants which were strawed to encourage foraging. In botanically rich rides, and especially where the straw was left, this was thought to smother woodland plants, enrich soils and bring weeds into the woods (*Robertson 1992). The use of hoppers for feeding is much more common today (see 2.D). Rides are considered a priority in forest management for conservation (Ferris & Carter 2000). Game managers maintain rides for access, as open areas for pheasants, and on some shoots as places to locate a line of guns.

Some of the techniques to improve woodlands for pheasants e.g. coppicing and skylighting are considered to be beneficial to other wildlife particularly birds (Amar et al. 2006; Fuller 2005; Fuller & Henderson 1992; *Ludolf *et al*. 1989a). ***Short *et al*. (1994) reported that 58%, 36% and 41% of 261 holdings that released pheasants managed rides, coppiced trees and planted shrubs respectively. Equivalent figures for non-release holdings were 8%, 5% and 10%. ***Cox *et al.* (1996) reported higher instances of coppicing (36% vs. 4%), ride management (52% vs. 6%), selective tree planting (46% vs. 9%) and shrub planting (39% vs. 8%) on farms with shooting that released vs. did not release pheasants.

In 2006, 159 lowland woods were surveyed in southern and eastern England during spring-summer, including their rides. Half the woods were managed for game while the other half had no releasing or feeding of game for at least 25 years. 40% of study sites were randomly selected while others were from the GWCT contacts database (***Hoodless & Draycott 2008, ***Capstick *et al*. 2019b). Tree species composition did not vary between the two woodland types. Rides were not longer, but were about 20% wider in game woods (10.5 m) compared to non-game woods (8.8m). As a consequence rides occupied a higher proportion of woodland area in game woods (13%) than non-game woods (8%). 40% of rides in game woods had an open canopy compared to 30% in control woods.

Hedgerows are widely used by game managers to link woodland release pens to holding cover, usually game crops, to facilitate shooting (GCT 1988). It is likely that many hedgerows today, like woodlands, were planted or retained for game interests in the past. ***Firbank (1999) reported more hedges, more complete hedgerow networks and greater connectivity between hedges and woods on game areas than on non-game areas. Common farmland/hedgerow birds and butterflies were 10% more abundant on game areas per unit area of habitat than on non-game areas in this study. ***Draycott et al. (2012) looked at the abundance of hedgerows on 97 release-based pheasant shoots and on 53 farms with no releasing using satellite imagery. The shooting estates had between 10% and 65% more hedgerow per square kilometre than farms with no releasing. Using ground surveys, the size of their hedges was similar on game and non-game woods but on shooting estates hedges were more frequently buffered from adjacent fields by grass margins or other uncultivated strips. ***Howard & Carroll’s (2001) survey of Essex farmers reported that 83% of farms with shoots retained and managed hedges compared to 52% of farms without shoots.

Game managers surveyed in the **PACEC (2006) report said that they would change the way that they managed land and wildlife, including woodland planting and maintenance, if sporting shooting stopped. 59% said that they would manage their habitats differently or very differently, while 20% said that they would stop all habitat and wildlife management. We are not aware of any studies that confirm whether such actions actually occur at the cessation of shooting on an estate or farm.

D) Associated effects of supplementary feeding on small vertebrates

Providing supplementary winter food for released gamebirds through feeders is practised on most release-based shoots and this may additionally support small vertebrates. For pheasant and red-legged partridge, this typically involves the provision of grains dispensed from feeders or scattered on straw in woodland rides. Supplementary feeding for released mallard typically occurs via direct spreading of cereals and root crops either into ponds or along their shores. In a survey of the 88 participating shoots (**Teanby *et al*. 2019), 86% feed their birds after the shooting season and 14% have taken up a supplementary feeding agri-environment scheme. A survey in 2019-20 of 965 keepers revealed that ~85% of those on lowland shoots reported continuing to provide supplementary feed after the shooting season, mainly stopping in April (**Ewald & Gibbs 2020). The mean amounts of feed supplied per shoot comprised 40 tonnes/(lowland) shoot during the shooting season and 6.5-8 tonnes after the shooting season. This was distributed via a network of feeders set at densities of 3.1-14.4/100ha. **Larkmann et al. 2015 calculate that shoots across the UK supplement 566,000 tonnes of grain/year and additionally produce 180,000 tonnes of seeds from cover and mix crops annually.

**Sanchez-Garcia *et al*. (2015) used trail cameras on 260 spiral dispenser drum feeders at three sites with modest pheasant releases plus wild grey and red-legged partridges. They found that pheasants accounted for around 20% of all photos taken of animals using the feeders, grey partridges 5% and red-legged partridge 2%. Of the other birds using the feeders wood pigeon *Columba palumbus* accounted for 17% of the photos, blackbird *Turdus merula* and dunnock *Prunella modularis* 5% each and yellowhammer, which occurred on all three sites, 4%. In total 33 bird species were recorded. Other UK BAP (1995) species recorded included house sparrow, linnet *Linaria cannabina*, song thrush *Turdus philomelos*, and starling *Sturnus vulgaris*. Fourteen mammal species were also recorded using the feeders Sanchez-Garcia *et al*. (2015). The commonest were brown rat *Rattus norvegicus* (17%) and mice (10%). Others included deer, hares *Lepus europaeus*, rabbits and grey squirrels *Sciurus carolinensis* plus the occasional stoat *Mustela erminea* or hedgehog *Erinaceus europaeus*. Overall, the feeders were used slightly less than half the time by gamebirds and songbirds and slightly more than half the time by non-target birds and mammals. Sanchez-Garcia et al. (2015) provide management suggestions for minimising use of feeders by pest or other non-target species especially brown rat, which use feeders only along hedges or wood edges and take more time than birds to find feeders again when they are moved.

In his study of farmland birds in Sussex ***Brickle (1997) looked at use of winter habitats and of game feeders located in wood edge / scrub and game crop habitats. He found that yellowhammer, corn bunting, reed bunting *Emberiza schoeniclus* and linnet all spent more time around feeders located in those habitats than other available habitat patches without feeders.

In Spain, grain is commonly provided on release based red-legged partridge shoots. ***Caro et al. (2015) compared the abundance of a variety of wild birds on 12 shooting estates (8 released red-legged partridge shoots) and 12 matching areas with similar sizes and land uses but no game management. All estates provided feed, while some also established game crops and controlled predators. Game management was associated with higher abundance of raptors and ground-nesting birds. There was no relationship for other guilds and species. In another study

(***Estrada *et al.* 2015) at 54 red-legged partridge shooting estates in Spain the abundance of granivorous species (sandgrouse) increased significantly with the density of feeders.

*Siriwardena *et al*. (2007, 2008) looked at relationships between farmland birds and seed provided on the ground experimentally (i.e. not game crops) at 110 sites in England over two years. They found that the peak use of supplementary food was in January or February for most farmland bird species. Declines for yellowhammer, robin *Erithacus rubecula* and dunnock were less steep on sites with more supplementary feeding and numbers of several other species appeared to increase. The study did not show direct improvements in breeding success in fed areas compared to controls. *Siriwardena *et al*. (2006) indicated that most birds benefitted if the food resources were widely distributed i.e. more than 1km apart especially if standing crops (such as game crops) were used rather than food patches. They also conclude that current farming practices, including the agri-environment prescriptions of the time (without a game interest and associated game feeding) do not provide enough food in late winter for these birds. This suggests that if shooting estates maintain feed points following shooting, as required by the Code of Good Shooting Practice, over-winter survival and subsequent breeding numbers of seed-eating farmland birds are likely to increase.

***Larkman *et al*. (2015) correlated changes in the populations of 14 species of seed-eating farmland birds with changes in the numbers of released pheasants between 1966 and 2010s. During this time, when pheasant releases increased, small native seed-eating species declined while large native seed-eating species populations increased. Declines in seed eating populations were more weakly correlated with an index of general agricultural intensification. They then conducted functional response modelling of foraging on seed supplies at low density (as might be found naturally from plants) and high density (as might be found at feeders). They concluded that pheasants (and large seed-eating native species) might benefit from high density seed deposition and also depredate low density seed sources, whereas small seed-eating native species might not benefit from high density seed sources (i.e. feeders) and faced landscapes in which low density sources might have been depleted by large seed-eaters. The authors propose that this could explain population declines in small seed-eating species.

E) Associated effects of predator control on predators

Game managers commonly control predators likely to kill their quarry species. Lethal control may be practised over the entire game shoot area throughout the year to reduce predation of the quarry species and thus improve the breeding success of wild-breeding gamebirds. This was certainly the case historically (Tapper 1992), but with increased attention paid to gamebird releases over wild bird management, control may be more focused on protecting released gamebirds when they are young and naïve, with efforts concentrated around the release pens and time when releases happen. Data on control effort may reflect both motivations and it is may be difficult to establish the precise influence that gamebird release alone exerts on predator control. As described above (see 1.H), the common predators of released gamebirds are foxes, which may be legally controlled, and raptors, which may not be legally controlled without specific licence.

i) *Legal Predator Control*

*Ewald and Gibbs (2020) survey of UK gamekeepers (which did not distinguish between those releasing gamebirds or managing wild game populations) reported that 682 respondents spent an average of 1596 (±153) hours annually on predator control. A larger sample of 740 respondents reported how their predator control efforts had changed over the past five years, with 50% reporting no change, 39% that it had increased (by an unspecified amount) and 11% that it had decreased. More than 90% of respondents with common predators (e.g. fox, rat, grey squirrel) on their land actively controlled them. There have been shifts in attention to different predator species over the last five years, with decreased reporting of culling of stoats *Mustela erminea* and weasels *M. nivalis* and increased reporting of culling jackdaws *Corvus monedula* and rooks *C. frugilegus.*

It has been demonstrated that fox control by gamekeepers can reduce foxes locally (e.g. Reynolds et al. 1993; Tapper et al. 1996). More recently **Porteus *et al*. (2019) modelled fox population dynamics in relation to culling effort at 22 estates (some of which released gamebirds). Their models were verified and showed that all estates successfully suppressed the fox population, reducing on average pre-breeding (late winter) density to about half (range 20 – 90%) of estimated carrying capacity. Immigration rates were also very variable but at most sites it was rapid so effort was needed to maintain reductions. Those sites that did this also had reduced fox population during the spring/summer fox breeding period.

There is also some evidence that coordinated collaborative fox control can result in reducing the population of foxes regionally. **Heydon *et al*. (2000) and **Heydon & Reynolds (2000) used spotlight transect counts and compared fox abundance in East Anglia, where wild gamebird management is relatively common, with two other UK regions. They concluded that culling (mainly shooting and snaring by gamekeepers) had probably reduced the fox population to below that predicted by landscape in East Anglia but not in the other regions. This difference may be because many release-based shoots (typical of the other two regions) undertake relatively little or no predator control but there is little quantification of this. In **Heydon *et al*. (2000) the effect of regional fox control in the East Midlands, where there is relatively little interest in wild gamebird management, yet a similar amount of fox culling activity as seen in East Anglia, there was no regional reduction in the fox population level below that predicted based on landscape. This suggests culling in the East Midlands region is generally less effective than in East Anglia.

Five regionally-disparate estates for which data on gamebird releasing were available and which carried out fox control (primarily by lamping and shooting with a rifle,) also monitored foxes weekly as part of The GWCT's Fox Monitoring Scheme in the late 1990s (***Porteus 2015). These few data points suggested that the effectiveness of fox control declined as the size of the release went up. The site with the least suppressed fox population had the highest density release and vice versa. In his PhD thesis, ***Porteus (2015) suggested that the larger releases in his study supported more foxes (see Section 6.2) which were then ineffectively controlled.

ii) Illegal Predator Control

Some game managers may also illegally kill predators of released gamebirds. Such illegal killing is likely motivated by the desire to reduce losses of released gamebirds to protected predators. These would most notably be raptors. In their review of raptors and gamebirds, **Park *et al*. (2008), citing primarily **Kenward *et al*. (2001), found that that numbers taken by raptors at release pens varied considerably and in only a few cases had raptors been documented killing relatively large numbers.

According to 156 game managers who responded to a questionnaire survey (**Lloyd 1976) tawny owls *Strix aluco*, sparrowhawk *Accipiter nicus* and buzzard (in that order) were most commonly involved in attacks on pheasant release pens in woodland. This survey report calculated an overall loss of pheasant poults of just under 1 % on average. It also concluded that mass kills reported to be by raptors were probably by foxes. In a later survey, the same three species were involved in the majority of incidents but in a different order - sparrowhawk (36% of respondents), buzzard (20%) and tawny owl (17%) (**Harradine *et al*. 1997). This change in emphasis may be a result of population change over the intervening 20 years. Summaries of these and other studies that have quantified 'bird of prey conflicts with pheasants at release pens' are reviewed in *FERA (2012). The report concluded that losses of released pheasant poults at release pen sites to raptor predation was <1% and >90% of sites but that some sites do suffer much higher levels. The report (and Lloyd 1976) identified factors which may have caused higher predation (younger birds, released earlier into large pens with poor shrub cover) and then considered a variety of legal mitigation measures.

Recently, particular attention has been paid to predation of released gamebirds by buzzards. **Kenward *et al*. (2001) studied the effect of buzzard predation on pheasants at release pens. They documented the predatory behaviour of 40 radio tagged buzzards living close to pheasant release pens in southern England. On average 4.3 % of poults were killed by buzzards in those pens and another 5.2% by other predators. Only a minority of buzzards associated regularly with pens, and predation was heavy only at a minority of sites (only 20% of releases lost more than 2 pheasants). Buzzard predation was found to be associated with pens with little shrub cover (and hence more ground cover), deciduous canopies and large number of pheasants suggesting that occasional heavy losses can be avoided by proper management of the pens and the woodland in them. *Swan (2017) found support for the idea that there are some buzzards that specialise in taking pheasant poults. Other studies report little direct predation by buzzards on released gamebirds (**Turner & Sage 2003; **Lees *et al*. 2013).

The predation of released gamebirds by buzzards may provoke the killing of those raptors. This may rarely be achieved legally. Natural England have issued nine licences (2016 – 4; 2017 – 4; 2018; - 1) to control buzzards to prevent predation of young pheasants (as compared to a total of 70 such licences to kill buzzards for air safety issued since 2014) (**Diamond 2019). 81% of 563 keepers responding to a recent GWCT/NGO survey perceived that buzzards had a negative effect on game (confounding wild and released gamebirds), being a slight increase since 2011 when 76% of keepers had this perception (**Ewald & Gibbs 2020). These perceived effects may extend beyond the obvious killing of released gamebirds and may include the effects of disturbance to the released birds, stimulating them to disperse from the area containing the raptor. **Kenward (2001) provides the main source of evidence of buzzard being killed specifically in association with pheasant release pens. Of 136 radio-tagged buzzards in his studies between 1991 and 1995, 38 birds died of which 12 were shot or poisoned, mostly near to pheasant release pens.

Other raptors may also predate released gamebirds and so be killed. In an early study at a large release pheasant shoot in Sweden **Kenward (1977) found that goshawks took 19% of 4300 pheasants and made no selection for handicapped or under-average condition birds. Kenward (2001) concluded that goshawk can present a serious threat to gamebird releases but it is relatively uncommon in English woodlands. In their comprehensive look at goshawk in Britain, **Marquiss & Newton (1982) report two slightly conflicting evidence bases relevant to this review. The first set of data indicates that between 1975 and 1980, 101 nestling goshawks were ringed in Britain, and 14 were recovered. Eight of these had been shot, trapped or poisoned, but there was no information from this sample if this was done in association with releasing gamebirds. The second set of data is of a total of 49 goshawks, ringed or not, recorded as killed by man between 1971 and 1980. Of these eight were shot or pole-trapped at or near to pheasant release sites.

In their recent review of management and gamebirds Mustin et al. (2018) described illegal killing of predators as an important negative effect of mainly wild game-bird management. Only one of the seven papers in their review related to released game (Beja et al. 2009) while five of the remaining six concerned raptors and grouse moors. ***Beja *et al.* (2009) measured numbers of raptors, amongst other things, on 12 shooting estates and 12 similar areas in Portugal. Most (it wasn't stated how many) shooting estates released red-legged partridges. They reported only kestrel as being less common on shooting estates. Within shooting estates, the overall abundance of raptors varied inversely with gamekeeper density except for buzzard which increased.

The Europe-wide REGHAB project undertaken 20 years ago aimed to 'reconcile gamebird hunting and biodiversity conservation' (Arroyo & Beja 2002; Manosa 2002) and there was a focus on predator control and the illegal killing of raptors. Without providing a great deal of evidence relevant here, the study concluded that raptor predation was seen by game managers as being more important where shooting depends on wild breeding populations than where releases for immediate shooting are performed, that illegal killing of raptors is less common in association with releasing than with other forms of game management and that it had declined substantially across Europe.

The RSPB's Annual Bird Crime report documents illegal killing of raptors in Britain (*RSPB 2019). The great majority of these cases tend to focus on illegal killing in association with grouse-moor activities rather than lowland gamebird release. The reports concentrate on 'confirmed' incidents which have 'high evidential weighting' but points out that many incidents are not detected or reported because they often take place in remote and private locations. Since 1990, two thirds of 180 prosecutions have involved gamekeepers. Simply looking at the locations where crimes were committed may tell us little about the motivation for the killing. However, in recent years the reports contain case studies and other general information while an appendix to each report document incident cases and look at spatial distribution and other context. We have looked at several recent example reports sent to us by the RSPB. The appendices do not routinely provide information on the context for incidents so it is hard to extract data specifically about the rate of illegal killing at lowland release based shoots.

The 2018 report is not atypical of previous ones. There were 87 incidents reported. About two thirds of them involved buzzard or red kite *Milvus milvus* with a range of less common species involved in much smaller numbers of incidents. There were 16 prosecutions, lower than in some previous years. The UK region with the most incidents reported was North Yorkshire. The county list in Appendix 2 suggests that of the 67 incidents recorded in England, 29 occurred in counties where management of wild gamebirds is unlikely to occur and therefore the release of gamebirds may have been a motivating factor. The remaining 38 incidents were in parts of the country where wild bird game management occurs, either of grouse (generally northern counties of England) or lowland species such as grey partridge or wild pheasants and red-legged partridges (generally eastern counties of England). Released gamebird shooting also occurs in these counties, so it is hard to disentangle motivations here, but the map of incidents in Appendix 4 indicates that those occurring in the Northern counties were mainly on upland areas where wild bird grouse management is more common. We found details of a single case explicitly relevant to pheasant release in the 2008 report. In the 2018 report, one incident involved the use of a pheasant as a poisoned bait.

F) Associated effects of disturbance on soil, water and air

Management work on land to prepare or maintain it for gamebird release may cause physical damage. 87% of rides in game woods experienced some disturbance by vehicles compared to 55% of rides in control woods. 61% and 25% of rides in control woods had some erosion due to footfall or to horses respectively, compared to 7% and 10% of rides in game woods. There was 55% less bare ground in rides in game woods (***Hoodless & Draycott 2008, ***Capstick *et al*. 2019b). Construction work while building ponds to retain mallard may cause short term physical disturbance.

G) Associated effects of disturbance on small vertebrates

Wild animals may be disturbed by human recreation activity with detrimental fitness consequences (e.g. Beale & Monaghan 2004). Game managers may attempt to limit public access to release sites and there is a perception that release of gamebirds is associated with a reduction of access to the local countryside (*Cox *et al.* 1996). A reduction in access may be socially undesirable but may offer peace and refuge to local wildlife and reduce damage to habitats, especially if activities of free-roaming dogs are also reduced. However, a survey of the opinions of farm/estate owners that did (n = 157) vs. did not (n = 126) release game indicated that where releases occurred, 61% of game managers releasing game birds reported public use of footpaths > 1/week and 51% reported use of bridleways > 1/week compared to figures of 45% and 22% for non-releasers respectively (***Cox *et al.* 1996). This difference may be driven by higher levels of vigilance on releasing farms by game managers and other farm staff.

Disturbance may have non-lethal effects on gamebird and non-game species in the area. Wild-born mallard, that might be encountered by guns in mixed flocks with reared mallard, exhibited stronger tachycardia when exposed to human disturbance than did released birds and thus may be more susceptible to detrimental non-lethal effects of shooting than released birds. They may be more likely to encounter such disturbance in the presence of released mallard (***Heise 1989). Waterbirds may be susceptible to shooting disturbance in terms of altered behaviour, flight from the area and reluctance to forage in disturbed areas, but there is little evidence (primarily from the USA) of effects on population size resulting from such disturbance (*Madsen & Fox 1995), although shooting may alter waterbird assemblages in frequently disturbed areas (*Pallisson *et al*. 2002).

Similar patterns of disturbance were reported by *Casas *et al.* 2009 who studied the behaviour of lapwings *Vanellus vanellus*, golden plover *Pluvialis apricaria* and little bustards *Tetrax tetrax* (non-quarry species) on a 10km^2 agricultural area in France before, during and after the shooting season. Local guns practiced walked-up shooting of wild Galliformes and lagomorphs. During the days when shooting occurred, the focal non-quarry species spent more time vigilant and had increased flight frequency, but this did not detrimentally affect their time spent foraging or feeding compared to days when there was no hunting. Plover and lapwing also exhibited increased flight frequency on the day after shooting had occurred, suggesting that the effects of disturbance persisted for at least 24 hours. The authors suggest that this disturbance may have energetic costs for birds but did not demonstrate this. They also suggest that some species may be better able to cope with the disturbance and thus occupy areas where shooting occurs, whereas other species may avoid such areas and stay in shooting refuges. This suggests that focal species differ in their response to disturbance.

H) Associated effects of shooting on small vertebrates

The release of gamebirds may encourage or accidently facilitate the shooting of wild birds (both conspecifics and heterospecifics) during a day's harvest. The release of mallard may reduce shooting pressure on wild populations of that species (Swift & Laws 1982 in Callaghan 1996). Alternatively, their presence may stimulate greater shooting pressure that is detrimental to wild populations (Callaghan 1996). We are not aware of any evidence to support either suggestion. Releasing red-legged partridges facilitates shooting on farmland where wild partridges, red-legged or grey, are absent or in low numbers. In Spain at a sample of four sites ***Casas *et al*. (2016) found that where red legs were released, a greater number of wild red legs were shot, suggesting releasing could cause over shooting of wild stocks. In the UK, the release of reared pheasants or red-legged partridges could lead to overshooting of any wild-born, naturalised stocks but no work has been done to quantify this.

While shooting released gamebirds, guns may also (legally) shoot other wild-born game species. For the native grey partridge in the UK the potential for over-shooting alongside releasing has been quantified. **Watson *et al.* (2007) compared the effect of raptor predation and shooting on winter mortality of grey partridge on a study area on the South Downs in Sussex. Shooting there is based on the release of pheasants and red-legged partridges on about two thirds of the study area. Despite this, over the study area as a whole, 86 of 243 (35%) grey partridges present in the autumn were shot in 1999/2000. On two farms with large red-legged partridge releases around two-thirds of grey partridges present were shot. Overall shooting losses during the study was double that caused by raptor predation. Modelling suggested that if 50% or more of autumn grey partridge stocks are lost then the species will go locally extinct.

Watson et al. (2007) argued that where shooting is based on red-legged partridges, training guns to avoid grey partridges and implementing a warning system that alerts them when grey partridges are approaching could be effective. On one of the two farms in their study with a large red-legged partridge release, a policy of avoiding shooting grey partridges was introduced after the main study using these voluntary measures and the proportion of the autumn grey partridge stock shot dropped to 16%. In another GWCT grey partridge recovery demonstration project, both grey and red-legged partridges responded to a programme of wild game management (**Aebischer & Ewald 2010). Using similar measures under a GWCT programme, while a sustainable surplus of red-legged partridges were shot, losses of 5% of the autumn grey partridge stocks still occurred.

Venues that offer shooting of released birds may also permit the shooting of non-released game species and this may introduce higher level of shooting pressure to the area than would be supported by the wild game species alone. Of the 799 shoots advertising pheasant shooting on the Guns on Pegs Website (see above for Methodology), 193 (24%) offered woodcock *Scolopax rusticola* shooting, 105 (13%) offered wildfowl shooting and 25 (3%) offered snipe *Gallinago gallinago* shooting. Many shoots also offered pigeon shooting, but this was commonly described as a separate operation outside the game shooting season.

3. Indirect effects

A) Indirect effects of planting and management of woody plants on other plants

Management of woodland/hedges for game may inadvertently alter the floral composition. Pheasant-managed woods studied by ***Draycott et al. (2008a) had on average a more open canopy structure than unmanaged control woods. Probably as a consequence, the average ground cover in quadrats in the pheasant woods was 63% and in the control woods it was 48%. Around 80% of quadrats in the managed woods contained herbaceous plants compared to less than 60% in the control woods. The study did not look at more specific plant groups or species. There may be woodland plants, for example those that prefer shade, that would not benefit from game management. There were no differences in abundance or diversity of woody shrubs in the interior of pheasant and control woods. However, game managers tend to encourage shrubs in the woodland edge zone.

The edges of woods managed for pheasants had a more sloping profile, 1.3x greater shrub cover and fewer overhanging trees than non-game woods in East Anglia, but not in Hampshire (***Woodburn & Sage 2005). Game woods in East Anglia had 2.5x as many flowering shrubs as non-game woods. Shrub density 10m inside the wood was 1.7x higher in game woods than non-game woods in East Anglia but not Hampshire. On average, the edge zones of game woods appeared to have changed in both regions, but the effect was more pronounced in East Anglia. In Hampshire, differences between game and non-game woods were limited to the number of shrub species and understory cover inside the edge. Historically, forest fragmentation and silvicultural regimes in the UK such as coppicing has favoured woodland edge species which are today of conservation importance (Ferris & Carter 2000).

In woodland rides managed for game there were 21% more herbaceous and 31% more grass species than non-game woods (***Capstick 2019b). There also were more ruderal species (27%) and species of fertile soil (53%) in game woods. There were similar numbers of ancient woodland indicator species in each woodland type. There were 41% more shrub species in rides in game woods compared to non-game woods in the southern region.

B) Indirect effects of planting and management of woody plants on invertebrates

Management of woodland/hedges for gamebirds and their release may alter its attractiveness to invertebrates. The woodland edge zone is the area where pheasants spend most of their time in winter and spring (see 2.C). In 2005, most of the woods used in the sample by ***Draycott & Hoodless (2005) and ***Draycott et al. (2008a) and described in (2.C) were used in a follow up study of habitat quality and butterfly and bee abundance in the woodland edge zone ***(Woodburn & Sage 2005), defined as c.20 m into the wood. During the summer, the profile of the woodland edge, shrub flowering and shrub density was measured and counts of butterflies and bees undertaken. Butterfly numbers were 2.2x higher and the number of species 1.5x higher in the edge zone of game woods than non-game woods in East Anglia but not in Hampshire. No relationship was detected between game management and numbers of bumblebees in either region.

Butterfly abundance and diversity was not significantly different in rides between game and non-game woods in ***Capstick et al. (2019b). Butterflies are looking for a combination of sun and shelter in these habitats (Warren & Fuller 1993). Robertson *et al.* (1988) found more butterflies in game-managed areas of woodland than other areas of the same woodland and suggested that the presence of rides and other open areas were the main reason.

C) Indirect effects of planting and management of non-woody plants on small vertebrates

Winter and summer game crops are planted in relatively small plots (see 2.B) and hence concentrate birds. Nevertheless, these patches of game crops could support high numbers and local abundances of wintering and breeding songbirds using those plots and the adjacent land. This applies to both generalist farmland/wood-edge species as well as some declining farmland birds. In regions of the UK where there is no arable cropping, game crops can be the only seed crops available to farmland birds.

Game crops, buffer strips and mix cover options planted by game managers motivated by gamebird release may prove to be attractive to small vertebrates. In ***Sage *et al*. (2005b) 30 winter game crop plots contained more than 10 songbirds per ha in most months between October and January, while the adjacent 30 arable field plots contained less than one. Densities in kale *Brassica oleracea* spp. and quinoa *Chenopodium quinoa* game crops were higher than in cereal-based game crops. Of the 26 bird species recorded in the winter game crops, 10 have undergone rapid declines over the last 30 years. In all three winter game crop types, songbird numbers declined significantly in the second half of the winter (January to March) while numbers in the arable fields did not. In Eastern Scotland ***Parish and Sotherton (2004) found a similar magnitude of difference when comparing overall bird numbers in 20 kale and cereal game crop plots with 20 nearby stubbles and conventional arable fields.

In a study of winter bird crops (i.e. game crops) at 192 farmland sites **Henderson *et al*. (2003) found 12 times as many birds per ha compared to conventional crops. Kale was the most attractive game crop overall for the 18 species of bird recorded. The authors thought that this was due partly to the kale's seed-bearing properties combined with its soil-moisture retention properties which would benefit snails, worms and other invertebrates. Henderson et al. also found that kale and quinoa retained seed better as the winter progressed compared to most other crops they looked at. For most of these crops, larger plots of 1 ha or more retained seeds for longer. Stoate and Szczur (2001) provide an overview of the of benefits of game management including game crops to birds on the GWCT demonstration farm. *Stoate *et al*. (2003) found that kale and quinoa were the most attractive game crops for wintering birds including tree sparrow *Passer montanus*, bullfinch *Pyrrhula pyrrhula*, reed bunting, yellowhammer and grey partridge which have all shown significant declines on UK farmland.

Breeding birds using hedgerows near winter game crop plots were counted in the spring on three Exmoor shooting estates and the numbers compared to those counted using hedgerows on three farms without shooting (***Sage 2018 a,b). The number of breeding resident birds was 2.5x as high in the hedgerows on shooting estates as on non-shooting farms, while the number of migrant birds, which arrive on the sites in spring, were not different. On the shooting estates, hedges within 200 m of game crops had more breeding resident songbirds in spring than the hedges further away. Another study in Eastern Scotland compared birds using game crops in grasslands and arable areas at nine sites over two years (**Parish & Sotherton 2008). They found that game crop plots in grassland landscapes had just over double the number of birds in winter than similar game crops in arable areas. Individual yellowhammers were recorded flying 2km to game crops. Away from the gamecrops, the grassland fields had just 14% of the birds found in the conventional arable crops.

D) Indirect effects of planting and management of woody plants on small vertebrates

i) Birds

Most studies in this area have focused on numbers of bird other than game species living in woods on shooting estates during the breeding season. Several woodland specialists were found to be more abundant in game-managed woodland areas compared to areas of closed canopy woodland areas nearby (***Robertson et al. 1988; **Woodburn & Robertson 1990; **Robertson 1992).

The interior of 159 lowland woods were surveyed in southern and eastern England during spring-summer 2004 to determine the effect of pheasant management on vegetation structure and composition and on songbird abundance (***Draycott *et al*. 2008a). Half the woods were managed for game while the other half had no releasing or feeding of game for at least 25 years. Surveys were undertaken away from release pens. 40% of study sites were randomly selected while others were from the GWCT contacts database. There were between 22 and 32% more songbirds in pheasant woods than control woods. Each 4-ha survey plot contained on average two warbler territories in the game woods and 1.3 territories in the control woods. Warblers depend on the presence of a ground layer of vegetation for nesting. The work suggests that active woodland management for released pheasants can have positive effects on woodland vegetation and some songbird species.

Davey (2008) looked at the influence of game management variables on the abundance of songbirds in 20 woods with release pens, taking account of bird detectability. The abundance of four granivorous birds (blue tit *Cyanistes caeruleus*, robin, nuthatch *Sitta europaea*, dunnock) and two primarily insectivorous species (blackbird and wren *Troglodytes troglodytes*) was positively related to the density of feed hoppers. Other granivorous species such as chaffinch *Fringilla coelebs* showed no relationship with food provision. Two insectivorous species, song thrush and willow warbler *Phylloscopus trochilus* showed negative relationships with the density of feed hoppers (no mechanism for this effect was offered). There were no relationships between songbird abundances and the density of pheasants in the woodlands. Davey (2008) did not include non-game woods in her field-based work so she looked at a sample of 40 game (i.e. with release pen) and 40 non-game woods from the RSPB/BTO Repeat Woodland Bird Survey (RWBS) dataset (Amar *et al*. 2006). Every few years, the RWBS uses point counts for birds in around 400 woods and measures potential explanatory factors including game management. Of 20 bird species considered, none showed a difference in abundance between game and non-game woods for this sample.

Effects on non-game bird populations appear to be consistent across woodland type. In a sample of 26 conifer woodlands on four shooting estates in the Exmoor region ***Sage (2018a,b) measured the structure of the woodland with and without game management. This indicated that the lower and upper tree canopy in conifer woods managed for game was about 25% more open than in non-game ones. There was 30% more bracken *Pteridium* spp. in game woods and a tendency towards more bramble *Rubus fruticosus* and grasses. The abundance of herbaceous vegetation was not different. On average 18 (non-game) birds were encountered per survey transect in the game conifer woods, significantly more than in the non-game woods with, on average, 10 birds per survey. Conifer woods with the habitat differences identified were probably selected for game management purposes and then further improved through management for game. Many upland pheasant releases are located in conifer plantations.

Effects on non-game bird populations are also seen in winter. For most resident woodland birds, while knowledge of their winter habitat use is poor compared to that of their breeding requirements, shelter as well as food will be an important factor (e.g. Vanhinsbergh et al. 2002). Managing woods for game might be expected to alter winter habitat provision and food supply for (non-game) birds (*Robertson *et al*. 1993a). In 2005, non-game birds were counted along 1-km transects in 70 semi-natural oak and ash woods in central southern England (***Hoodless *et al*. 2006). Half of the woods had pheasant release pens and game feeding. Vegetative cover was measured in the field, shrub understorey and canopy layers. Bird numbers in November-December were 1.5x higher in woods where pheasants were released compared to the sample of 35 non-game woods. On average, 13 species were recorded in game woods compared to 10.4 species in non-game woods. Bird communities of game woods contained higher numbers of finches, tits, shrub species as a group and woodpigeons than those of non-game woods. Thrush and woodpecker numbers were no different. There was a relationship across all woodlands in the study whereby bird numbers increased as canopy cover decreased and average canopy cover was lower in game woods than non-game woods (37% and 45%). This suggested that thinning or skylighting in the game managed woods may have been benefitting birds. Feeding in pheasant woods in winter may also be a component (see 2.D).

Effects of game management mediated by woody plants may be seen outside woodland. ***Sage *et al*. (2009) conducted transects leading away from release points at over 100 shooting estates in southern and eastern England in 2002 and 2003, considering numbers of gamebirds released and distance from hedge to the release pen. They found that overall, the abundance of (non-game) birds in hedgerows was not affected by the proximity of a hedge to a release site. However, there were typically around a third fewer songbirds in hedgerows near to release sites that released more than 1500 gamebirds. This was probably because of the reduced shrub structure and flora in these hedgerow areas identified above (see 1.B).

ii) Small mammals

A smaller number of studies have considered differences in small mammal populations due to woodland management for game.

Data collected from a grid of 160 baited traps at between 8 and 16 releasing woods (so no non-releasing controls) in south-west England enabled an investigation of relationships between small mammals and pheasant releasing and feeding intensity, while considering other spatial and habitat factors (***Davey 2008). 2100 mammals were caught, including six rodent species (wood mouse *Apodemus sylvaticus*, yellow-necked mouse *Apodemus flavicollis*, harvest mouse *Micromys minutus* and house mouse *Mus musculus*, bank vole *Myodes glareolus* and field vole *Microtus agrestis*), and three shrew species (common *Sorex araneus*, pygmy *Sorex minutus* and water *Neomys fodiens*), although we don't know how abundant individual species were within sites. Habitat variables were most important in explaining the number of mammals caught but game management had an effect once these were controlled for. Numbers of bank voles and wood mice caught were consistently higher at sites with feed hoppers all year. However, where excess (spilt) grain was greatest fewer of these mammals were caught, suggesting competition between small mammals or predation effects might be important. The distribution of bank voles and common shrews caught was not found to be related to feed hoppers but that of wood mice was. Wood mice were more abundant close to release pens in autumn but more abundant further away from those pens in spring. Bank voles were more common near release pens in Spring. Common shrew was the only species that was less commonly caught near to release pens after the pheasants were released and the suggestion was that this was due to habitat disturbance. These results were used to conclude that game management tended to be positive for woodland small mammals. The study found no evidence that the pheasants themselves affected the small mammals either directly or through habitat effects. Wood mice and, to a lesser extent, bank voles are robust common species found in a range of habitats. Common and pygmy shrews are adaptable but insectivorous

(Harris and Yalden 2008) and it is possible that effects of pheasants on invertebrate communities (e.g. Neumann et al. 2015, see 1.F) in release pens may be detrimental to that group. Yellow-necked mice did not form part of Davey's (2008) data analysis but may be more vulnerable as a species. Dormice *Gliridae* spp. were not caught in the study but need a diverse shrub layer. Game management may benefit dormice in this respect but they may also be sensitive to other game related activities in woods (Bright & Morris 1990).

Grey squirrels are sometimes reported as being more common in woodlands with pheasant feeders but there have been no dedicated studies. ***Draycott & Hoodless (2005) counted squirrel encounters during other spring and summer surveying episodes in their sample of 159 half game and half non-game woods (described in 2.C) and found no difference.

E) Indirect effects of predator control on small vertebrates

The local removal of generalist predators may reduce predation on a suite of wild gamebird or non-game species. For example, an experimental six-year study of grey partridges in the 1980's employed a gamekeeper to control foxes, crows, magpies, stoats and weasels on two areas of Salisbury Plain in Southern England, one after the other, in spring and summer. The partridges showed on average a 2.6x increase in breeding density and 3.5x increase in autumn density after three years of summer predator control in each plot (Tapper et al. 1996). The study demonstrated how controlling one or more of these predators was playing a key role in improving breeding abundance and output in this ground-nesting bird. Reynolds & Tapper (1996), describe and discuss how control of mammalian predators to benefit small game has been practiced in the UK for 200 years and has played a key role in shaping the UK fauna today. More recently, by comparing the effects of management on avian abundance at the GWCT and RSPB demonstration farms over 25 and 15 years respectively, Aebischer et al. (2016) concluded that where predator densities were high (>3 corvids and 1 fox per km^2), recovery of declining farmland birds including some passerines may require predator control as well as habitat management.

These game related studies and many others that have attempted to quantify the effect of predators or of predator control on birds have been reviewed recently (*Roos *et al*. 2018). Their synthesis and analysis provided evidence that for three of the four main groups of birds (seabirds, gamebirds, waders) numbers were limited by predators, and in an experimental predator control sub-sample, there was evidence of this being effective for passerines too. Roos et al. (2018) concluded that predator management aimed at foxes and corvids simultaneously (as applied by many lowland shoots) is more likely to be effective and that lethal control or fencing should be considered when managing vulnerable species. In their review *Mustin *et al*. (2018) concluded, from 13 studies focussed largely on abundance and breeding success of birds, that legal predator control associated with game management had a positive effect on some other wildlife. Two of these studies occurred at specific sites where gamebirds were being released, but the analysis did not consider how predator control was explicitly related to these releases.

An analysis of 11 years of nest data from 6 songbird species on 3 lowland farms enabled a comparison of the effect of systematic mammal and corvid reduction and sporadic corvid reduction on nest success in songbirds (*White *et al*. 2014). There was a positive effect of systematic predator reduction on common blackbird, chaffinch, dunnock, song thrush, and yellowhammer but not common whitethroat *Sylvia communis*. Daily nest survival improved by a factor of between 1.59 and 1.89. Sporadic corvid reduction had a positive effect on nest survival for common blackbird (at the nestling stage) and a negative effect for yellowhammer (across both stages). The comparison might be useful here because the systematic predator control described is usually undertaken on wild-bird shooting estates while sporadic corvid control is more likely on release-based shoots.

The studies described above have concerned predator control generally although it may be motivated by shooting interests. More specifically, two studies have considered sites where gamebird release occurs. **Sage *et al*. (2018c) combined the results of seven spring and summer pheasant radio-tracking studies. Three of the sites classified as having high-level predator control (and two of these released pheasants) had improved survival of adult gamebirds during the spring compared to four with low-level control. In a sub-sample of six of these sites, **Draycott *et al.* (2008b) documented improved nest survival at two sites with high-level predator control, one of which released. These two studies of pheasants suggest that at least some release-based shoots suppress predators sufficiently to have an effect on wild breeding birds. To conclude that, generally, shoots releasing gamebirds can provide the same benefits to wild breeding birds, it is necessary to consider the form, duration, extent and efficiency of predator control practiced by such shoots (see 2.E).

F) Indirect effects of carcass availability on small vertebrate populations

Bicknell et al. (2010) and others (e.g. Roos et al. 2018) discuss the idea that generalist predators, attracted to or sustained by high numbers of released game during the winter, remain on site and switch to other prey such as ground-nesting waders when the numbers of released birds decline in the following spring. This is a reasonable hypothesis and has attracted much attention but we are not aware of any evidence of effects on non-game prey populations linked to releases that would be appropriate to use to support or refute it. We discuss ways to address this Knowledge Gap in a later section.

Some evidence for mediating factors

The consequences of the effects that we have described in the section above are highly dependent on where gamebird releases occur and the numbers or density of birds involved in those releases. Many studies in the Section above report that the effects observed become larger in areas where high densities of gamebirds are released (e.g. Sage et al. 2005a, Gortazar et al. 2006, Pressland et al. 2009, Neumann et al. 2015, Porteus 2015, Capstick et al. 2019, but see Davey 2008). The effects occurring outside the release pen may also depend on the density of released birds in the wider landscape and that is a product of the numbers of birds being released, but also the distance from the release pen, the dispersal behaviour of those birds, the time since release and the local mortality patterns that the released birds suffer. Other studies report that effects are larger close to the sites of release (e.g. Ewald & Touyerus 2002, Sage et al. 2009). Finally, there is evidence that the effects are not homogenous across sites, but instead are influenced by local ecological or social factors even when release sizes and densities may be comparable (e.g. Cox et al. 1996, Heydon et al. 2000). These mediating factors may also apply to the Associated Effects. It appears likely that on shoots where more birds are released, larger areas of habitat are managed, and perhaps greater investment is made in supplementary feeding and predator control (Porteus 2015, Sage 2018b, Teanby et al. 2017). These associated effects may also scale with spatial distance to release sites with increased management occurring at the core of a shoot near the release sites.

Therefore, to understand the size of any effects both locally and nationally, and in order to compute net cost and benefits of releasing gamebirds, it is necessary to determine and account for how many birds are released, where they are released, what area they are released over and at what densities. In our Conceptual Model, we describe these as Mediating Factors. In the following section we summarise evidence and present some novel data that describes the scale and extent of gamebird releases set in an historical context, and data regarding the movement and survival of released gamebirds that together determine their densities post-release.

A) Historic patterns of game management and gamebird occurrence and abundance

By understanding changes in spatial and numerical patterns of gamebird shooting and associated release we may detect areas where effects of gamebird release and associated land management are the most well established or where we might expect to see most recent changes (either because gamebird releases have increased or decreased in recent years) or where we might consider to be comparator areas where gamebird releases and associated management has not been practiced.

i) Pheasants and Red-legged Partridges

Pheasant were commonly being hunted and eaten by the 12th Century, indicating that they had been introduced and naturalised in England by that time (Lever 1977). Red-legged partridges were successfully introduced to the UK for shooting in the late

18 Century, although earlier attempts had been made (Lever 1977). As technology and fashions changed, they moved from being the quarry of falconers to being shot, and the scale of this shooting increased until the late 19th/early 20th Century. At this time, around 50% of the agricultural land of England and Wales was managed for shooting with around 21,000 people employed as gamekeepers (Tapper 1992). This activity was spatially heterogeneous, with highest levels of game preservation in East Anglia (Norfolk, Suffolk), the south coast (Hampshire, Sussex, Kent, Dorset), and the Home counties (Surrey, Berkshire, Hertfordshire), with high levels in Wales (Flintshire, Anglesey and Denbeighshire) and the borders (Shropshire) (Martin 2012). Much of this effort was directed at preserving grey partridge, but pheasant also comprised large proportions of the bag in many areas. These high bags were achieved through intensive land management including planting and tending of woodland and hedges and predator control. At this time, natural populations of pheasants were supplemented by eggs either produced from female pheasants that had been caught from the wild or taken from wild nests that were then hatched under broody hens. Commercial game farms did exist at this time, but they produced only around 100,000 chicks (mainly pheasants) annually with estate owners producing around 300-400,000 chicks annually [Lillywhite pers comm in Martin 2011). On estates operating high volume game shooting, birds were released at a density of about 30 birds/km^2 (**Tapper 1992).

From a peak in game management in the 1930s, the management of gamebirds fell dramatically over the next 50 years. This was due to political and economic changes in the 1930s and 1940s, which were accompanied by a fall in the number of gamekeepers from ~11,000 to 4,391 in 1951 (Martin 2011). This number of (full time) keepers continued to fall to around 2,500 in 1981, during which time it is estimated that less than 8% of farmland was keepered. Since then, there appears to have been a small increase in the number of keepers, with the National Gamekeepers Organisation reporting in 2020 that there were around 3000 full-time keepers and a similar number of part-time keepers (https://www.nationalgamekeepers.org.uk/about-gamekeeping). An initial impetus for this decline was the prohibition in 1940 of rearing gamebirds as part of the Defence Regulations, coupled with Keepers and other estate staff enlisting, and a change in land use to provide food supplies. The prohibition on rearing was lifted in 1949, but the effects of reduced management (and perhaps poaching to overcome rationing) on the population of game were noticeable, with post-war harvests being only around 30-40% of the pre-war ones (Martin 2012).

This decline in game management did not perfectly match patterns of gamebird release. Although the number of keepers, and perhaps the area of land under game management was at a low point in the early 1980s, the number of pheasants and red-legged partridges being released had started to increase since the early 1960s (**Robertson *et al*. 2017). By the early 1980s, the index of release density for pheasants was around three times higher than in 1961. This increase continued at a rate of 4.3%/year so that in the 2010s, the index of release density for pheasants was around nine times higher than in 1961 (Robertson *et al.* 2017). This increase in release density index has been accompanied by a fall in the contribution to the bag of birds that were not released that year. Pre 1990s, the bag comprised around 30 birds/10km^2 that had not been released on the participating estate, and a correlation between this value and the productivity rates of wild grey partridge suggests that those birds not released were wild-born. The number of pheasants in the bag that had not been released declined post 1990 to around 10 birds/10km^2 and the relationship with the grey partridge chick index fell markedly, indicating that wild-born pheasants no longer made up a substantial part of the harvest but that shoots were more reliant on their released birds (Robertson *et al*. 2017). A similar pattern is seen for red-legged partridges, with the release index in 2016 being around 200 times greater than in 1961, although the bag index is only nine times higher than in 1961 (Aebischer 2019).

Wild pheasants and red-legged partridges were initially most commonly shot in the South and East of England, with highest bag densities (>100birds killed/km^2) recorded in Norfolk, Suffolk, Cambridgeshire, Bedfordshire, Hertfordshire, Surrey and East Sussex between 1900 and 1938 (Tapper 1992). By 1961-1985, similar bag levels were obtained in all SE counties as well as in Wiltshire, Gloucestershire and Staffordshire. Current patterns of reported release and shooting indicate a continued westward expansion of the area where gamebirds are shot, especially for pheasants (Fig. 2). Because of differences in sampling and reporting, and the underlying assumption that patterns of shooting reveals patterns of release which may not have been the case historically, these maps cannot give quantitative differences but rather illustrate general shifts in likely intensities of gamebird shooting and release.

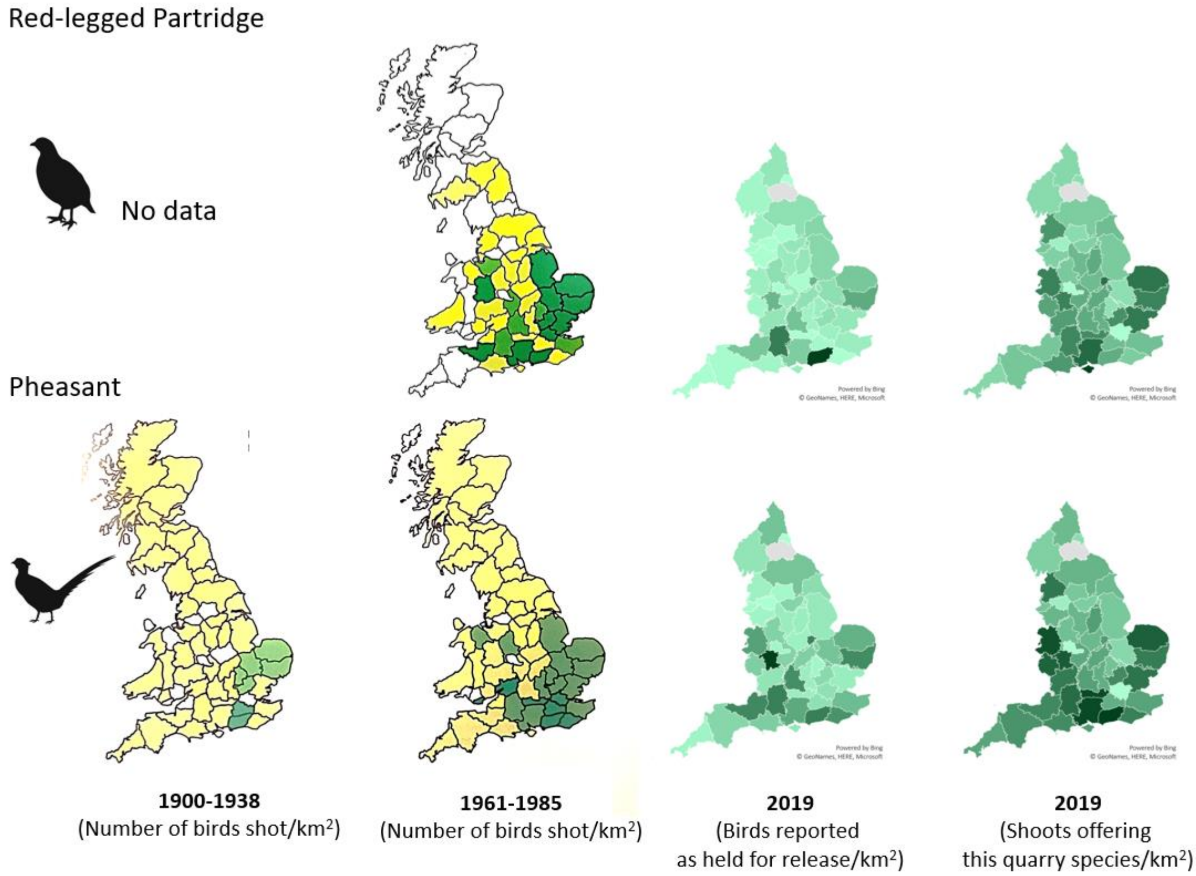


Figure 2. Changes in patterns of gamebird (red-legged partridge and pheasant) shooting and release since 1900. Maps for 1900-1985 taken form Tapper 1992 showing the numbers of birds recorded as being shot per km2. Maps for 2019 derived from the APHA Poultry register (numbers of birds reported as being held for releasing per km2) and the Guns on Pegs website (numbers of shoots offering that

quarry species per km2). Darker green indicates higher numbers. Because all measures differ, we do not give absolute values but rather the maps allow comparison of relative differences between counties.

ii) Mallard

Releasing mallard is believed to have started in the UK in 1890 (Sellers & Greenwood 2019). It was common practice to supplement wild populations by the 1950s (Boyd 1957). In the 21st Century, captive-reared mallard are typically released on inland shoots that also release pheasant and red-legged partridge. The mallard are introduced in late summer (like pheasants and partridge) to ponds which are fed and sometimes protected from terrestrial predators by fencing etc. The mallard settle on these ponds and appear to move little from them. They may attract other wild mallard to roost or feed on the ponds, meaning that even on stocked ponds, the population that is hunted will contain both reared and wild-born mallards, further complicating discrimination of origins. Both released and wild-born mallard, including winter immigrants from Europe, are shot in the UK. The total annual bag in 2016 was estimated at 940,000 (95% CI 710,000-1,200,000) (Aebischer 2019). This is a change of +202% (95% CI 82-395) since 1966, +26% (-15-91) since 1991 and +18% (-11-54) since 2004) (Aebischer 2019). Determining how many of these harvested birds were reared and released as opposed to being wild-born is difficult. Released mallard are typically born from breeding stock maintained in farms for a number of generations and although they do differ slightly from wild-born birds both genetically and in terms of their behaviour and morphology, these subtle differences are not reported by shoots and little, if any, attempt is made to discriminate their origins (refs in Callaghan 1996).

B) Current Scale of Release

Various estimates of the total size of gamebird release have been made. We will not attempt here to accurately determine the number in this Review, but we present a range of values suggested by various publications and relate these to the set of data sources that we described above. Within an order of magnitude, it is probable that some tens of millions of gamebirds are released annually for shooting in the UK. Elsewhere, the numbers of gamebirds released is smaller but still substantial. In France more than 10 million pheasants and 2.5 million red-legged partridges are reared each year (ONCFS 2013). In the United States an estimated 10 million pheasants (as well as 37 million quails (*Coturnix coturnix*), one million mallards and 200 thousand turkeys (*Meleagris gallopavo*)) are reared each year (Burden 2013).

i) Pheasants

The most up-to-date published estimate is that in 2016, 47 million (95% CI: 39-57 million) pheasant were released in the UK (**Aebischer 2019). This estimate was based on calibration of NGC bag indices against PACEC aggregate totals from 2004 and 2012 estimates presented in PACEC reports. The APHA Poultry Register records for 2019 show that 11,196,463 pheasants were reported as reared for shooting and 10,039,379 were reported as held for release for shooting (90% of reared birds being released). If the estimate of Aebischer (2019) is correct, then this suggests a compliance level with the required poultry register of <25%. The lack of compliance with the poultry register is emphasised by figures from the Import Register. Records for 2019 show that 19,778600 pheasant eggs were imported into the UK from Europe (France: 17,627,250; Poland: 1,242,460; Portugal: 142,630; Spain: 767,260) along with 5,395,706 live pheasants (France: 5,382,706; Belgium: 12,000; Ireland: 1,000) (Rutley 2019). Hatching rates of 72% (N = 216,288) at a commercial rearer were reported (Nagy et al. 2013). This suggests that of the imported eggs, around 14,243,760 would hatch and contribute to the released bird population. Therefore, even if eggs and live birds were only sourced from outside the UK, > 4 million reared pheasants remain unaccounted for in release data. An extrapolation of release populations could be made from the BTO breeding bird data survey. It reports 2,300,000 females in GB during 2016 breeding season (Woodward et al. 2020). Assuming that each hen has a single partner (generous, because pheasants are a polygynous breeding species with single males holding harems of several females, although other males may be classed as non-reproductive satellites), and that between 9 & 16% of released birds survive to the following breeding season (Madden et al. 2018), we might crudely estimate a release of 29-51million pheasants. Therefore, there is evidence for annual releases of pheasants of somewhere between 10-57 million.

ii) Red-legged Partridges

The most up-to-date published estimate is that in 2016, 10 million (95% CI: 8-13 million) red-legged partridges were released in the UK (**Aebischer 2019). This estimate was based on calibration of NGC bag indices against PACEC aggregate totals from 2004 and 2012 estimates presented in PACEC reports. The APHA Poultry Register for 2019 shows that 4,607,688 partridges were reported as reared for shooting and 3,819423 were reported as held for release for shooting (83% of reared birds being released). If the estimate of Aebischer et al 2019 is correct, this suggests a compliance level with the required poultry register of <40%. The figures are more congruent with data from the Import Register than were the pheasant data. Records for 2019 (**Rutley 2019) show that 2,187,437 live partridges were imported into the UK from Europe (France: 1,722,269; Spain: 465,168). All these may be included in the poultry register. An extrapolation of release populations could be made from the BTO breeding bird data survey. It reports 72,500 territories in GB during 2016 breeding season (Woodward et al. 2020). Assuming that all survivors pair and breed, that each territory comprises a single male and female, and that survival of released partridges to breeding matches that of pheasants (see above), we might crudely estimate a release of 900,000-1.6 million partridges. Therefore, there is evidence for annual releases of red-legged partridges of somewhere between 1-13 million.

iii) Mallard

There has been little effort to quantify the scale or extent of mallard release. The only published estimate that we are aware of giving absolute numbers of birds released is from 1985 and at that time it was believed to be around 500,000 (Harradine 1985). Data from the APHA Poultry Register (2019) shows that 195,811 ‘Duck’ (presumed mallard) were reared for shooting in the UK annually and 435,907 were held for release for shooting (meaning that around 2.2 times more birds were released as were reported to be reared in the UK). This in itself suggests a compliance level of ~45% with the register. Data from the National Gamebag Census (NGC) reveals that the index of releases has been increasing with a change up to 2016 of +590% (332-1086) since 1966, 90% (30-215) since 1991 and 34% (6-67) since 2004. If we conservatively assume that releases in 1991 were similar to those in 1985, then a

90% index increase from a figure of 500,000 derived in 1985 would mean that around 1,000,000 mallard are currently released annually. PACEC (2014) estimate that around 1,000,000 mallard are shot in the UK annually. We do not know the proportions of wild to released birds in the bag. Estimations from BTO breeding bird data are unlikely to be informative given the presence of wild, native birds included in the sample. However, for completeness, 59,000-140,000 breeding pairs were recorded in GB during 2016 breeding season (Woodward et al. 2020).

C) Current Extent of Release

i) Classes of Shoots

Shoots that release birds differ greatly in the area that they occupy, form of shooting practised, regularity of shooting and the size of the bag harvested on any day. These factors likely interact to influence the number of birds that the shoot releases, the management of those releases and the associated land management practiced in the area. For other shoots for which we may have information about only one or two factors, we may be able to roughly assign them to a particular class of shoot if we have some broad descriptors.

The most comprehensive description of shoot structure that we are aware of are reports by Savills, based on an annual voluntary survey of the economic and environmental aspects of UK shoots (most recent: Teanby et al. 2019). Participation is open and voluntary with participants receiving a detailed comparative report in exchange for participation. Despite this, the participation is moderate with <160 shoots contributing each year. In 2016/17 (the year with the largest response and most detailed reporting from 155 shoots across the UK which between them released >1.6 million birds and shot >3,300 days https://www.gwct.org.uk/media/664264/shoot-benchmarking-example-participants-benchmarking-report-fictional.pdf, **Teanby *et al*. 2017 crudely classified shoots into three classes. Small shoots release up to 3000 birds (species unspecified) with a mean release of 1,532 birds and shoot over an average of 964 acres for an average of nine days per season during which time they shoot an average of 80 birds per day. Medium shoots released 3000-10,000 birds with a mean release of 6,212 birds and shoot over an average of 1,795 acres for an average of 16 days per season during which time they shoot an average of 148 birds per day. Large shoots released more than 10,000 birds with a mean release of 26,241 birds and shoot over an average of 3,860 acres for an average of 41 days per season during which time they shoot an average of 232 birds per day.

The best record of release data that we are aware of is the APHA Poultry Register. Because of the way that records are reported, we cannot consider total birds of all species reported as being held for release as released on a shoot. Instead, we used the composition of releases reported from 22 shoots on the Guns on Pegs website which indicated that, of the birds released on an average shoot, 73% were pheasants, 12% were partridges and 14% were mallard. For pheasants, the distribution of releases was very skewed (Fig. 3A) with 6/3307 shoots reporting releasing >100,000 birds (111,000-200,000), but a median release size of 850 birds/shoot. Following the definitions taken from the Savills report, 2,464 (75%)

shoots are classed as small; 543 (16%) are classed as medium; and 300 (9%) are large. For red-legged partridges, the distribution of releases was also very skewed (Fig. 3B) with 2/1323 shoots reporting releasing >100,000 birds (180,000 & 250,000), but a median release size of 500 birds/shoot. Following the definitions taken from the Savills report, 536 (41%) shoots are classed as small; 344 (26%) are classed as medium; and 451 (34%) are large. For mallard one record was for 150,000 birds released at a single shoot. This was 25 times larger than the next value (6,000), so we assume that it was a typo and have excluded it from Fig. 3C, but NOT any of our summary statistics such that total and mean values may be overestimates. There was a skewed distribution of release sizes, with a few shoots responsible for a disproportionate number of mallard releases. Following the definitions taken from the Savills report, 413 (72%) shoots are classed as small; 119 (21%) are classed as medium; and 45 (8%) are large.

We compared the distribution of shoot types derived from the Poultry Register data with that derived from the Guns on Pegs website. Of the 512 English shoots for which a mean daily bag size could be calculated, 167 (33%) shot <114 and so were classed as small; 149 (29%) shot between 114-190 and so were classed as medium; and 206 (40%) shot >190 and so were classed as large. Of the 388 English shoots that reported the acreage that they shot over, 148 (38%) occupied <1375 acres and so were classed as small; 126 (32%) occupied between 1375-2825 acres and so were classed as medium; and 114 (29%) occupied > 2825 acres and so were classed as large.

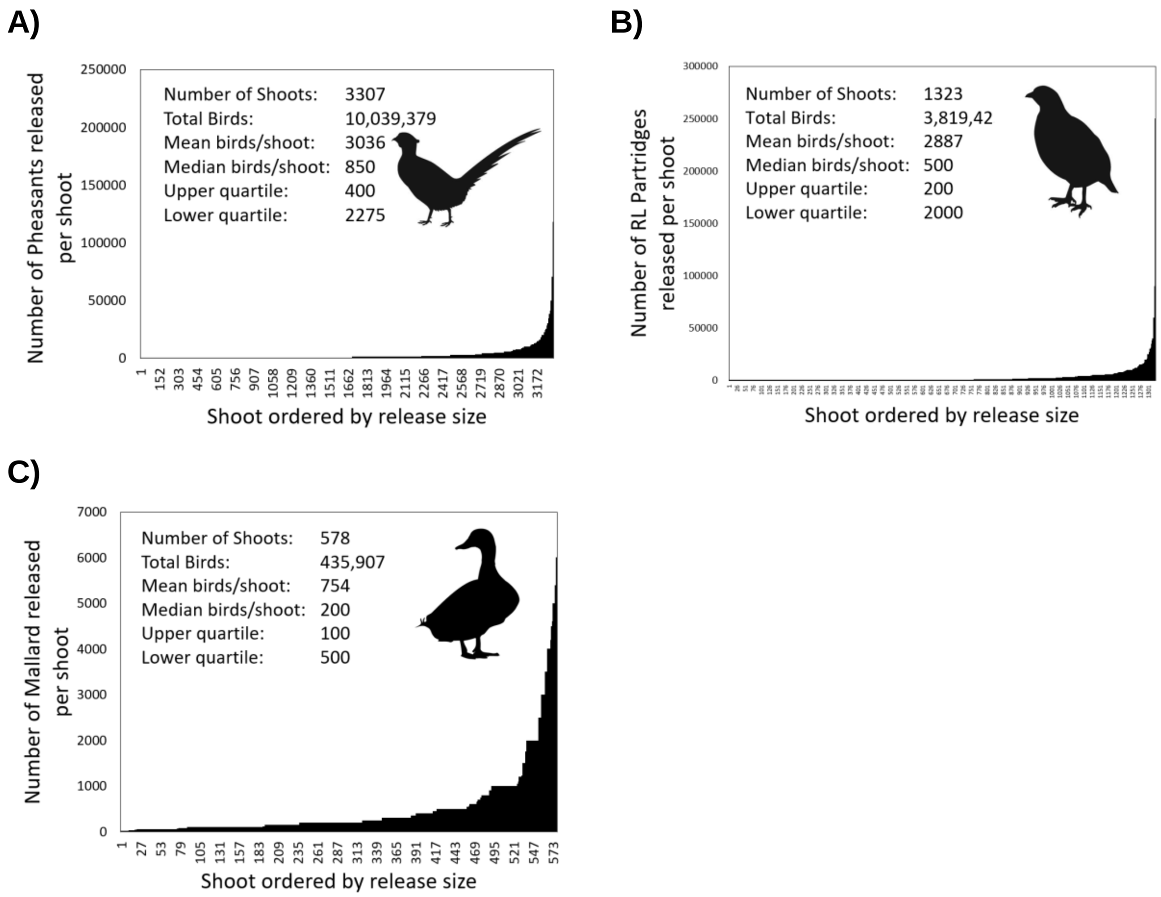


Figure 3. The distributions and summary statistics of three reared gamebird species (A) Pheasant; B) Red-legged partridge; C) Mallard) reported as being held for release for shooting by locations completing the APHA Poultry register.

Therefore, we conclude very crudely that approximately 50% of English shoots can be classified as small with 25% each being classified as medium and large. This classification may be helpful to understanding the ecological effects of gamebird release because larger shoots may also have disproportionately greater human activity and invest more financially and in terms of effort in land management activities such as game crop plantings, woodland management and supplementary feeding. They may also release birds at higher densities and/or be able to exercise more coordinated predator control over contiguous areas.

D) Current Geographic Distribution

The release of gamebirds is not homogenous across England. Historically, gamebird shooting was concentrated in regions where wild gamebirds were most prolific, namely the South and East of England. The practice of releasing gamebirds means that they can now be shot in areas less suitable for their breeding and survival. Identifying areas where gamebird releases are most numerous or intense can alert us to locations where we might expect to see their effects most markedly.

i) Patterns of release

In the absence of more precise fine-scale spatial release and shooting data, we plotted the total number of gamebirds of each species reported to be held in captivity for release in each county, divided by the total area of that county, to illustrate the national pattern of release (Fig. 4). Patterns differ between species of gamebird, but generally these maps suggest that counties with the densest releases are now more evenly spread than historically with concentrations in West Sussex, Suffolk, Wiltshire, Somerset, Oxfordshire, Worcestershire, Shropshire and Rutland.

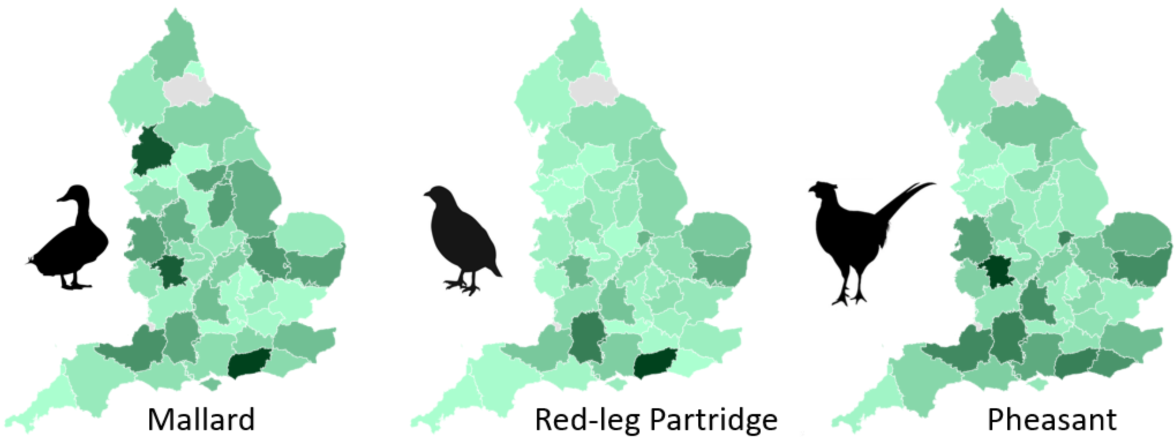


Figure 4. The number of gamebirds reported as held for release in the APHA Poultry register in each county, divided by the area of that county. Darker shades indicate higher numbers

ii) Patterns of shoot locations

We then plotted the distribution of the number of locations that reported per county (correcting for area) holding captive gamebirds for release to the APHA Poultry register and look to see how these corresponded to locations offering shooting of those species for sale on the Guns on Pegs website (Fig 5). These maps do not appear to correspond perfectly to maps of release, but again indicate that the highest densities of shooting are practiced in East Anglia and in a more westerly belt curving north from West Sussex up to Lancashire. The Midlands, apart from Rutland, appears to be less densely shot over.

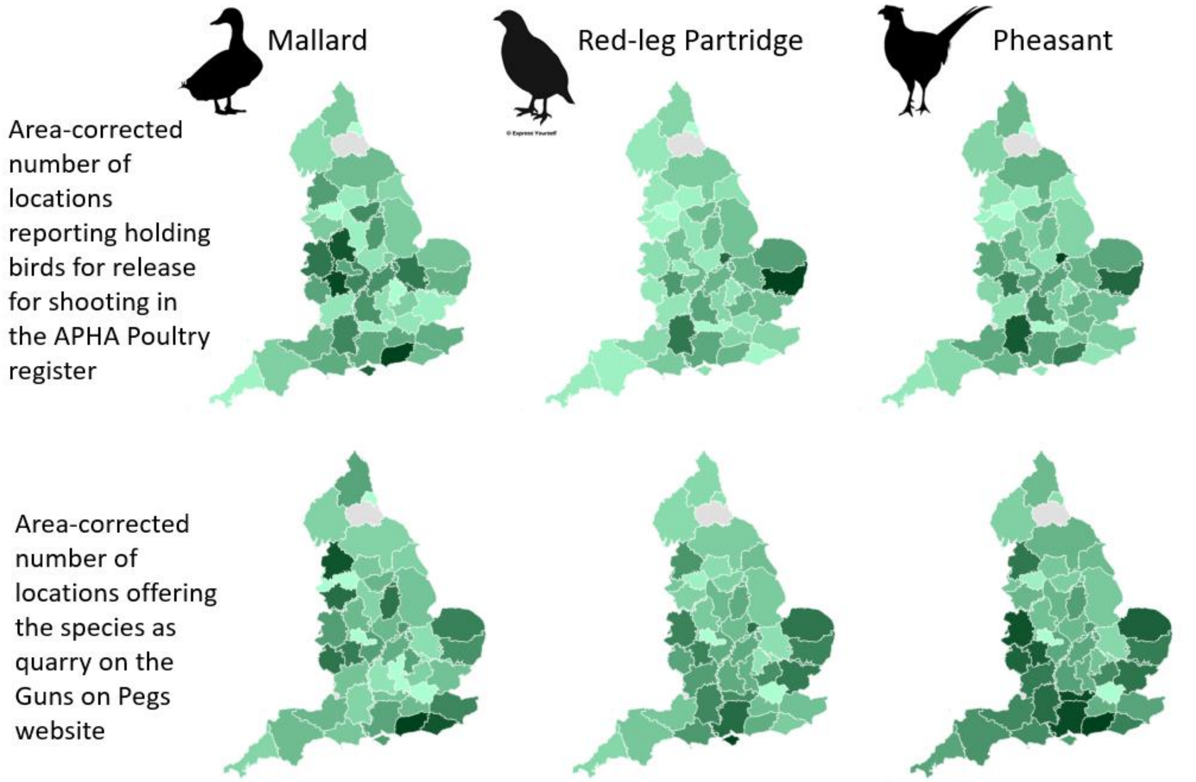


Figure 5. The area-corrected number of locations reporting to the APHA Poultry Register that they hold for release the three species of gamebirds (top line) and advertising shooting of those species on the Guns on Pegs website (bottom line). Darker shades indicate higher reported densities.

iii) Intensity of release

The extent of effects of release are likely to be related to local intensities of release and the associated management of woodlands and farmland. There are several indicators of intensity that we could plot. First, we considered the mean bag size (combined across all species) advertised by shoots on the Guns on Pegs website in a county (Fig. 6A). A large bag requires a large number of birds to be put over the guns. Highest bags were reported in Hertfordshire, Buckinghamshire, Gloucestershire, Leicestershire, Rutland and Cumbria all reporting mean bags >200 birds/day. We then plotted the mean size of shoots advertising on the Guns on Pegs website (Fig. 6B). Shoot area corresponds to shoot size (**Teanby *et al*. 2017) and this corresponds to the size of release, number of days shot and likely economic activity that may drive associated management practices. The largest shoots were seen in North and East Yorkshire and Leicestershire where mean shoot size was >3000 acres. There is likely to be an effect of land quality influencing these results. We combined the previous two Figures by considering the mean bag per day offered on a shoot divided by the mean area of shoots in that county (Fig. 6C). This might indicate where the highest density of releases occur with more birds being released on smaller areas. The highest such densities were seen in Southern England with Bedfordshire, East Sussex, Hertfordshire, Gloucestershire, Hampshire, Cornwall, Surrey, Cambridgeshire, Oxfordshire, and Worcestershire all reporting offering shooting of > 0.1 birds (of all species)/acre/shoot day. We considered the mean numbers of captive birds held for release per releasing location in each county of all birds reported in the APHA Poultry Register (Fig. 6D). The highest numbers held for release were seen in West & East Sussex, Somerset, East Yorkshire, Berkshire and Worcestershire that all reported sums of mean numbers held for release of >10,000 birds. Finally, we considered the total area-corrected number of gamebirds reported to be held for release (essentially combining and weighting the numbers used in Fig 4) and plotted these per county (Fig 6E). The counties of West Sussex, Worcestershire, Wiltshire, Rutland and Suffolk all reported releases of >500 birds/mile^2.

These attempts to illustrate release intensity show that it may be highly dependent on exact measures used with relatively little consistency in exact areas likely to be most affected. Crudely, release intensities appear to be highest in areas of Southern England, the Home Counties and West Midlands. It is in these areas that we might expect to see the greatest environmental effects of gamebird release.

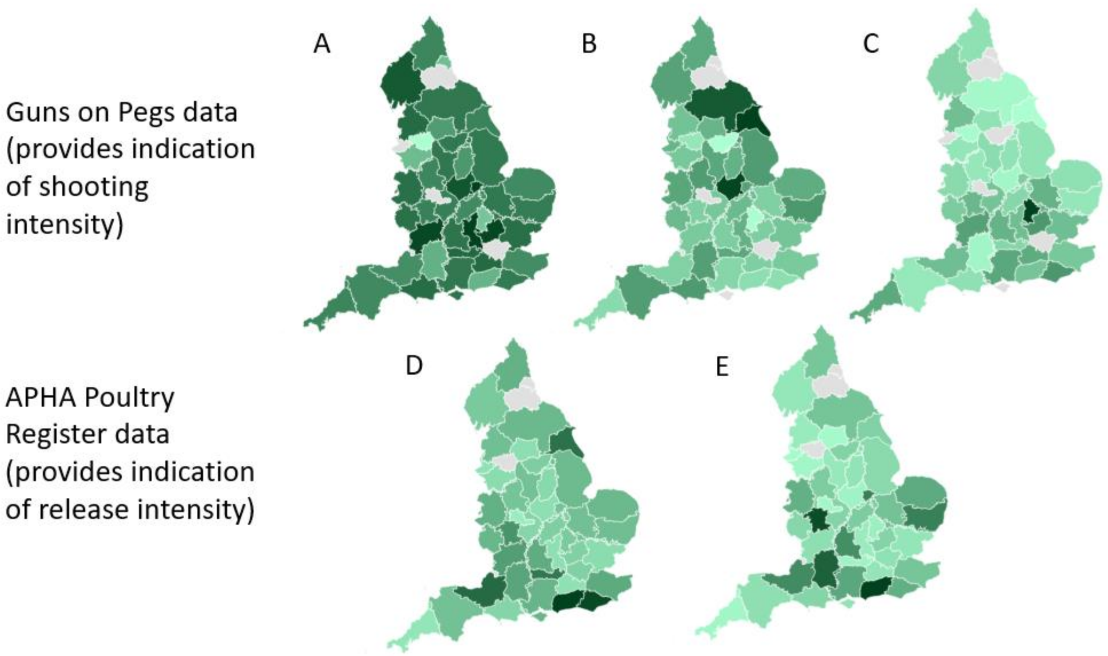


Figure 6. Indicators of release intensity. A) Mean daily bag size (all species) for shoots advertising on Guns on Pegs website. B) Mean acreage per shoot for shoots advertising on Guns on Pegs website. C) The mean daily bag divided by the mean acreage that it was shot over for shoots advertising on Guns on Pegs website. D) Sum across all three species of the mean number of birds per location as reported held for release for shooting in the APHA Poultry Register. E) Sum across all three species of the area-corrected number of birds as reported held for release for shooting in the APHA Poultry Register. Darker shades indicate higher intensity measures.

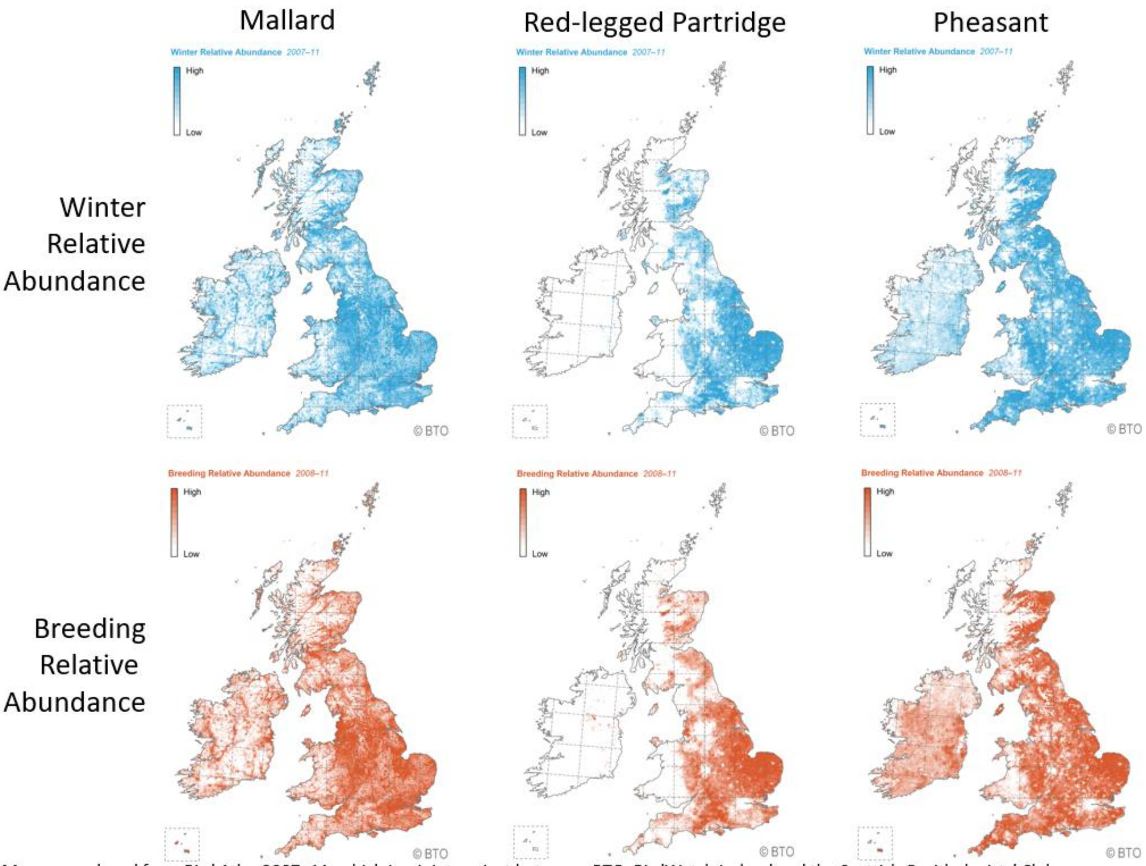
iv) Release Densities

The intensity of release is likely to be most marked at the release site where many birds may be introduced and encouraged to remain for several weeks. We found multiple papers describing how many Direct Effects of release including physical damage to soil and plants, and reductions in some invertebrates became evident or more marked when birds were released at high densities (e.g. Sage et al. 2005a, Gortazar et al. 2006, Pressland et al. 2009, Neumann et al. 2015, Porteus 2015, Capstick et al. 2019, but see Davey 2008). The Code of Good Shooting Practice (http://www.codeofgoodshootingpractice.org.uk/pdf/COGSP.pdf) advises that pheasants be released at densities of no more than 1000 birds/Ha generally or 700 birds/Ha when pens are situated in ancient semi-natural woodland. Several papers reported releases at higher densities than these (e.g. Neumann et al. 2015, Pressland 2009). There is some evidence that in the early 2000s, release densities declined. ***Sage *et al*. (2005) report release densities from two sample of about 50 pens each. In 1988, there was a mean release density of 2250 birds/Ha while in 2004 sample there was a mean release density of 1800 birds per hectare. They suggest that these declines were due to the withdrawal of the drug Emtryl, which stimulated the construction of larger pens to reduce disease build up, thus reducing densities.

v) Gamebird abundance after the shooting season

Little is known about the national distribution of released gamebirds after the end of the shooting season (but see section on dispersal below). Winter bird surveys by the BTO (that may include the latter part of the shooting season) (Fig. 7) (Balmer et al. 2013) show that pheasants were reported in 83% of surveyed quadrats, this being an increase of 11% since 1981-84; red-legged partridges were reported in 55% of surveyed quadrats, this being an increase of 75% since 1981-84; and mallard were reported in 93% of surveyed quadrats being an increase of 5% since 1981-84. From these surveys it was estimated that there were 665,000 mallard individuals present during winter 2012/13-2016/17 (Woodward et al. 2020).

Surveys later in the year during the breeding season by the BTO (Balmer et al. 2013) (Fig. 7) show that, for pheasants, 83% of GB 10x10-km quadrats have breeding records in 2008-2011 (7% possible, 20% probable, 56% confirmed). This is an increase of 6% since 1968-72 and an index increase of 0.08 since 1988-1991. For red-legged partridge, BTO breeding data show that 57% of GB 10x10-km quadrats have breeding records in 2008-2011 (8% possible, 25% probable, 24% confirmed). This is an increase of 78% since 1968-72 and an index increase of 0.21 since 1988-1991. For mallard, BTO breeding data shows that 94% of GB 10 km quadrats have breeding records in 2008-2011 (3% possible, 8% probable, 83% confirmed). This is an increase of 1% since 1968-72 and an index increase of 0.02 since 1988-1991.



Maps reproduced from Bird Atlas 2007–11, which is a joint project between, BTO, BirdWatch Ireland and the Scottish Ornithologists' Club.
Maps reproduced with permission from the British Trust for Ornithology.

Figure 7. Winter and breeding abundance of mallard, red-legged partridge and pheasant as detected during BTO led surveys form 2007-2011. These records include both released and wild-born individuals. Darker shades indicate higher relative abundances.

E) Gamebird Dispersal and Survival

The occurrence and abundance of released gamebirds in an area, determining their density and hence influencing their direct environmental effects, depends on their survival and dispersal and the time since their release. Game keepers are motivated to prevent released gamebirds from dispersing away from shoot areas and to manage their movements. A key tool is to provide appropriate habitat and food (GCT 1988; 1994; Robertson et al. 1993a) but game managers will also frequently herd dispersing birds back towards shoot areas (dogging-in).

i) Pheasant

**Turner (2007) studied released pheasant at six sites over three years in southern England. Sites were large shoots with at least one full time gamekeeper, driven shooting and with birds released into pens located in woodlands. She tagged and tracked 486 pheasants in total, between 24 and 30 individuals for each site/year combination. She tested the effect of stocking density, which was experimentally manipulated between years within sites, on fate and dispersal (no significant effects were found). She radio tracked birds for around 5 months but dispersal was investigated using data from the first three months when tracking was more frequent.

Overall, taking account of censored (lost) birds, 36 % of released pheasants were shot, 48 % died for other reasons (mostly predation) leaving 16 % still alive at the end of shooting (1 February). The overall average maximum distance moved was 913+82m. This is the average of the farthest distance each bird was recorded from the release point i.e. all other radio tracking locations were closer. Females moved further than males. Turner (2007) also estimated the home range area of individual radio-tagged birds for which there were sufficient data (at least 10 locations). Home ranges are routinely used in wildlife ecology to give an indication of the area of land used by an animal that isn't travelling. The mean overall home range size for male pheasants was 45+14 ha and for female pheasants it was 97+35 ha. This includes birds that did not remain faithful to one home-range area but wandered so that their home range enlarged. Turner presents these data graphically with an interquartile range which indicates that 75 % of females and 99 % of males had an overall home ranges of less than 250 ha. after 3 months. A circle with an area of 250 ha has a radius of 890 m. For the mean home range sizes described above, the radii are 380m for males and 560m for females.

***Madden et al. (2018) surveyed the literature (including material from outside Europe) and compared the survival of wild-born and released pheasants at different life stages. They constructed crude mortality curves and showed that although released pheasants survived well during rearing, once released, they died at higher rates than same-aged wild-born birds. They estimated that around 16% of released birds were still alive by the end of the shooting season.

**Beardsworth *et al.* (in prep). continuously tracked the movements of 168 released pheasants at a site in Devon in 2017). Their location was logged automatically every 4s. They plotted the mean value for all surviving individuals of their max distance away from the centre of the release pen from mid-August to the start of the shooting season (October 1). Birds slowly dispersed from the pen but almost all birds remained within 500m up to 1 October.

Mean dispersal distance from release pen to February catch-site in 24 reared and released pheasants on a shoot in Cambridgeshire was 266 ± 41 m (**Sage *et al.* 2001).

There are a few studies from elsewhere in Europe where small numbers of released pheasants have been radio-tagged and dispersal reported. In all cases the habitat and management programmes are very different to English shooting estates. A group of 20 or so captive reared and radio tracked pheasants stayed at the release site while a group of 10 caught wild pheasants moved around 1 km between March and August (***Bagliacca *et al*. 2008). **Ferretti *et al*. (2012) released 40 radio tagged pheasants into an agricultural and partially wooded area in Central Italy with no shooting in September and tracked them until April. They reported a mean dispersal distance (average maximum distance from pen) of about 410 ± 47 m and home ranges of about 12 ha.

**Turner (2007) found that on average around 15 % of released pheasants on the English shooting estates she studied survived until the end of shooting. Other radio tracking studies of hen pheasants during the spring in similar situations undertaken by the GWCT (*Sage *et al*. 2018c) indicate that around 60 % of these birds are then predated or scavenged between March and July. The (largely unpublished) movement data from these studies in Sage et al. (2018c) indicate that only a small proportion of these otherwise rapidly dwindling populations make any kind of further significant movements away from their release areas. Spring dispersal distances were reported for a group of 24 released pheasants in one of those studies at a shoot in Cambridgeshire in **Sage *et al*. (2001). Mean distance from release pen to nest sites was 503 ± 76 metres and from the February catch site to nest site was 350 ± 78 m.

ii) Red-legged Partridge

**Hesford (2012) (and others) studied released red-legged partridges at six sites (three in East Anglia and three in central southern England) over a 3-year period. All six sites had 1+ full time game keepers and driven shooting. Birds were released into pens located in game crops on farmland. In total 274 individuals were radio tagged, between 41 and 56 per site. Tracking per site was undertaken for around 5 months following release.

Overall fate of radio tagged birds was 38% shot, 34% died of other causes (mainly predation by foxes), 13% unknown (usually radio tag failure), leaving 15 % that survived beyond the end of the shooting season (31 January). Over this period the average final per-bird dispersal distance from the release pen was 408 m (this is not the same as Turners (2007) measure for pheasants which was the maximum dispersal location). 32 % of birds dispersed more than 500 m, 5 % more than 1 km and 1% more than 1.5 km.

There are a few studies from elsewhere in Europe where small numbers of released red-legged partridges or pheasants have been radio-tagged and dispersal reported. In all cases the habitat and management programmes are very different to English shooting estates. **Duarte (2010) tagged 20 red legs before release into a mountainous area, which showed poor survival (most died within one month) and a mean dispersal distance of 830m. **Alonso *et al*. (2005) reported mean dispersal distances of 378 m and similar poor survival for partridges released at six months old.

iii) Mallard

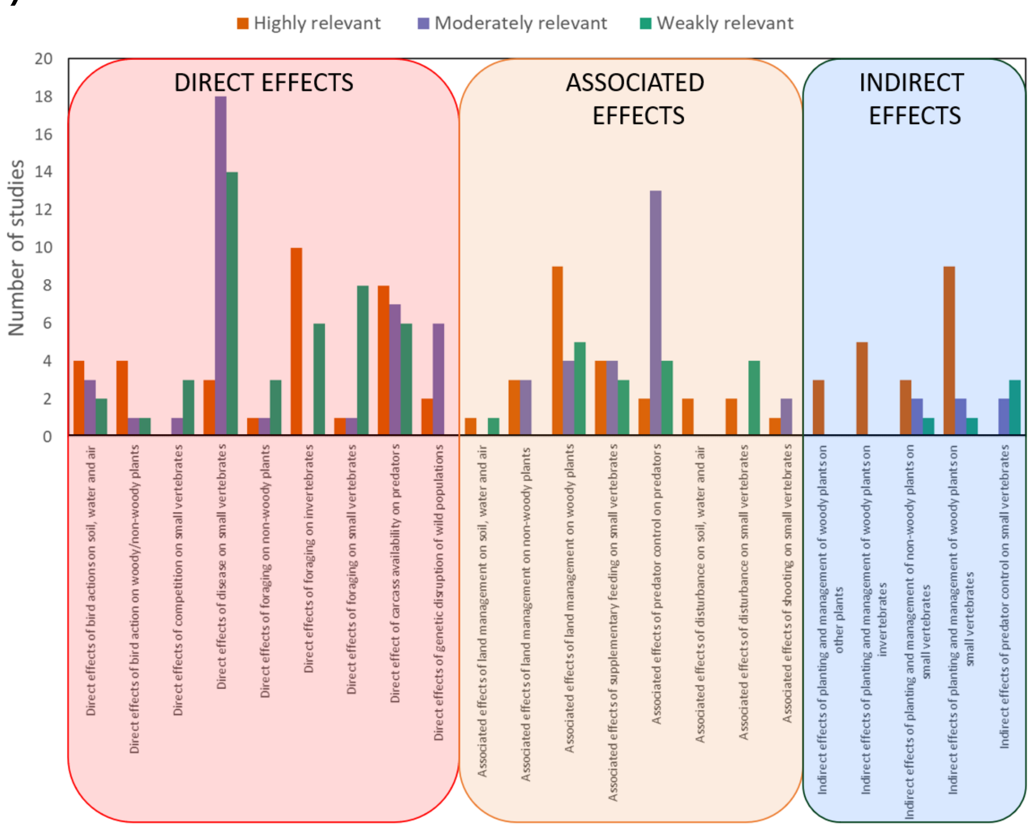
Released mallard survive less well than wild-born mallard (***Söderquist *et al*. 2013). For a Swedish sample, 77% of released mallard were dead in <1 year compared to 45% of wild-born mallard. For a Finnish sample, 90% of released mallard were dead in <1 year compared to 66% of wild-born mallard. This may be because reared mallard were in poorer body condition than wild-born mallard at the same site (***Champagnon *et al*. 2012).

Released mallard dispersed or migrated less well than wild-born mallard (***Söderquist *et al*. 2013). For a Swedish sample, released mallard were recovered a mean distance of 523 km (SE = 36) from ringing site whereas comparable measures for wild-born birds were 676 km (SE = 58). For a Finnish sample, released mallard were recovered a mean distance of 157 km (SE = 110) from ringing site whereas comparable measures for wild-born birds were 1,213 km (SE = 81). The authors note that these differences between early-life conditions may be partially driven by differences in survival (see above) and that differences between sites may depend on climate and landscape, but the general differences in dispersal distance appear to be consistent. Released and wild-born mallard did not differ in their behavioural time budgets for activity or foraging times in the wild (***Champagnon *et al.* 2012).

Knowledge gaps

Our Conceptual Model identifies several specific links that have no data or only anecdotal data often from just a few sites / years. Other links are fairly sparsely populated although are linked to (good quality) correlational data. Some other links are reasonably well populated with good studies (Figure 8, Appendix 3). Below, we briefly describe the sorts of data that are needed to calibrate such a model but which currently are missing, unreliable or insufficient.

A)



B)

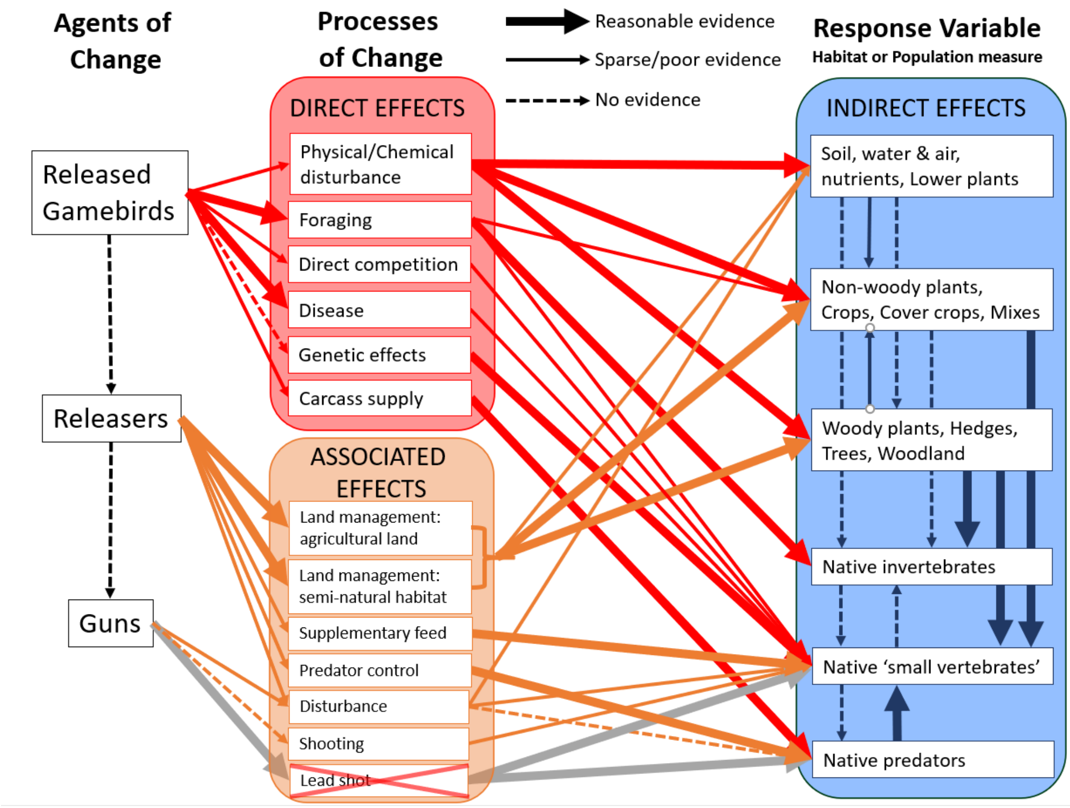


Figure 8. A summary of the number and relevance of studies considered in this Review with **A)** a direct comparison between studies of differing relevance within each Process of Change, and **B)** set within the framework of our Conceptual Model.

The size and location of gamebird releases

Data on the size and location of game releases is available from the APHA Poultry Register, voluntary participation in surveys such as the NGC or from mining data from advertisers of shooting such as the Guns on Pegs website. It appears from our review that there is poor compliance with the Poultry Register. This may lead to underestimates of numbers of birds released by studies that use these data. In addition, there is a risk of misinterpreting its data given ambiguities about breeding, rearing and release which may lead to double counting of birds. This may lead to overestimates of numbers of birds released. Similarly, participants in voluntary surveys or advertisers may not represent an unbiased sample of English game shoots. Advertisers may represent more commercial shoots that release larger numbers of gamebirds and/or have a larger financial base to conduct land management.

Therefore, it is unclear how accurate or reliable data regarding releases are. We advise that close attention is paid to how data about release sizes are gathered and interpreted and that future studies consider how to most accurately collect novel data and/or relate the available data to actual release patterns at a broader local or national scale.

We found that, in contrast to a reasonable body of work relating to pheasants and red-legged partridge, little research has been conducted on the environmental effects of released mallard on the habitats or wildlife of England. We relied on a body of work that is imperfectly related, or tangential, to the central question and which commonly confounded the effects of wild and released birds or was related to wildfowling practices (sometimes outside Europe) rather than the driven shooting of mallard as practiced on many English game shoots. Consequently, any conclusions drawn from this body of data should be treated with great caution. The literature that we include provides an indication of the kinds of effects that released mallard may have in England, but which require further, targeted study to elucidate effects reliably. The work done on the other gamebird species and some of the suggestion made for those above, could be uses as a reference point for future work on mallard.

The Direct Effects of released gamebirds

We found reasonable information for several sets of Direct Effects across a range of Processes of Change and Response Variables. However, several key knowledge gaps of likely ecological importance and public interest were obvious.

We found only patchy knowledge and poor understanding of the effects of diseases and parasites, introduced or harboured by released gamebirds, on native wildlife. Specifically, more information on diseases and parasites shared with other wildlife is required. It would also be helpful to compare disease or parasite loads in non-game species near and far from gamebird release locations.

In terms of direct effects on wildlife a study of impacts on reptiles is needed. There is a small amount of anecdotal evidence and a couple of unpublished and inconclusive studies that indicate that certain reptile species could be vulnerable to released pheasants. A targeted study of relationships between released gamebirds and sympatric reptile populations, perhaps similar to studies already conducted on invertebrate prey, is desirable.

While there are some population and behavioural data on foxes and some predatory birds, it is not clear how predator numbers respond to the release of gamebirds at the release site (through perhaps increased productivity or reduced territory size leading to higher densities) or more widely. The large quantities of biomass made available by the deaths of a substantial proportion of released gamebirds provides an opportunity for generalist predators. However, we also found evidence that game managers suppress these predators. Understanding the relationship between gamebird release and predator numbers is ecologically critical to determine net effects of gamebird release because the predators may also consume non-game species, perhaps especially so when released gamebird populations are low during the spring and summer. Such a study might include an assessment of changes in predator population, productivity, movement and foraging behaviour in response to the introduction and then declines of released gamebirds each year. Such work would need to consider the temporal patterns and local efficiencies of culling by game managers at and around release locations. Crucially, it will need to try an identify the contribution of pheasant releases in the context of the other land management practices, including many agricultural related activities and the supply of anthropogenic food sources and disturbance, which will also be influencing predators.

Despite the genetic effects of released mallard on wild conspecifics being reasonably well understood outside the UK, there are no data that we are aware of detailing effects locally or nationally within England. This may be difficult, given the migratory nature of many mallard. It is not clear what ecological effects any such genetic introgression may have.

The Associated Effects related to released gamebirds

We found some good and other reasonable evidence to link several aspects of Associated Effects to gamebird release. These include management of agricultural land including gamecrops and field edge habitats and management of semi-natural habitats such as woodland and hedges, supplementary feeding and predator control. However, this evidence is subject to some knowledge gaps and the link between game release and land management in the literature was sometimes implicit rather than explicit.

First, there is a need to quantify more carefully the scale and extent of woodland, hedgerow and edge habitat management on farmland and in semi-natural habitats and relate it with different scales of releasing. We have some good data on this in relation to game crops. The data from Teanby et al. 2017 suggest that as release size increases, so too does the size of area under game management, but the shape of this relationship, including saturation points or phase changes, needs clarifying. It is unclear how patterns of supplementary feeding and predator control effort and efficiency vary with different release sizes.

Second, there is a mixture of information about the size and location of planted or maintained habitats. Some is made using remote sensing methods and this may permit random or other robust site selection processes. Other data are self-reported. Self-report data may not always correspond to activity on the ground and self-reported probabilities of future behaviour (perhaps in response to changes in release practices) may be unreliable. Future efforts should attempt to better link remotely sensed habitat data to particular shoots with known release and land management practices. Future efforts should also establish the reliability of self-report data both for current management practice and predicted future behaviour in the face of changed circumstances.

Third, some of the data on woodland management activity rates, woodland planting and retention is out of date, being collected >20 years ago, although other studies have been undertaken within the last decade. The scale and practice of gamebird release has changed markedly over these periods. We advise that an effort is made to link the current scale and extent of land management, supplementary feeding and predator control to current patterns of gamebird release. This is essential if an evaluation is to be made of the net effects of gamebird release and to predict future changes to Response Variables of interest if patterns of gamebird release were to change. Without such data, future increases in releases (perhaps driven by the economic interests of game shoots) or decreases in releases (perhaps driven by public objections to the release and shooting of gamebirds) could risk initiating a series of net negative ecological effects.

We found very sparse data on the effects of disturbance by game managers or guns across Response Variables. Future work should consider both possible negative and positive effects of disturbance (for example consequences of restrictions or discouragement of public access) leading to changes in the occurrence, abundance, productivity and behaviour of non-game species.

We found few data relating to the behaviour of guns and beaters involved in released gamebird shooting. Areas of interest for future work could include the intentional or unintentional shooting of species other than released gamebirds while on a game shoot and the disturbance caused by the guns, beaters and dogs during a shooting day. It was notable that the majority of papers that collected data from game managers or guns and those papers that compared the behaviour of game managers to other people were at least 20 years old.

The extent of illegal raptor killing in association with releasing for shooting is unclear, and certainly much less clear than its association with some wild game management. Because the data are hard to collect, it is unclear whether the scale of any illegal raptor killing has ecological effects. Therefore, whether it is a key research strand necessary to inform future assessments of net ecological effects is debatable. Our (JRM & RS) opinion is that it is simply illegal, should be subject to the law and should be stamped out immediately. We found weak evidence to suggest that illegal killing of raptors was related to gamebird release, although we acknowledge that data about individual crimes is hard to obtain and the aim of prosecution is usually to determine guilt rather than motivation. We advise that data concerning the spatial and temporal distribution of such crimes be compared to data relating to patterns of gamebird release so as to better understand whether gamebird release prompts illegal raptor killing.

The Indirect Effects arising from the release of gamebirds

We found a poor understanding of most indirect effects. There is good evidence that other birds use game crops planted for released gamebirds during the winter, but while there is some evidence that this supports local breeding populations or increases productivity it is less good. Similar evidence exists for birds, invertebrates and small mammals in woodland. It is unclear whether these higher numbers are the result of enhanced local productivity and recruitment within these populations, or of immigration from other areas into preferred habitats. We suggest that future work should explore the consequences of game management practices not just on occurrence and abundance of species of interest within the game shoot, but also on local productivity and movement and distribution of species in the immediate and surrounding area.

One particular Indirect Effect that has attracted much interest and speculation is the suggestion that large-scale late-summer releases in the UK enhance the abundance of generalist predators, which go on to have detrimental impacts on breeding farmland birds in the following spring. We found no good evidence to support or refute this (including our interpretation of Pringle et al. (2018)). We suggest that this is a question of broad interest and ecological significance and deserving of prompt future study. Answering it requires an understanding of how predator numbers, both locally and more widely, change in response to the release of gamebirds on a shoot and the land management, including farming activities, and predator control that accompanies that release. It also requires an understanding of how the behavior and diet of those predators changes over the year as gamebird numbers decline. Finally, this work should be accompanied by study of likely alternative prey species, perhaps identified from predator diet, considering their abundance and productivity, with the effects considered both within and beyond game shoots.

Summary and conclusions

The ecological consequences of gamebird release form a significant part of the debate about lowland gamebird shooting in the UK, where more birds are released than in any other country. This review focused on ecological effects directly attributable to releasing, managing and shooting pheasants and red-legged partridges and the creation and management of the habitats they occupy. The review used a formal search and review procedure to identify primary and other literature relevant to answer the question: what are the effects of releasing gamebirds on the habitats and wildlife of England? We have examined this material and briefly summarized and crudely assessed it so as to provide an insight into current knowledge of the effects of gamebird releasing for shooting and its associated management on habitats and wildlife in England. This examination has had two broad aims. First, to summarize and organise key findings from a diverse literature base, including unpublished and grey literature, into sections so that the review can be used as a future source of information or as a route into the literature. Second it provides the basis for an overview and summary of effects set within a holistic conceptual model which also facilitates an appraisal of where knowledge exists and the identification of knowledge gaps.

In total, 229 papers contained evidence or data directly relevant to the question. These are identified in sub-topic sections in the Critical Appraisal and in Mediating Factors and contain summarized details of key findings from references that are not always available in an abstract. While gamebird release occurs elsewhere in Europe, the scale of release is lower and habitats, wildlife and shooting and management practices do not always match those in England. Work conducted in other countries may prove indicative of ecological effects likely to be seen in England, but it should be treated with caution. Much of the material that we present is drawn from the grey or unpublished literature. In our view, many of the grey literature studies were not necessarily of inferior quality to published work and often represented the only information about particular processes. The field of gamebird release and land management has attracted relatively little academic interest, so much work has been driven by industry, government or NGOs and this may explain the relative wealth of grey studies compared to the published literature. The previously published Reviews (see Introduction) have often missed out such material.

The papers that we considered fell into three broad categories. The first set of 58 papers (which we showed indicated by ***) permitted us to formally compare measures of interest relevant to releasing and management between a treatment group and a control group, or to explore relationships between such measures and variation in release size or across spatial scales. Most of these papers made use of ‘natural experiments’ in which some areas hosted game shoots while others did not. In some cases, an attempt was made to pair treatment and control sites. Site selection often (but not always) relied on voluntary participation both by game shoots and ‘control’ areas. Likewise, studies of the behaviour of game managers was often compared with a control group of land managers (i.e. farmers) that did not host shoots. We found no that deliberately experimentally manipulated gamebird release patterns and monitored associated change. Such experimental manipulations are highly desirable in order to determine causality in complex ecological systems and, while we acknowledge that this may be logistically problematic, we suggest that for key knowledge gaps and processes of change identified as likely being particularly influential, the possibility of conducting landscape scale experiments is considered.

Although sample sizes varied between papers, cumulatively the findings that we considered were obtained from field-work undertaken at many hundreds of different release-based shoots over several decades. Such field studies are often dependent on the (voluntary) participation of the land owner or game manager and as such may not be an unbiased representation of the size or structure of English shoots or of management practices conducted nationally, but we have no evidence to support this concern. Therefore, we believe that until further data are forthcoming, the findings of the review should be interpreted as representing a median type of shoot in terms of size and adherence to good practice over that period. However, during the period covered by our Review (and especially since the mid 1990’s), releasing numbers have steadily increased (Robertson et al. 2017). The pace of change in gamebird release, game management and shooting is such that the relevance of the (relatively small amount of) earlier work is less certain than that of more recent work. However, we have not attempted to weight studies by their date of completion but advise that older studies are examined carefully for their current relevance. The fact that several studies reported spatial variation in their results (e.g. Cox et al. 1996, Heydon et al. 2000, Woodburn & Sage 2005, Devlin 2019) indicates that findings from one area may not simply transfer to other areas. There are currently insufficient studies to permit us to account for these spatial variations in a robust and formal manner. We have not attempted to weight or account for spatial variation but we advise that drawing national conclusions based on studies from single sites is risky and we encourage attempts to replicate findings at multiple sites.

Some previous Reviews (e.g. Bicknell et al. 2010, Gallo & Pejchar 2016, Mustin et al. 2018) defined the various effects of a released gamebird or the game manager as positive or negative. We believe that whether a Direct, Associated or Indirect Effect is classed as positive or negative may sometimes be subjective and/or context dependent. For example, the availability of gamebird carcasses may support a higher population of predators. If the predator species is rare or endangered, then an increase in their numbers may be desirable and seen as a positive outcome of gamebird release. Alternatively, if the predator is a generalist and it’s increase leads to depletion of non-gamebird species then this may be undesirable and can be seen as a negative outcome of gamebird release. Another example: supplementary feeding may support larger populations of overwintering passerines but may simultaneously support increased rodent numbers. Such rodents may be pests but may also be desirable species of conservation concern or provide prey for predators of conservation concern. Should supplementary feeding be considered to exert a positive or negative effect? Therefore, we have avoided defining the direction of each individual effect on the Response Variables. We advise that future work needs to clearly determine the specific ecological outcomes that are of interest and carefully consider and assign the direction of each effects in order to arrive at meaningful net outcomes. The net direction of effect (positive or negative) for any one Response variable may only become apparent when the cumulative effects of multiple Processes of Change and Mediating Factors are considered using the Conceptual Model we propose or similar.

We note that there are some coarse patterns relating to the effects of releasing and managing gamebirds. In general, effects that we might consider to be subjectively positive are usually a consequence of gamebird management activities (Associated Effects) and most effects that we might consider negative are caused by the released birds themselves (Direct Effects). We found reasonable evidence for physical disturbance of soil, nutrient enrichment of water and soil, reductions in non-woody plants (especially those of conservation interest) due to damage or enrichment and reductions in abundance and/or diversity of at least some invertebrate species at or close to release sites. We found weaker, less or more ambiguous evidence that the released birds predated small vertebrates (reptiles), posed a direct competition to non-game species, spread disease to non-game species, influenced the genotypes of wild conspecifics (in England) or that their carcasses supported increases in generalist predators. Some negative Direct Effects have relatively straightforward management solutions and these are discussed in the relevant sub-sections.

We are aware of a small body of literature that describes how some of these Direct Effects may be moderated by changing the behaviour of the released birds. The early rearing conditions that a bird experiences early in life during artificial rearing may determine its post release vigilance, foraging choices, response to predators, dispersal behaviour and ultimately its survival (e.g. Miilan et al. 2003, Gaudioso et al. 2012, Whiteside et al. 2015, 2016, Hall in prep). By adopting such enhanced rearing conditions, it may be possible to reduce some of the negative Direct Effects such as the number of gamebirds being predated (thus sustaining high predator populations), or their risk of moving far from release pens into ecologically sensitive habitats, or predation of invertebrate prey. These solutions need further exploration. Other Direct Effects may be moderated by changing the scale and location of releases. Some negative effects such as effects on reptiles or sensitive lichen communities involve very specific conflicts with nature conservation interests which can likely be reduced or eliminated if sensitive sites are identified and avoided when releasing gamebirds. Several studies revealed that the strength of these (negative) effects grew stronger as the density of released birds in the pen and surrounding area increased (e.g. Sage et al. 2005a, Gortazar et al. 2006, Pressland et al. 2009, Neumann et al. 2015, Porteus 2015, Capstick et al. 2019, but see Davey 2008). Working within the normal range of releases described in the papers we reviewed (a few hundred birds to a few thousand gamebirds in any one pen) it was a consistent result across studies that smaller releases had a reduced effect. This has been previously suggested and it forms the basis of the Code of Good Practice, advising that no more than 1000 birds be released per Ha of pen and that only around one third of woodland on a shoot be used as a pen. However, it is evident from the densities reported in the papers that, while average pen densities have in general gone down in recent years (presumably because pens have become larger), this advice is still often ignored.

The actions of game managers had a range of effects that we considered to be associated with, or motivated by, gamebird release. We found reasonable to good evidence that they engaged in land management of agricultural and semi-natural habitats which involves planting, maintaining and retaining areas of woodland, rough ground, ponds, hedges and game crops at levels higher than other land owners. This land management was typically accompanied by increases in numbers or diversity of plants, invertebrates and non-game vertebrates in those areas of the game shoot. Several of these were species of conservation interest. It is important to better understand whether and to what extent these benefits are spread more widely to areas of the game shoot that are not so intensively managed and further out to neighbouring areas where release and shooting does not take place. While it is usually clear that releasing provides the motivation and economic underpinning of the management of these habitats, it would be useful to define more specifically the motivations and actions of land managers with respect to gamebird release using techniques other than questionnaire or similar surveys. This would allow us better understand the link between reported intended behaviour and actual actions arising from changes in release patterns, which would permit more accurate predictions of management changes and accompanying ecological effects in response to any future alterations in gamebird releases.

The legal control of generalist predators led to lower than expected numbers of such predators locally, although the evidence for this was weak and this may reflect the mobility of predators or the inefficiency of much predator control. We found little evidence that predator control associated with gamebird release itself led to increases in populations of non-game small vertebrates. Predator control can help many species of for example birds, especially when breeding (Roos et al. 2018) but it's the effectiveness of predator control in association with releasing, which usually occurs after the end of the breeding season that isn't so clear. Another related Indirect Effect proposed in several papers and commonly asserted in the non-scientific literature is that generalist predators thrive on abundant gamebird carcasses (see above) and this leads to overall decreases in non-game species. This is a reasonable hypothesis, but we found no studies offering evidence to support it. Because this is a potentially large scale effect on a wide range of vertebrate species of conservation concern, we advise that efforts be made to explore, experimentally and in the context of other contributing factors, such as farming activities, whether one indirect effect of gamebird release is the reduction in abundance or diversity of non-game species due to prolific predators. We found weak evidence to suggest that illegal killing of predators was related to gamebird release, although we acknowledge that data about individual crimes is hard to obtain and the aim of prosecution is usually to determine guilt rather than motivation. We advise that data concerning the spatial and temporal distribution of such crimes be compared to data relating to patterns of gamebird release so as to better understand the extent to which gamebird release prompts illegal raptor killing.

The provision of supplementary feed was accompanied by increases in a range of non-game small vertebrates and while many birds and mammals benefit, it is sometimes a matter of perspective as to whether this is ecologically positive or not, depending on the species considered. We found weak evidence that the actions of the guns and beaters during shooting led to disturbance of and potentially unintentional killing of non-game species. The effects of guns and beaters (disturbance, killing of wild species, use of lead shot) is not inevitably linked to gamebird release; it occurs when shooting wild game too. Likewise, the use of lead shot is not restricted to the shooting of released gamebirds. We did not consider the environmental effects of lead shot in this Review, but the work by Pain et al. (2019) describes them as uniformly negative. We advise continued monitoring of these effects as the proposed phasing out of lead shot takes place over the next five years.

In order to calculate the net effects of gamebird release either at a local or national scale, it is necessary to have reliable data about the size, location and past history of releases and the size of area over which released birds disperse and associated game management occurs (some of what we describe as the mediating factors). At present, none of these data are easily available. The numbers of birds released are not documented, nor are the locations where they are initially placed. While individual shoots may be willing to report this data and provide information about where they practice land management or the extent to which they engage in supplementary feeding or predator control, these measures appear to be highly heterogenous and it is not always clear that data from shoots that participate in research can be extrapolated to other shoots or to regional or national scales. From a research perspective, such data would be desirable, but reliable mechanisms to collect these data do not currently exist. These data are critical in order to robustly evaluate net effects because the spatial scale of the different effects varies and may co-vary with release numbers or density (see above).

Most documented negative effects are very spatially confined, usually at the release site or feed point although others, in particular disease issues and the effect of releasing on generalist predators, may occur at a landscape scale. In contrast, most of the positive effects of management for releases occur at the scale of a whole woodland or across an estate or farm. Therefore, to determine net effects it is necessary to weigh small areas of apparently high levels of damage (i.e. within release pens) vs large areas where moderate benefits might accrue (i.e. managed landscapes). Depending on the Response Variable of interest, sometimes both scales and effects need to be considered simultaneously. E.g. is it worth accepting reductions of woodland invertebrates in a 1 Ha release pen in order to motivate the creation or management of 10 Ha of woodland elsewhere that benefits those same invertebrates? Do those benefits outside the pen still persist if the release density is high and gamebirds disperse or are drawn (by the management actions) into that other woodland? Are the benefits of local reductions in predator numbers due to (legal) control worth the risk of boosting numbers of generalist predators over a larger area beyond the game shoot? It can become more problematic when comparing different ecological currencies. E.g. is it worth accepting the risk of disease transfer from released gamebirds collecting at feeders in exchange for the probability that songbirds may gain nutritional benefits from supplementary feeders over winter? This illustrates the need to consider all the evidence that we have evaluated, and any that is collected in the future, in a holistic manner, perhaps using our Conceptual Model as a framework.

Our Review has identified a series of Processes of Change that can lead to damaging Direct and Associated Effects on habitats and wildlife of England related to gamebird release. The evidence suggests that at least some of these effects can be ameliorated by following best practice relating to release sizes and densities and by consideration of release site locations and the rearing conditions of gamebirds for release. Potential solutions to many of these damaging effects exist, and we have indicated some of these in our Review. Equally, our Review has also identified a series of Processes of Change that can lead to beneficial Direct and Associated Effects on habitats and wildlife of England related to gamebird release. These actions such as habitat management of woodlands and field edges, careful and timely supplementary feeding and appropriate predator control, can be enhanced through the deployment of best practice. Ultimately, either an increase in damaging effects arising from poor or excessive game management, or a decrease in beneficial effects arising from the reduction or cessation of gamebird release could contribute to net negative ecological outcomes. To achieve net positive ecological outcomes for the habitats and wildlife of England, it is necessary to carefully consider and act to simultaneously reduce the negative and enhance the positive effects of gamebird release, both today and in the future.

APPENDIX 1: Letter requesting additional material from interested researchers

From: Rufus Sage <rsage@gwct.org.uk>
Sent: 20 January 2020 11:48

Dear All,

In November, along with others, Joah Madden (University of Exeter) and I were approached by Natural England and BASC to provide a quote for undertaking a literature review of evidence of the ecological effects of gamebird release in England. Our combined proposal was accepted in December and we met with NE/BASC two weeks ago to start that work. It will be completed by early March.

The evidence from conventional published literature is patchy and therefore NE/BASC have asked us to draw on all potential sources in the grey literature. The scope of the review is fairly tightly defined by them and we will not be eliciting 'expert opinion'. However given your expertise in the field we should like to ask you if you have any suggestions of grey literature or forthcoming publications that may be relevant**.** We are aware of some material but would rather be sent repeated suggestions than miss anything.

This call would cover any work pertinent to the ecological effects of gamebird releasing (pheasant, red leg and mallard, not grey partridge) with a particular focus on effects in the UK. The Review is not expected to produce policy recommendations or even draw definitive conclusions, but rather lay out the current state of knowledge and highlight gaps. Our review will follow a strict protocol including screening for relevance and critical appraisal, following guidelines supplied by NE. Your contributions will go through the same process as all other material. We will acknowledge any such contributions in the final Review report. We'd like any responses by Friday 7 February - apologies for the tight deadline.

If you would like to suggest grey literature for consideration, please either send us a weblink to an open access version of the study, or a scan/pdf on email. If necessary, please send us a hard copy to GWCT Fordingbridge address.

If you would like to submit unpublished data or draft manuscripts for consideration, please deposit the work on a pre-print server and send us the DOI/link where applicable.

We'd be extremely grateful if you can contribute unpublished work or suggest other grey literature that you may be aware of for consideration in the Review. If necessary feel free to pass this on to a colleague within your organisation who you may feel can also, or is better placed, to respond.

Rufus Sage and Joah Madden

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APPENDIX 2: Summary of the 58 papers deemed highly relevant to this review

See Excel file Appendix 2

APPENDIX 3: Summary of the number of papers of differing relevance to the various links within our Conceptual Model

See Excel file Appendix 3

References

All weblinks checked 9 March 2020

Abbasi, S., Momtaz, H., Rahimi, E., Momeni, M., & Riahi, M. (2012). Detection and assessment of antimicrobial resistance properties in *Escherichia coli* O157 isolated from pheasant, partridge, duck and goose meat. *Pajoohandeh Journal* 17, 210-214.

Aebischer, N.J. (2019). Fifty-year trends in hunting bags of birds and mammals and calibrated estimation of national bag size using GWCT's National Gamebag Census. *European Journal of Wildlife Research* 65, 64-77.

Aebischer, N.J., Bailey, C.M., Gibbons, D.W., Morris, A.J., Peach, W.J. & Stoate C. (2016). Twenty years of local farmland bird conservation: the effects of management on avian abundance at two UK demonstration sites. *Bird Study* 63, 10-30.

Aebischer, N.J. & Ewald, J.A. (2010). Grey Partridge *Perdix perdix* in the UK: recovery status, set-aside and shooting. *Ibis* 152, 530–542.

Aldous, E.W. & Alexander, D.J. (2008). Newcastle disease in pheasants (*Phasianus colchicus*): a review. *The Veterinary Journal* 175, 181-185.

Alonso, M.E., Pérez, J.A., Gaudioso, V.R., Díez, C., & Prieto, R. (2005). Study of survival, dispersal and home range of autumn-released red-legged partridges (*Alectoris rufa*). *British Poultry Science*, 46, 401-406.

Alsop J., Goldberg E. (2018). *Synthesis of evidence and statement of rationale: Cessation of pheasant (Phasianus colchicus) feeding and game driving activities within Meadow Place Wood on the Derbyshire Dales NNR.* Natural England report NE2018-DDNNR-MPW-PE003.

Amar, A., Hewson, C.M., Thewlis, R.M., Smith, K.W., Fuller, R.J., Lindsell, J.A., Conway, G., Butler, S. & MacDonald, M. (2006). *What is happening to our woodland birds? Long-term changes in the populations of woodland birds.* RSPB Research Report Number 19, BTO Research Report Number 169.

Anderson, R.C. (2000). *Nematode parasites of vertebrates: their development and transmission*. Second edition. – CAB International, Wallingford, Oxon, UK.

Anon (2015). *The personal value of shooting*. https://basc.org.uk/shooting-sports-is-good-for-mental-health/

Anon (2019). *Poultry (including game birds): registration rules and forms*. https://www.gov.uk/government/publications/poultry-including-game-birds-registration-rules-and-forms

Arroyo, Beatriz; Beja, Pedro (2002). *Impact of hunting management practices on biodiversity. Renconciling gamebird hunting and biodiversity (REGHAB)* report 78pp. Centre for Ecology and Hydrology.

Avery, M. (2019). The Common Pheasant: its status in the UK and the potential impacts of an abundant non-native. *British Birds* 112, 372-389.

Bagliacca, M., Falcini, F., Porrini, S., Zalli, F., & Fronte, B. (2008). Pheasant (*Phasianus colchicus*) hens of different origin. Dispersion and habitat use after release. *Italian Journal of Animal Science* 7, 321-333.

Balmer, D.E., Gillings, S., Caffrey, B.J., Swann, R.L., Downie, I.S. & Fuller, R.J. (2013). *Bird Atlas 2007-2011: the breeding and wintering birds of Britain and Ireland.* BTO Books, Thetford.

Bandelj, P., Blagus, R., Trilar, T., Vengust, M. & Rataj, A.V. (2015). Influence of phylogeny, migration and type of diet on the presence of intestinal parasites in the faeces of European passerine birds (*Passeriformes*). *Wildlife Biology* 21, 227-233.

Barbanera, F., Pergams, O.R.W., Guerrini, M., Forcina, G., Panayides, P. & Dini, F. (2010) Genetic consequences of intensive management in game birds. *Biological Conservation* 143, 1259–1268.

Barbanera, F., Forcina, G., Cappello, A., Guerrini, M., Grouw, H.V. & Aebischer, N.J. (2015). Introductions over introductions: the genomic adulteration of an early genetically valuable alien species in the United Kingdom. *Biological Invasions* 17, 409–422.

Beale, C.M., & Monaghan, P. (2004). Human disturbance: people as predation-free predators?. *Journal of Applied Ecology* 41, 335-343.

Beardsworth, C. Capstick, L., Langley, E., Nathan, R. Orchan, Y., Toledo, S., van Horik, J., Whiteside, M. & Madden, J. (in prep). Spatial cognitive ability predicts the development and structure of movement paths in a novel landscape.

Beebee, T.J.C. & Griffiths, R.A. (2000). *Amphibians and reptiles: A natural history of the British Herpetofauna*. Harper Collins, London.

Beer, J.V. (1988). *Nutrient requirements of gamebirds*. Pages 195-203 in Recent Advances in Animal Nutrition. D.J.A. Cole and W. Haresign, ed. Butterworths, London.

Beja, P., Gordinho, L., Reino, L. et al. (2009) Predator abundance in relation to small game management in southern Portugal: conservation implications. *European Journal of Wildlife Research* 55, 227-238.

Berthon, G. (2014). *Reptiles and pheasants.* Unpublished MSc project report. GWCT.

Bertran, K., Dolz, R. & Majó, N. (2014). Pathobiology of avian influenza virus infection in minor gallinaceous species: a review. *Avian Pathology* 43, 9-25.

Bicknell, J., Smart, J., Hoccom, D., Amar, A., Evans, A., Walton, P. & Knott j. (2010). *Impact of non-native gamebird release in the UK: a review*. RSPB Research Report Number 40.

Blackburn, T.M. & Gaston, K.J. (2018). Abundance and energy use of native and alien breeding birds in Britain. *Biological Invasions* 20, 3563–3573.

Blanco-Aguiar, J.A., Gonzalez-Jara, P., Ferrero, M.E., Sanchez-Barbudo, I., Virgos, E., Villafuerte, R., Davila, J.A. (2008). Assessment of game restocking contributions to anthropogenic hybridization: the case of the Iberian red-legged partridge. *Animal Conservation* 11, 535–54.

Blanke, I. & Fearnley, H. (2015). *The sand lizard: Between light and shadow*. Laurenti Verlag.

Bosanquet S. (2018). *Lichens and N pollution at Allt-y-gest SSSI – implications for pheasant rearing.* Natural Resources Wales Evidence Report No 295 23pp.

Boyd, H. (1957). The use of hand-reared ducks for supplementing wild populations. *Wildfowl*, 8, 91-95.

Brickle, N. (1997). *The use of game cover and game feeders by songbirds in winter*. Proceedings of the 1997 Brighton Crop Protection Conference, BCPC, 1185-1190.

Bright, P.W. & Morris, P.A. (1990). Habitat requirements of dormice *Muscardinus avellanarius* in relation to woodland management in southwest England. *Biological Conservation* 54, 307-326.

Burden, D. (2013). *Game-Bird Preserve Business Development Guide.* https://lib.dr.iastate.edu/cgi/viewcontent.cgi?article=1062&context=leopold_pubspapers

Callaghan, D. (1996). *The release of water fowl for hunting and the implications for biodiversity and sustainability*. In The introduction and naturalisation of birds. Proceedings of a conference organised jointly by the British Ornithologists Union and the UK Joint Nature Conservation Committee. The Stationary Office, London (pp. 37-48).

Callegari, S.E. (2006a). *Impact of released gamebirds on chalk grassland*. The Game Conservancy Review of 2005, 42-43.

Callegari, S.E. (2006b). *The Impact of Released Gamebirds on the Nature Conservation Value of Chalk Grassland in Central Southern England*. PhD thesis University of Reading.

Callegari, S.E., Bonham, E., Hoodless, A.N., Sage, R.B. & Holloway, G.J. (2014). Impact of game bird release on the Adonis blue butterfly *Polyommatus bellargus* (Lepidoptera Lycaenidae) on chalk grassland. *European Journal Wildlife Research* 60, 781–787.

Capstick, L., Sage, R.B. & Hoodless, A.N. (2019a). Ground flora recovery in disused pheasant pens is limited and affected by pheasant release density. *Biological Conservation*, 231, 181-188.

Capstick, L.A., Draycott, R.A.H., Wheelwright, C.M., Ling, D.E., Sage, R.B. & Hoodless, A.N. (2019b). Changes in flora and butterfly communities of woodland rides as a result of management for pheasant releasing. *Forest Ecology & Management*. 454, doi.org/10.1016/j.foreco.2019.03.034.

Caro, J., Delibes-Mateos, M., Estrada, A., Borralho, R., Gordinho, L., Reino, L., Beja, P. & Arroyo, B. (2015). Effects of hunting management on Mediterranean farmland birds. *Bird Conservation International*. 25, doi:10.1017/S0959270914000197

Carson, M., Meredith, A.L., Shaw, D.J., Giotis, E.S., Lloyd, D.H., & Loeffler, A. (2012). Foxes as a potential wildlife reservoir for mecA-positive staphylococci. *Vector-Borne and Zoonotic Diseases*, 12, 583-587.

Casas, F., Mougeot, F., Viñuela, J., & Bretagnolle, V. (2009). Effects of hunting on the behaviour and spatial distribution of farmland birds: importance of hunting-free refuges in agricultural areas. *Animal Conservation*, 12, 346-354.

Casas, F., Mougeot, F., Sanchez-Barbudo, I., Davila, J.A. & Vinuela, J. (2012). Fitness consequences of anthropogenic hybridization in wild red-legged partridge (*Alectoris rufa*, Phasianidae) populations. *Biological Invasions* 14, 295–305.

Casas F., Arroyo B., Viñuela J., Guzmán J.L. & Mougeot F. (2016). Are farm-reared red-legged partridge releases increasing hunting pressure on wild breeding partridges in central Spain? *European Journal of Wildlife Research* 62, 79–84.

Catelli, E., De Marco, M.A., Delogu, M., Terregino, C. & Guberti, V. (2001). Serological evidence of avian pneumovirus infection in reared and free-living pheasants. *Veterinary Record* 149, 56-58.

Cavanagh, D., Mawditt, K., Welchman, D. de B., Britton, P. & Gough, R.E. (2002). Coronaviruses from pheasants (*Phasianus colchicus*) are closely related to coronaviruses of domestic fowl (infectious bronchitis virus) and turkeys. *Avian Pathology* 31, 81-93.

Chaichana, R., Leah, R., & Moss, B. (2010). Birds as eutrophicating agents: a nutrient budget for a small lake in a protected area. *Hydrobiologia*, 646, 111-121.

Champagnon, J., Guillemain, M., Elmberg, J., Massez, G., Cavallo, F., & Gauthier-Clerc, M. (2012). Low survival after release into the wild: assessing “the burden of captivity” on Mallard physiology and behaviour. *European Journal of Wildlife Research*, 58, 255-267.

Champagnon, J., Crochet, P.A., Kreisinger, J., Čížková, D., Gauthier-Clerc, M., Massez, G., ... & Guillemain, M. (2013). Assessing the genetic impact of massive restocking on wild mallard. *Animal Conservation*, 16, 295-305.

Champagnon, J., Legagneux, P., Souchay, G., Inchausti, P., Bretagnolle, V., Bourguemestre, F., ... & Guillemain, M. (2016). Robust estimation of survival and contribution of captive-bred Mallards *Anas platyrhynchos* to a wild population in a large-scale release programme. *Ibis*, 158, 343-352.

Chapman, P. (in prep). *The direct ecological effects of non-native gamebird release in the lowland UK: an evidence synthesis*. PhD chapter, Imperial College London

Cheng, K.M., Shoffner, R.N., Phillips, R.E. & Lee, F.B. (1978). Mate preference in wild and domesticated (game-farm) Mallards (Anas platyrhynchos): I. Initial preference. *Animal Behaviour* 26, 996–1003.

Clapham, P.A. (1957). Helminth parasites in some wild birds. *Bird Study* 4, 193-196.

Clapham, P.A. (1961). Recent observations on Helminthiasis in some British Game Birds. *Journal of Helminthology,* 35-40.

Clarke, S.A., Robertson, P.A. (1993). The relative effects of woodland management and pheasant *Phasianus colchicus* predation on the survival of pearl-bordered and small pearl bordered fritillaries *Boloria euphrosyne* and *B. selene* in the south of England. *Biological Conservation* 65, 199-203.

Cobham Resource Consultants (1983). *Countryside sports, their economic significance*. Standing conference on Countryside Sports, Reading University.

Collins, A., Coughlin, D., Miller, J., & Kirk, S. (2015). *The production of quick scoping reviews and rapid evidence assessments: a how to guide*. NERC, UK

Corke, D. (1989). Of Pheasants and Fritillaries: Is predation by pheasants (*Phasianus colchicus*) a cause of the decline in some British butterfly species? *British Journal of Entomology and Natural History* 2, 1-14.

Cox, G., Watkins, C., & Winter, M. (1996). *Game management in England: Implications for public access, the rural economy and the environment*. Arkleton Centre for Rural Development Research.

Craine, N.G., Nuttall, P.A., Marriott, A.C., & Randolph, S.E. (1997). Role of grey squirrels and pheasants in the transmission of *Borrelia burgdorferi* sensu lato, the Lyme disease spirochaete, in the UK. *Folia Parasitologica*, 44, 155-160.

Curland, N., Gethöffer, F., van Neeret. A (and 20 others) (2018). Investigation into diseases in free-ranging ring-necked pheasants (*Phasianus colchicus*) in northwestern Germany during population decline with special reference to infectious pathogens. *European Journal of Wildlife Research* 64, doi.org/10.1007/s10344-018-1173-2

Dalke, P. D. (1935). *Dropping analysis as an indication of pheasant food habits.* In Transactions of the 21st American Game Conference (pp. 387-391).

Davey, C.M. (2008). *The impact of game management for pheasant (Phasianus colchicus) shooting on vertebrate biodiversity in British woodlands*. PhD thesis. University of Bristol.

Dessborn, L., Brochet, A.L., Elmberg, J., Legagneux, P., Gauthier-Clerc, M., & Guillemain, M. (2011a). Geographical and temporal patterns in the diet of pintail *Anas acuta*, wigeon *Anas penelope*, mallard *Anas platyrhynchos* and teal *Anas crecca* in the Western Palearctic. *European Journal of Wildlife Research*, 57, 1119-1129.

Dessborn, L., Elmberg, J., & Englund, G. (2011b). Pike predation affects breeding success and habitat selection of ducks. *Freshwater Biology*, 56, 579-589.

Devlin, J.J. (2019). *Common pheasant (Phasianus colchicus) densities in upland Wales and their impact on invertebrate communities.* MRes Thesis. Cardiff University.

Devriese, L.A., Ieven, M., Goossens, H., Vandamme, P., Pot, B., Hommez, J., & Haesebrouck, F. (1996). Presence of vancomycin-resistant enterococci in farm and pet animals. *Antimicrobial Agents and Chemotherapy*, 40, 2285-2287.

Diamond, J. (2019). *Latest update: Wild bird licencing.* https://naturalengland.blog.gov.uk/2019/04/08/latest-update-wild-bird-licensing-2/

Díaz-Sánchez, S., Sánchez, S., Ewers, C. & Höfle, U. (2012a). Occurrence of avian pathogenic *Escherichia coli* and antimicrobial-resistant *E. coli* in red-legged partridges (*Alectoris rufa*); sanitary concerns of farming. *Avian Pathology* 41, 337-344.

Diaz-Sanchez, S; Moriones, AM; Casas, F; Hofle, U (2012b) Prevalence of Escherichia coli, Salmonella sp and Campylobacter sp in the intestinal flora of farm-reared, restocked and wild red-legged partridges (*Alectoris rufa*): is restocking using farm-reared birds a risk? *European Journal of Wildlife Research* 58, 99-105.

Dimond, R., Warner, N., Wheeler, M.J. & Westbury, D.B. (2013) *An investigation into the relationship between pheasants (Phasianus colchicus) and reptiles as prey*. Report, Nottingham University.

Draycott, R.A.H. (2013). *The causes of wild pheasant mortality*. The Game & Wildlife Conservation Trust Review of 2012, 22-23. Fordingbridge.

Draycott, R.A.H., Parish, D.M.B., Woodburn, M.I.A & Carroll, J.P. (2000). Survey of selected parasites of wild-living pheasants in Britain. *Veterinary Record* 147, 245-246.

Draycott, R.A.H., Parish, D., Woodburn, M.I.A. & Carroll, J.P. (2002). Spring body condition of pheasants in Great Britain. *Wildlife Biology* 8, 261-266.

Draycott, R.A.H. & Hoodless, A.N. (2005). *Effect of pheasant management on wildlife in woods.* The Game Conservancy Trust Review of 2004, 38-39. Fordingbridge.

Draycott, R.A.H., Woodburn, M.I.A., Ling, D.E. & Sage, R.B. (2006). The effect of an indirect anthelmintic treatment on the parasite burden and breeding success of free-living pheasants. *Journal of Helminthology* 80, 409-415.

Draycott, R.A.H., Hoodless, A.N. & Sage, R.B. (2008a). Effects of pheasant management on vegetation and birds in lowland woodlands. *Journal of Applied Ecology* 45, 334-341.

Draycott, R.A.H. Hoodless, A.N., Woodburn, M.I.A. & Sage, R.B. (2008b). Nest predation of common pheasant *Phasianus colchicus*. *Ibis* 150, 37-44.

Draycott, R.A.H., Hoodless, A.N., Cooke, M. & Sage, R.B. (2012). The influence of pheasant releasing and associated management on farmland hedgerows and birds in England. *European Journal of Wildlife Research* 58, 227-234.

Duckworth, J.C., Firbank, L.G., Stuart, R.C., & Yamamoto, S. (2003). Changes in land cover and parcel size of British lowland woodlands over the last century in relation to game management. *Landscape Research* 28, 171-182.

Duarte, J., Farfán, M.Á., & Vargas, J.M. (2011). New data on mortality, home range, and dispersal of red-legged partridges (*Alectoris rufa*) released in a mountain range. *European Journal of Wildlife Research* 57, 675-678.

Edgar, P., Foster, J., & Baker, J. (2010). *Reptile habitat management handbook*. Bournemouth: Amphibian and Reptile Conservation.

Ehrmann, S., Ruyts, S.C., Scherer-Lorenzen, M., (and 22 others) (2018). Habitat properties are key drivers of *Borrelia burgdorfeir* (s.l.) prevalence in *Ixodes ricinus* populations of deciduous forest fragments. *Parasites & Vectors* 11, 10.1186/s13071-017-2590-x.

Estrada, A., Delibes-Mateos, M., Caro, J., Viñuela, J., Díaz-Fernández, S., Casas, F. & Arroyo, B. (2015), Game management and conservation. *Animal Conservation* 18, 567-575.

Ewald, J.A. & Touyéras, H. (2002). Examining the spatial relationship between pheasant (*Phasianus colchicus*) release pens and grey partridge (*Perdix perdix*) population parameters. *Zeitschrift für Jagdwissenschaft* 48, 354-363.

Ewald, J. & Gibbs, S. (2020). *Gamekeepers: conservation and wildlife*. https://www.gwct.org.uk/media/1095291/NGOGWCT-Survey2019-final.pdf

Feber, R.E., Johnson, P.J., & Macdonald, D.W. (In press). Shooting pheasants for sport: What does the death of Cecil tell us? *People and Nature*.

FERA (2012). *Reviews of selected wildlife conflicts and their management. Annex B: Approaches to mitigating bird of prey conflicts with pheasants at release pens, outdoor poultry and lambs.* The Food and Environment Research Agency report No. WM0415. 112 pp.

Ferretti, M., Falcini, F., Paci, G., & Bagliacca, M. (2012). Captive rearing technologies and survival of pheasants (*Phasianus colchicus* L.) after release. *Italian Journal of Animal Science*, *11*, DOI: 10.4081/ijas.2012.e29

Ferris, R. & Carter, C. (2000). *Managing Rides, Roadsides and Edge habitats in Lowland Forests*. Forestry Commission Bulletin 123.

Firbank, L.G. (1999). *Lowland Game Shooting Study*. Institute of Terrestrial Ecology & Centre of Ecology and Hydrology. Grange over Sands, Cumbria.

Fried, L.A. (1940). The food habits of the ring-necked pheasant in Minnesota. *The Journal of Wildlife Management*, 4, 27-36.

Fuller, R.J. & Henderson, A.C.B. (1992). Distribution of breeding songbirds in Bradfield Woods, Suffolk, in relation to vegetation and coppice management. *Bird Study* 39, 73-88.

Fuller, R.J., Noble, D.G., Smith, K.W. & Vanhinsbergh, D. (2005). Recent declines in populations of woodland birds in Britain: a review of possible causes. *British Birds* 98, 116-143.

Game Conservancy Trust (1988). *Woodlands for Pheasants*. Game Conservancy Ltd., Fordingbridge, Hampshire.

Game Conservancy Trust (1994). *Game and Shooting Crops*. Game Conservancy Ltd., Fordingbridge, Hampshire.

Game Conservancy Trust (1996). *Gamebird releasing*. Game Conservancy Ltd., Fordingbridge, Hampshire.

Gaudioso, V.R., Pérez, J.A., Sánchez-García, C., Armenteros, J.A., Lomillos, J.M., & Alonso, M.E. (2011). Isolation from predators: a key factor in the failed release of farmed red-legged partridges (*Alectoris rufa*) to the wild? *British Poultry Science* 52, 155-162.

Gethings, O.J. (2018). *The influence of Syngamus trachea on pheasant populations*. Doctoral dissertation, Harper Adams University.

Gethings, O.J., Sage, R.B. & Leather, S.R. (2015). Spatio-temporal factors influencing the occurrence of *Syngamus trachea* within release pens in the South West of England. *Veterinary Parasitology* 207, 64-71.

Gethings, O.J., Sage, R.B., Morgan, E.R. & Leather, S.R. (2016a). Body condition is negatively associated with infection with Syngamus trachea in the Ring-necked Pheasant *Phasianus colchicus*. *Vetirinary Parasitology* 228, 1-5.

Gethings, O.J., Sage, R.B. & Leather, S.R. (2016b). Density-dependent regulation of fecundity in Syngamus trachea infrapopulations in semi-naturally occurring ring-necked pheasants (*Phasianus colchicus*) and wild Carrion Crows (*Corvus corone*). *Parasitology* 143, 716-22.

Gilbert, J. (2007). *National Inventory of woodland and trees (1995-1999): Analysis of management and biodiversity data*. Forest Research, Forestry Commission.

Gilliver, M.A., Bennett, M., Begon, M., Hazel, S.M., & Hart, C.A. (1999). Enterobacteria: antibiotic resistance found in wild rodents. *Nature*, 401, 233-234.

Gortázar, C., Acevedo, P., Ruiz-Fons, F., & Vicente, J. (2006). Disease risks and overabundance of game species. *European Journal of Wildlife Research* 52, 81-87.

Green, A.J. (2016). The importance of waterbirds as an overlooked pathway of invasion for alien species. *Diversity and Distributions* 22, 239-247.

Greenall, T. (2007). *Management of gamebird shooting in lowland Britain: Social attitudes, biodiversity benefits and willingness-to-pay*. Doctoral dissertation, The University of Kent.

Guerrero-Ramos, E., Cordero, J., Molina-González, D., Poeta, P., Igrejas, G., Alonso-Calleja, C., & Capita, R. (2016). Antimicrobial resistance and virulence genes in enterococci from wild game meat in Spain. *Food Microbiology* 53, 156-164.

Guillemain, M., & Fritz, H. (2002). Temporal variation in feeding tactics: exploring the role of competition and predators in wintering dabbling ducks. *Wildlife Biology* 8, 81-90.

Hall, A. (in prep) *The use of enhanced rearing techniques to improve post-release survival of released pheasants.* PhD Thesis. University of Exeter.

Hand, N. (2020) *The adder (Vipera berus) and the cultivation of the ring neck pheasant (Phasianus colchicus). Field observations from long term monitoring and adder radio telemetry projects across the Midlands and Southern England*. Nigel Hand Central Ecology, Ledbury.

Harradine, J. (1985). Duck shooting in the United Kingdom. *Wildfowl*, 36, 81-94.

Harradine, J., Reynolds, N. & Laws, T. 1997. *Raptors and Gamebirds. A Survey of Game Managers Affected by Raptors*. Wrexham: British Association for Shooting and Conservation.

Harris, S. & Yalden, D.W. (2008). *Mammals of the British Isles: Handbook*. Mammal Society. Southampton.

Heise, M. (1989). Human-induced tachycardia in wild and tame Mallard (*Anas platyrhynchos*). *Comparative Biochemistry and Physiology Part A: Physiology*, 92, 125-128.

Henderson, I.G., Vickery, J.A. & Carter, N. (2003). The use of winter bird crops by farmland birds in lowland England. *Biological Conservation* 118, 21-32.

Hesford N.J. (2012) *Fate and survival of hand-reared red-legged partridges released for sport on farmland in the UK.* MSc thesis. GWCT/Cardiff University.

Heydon, M.J. & Reynolds, J.C. & Short, M.J. (2000). Variation in abundance of foxes (*Vulpes vulpes*) between three regions of rural Britian, in relation to landscape and other variables. *Journal of Zoology 251*, 253-264.

Heydon, M.J. & Reynolds, J.C. (2000). Demography of rural foxes (*Vulpes vulpes*) in relation to cull intensity in three contrasting regions of Britain. *Journal of Zoology* 251, 261-276.

Hill, D.A., & Robertson, P.A. (1988). *The pheasant: ecology, management, and conservation.* BSP Professional Books.

Holand. H., Jensen, H., Tufto, J., Parn, H., Saether, B.E. & Ringsby, T.H. (2015). Endoparasite infection has both short- and long-term negative effects on reproductive success of female house sparrows, as revealed by faecal parasitic egg counts. *PLos One* 10(5), e0125773.

Hoodless, A.N. & Draycott, R.A.H. (2008)*. Pheasant releasing and woodland rides*. The Game Conservancy Trust Review of 2007, 16-17. Fordingbridge.

Hoodless, A.N., Kurtenbach, K., Peacey, M., Nuttall, P.A. & Randolph, S.E. (1998). The role of pheasants (*Phasianus colchicus*) as hosts for ticks (*Ixodes ricinus*) and Lyme disease spirochaetes (*Borrelia burgdorferi*) in Southern England. *Gibier Faune Sauvage* 15, 477-489.

Hoodless, A.N., Lewis, R. & Palmer, J. (2006). *Songbird use of pheasant woods in winter*. The Game Conservancy Trust Review of 2005, 28-29. Fordingbridge.

Howard, N.S., & Carroll, J.P. (2001). Driven game shooting on farms in Essex. *Game and Wildlife Science* 18, 157-169.

Jensen, P.M., Madsen, P., Jensen, L. & Christian, P. (2012). Differences in carbon and nitrogen stable isotope signatures amongst wild and released pheasant populations. *European Journal of Wildlife Research* 58, 755–760..

Jones, E.L. (2009). The environmental effects of blood sports in lowland England since 1750. *Rural History* 20, 51-66.

Kenward, R.E. (1977). Predation on released pheasants (*Phasianus colchicus*) by Goshawks (*Accipiter gentillis*) in central Sweden. *Swedish Game Research* 10, 79-112.

Kenward, R.E., Arraut, E.M., Robertson, P.A., Walls, S.S., Casey, N.M. & Aebischer, N.J. (2018). Resource-Area-Dependence Analysis: Inferring animal resource needs from home-range and mapping data. *PLoS ONE* 13(10), e0206354.

Kenward, R.E. Hall, D.G. Walls, S.S. & Hodder, K.H. (2001). Factors affecting predation by buzzards *Buteo buteo* on released pheasants *Phasianus colchicus*. *Journal of Applied Ecology* 38, 813–822.

Kenward, R.E., Marcström, V. & Karlbom, M. (1981). Goshawk winter ecology in Swedish pheasant habitats. *The Journal of Wildlife Management* 45, 397–408.

Kleyheeg, E., Klaassen, M., & Soons, M.B. (2016). Seed dispersal potential by wild mallard duck as estimated from digestive tract analysis. *Freshwater Biology* 61, 1746-1758.

Kurtenbach, K., Carey, D., Hoodless, A.N., Nuttall, P.A. & Randolph, S.E. (1998). Competence of pheasants as reservoirs for Lyme disease spirochetes. *Journal of Medical Entomology* 35, 77-81.

Larkman, A., Newton, I., Feber, R.E. & Macdonald, D.W. (2015). Small farmland bird declines, gamebird releases, and changes in seed sources. In: *Wildlife Conservation on Farmland*. Eds: D.W. Macdonald & R. E. Feber. Oxford University Press.

Lees, A.C., Newton. I. & Balmford, A. (2013). Pheasants, buzzards, and trophic cascades. *Conservation Letters* 6, 141–144.

Lever, C. (1977). *The naturalized animals of the British Isles*. Hutchinson, London.

Lewis-Phillips, J., Brooks, S., Sayer, C.D., McCrea, R., Siriwardena, G., & Axmacher, J.C. (2019). Pond management enhances the local abundance and species richness of farmland bird communities. *Agriculture, Ecosystems & Environment* 273, 130-140.

Lister, S.A., Beer, J.V., Gough, R.E., Holmes, R.G., Jones, J.M.W. & Orton, R.G. (1985). Outbreaks of nephritis in pheasants (*Phasianus colchicus*) with a possible coronavirus aetiology. *Veterinary Record* 117, 612-613.

Lloyd D.E.B. (1976) *Avian predation of reared pheasants*. Unpublished report for the British Field Sports Society, The Game Conservancy, The RSPB and the Wildfowlers Association of GB and Ireland. 84pp.

Ludolf, I.C., Robertson, P.A. & Woodburn, M.I.A. (1989a). Changes in the ground flora and butterfly populations of woodlands managed to encourage pheasants. In: Buckley, PJ (Ed), *Biological Habitat Reconstruction*. Belhaven Press, London, pp. 312-327.

Ludolf, I.C., Robertson, P.A. & Payne, S. (1989b). *The effect of pheasant release pens and strawed feed rides on the ground flora of ancient woodlands*. Nature Conservancy Council, Peterborough.

Madden, J.R., & Perkins, S.E. (2017). Why did the pheasant cross the road? Long-term road mortality patterns in relation to management changes. *Royal Society Open Science*, 4, e170617.

Madden, J. R., Hall, A., & Whiteside, M. A. (2018). Why do many pheasants released in the UK die, and how can we best reduce their natural mortality? *European Journal of Wildlife Research*, 64, 40. doi.org/10.1007/s10344-018-1199-5

Madden, J.R., Santilli, F. & Whiteside, M.A. (2020). The welfare of game birds destined for release into the wild: a balance between early life care and preparation for future natural hazards. *Animal Welfare* 29, 1-18.

Madden, J.R., Whiteside, M.A. & Perkins, S.E. (in prep). Relationships between numbers of released pheasants and foxes in the UK derived from roadkill reports

Madsen, J., & Fox, A.D. (1995). Impacts of hunting disturbance on waterbirds - a review. *Wildlife Biology* 1, 193-207.

Madge, S. & McGowan, P. (2002). *Pheasants, partridges and grouse*. A and C Black Ltd., London.

Manny, B.A., Johnson, W.C., & Wetzel, R.G. (1994). Nutrient additions by waterfowl to lakes and reservoirs: predicting their effects on productivity and water quality. In *Aquatic Birds in the Trophic Web of Lakes* (pp. 121-132). Springer, Dordrecht.

Manosa S. (2002) *The conflict between gamebird hunting and raptors in Europe. Renconciling gamebird hunting and biodiversity (REGHAB)* report 79pp. University of Barcelona.

Marquiss, M., Newton, I. (1982) The goshawk in Britain. *British Birds* 75, 243-260.

Martin, J. (2011). The transformation of lowland game shooting in England and Wales since the Second World War: the supply side revolution. *Rural History* 22, 207-226.

Martin, J. (2012). The Transformation of Lowland Game Shooting in England and Wales in the Twentieth Century: The Neglected Metamorphosis. *The International Journal of the History of Sport* 29, 1141-1158.

Mitchell, R.J., Sutton, M.A., Truscott, A.M., Leith, I.D., Cape, J.N., Pitcairn, C.E.R. & Van Dijk, N. (2004). Growth and tissue nitrogen of epiphytic Atlantic bryophytes: effects of increased and decreased atmospheric N deposition. *Functional Ecology* 18, 322-329

Millan, J., Gortazar, C., Buenestado, FJ., Rodrıguez, P., Tortosa, F. S., & Villafuerte, R. (2003). Effects of a fiber-rich diet on physiology and survival of farm-reared red-legged partridges (*Alectoris rufa*). *Comparative Biochemistry and Physiology Part A: Molecular & Integrative* Physiology 134, 85-91.

Millan, J., Gortazar, C. & Villafuerte, R.A. (2004) A comparison of the helminth faunas of wild and farm-reared red-legged partridge. *The Journal of Wildlife Management* 68, 701 – 707.

Millins, C., Gilbert, L., Medlock, J., Hansford, K., Thompson, D.B.A., & Biek, R. (2017). Effects of conservation management of landscapes and vertebrate communities on Lyme borreliosis risk in the UK. *Philosophical Transactions Royal Society Bulletin* 372, doi.org/10.1098/rstb.20160123

Mustin, K., Arroyo, B., Beja, P., Newey, S., Justin Irvine, R., Kestler, J. & Redpath, S.M. (2018). Consequences of gamebird management for non-game species in Europe. *Journal of Applied Ecology* 55, 2285-2295.

Nagy, I., Ristić, Z.A., Marković, V., Kovačević, M., Matejević, M., & Jovanović, T. (2013). The significance of egg shell color on the pheasant hatching production results. *Pakistan Journal of Zoology*, 45, 1549-1553.

Neumann, J.L., Holloway, G,J, Sage, R.B. & Hoodless, A.N. (2015). Releasing of pheasants for shooting in the UK alters woodland invertebrate communities. *Biological Conservation* 191, 50-59.

ONCFS (2013). *Le faisan commun*, http://www.oncfs.gouv.fr/Connaitre-les-especes-ru73/Le-Faisan-commun-ar1227

Oldfield, T.E.E., Smith, R.J., Harrop, S.R. & Leader-Williams, N. (2003). Field sports and conservation in the United Kingdom. *Nature* 423, 531-533.

Opermanis, O., Mednis, A., & Bauga, I. (2001). Duck nests and predators: interaction, specialisation and possible management. *Wildlife Biology* 7, 87-96.

Ostfeld, R.S., Levi, T., Keesing, F., Oggenfuss, K. & Canham, C.D. (2018). Tick-borne disease risk in a forest food web. *Ecology* 99, 1562-1573.

Otter, A., Payne, J., & Carr, S. (2018). Botulism in farmed ducks and pheasants. *The Veterinary Record*, 183, 452.

PACEC. (2006). *The economic and environmental effect of sporting shooting*. A report prepared by public and corporate economic consultants (PACEC) on behalf of BASC, CA, and CLA and in association with GCT.

PACEC. (2014). *The value of shooting: The economic, environmental and social contribution of shooting sports to the UK.* http://www.shootingfacts.co.uk/pdf/The-Value-of-Shooting-2014.pdf

Padyšáková, E., Šálek, M., Poledník, L., Sedláček, F., & Albrecht, T. (2011). Predation on simulated duck nests in relation to nest density and landscape structure. *Wildlife Research* 37, 597-603.

Pain, D.J., Mateo, R., & Green, R.E. (2019). Effects of lead from ammunition on birds and other wildlife: A review and update. *Ambio* 48, 935-953.

Paillisson, J. M., Reeber, S., & Marion, L. (2002). Bird assemblages as bio-indicators of water regime management and hunting disturbance in natural wet grasslands. *Biological Conservation* 106, 115-127.

Parish, D.M.B. & Sotherton, N.W. (2004). Game crops and threatened farmland songbirds in Scotland: a step towards halting population declines. *Bird Study* 51, 107-112.

Parish, D.M.B. & Sotherton, N.W. (2008). Landscape-dependent use of a seed-rich habitat by farmland passerines: relative importance of game cover crops in a grassland versus an arable region of Scotland. *Bird Study* 55, 118-123.

Park K.J., Graham K.E., Calladine J. & Wernham C.W. (2008). Impacts of birds of prey on gamebirds in the UK: a review. *Ibis* 150, 9-26.

Parkin, D.T. & Knox, A.G. (2010). *The Status of Birds in Britain and Ireland*. Christopher Helm, London.

Parrott, D. (2015). Impacts and management of common buzzards *Buteo buteo* at pheasant *Phasianus colchicus* release pens in the UK: a review. *European Journal of Wildlife Research* 61, 181-197.

Pennycott, T.W., Dare, C.M., Yavari, C.A. & Bradbury, J.M. (2005). *Mycoplasma sturni* and *Mycoplasma gallisepticum* in wild birds in Scotland. *Veterinary Record* 156, 513-515.

Perez, G., Bastian, S., Agoulon, A., Bouju, A., Durand, A., Faille, F., Lebert, I., Rantier, Y., Plantard, O., & Butet, A. (2016). Effect of landscape features on the relationship between *Ixodes ricinus* ticks and their small mammal hosts. *Parasites & Vectors* 9:20. doi: 10.1186/s13071-016-1296-9

Peterken, G.F. & Game, M. (1984). Historical factors affecting the number and distribution of vascular plant species in the woodlands of central Lincolnshire. *Journal of Ecology* 72, 155-182.

Porter, K. (1981). *The population dynamics of small colonies of the butterfly* Euphydryas aurinia. PhD thesis, Oxford University.

Porteus, T.A. (2015). *Evaluation of restricted area culling strategies to control local red fox density*. PhD thesis, University of British Columbia.

Porteus, T.A., Reynolds, J.C. & McAllister, M.K. (2018). Quantifying the rate of replacement by immigration during restricted-area control of red fox in different landscapes. *Wildlife Biology* 1, 1-9.

Porteus, T.A., Reynolds, J.C. & McAllister, M.K. (2019) Population dynamics of foxes during restricted-area culling in Britain: Advancing understanding through state-space modelling of culling records. *PLoS ONE* 14, e0225201.

Potts, G.R. (2009). Long-term changes in the prevalences of caecal nematodes and histomonosis in gamebirds in the UK and the interaction with poultry. *Veterinary Record* 164, 715-718.

Potts, G.R. (2010). Pheasants, parasites and partridges. *British Birds* 103, 123

Potts, G.R. (2012). *Partridges.* London: Collins.

Pressland, C.L. (2009). *The impact of releasing pheasants for shooting on invertebrates in British woodlands*. PhD thesis, University of Bristol.

Pringle, H., Wilson, M., Calladine, J. & Siriwanrdena, G. (2019). Associations between gamebird releases and generalist predators. *Journal of Applied Ecology* 56, 2102-2113.

Proctor, H., & Owens, I. (2000). Mites and birds: diversity, parasitism and coevolution. *Trends in Ecology & Evolution* 15, 358-364.

Rackham, O. (2003). Ancient Woodland: *Its history, vegetation and uses in England*. Castlepoint Press, UK.

Radhouani, H., Pinto, L., Coelho, C., Sargo, R., Araujo, C., Lopez, M., ... & Poeta, P. (2010). MLST and a genetic study of antibiotic resistance and virulence factors in vanA-containing *Enterococcus* from buzzards (*Buteo buteo*). *Letters in Applied Microbiology* 50, 537-541.

Reynolds, J.C. (1994). *Are foxes on the increase?* Game Conservancy Trust Review, 94-96. Fordingbirdge.

Reynolds, J.C., Goddard, H.N. & Brockless, M.H. (1993). The impact of local fox (*Vulpes vulpes*) removal on fox populations at two sites in southern England. *Gibier Faune Sauvage* 10, 319-334.

Reynolds, J.C. & Tapper, S.C. (1996). Control of mammalian predators in game management and conservation. *Mammal Review* 26, 127-156.

Rice, C.N. (2016). *Abundance, impacts and resident perceptions of non-native common pheasants (Phasianus colchicus) in Jersey, UK Channel Islands*. Doctoral dissertation, University of Kent.

Robertson, P.A. (1986). *The ecology and management of hand-reared and wild pheasants Phasianus colchicus in Ireland*. PhD thesis, University College Dublin.

Robertson, P.A. (1988). Survival of released pheasants, *Phasianus colchicus*, in Ireland. *Journal of Zoology* 214, 683-695.

Robertson, P.A. (1992). *Woodland management for pheasants*. Forestry Commission Bulletin 106.

Robertson, P.A., Woodburn, M.I.A. & Hill, D.A. (1988). The effects of woodland management for pheasants on the abundance of butterflies in Dorset, England. *Biological Conservation* 45, 159-167.

Robertson, P.A. & Dowell, S.D. (1990). *The effects of hand-rearing on wild gamebird populations*. In: The future of wild galliformes in The Netherlands. Eds: Lumeij, J.T. and Hoogeveen, Y.R.

Robertson, P.A. & Woodburn, M.I.A. (1991). *Woodland Rides for Pheasants*. The Game Conservancy Review of 1990, 122-123. Fordingbridge.

Robertson, P.A., Woodburn, M.I.A. & Hill, D.A. (1993a). Factors affecting winter pheasant density in British woodlands. *Journal of Applied Ecology* 30, 459-464.

Robertson, P.A., Woodburn, M.I.A., Neutel, W. & Bealey, C.E. (1993b). Effects of land-use on breeding pheasant density*. Journal of Applied Ecology* 30, 465-477.

Robertson, P.A., Mill, A.C., Rushton S.P., McKenzie A.J., Sage R.B. & Aebischer, N.J. (2017). Pheasant release in Great Britain: long-term and large-scale changes in the survival of a managed bird. *European Journal of Wildlife Research* 63, 100. Doi: 10.1007/s10344-017-1157-7.

Roos, S., Smart, J., Gibbons, D.W. & Wilson, J.D. (2018). A review of predation as a limiting factor for bird populations in mesopredator-rich landscapes: a case study of the UK. *Biological Reviews* 93, Doi:10.1111/brv.12426.

Rothero, G. (2006). *Baseline Surveys of Tortula leucostoma and Athalamia hyalina on Craig Leek SSSI.* Scottish Natural Heritage. Report No. 176.

RSPB (2019). *Birdcrime 2018.* https://www.rspb.org.uk/birds-and-wildlife/advice/wildlife-and-the-law/wild-bird-crime/birdcrime-2018/

Rutley D. (2019). *Partridges: Imports: Written question – 251254.* https://www.parliament.uk/business/publications/written-questions-answers-statements/written-question/Commons/2019-05-07/251254/?fbclid=IwAR07jCjtzI0ETMAUqFstgN_C6cZ1jKIT0XEtiulXy_9xH65YIPt6hmIS_KY

Sage, R.B., Robertson, P.A., & Wise, D.R. (2001). Survival and breeding success of two pheasant (*Phasianus colchicus*) strains released into the wild. *Game and Wildlife Science* 18, 331-340.

Sage, R.B., Woodburn, M.I.A., Davis, C. & Aebischer, N.J. (2002). The effect of an experimental infection of the nematode *Heterakis gallinarum* on hand-reared grey partridges *Perdix perdix*. *Parasitology* 124, 529-535.

Sage, R.B., Ludolf, C. & Robertson, P.A., (2005a). The ground flora of ancient semi-natural woodlands in pheasant release pens in England. *Biological Conservation* 122, 243-252.

Sage, R.B., Parish, D.M.B., Woodburn, M.I.A. & Thompson, P.G.L. (2005b). Songbirds using crops planted on farmland as cover for gamebirds. *European Journal of Wildlife Research* 51, 248-253.

Sage, R.B., Woodburn, M.I.A., Draycott, R.A.H., Hoodless, A.N. & Clarke, S. (2009). The flora and structure of farmland hedges and hedgebanks near to pheasant release pens compared with other hedges. *Biological Conservation* 142, 1362–1369.

Sage, R.B. (2018a). *Impacts of pheasant releasing for shooting on habitats and wildlife on the South Exmoor estates*. GWCT report. Fordingbridge.

Sage, R.B. (2108b). *Ecological impact of releasing pheasants on Exmoor shoots*. The Game & Wildlife Conservation Trust Review of 2017, 24-25, Fordingbridge.

Sage, R.B., Woodburn, M.I.A., Hoodless, A.N., Turner, C.V., Draycott, R.A.H., Sotherton, N.W. (2018c). Fate of released common pheasants Phasianus colchicus on UK lowland farmland and the effect of predator control. *European Journal of Wildlife Research* 64, doi 10.1007/s10344-018-1174-1.

Sanchez-Garcia, C., Buner, F.D. & Aebischer, N.J. (2015). Supplementary Winter Food for Gamebirds Through Feeders: Which Species Actually Benefit? *The Journal of Wildlife Management* 79, 832–845.

Sellers, R.M., & Greenwood, J.J. (2019). Sir Richard Graham and the marking of ducks at Netherby, Cumberland, 1908–1933–an early bird-ringing project. *Ringing & Migration* DOI: 10.1080/03078698.2018.1629151

Short, C. (1994). *Implications of game management for woodland management, landscape and conservation*. Centre for Rural Studies, Royal Agricultural College, Cirencester.

Siriwardena, G.M., Calbrade, N.A. & Vickery, J.A. (2008). Farmland birds and late winter food: Does supply fail to meet demand? *Ibis* 150, 585-595.

Siriwardena, G.M., Calbrade, N.A., Vickery, J.A. & Sutherland, W.J. (2006). The effect of spatial distribution of winter seed food resources on their use by farmland birds. *Journal of Applied Ecology* 43, 628-639.

Siriwardena, G.M., Stevens, D.K., Anderson, G.Q.A., Vickery, J.A. Calbrade, N.A. & Dodd, S. (2007). The effect of supplementary winter seed food on breeding populations of farmland birds: Evidence from two large scale experiments. *Journal of Applied Ecology* 44, 920-932.

Smith P (2014). *Effects of fixed nitrogen Eutrophication on lichen floras – does pheasant rearing represent a threat to the lichen flora of Croxton Park?* Natural England Technical report.

Söderquist, P., Gunnarsson, G. & Elmberg, J. (2013). Longevity and migration distance differ between wild and hand-reared mallards *Anas platyrhynchos* in Northern Europe. *European Journal of Wildlife Research* 59, 159-166.

Söderquist, P., Elmberg, J., Gunnarsson, G., Thulin, C. G., Champagnon, J., Guillemain, M., ... & Kraus, R. H. (2017). Admixture between released and wild game birds: a changing genetic landscape in European mallards (*Anas platyrhynchos*). *European Journal of Wildlife Research*, 63, 10.1007/s10344-017-1156-8.

Solomon, M.E. (1949). The natural control of animal populations. *Journal of Animal Ecology* 18, 1-35.

Stallknecht, D.E. & Brown, J.D. (2007). Wild birds and the epidemiology of avian influenza. *Journal of Wildlife Diseases* 43, S15-S20.

Stoate, C. & Szczur, J. (2001). Could game management have a role in the conservation of farmland passerines? A case study from a Leicestershire farm. *Bird Study* 48, 279-292.

Stoate, C., Szczur, J. & Aebischer, N.J. (2003). Winter use of wild bird cover crops by passerines on farmland in northeast England. *Bird Study* 50, 15-21.

Stromborg, K.L. (1979). Pheasant food habits in spring and consumption of seed treatment pesticides. *The Journal of Wildlife Management* 43, 214-219.

Swan, G.J.F. (2017). *Understanding conservation conflicts surrounding predation and game shooting interests*. PhD thesis, University of Exeter.

Tapper, S.C. (1992). *Game Heritage*. The Game Conservancy Trust, Fordingbridge.

Tapper, S.C. (1999). *A Question of Balance.* The Game & Wildlife Conservation Trust, Fordingbridge.

Tapper, S.C., Potts, G.R. & Brockless, M.H. (1996). The effect of an experimental reduction in predation pressure on the breeding success and population density of grey partridges *Perdix perdix*. *Journal of Applied Ecology* 33, 965-978.

Teanby, A., Norton, E., Steel, D. & Draycott, R. (2019). *Game and conservation benchmarking*. https://pdf.euro.savills.co.uk/uk/rural---other/savills-spotlight-game-lo.pdf

Teanby, A., Steel, D. & Draycott, R. (2017). *Shoot Benchmarking 2017 – Demonstration report*. https://www.gwct.org.uk/media/664264/shoot-benchmarking-example-participants-benchmarking-report-fictional.pdf

Therkildsen, O.R., Jensen, T.H., Handberg, K.J., Bragstad, K., & Jørgensen, P.H. (2011). Low pathogenic avian influenza (H7N1) transmission between wild ducks and domestic ducks. *Zoonoses and Public Health* 58, 312-317.

Tompkins, D.M., Greenman, J.V. & Hudson, P.J. (2001). Differential impact of a shared nematode parasite on two gamebird hosts: implications for apparent competition. *Parasitology* 122, 187-93.

Tompkins, D.M., Greenman, J.V., Robertson, P.A. & Hudson, P.J. (2000). The role of shared parasites in the exclusion of wildlife hosts: *Heterakis gallinarum* in the ring-necked pheasant and the grey partridge. *Journal of Animal Ecology* 69, 829-840.

Tompkins, D.M., Parish, D.M.B. & Hudson, P.J. (2002). Parasite-mediated competition among red-legged partridges and other lowland gamebirds. *The Journal of Wildlife Management* 66, 445-450.

Turner, C. & Sage, R.B. (2003). *Fate of released pheasants*. The Game Conservancy Trust Review of 2002, 74–75. Fordingbridge

Turner, C. (2007) *The fate and management of pheasants (Phasianus colchicus) released in the UK.* Unpublished PhD thesis, University of London.

UK-VARSS (2019).*UK Veterinary Antibiotic Resistance and Sales Surveillance Report (UK-VARSS 2018)*. New Haw, Addlestone: Veterinary Medicines Directorate

Vanhinsbergh, D., Gough, S., Fuller, R.J. & Brierley, E.D.R. (2002). Summer and winter bird communities in recently established farm woodlands in lowland England. *Agriculture, Ecosystems and Environment* 92, 123-136.

Warren, M.S. & Fuller, R.J. (1993). *Woodland rides and glades: their management for wildlife.* JNCC, Peterborough.

Watson, M., Aebischer, N.J., Potts, G.R. & Ewald, J.A. (2007). The relative effects of raptor predation and shooting on overwinter mortality of grey partridges in the United Kingdom. *Journal of Applied Ecology* 44, 972–982.

Welchman, D. de B., Bradbury, J.M., Cavanagh, D. & Aebischer, N.J. (2002). Infectious agents associated with respiratory disease in pheasants. *Veterinary Record* 150, 658-664.

Welchman, D. de B., Ainsworth, H.L., Jensen, T.K., Boye, M., King, S.A., Koylass, M.S., Whatmore, A.M., Manvell, R.J., Ayling, R.D. & Dalton, J.R. (2013). Demonstration of *Ornithobacterium rhinotracheale* in pheasants (*Phasianus colchicus*) with pneumonia and airsacculitis. *Avian Pathology* 42, 171-178.

White, P.J.C., Stoate, C., Szczur, J., & Norris, K.J (2014). Predator reduction with habitat management can improve songbird nest success. *Journal of Wildlife Management* 78, 402-412.

Whiteside, M. A. (2015). *How early rearing conditions influence behaviour and survival of pheasants released into the wild?* PhD Thesis, University of Exeter.

Whiteside, M.A., Sage, R. & Madden, J.R. (2015). Diet complexity in early life affects survival in released pheasants by altering foraging efficiency, food choice, handling skills and gut morphology. *Journal of Animal Ecology* 84, 1480-1489.

Whiteside, M.A., Sage, R. & Madden, J.R. (2016). Multiple behavioural, morphological and cognitive developmental changes arise from a single alteration to early life spatial environment, resulting in fitness consequences for released pheasants. *Royal Society Open Science*, **3,** e160008.

Woodburn, M.I.A. (1999). *Comparative population dynamics of wild and reared pheasants*. DPhil. Thesis, University of Southampton.

Woodburn,. M.I.A. & Robertson, P.A. (1990). *Woodland management for pheasants: economics and conservation effects*. In: The Future of Wild Galliformes in the Netherlands. Eds J.T. Lumeij & Y.R. Hoogeveen. Gegevens Koninklijke Bibliotheek, The Hague, 185-198.

Woodburn, M. & Sage, R. (2005). *Effect of pheasant releasing on edge habitats.* The Game and Wildlife Conservation Trust Review of 2004, 36-37. Fordingbridge.

Woodward, I.D., Aebischer, N.J., Burnell, D., Eaton, M.A., Frost, T.M., Hall, C., Stroud, D.A. & Noble, D.G. (2020). Population estimates of birds in Great Britain and the United Kingdom. *British Birds* 113, 69-104.

Wright, T. (1941). A study of the fall food supply of the ring-necked pheasant and the bob-white quail in Washington County, Rhode Island. *The Journal of Wildlife Management*, *5*, 279-296.

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