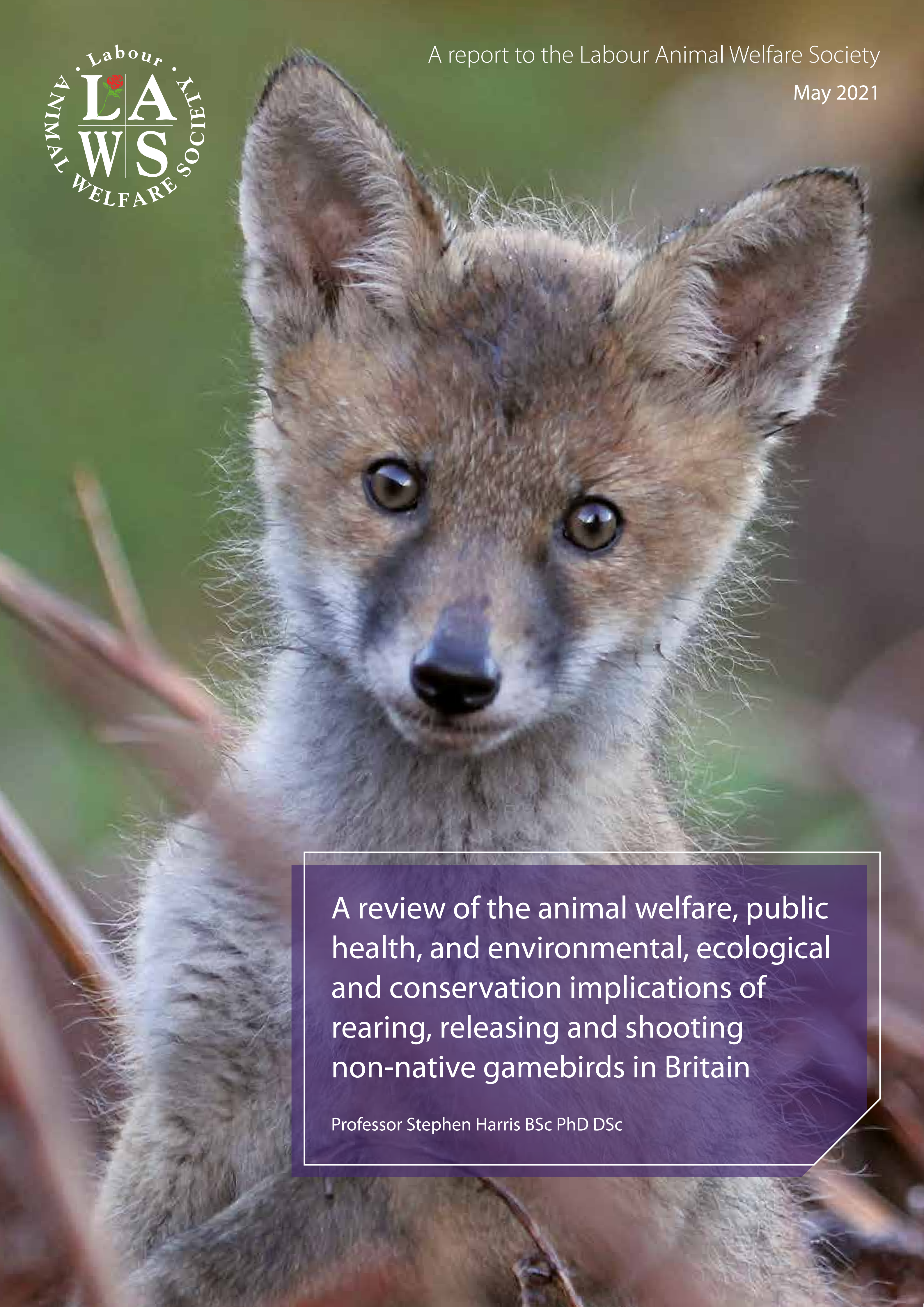


LAWS
Labour Animal Welfare Society

A report to the Labour Animal Welfare Society

May 2021

A review of the animal welfare, public health, and environmental, ecological and conservation implications of rearing, releasing and shooting non-native gamebirds in Britain

Professor Stephen Harris BSc PhD DSc

A review of the animal welfare, public health, and environmental, ecological and conservation implications of rearing, releasing and shooting non-native gamebirds in Britain

A report to the Labour Animal Welfare Society

Professor Stephen Harris BSc PhD DSc

May 2021

Instructions

I was asked to review the scientific evidence on:-

1. The potential welfare and public health issues associated with rearing, releasing and shooting large numbers of non-native gamebirds in Britain
2. Whether rearing and releasing large numbers of non-native gamebirds in Britain, and associated predator-control activities, have an impact on the numbers of different species of avian and mammalian predators, and the character and extent of any possible species interactions
3. Whether rearing and releasing large numbers of non-native gamebirds in Britain affects the numbers of foxes and other predatory species of birds and mammals, and the spatial scale of these influences
4. Whether rearing and releasing large numbers of non-native gamebirds in Britain has an influence on the population dynamics of foxes and other predators, and the spatial scale of these influences
5. The potential impacts of releasing large numbers of non-native gamebirds in Britain on the predation pressure on species of conservation concern, particularly ground-nesting birds, and the scale of these impacts
6. Other potential environmental, ecological and/or conservation impacts of rearing and releasing large numbers of non-native gamebirds in Britain

I was told that:-

- I should identify any animal welfare concerns and potential effects on wildlife and public health
- I should identify the actual or potential direct and indirect effects of activities associated with rearing, releasing and shooting gamebirds, and the distances that may be required for any precautionary actions, prohibitions or mitigation
- I should consider any potential environmental, ecological and/or conservation impacts of widespread supplementary feeding by the gamebird-shooting industry
- I should evaluate Natural England's gamebird release rapid assessment and identify potential omissions and oversights
- It is of particular importance to consider the potential impacts of widespread killing of foxes on fox population dynamics, species interactions and food webs more generally
- It is my decision as to what evidence I wished to include in my review and, while the review relates particularly to the situation in Britain, I was free to refer to any studies from other countries that I considered to be relevant

Contents

Summary of the key points

I review the environmental, ecological and conservation impacts of rearing, releasing and shooting non-native gamebirds in Britain, and the wider impacts of the gamebird-shooting industry on animal welfare and public health. I show that:-

1. *Each year the gamebird-shooting industry releases 47 million ring-necked pheasants and 10 million red-legged partridges (both non-native species) to provide an opportunity for participants to pay to kill large numbers of gamebirds in a single day. While the number of gamebirds released each year has increased, the proportion that are shot has been declining. The density of pheasants reared, released and shot in Britain, and in particular in England, greatly exceeds that of any other European country. The quality of the experience when shooting driven gamebirds in Britain appears to be equated with the number of birds that are killed. Financial interests override the environmental, ecological, conservation and animal welfare and public health impacts of the industry.*
2. *The legal status of ring-necked pheasants and red-legged partridges in Britain is confused, and the level of protection they receive is inconsistent with their status as an introduced species. While the gamebird-shooting industry portrays pheasants and red-legged partridges as naturalised, there is no evidence that either species would survive in Britain without extensive releases each year and a variety of other interventions designed to support released gamebirds. Both ring-necked pheasants and red-legged partridges are non-native species, and a review of the legal protection they are afforded in Britain is long overdue.*
3. *There are a number of significant welfare issues associated with rearing and shooting gamebirds in Britain. These include: high predation rates while poults are in their release pens; high levels of mortality on the roads following release; high levels of disease; high levels of wounding; and the number of wounded gamebirds that are not recovered. Self-regulation by the gamebird-shooting industry is ineffective, and there is no evidence that existing codes of practice are addressing these welfare issues.*
4. *There are significant welfare issues associated with the widespread killing of possibly/probably milions of predominantly native birds and mammals each year to protect the commercial interests of the gamebird-shooting industry. These include the widespread use of poisons, snares, traps and different forms of shooting to kill native predators and scavengers, all of which have significant, and well-established, welfare concerns. The ecological impacts of killing so many native predators and scavengers have not been quantified but are likely to be substantial.*
5. *It is anomalous that released non-native gamebirds have a close season while a wide range of native predators and scavengers are routinely killed during their breeding season. Recognising and respecting animal sentience in Britain is long overdue, whether or not a species is a predator and/or scavenger.*
6. *The only stratified survey of fox numbers in Britain was undertaken at the turn of the century. Since then, a long-term monitoring scheme run by the British Trust for Ornithology has shown that the number of foxes in Britain has been declining, possibly because of an even greater decrease in the numbers of rabbits due to rabbit haemorrhagic disease. Rabbits were a major food source for foxes in Britain during the second half of last century. The spread of sarcoptic mange may also have contributed to the decline in fox numbers. The scientific evidence does not support claims that fox numbers have been increasing or that fox densities in Britain are higher than the rest of Europe.*
7. *The gamebird-shooting industry produces substantial amounts of surplus food for predators/scavengers in the form of captive-reared birds that are easy to catch and kill, and carcasses of gamebirds that die of disease, accidents, or are wounded or killed by shooters but not recovered. Industry figures show that 33.2 million kg of surplus pheasants and 2.6 million kg of surplus red-legged partridges (i.e., gamebirds that are reared and released but not shot) enter the ecosystem each year. This has widespread environmental, ecological and conservation impacts, and has a substantial impact on the numbers of predators and scavengers in Britain.*

Many of the pheasants shot each year are surplus to requirements

8. *A number of studies have shown that 40% of released pheasants and red-legged partridges (and possibly more) are predated by foxes, i.e., of the 35.8 million kg total biomass of surplus gamebirds released in Britain each year, around 14.3 million kg is predated by foxes. Since an adult fox requires 180 kg of meat to support itself for a year, data from the gamebird-shooting industry show that predation on pheasants and red-legged partridges provides enough food to support 80,000 foxes for a full year. The availability of carrion from gamebirds that die of other causes could support anything up to a further 120,000 foxes for a year, although it is not possible to determine the proportion of the available gamebird carrion that is consumed by foxes, how much is consumed by other scavengers, and how much decomposes. The number of foxes supported by predating and/or scavenging non-native gamebirds has increased 10-fold since the turn of the century.*
9. *Gamebird shooting is widespread in Britain and its environmental, ecological and conservation impacts affect much, if not all, of lowland Britain. The 'surplus' gamebirds (i.e., easy-to-catch prey and carrion) reared and released each year by the gamebird-shooting industry are available throughout much of the year. Food availability is a major factor that determines fox numbers, and there would be significantly fewer foxes in Britain if the gamebird-shooting industry stopped providing the supplementary food that supports*

large numbers of predators and scavengers. The gamebird-shooting industry has played a major role in supporting the British fox population after the decline in rabbit numbers following the spread of rabbit haemorrhagic disease.

10. *The ecological impacts of supplementary feeding by the gamebird-shooting industry on populations of other small and medium-sized predators are currently unclear. Changes in fox numbers will have significant effects on the numbers of other predators and scavengers. Since foxes disperse over large areas, the effects of supplementary feeding by the gamebird-shooting industry on predator numbers and community structure, and the consequential environmental, ecological and conservation impacts, cover large spatial scales.*

11. *A number of studies in Britain and elsewhere have shown that the availability of carrion, particularly predictable supplies of carrion, results in high densities of predators and scavengers, thereby locally increasing the predation pressure on ground-nesting birds. This problem may be exacerbated in Britain because the amount of easy-to-catch prey and carrion produced by the gamebird-shooting industry supports large numbers of foxes through the winter, but the availability of this supplementary food is lowest in spring and early summer, during the bird nesting season. Providing large amounts of supplementary food at predictable times and locations is contrary to best-practice guidelines.*

12. *Attracting predators and scavengers to predictable sources of carrion enhances the risk of disease spread, both within and between various species of wildlife, and by spreading diseases and parasites of livestock and gamebirds. The role of gamebirds in spreading avian influenza, coronaviruses and other diseases to wildlife is currently unknown.*

13. *Each year the gamebird shooting industry uses 376,000 tons of feed (94,000 tons of grower's pellets, 282,000 tons of wheat) to rear 47 million pheasants for shooting, i.e., nearly 2% of the UK's annual wheat production is used to rear pheasants for shooting. The impact of all this supplementary food on wildlife populations is unknown. Eight million pheasants are left alive at the end of the shooting season, when many shoots stop feeding their released birds. This is the time of the year when natural food supplies are at their minimum: the impact of this high level of competition on wild bird populations is unknown. The various forms of supplementary feeding practised by the gamebird-shooting industry have widespread effects on food webs and ecological processes in Britain.*

14. *Rearing and releasing large numbers of non-native gamebirds in confined conditions outside their native ranges is associated with a wide range of health issues. The gamebird industry uses a disproportionate amount of antibiotics when compared to all the other animal-production systems in Britain. This is of particular concern because antibiotics continue to be administered to gamebirds after they have been transferred to their release pens, where they are highly vulnerable to predation. Over the past twenty years, many tonnes of the antibiotics used by the gamebird-rearing industry will have entered the British ecosystem through predation and scavenging on gamebirds. In addition, many species of wildlife has access to the feeders and drinkers that are placed in and around pheasant and red-legged partridge release pens to administer these antibiotics. The gamebird rearing and shooting industry is likely to be a major source of antibiotic resistant bacteria in British wildlife. This is a significant 'One Health' issue.*

15. *A wide range of other pharmaceuticals used to treat gamebirds for a diversity of diseases also enter the ecosystem because many species of wildlife have access to the feeders and drinkers placed in release pens to administer the pharmaceuticals, and through predation and scavenging on gamebirds before the onset of the shooting season. The environmental, ecological and conservation impacts of disseminating these pharmaceuticals in British wildlife and the ecosystem are unknown.*

16. *A significant amount of lead shot is used to kill non-native gamebirds each year. A minimum estimate is 2500 tonnes, but it may be significantly higher. A substantial amount of lead shot is also used to kill (predominantly) native species of birds and mammals that are considered to be detrimental to the interests of the gamebird-shooting industry. Since most gamebird shooting is done from stands that are re-used several times each year, and often for many consecutive years, the billions of shotgun pellets deposited each year generates very high levels of lead pollution in areas where gamebirds are being reared and shot.*

17. *In addition to the contamination of large numbers of pheasants and other birds that ingest lead shotgun pellets each year, scavenging of shot gamebirds that were missed by the pickers-up, and predation of gamebirds that were wounded but escaped, means that a wide range of avian and mammalian predators and scavengers on and around gamebird shoots ingest significant amounts of lead shot each year. This level of lead pollution is another 'One Health' issue associated with the gamebird-shooting industry. The environmental, ecological, conservation and welfare consequences of depositing so much lead into the British countryside are currently unclear.*

18. *While supplementary feeding by the gamebird-shooting industry supports between 80,000 and 200,000 foxes a year, an estimated 89,000 (± 76,000-100,000 95% CIs) foxes were killed on game shooting estates in 2016, i.e., the gamebird-shooting industry is killing as many foxes each year as they support by supplementary feeding. Industry arguments that they need to kill large numbers of foxes each year to support their interests are, at best, incongruous. Furthermore, there is widespread agreement from studies in Britain and elsewhere that any perceived benefits of killing large numbers of foxes are, at best, temporary. An end to supplementary feeding of predators and scavengers by the gamebird-shooting industry is likely to have a much more substantial, and long-term, effect on fox numbers and population dynamics.*

19. *There is no clear evidence that the large-scale killing of foxes has significant population benefits for ground-nesting birds. Killing one species of predator without considering the impacts on populations of other species of small and medium-sized predators, and the consequential impacts on ground-nesting birds, is naïve. An objective assessment of the need for, and consequences of, predator control in Britain, based on environmental, ecological, conservation and welfare criteria rather than the commercial interests of the gamebird-shooting industry, is long overdue.*

20. *Rearing and releasing large numbers of non-native gamebirds has a range of environmental, ecological and conservation impacts at a variety of spatial scales: (i) on shooting estates, especially estates that contain European Protected Sites, Sites of Special Scientific Interest, and functionally linked land; (ii) on land immediately adjacent to shooting estates; and (iii) in the wider countryside. A recent review by Defra only considered the local effects of gamebird releases, whereas the gamebird-shooting industry has a number of significant (perhaps more significant) landscape-scale impacts on European Protected Sites, Sites of Special Scientific Interest, functionally linked land, and conservation in Britain more generally. There are also significant public health and safety, and animal welfare concerns associated with rearing and releasing large numbers of non-native gamebirds.*

Sources of information

My brief was to look at the impacts of rearing, releasing and shooting non-native gamebirds in Britain, and in particular any environmental, ecological, conservation, welfare and public health concerns.

I have focussed on issues that relate to gamebirds once they have been transferred to their release pens. In particular, the impacts of large-scale releases of non-native gamebirds on European Protected Sites (EPSs) and Sites of Special Scientific Interest (SSSIs) need to be better understood [1,2]. Most pheasant shooting occurs in south and east England [3]. In the mid-1990s, it was estimated that there were 94,000 providers of *game, wildfowl and rough shooting* in Britain, of which 80,000 were in England and Wales [4]. However, this appears to have been an over-estimate: in 2004 the Public and Corporate Economic Consultants (PACEC) estimated that there were 61,000 shooting providers in Britain, of which 83% relied on released ring-necked pheasants (*Phasianus colchicus*) or partridges, especially in England [5]. A subsequent review by PACEC suggested that shooting providers have management responsibilities for some 14 million hectares, i.e., about two-thirds of the UK's rural land area, with active shoot management undertaken on nearly 2 million hectares, which represents about 12% of the UK's rural land [6].

In undertaking this review, a major limitation was the lack of data on the extent of the gamebird rearing and shooting industry in Britain. There are, for instance, very different estimates of the numbers of pheasants and red-legged partridges (*Alectoris rufa*) being released, and so at various points I explain which data sources I use in my analyses. I was asked to focus in particular on the impacts of foxes (*Vulpes vulpes*), since they are believed to be the major predator of released gamebirds and on native ground-nesting birds, and other reviews have already considered the impacts of gamebird releases on avian predators and scavengers, e.g. [7,8]. When looking at the ecological effects of carrion availability and supplementary feeding, I have used data from other countries where appropriate, as I have when looking at the effects of supplementary feeding on fox demography and the effects of culling on red fox populations.

Pheasants on roads are a significant safety issue

Introduction

In 2016 47 million ring-necked pheasants were released in Britain

Recreational shooting has long been promoted world-wide as a means of generating conservation benefits such as controlling populations of overabundant species and restoring ecosystems. However, there is a widespread, and growing, concern about the introduction of non-native (alien) species for shooting purposes and their impacts on ecosystems, and that recreational shooting involves killing native predators to ensure that non-native species of game can be maintained at artificially high densities. A recent review [9] highlighted the lack of information to assess these issues at both national and global scales, and in particular the need for more information on the impact of recreational shooting on biodiversity.

Over the past twenty years a number of reviews have discussed various aspects of the environmental and ecological impacts and conservation gains of gamebird shooting in Britain, e.g. [7-30]. However, most of these reports and papers ignore the welfare issues and only consider the effects of gamebird shooting on specific species or groups of species, or on landscape structure: they generally relied on limited and/or species-specific sources of information and often failed to consider the wider scientific literature. This is a significant weakness of earlier reviews. Here I consider the wider welfare, human health, environmental, ecological and conservation impacts of rearing, releasing and shooting large numbers of non-native species of gamebirds in Britain. I focus on issues not addressed in earlier reviews. Wherever possible, I have relied on peer-reviewed papers, but of necessity background information has been taken from industry and other reports, and popular and press articles where these are relevant.

The legal status of non-native gamebirds in Britain

Grey partridges (*Perdix perdix*) are native to Britain; however, their numbers have been in steep decline since World War II, and efforts to reverse this decline have been unsuccessful [31,32]. Relatively small numbers of grey partridges are released each year: an estimated 180,000 ± 97,000-290,000 (95% CIs) were released in 2004, 200,000 ± 100,000-400,000 in 2012, and 190,000 ± 97,000-390,000 in 2016 [33]. Over-winter survival rates of these released birds are low [34]. Since grey partridges are a native species, they are not included in this review.

The vast majority of gamebirds released in the UK are ring-necked pheasants (hereafter pheasants) and red-legged partridges. Both are introduced, i.e., non-native species, and their legal status can best be described as perplexing. The Game Conservancy Trust (subsequently renamed the Game & Wildlife Conservation Trust, hereafter GWCT), for instance, stated that *the conservation of the pheasant is important because of its long history of naturalisation and importance as a symbol of our traditional countryside* [35].

However, the term *naturalised* is widely misused and misunderstood. Neither pheasants nor red-legged partridges fulfil the criteria to be described as naturalised. *Naturalisation starts when abiotic and biotic barriers to survival are surmounted and when various barriers to regular reproduction are overcome* [36], i.e., by the species, not by human intervention. This is not the case for pheasants or red-legged partridges in Britain. Data from studies by the GWCT, quoted throughout this report, show that very few released pheasants and red-legged partridges survive long enough to breed successfully. It is unclear whether, or for how long, either species would persist in Britain without the extensive measures necessary to support their releases. These include annual restocking with large numbers of captive-reared birds, the large-scale killing of (predominantly) native species of predators and scavengers, frequent medication during the rearing and releasing processes, extensive supplementary feeding, and widespread habitat manipulation.

As an indication of the magnitude of these interventions, it has been estimated that a minimum of 1.5 million birds and 3 million mammals are killed in Britain because they are perceived to be detrimental to the interests of the gamebird-shooting industry [37]. I refer to these as *predominantly* native species because the total includes non-native species such as brown rats (*Rattus norvegicus*), feral cats (*Felis catus*), feral ferrets (*Mustela putorius furo*) and rabbits (*Oryctolagus cuniculus*), as well as a wide range of native avian and mammalian predators and scavengers.

While this estimate may appear to be high, a study on Scottish grouse moors calculated that 15 animals per km^2 were snared and trapped each year [38]; this calculation did not include the mammals that were shot or poisoned, or targeted shooting and trapping of birds, both illegal and illegal. So it is indisputable that large numbers of native birds and mammals, i.e., possibly/probably into the millions, are being killed each year to support the gamebird-shooting industry, and that many, if not most, of these are killed either illegally or in contravention of existing codes of practice [38]. Since many of the welfare issues associated with killing large numbers of predators and scavengers by the gamebird-shooting industry have already been reviewed [38], I focus on the environmental, ecological and conservation impacts of killing large numbers of (predominantly) native birds and mammals.

Although the non-native gamebirds used by the shooting industry are bred in captivity using intensive-production systems comparable to those used for poultry, the Welfare of Farmed Animals (England) Regulations 2007 (and similar legislation in Scotland and Wales) defines a farmed animal as one that is bred or kept for the production of food, wool or skin, or other farming purposes, but not including *an animal whilst at, or solely intended for use in, a competition, show or cultural or sporting event or activity* [39], i.e., non-native gamebirds are not considered to be farmed animals.

However, captive gamebirds are covered, up to their time of release, by welfare legislation similar to that which applies to poultry production, including the Cruelty to Animals Act 1911, the Animal Welfare Act 2006, and other welfare and transport legislation [40]. This is why the *Code of practice for the welfare of gamebirds reared for sporting purposes* [41], issued by the Department for Environment, Food and Rural Affairs (hereafter Defra), only applies to gamebirds *up to and including the period when they are confined to the release pens*. Pheasant poults are moved to a release pen at six weeks of age and, once they have acclimatised, pop holes are opened so that they can forage during the day but return at night [42]. Red-legged partridges are typically placed in release pens around eight weeks of age, and held there for two to four weeks before being 'trickle' released, i.e., small numbers of birds are released at a time, and food is provided in the pen to keep them in the vicinity, although sometimes all the birds may be released at the same time [43].

While the *Code of practice for the welfare of gamebirds reared for sporting purposes* ceases to apply to gamebirds once they are able to leave and re-enter the pens voluntarily [41], Defra's code of practice goes on to say that *It is recognised, however, that as keepers will retain some responsibility for the welfare of the birds immediately post release and until they have adjusted to a free-living existence, the suitability of the release environment to meet the needs of birds must be considered*, and *All birds should be adequately protected from predators* [41]. Since 25-30% of poults are killed by predators (mostly foxes) between the time they have free access to and from their release pen to the start of the shooting season [44], it is unclear how knowingly releasing poults into a situation where they are exposed to high levels of predation fulfils a landowner's, shoot-manager's and/or gamekeeper's obligations under section 9(1) of the Animal Welfare Act 2006 to ensure that he/she meets the needs of the animals for which he/she is responsible.

Since gamebirds are susceptible to avian influenza and Newcastle disease, they are covered by notifiable disease legislation such as the Avian Influenza and Newcastle Disease (England and Wales) Order 2003 and the Avian Influenza (Preventative Measures) (England) Regulations 2006 (and the equivalent legislation in Scotland, Wales and Northern Ireland). The Avian Influenza (Preventive Measures) (England) Regulations 2006 make it a legal requirement for anyone with 50 or more gamebirds to register their flock [40].

Unlike other non-native species of pheasants and partridges, ring-necked pheasants and red-legged partridges are not included in schedule 9 of the Wildlife & Countryside Act 1981. However, neither do they enjoy the comprehensive protection given to wild birds in section 1 of the Wildlife & Countryside Act: the definition of a *wild bird* used in the Act excludes pheasants and red-legged partridges, except in relation to the use of *Prohibited Methods* of killing (section 5 of the Act) and *Licensing* (section 16 of the Act). This means that gamebirds, their eggs and their nests are protected under this legislation from the use of certain methods of killing and taking [45].

While the shooting season for pheasants is from 1st October to 1st February, the British Association for Shooting and Conservation (BASC) advises that most shoots should not start before 1st November as birds reared in Britain are not normally suitable for shooting until the end of October. Commercial operations that want to organise shoots in October are more likely to buy in chicks and poults from France to enable them to get off to an earlier start [46]. The shooting season for red-legged partridges in England is from 1st September to 1st February. BASC's code of good shooting practice says that red-legged partridges should be at least 15 to 16 weeks old before shooting to ensure that they are fully mature, healthy and marketable [47].

However, it is unclear why two non-native species of gamebirds are afforded this level of protection: twenty years ago it was pointed out that their close seasons simply reflect the *existence of*

Unlike released gamebirds, foxes have no close season and can be killed at any time of the year

outdated game laws, divorced from modern conservation legislation that perpetuates the status quo so that the demands of gamebird management take precedent over wider environmental issues [37]. This remains the position today. A reassessment of the legal status of non-native gamebirds is long overdue. In particular, landowners who do not rear, release, or shoot non-native gamebirds should be allowed to cull these introduced species whenever they so wish.

Non-native gamebirds are essentially a private resource released by a landowner and/or his/her agents who largely control access to that resource. However, the landowner and/or person who releases non-native gamebirds has minimum responsibility for the impacts of their actions, whether they be environmental, ecological or on conservation, or for any injuries, deaths or effects on human health, although a neighbour or tenant may claim for damages to their crops. This anomalous situation needs to be addressed. The polluter-pays-principle is enshrined in England in the Environmental Damage (Prevention and Remediation) Regulations 2009, and it is unclear why this principle is not applied to people who release non-native gamebirds. Section 4 (1) of these Regulations states that they *apply in relation to the prevention and remediation of environmental damage*; and that environmental damage includes *protected species or natural habitats, or a site of special scientific interest*.

Since gamebirds typically disperse up to 500 m before the onset of the shooting season (see below), introducing legislation to prohibit positioning gamebird release pens within 500 m of a road would have a substantial impact on the number of accidents and human fatalities due to collisions with released pheasants. Between 1999 and 2003 collisions with pheasants were implicated in 65 accidents per year that led to human injury, with approximately 6% of these leading to serious injury or death [48]. Insurance-industry figures suggest that more than half of all drivers have either hit or had a near miss with an animal on UK roads, and that large gamebirds such as pheasants are the third most likely animals to be involved, accounting for 20% of all incidents [49]. Fatal road accidents with pheasants most commonly involve motor-cyclists, e.g. [50-56], although car drivers can also be involved in fatal collisions with pheasants, e.g. [57-58].

Such a simple provision would also (i) lead to a significant improvement in animal welfare standards by minimising the number of pheasants killed on British roads, and (ii) reduce the number of gamebird carcasses available to scavengers, and the consequential environmental, ecological and conservation impacts that I discuss below. It should also be a legal requirement for released pheasants and red-legged partridges to be fitted with a numbered leg ring so that, where necessary, the person who released the birds can be traced and held responsible for their actions.

Good-practice guidelines for gamebird shooting

A number of guidelines have been issued on best practices when rearing gamebirds for shooting. However, there is little official surveillance or monitoring of premises used for rearing gamebirds because they are not considered to be farmed animals [39], and so gamebird premises are not selected for risk-based or random inspection by the Animal and Plant Health Agency.

In 2010 Defra published its *Code of practice for the welfare of gamebirds reared for sporting purposes* [41]. The aim of this code was *to provide practical guidance in relation to section 9 of the Animal Welfare Act 2006 affecting birds bred and reared under controlled conditions for the purpose of release for sport shooting, together with birds retained for breeding purposes* [41]. The sections of the code that are particularly relevant to this review are:-

- *During the production and rearing process, birds are protected from unnecessary suffering by section 4 of the* (Animal Welfare) *Act*
- Section 9(1) of the Animal Welfare Act 2006 states that *A person commits an offence if he does not take such steps as are reasonable in all circumstances to ensure that the needs of an animal for which he is responsible are met to the extent required by good practice*
- Section 9(2) states that *For the purposes of this Act, an animal's needs shall be taken to include – its needs for a suitable environment* and *its needs to be protected from pain, suffering, injury and disease. Should any of these occur a rapid response is required, including diagnosis, remedial action and, where applicable, the correct use of medication*
- *All birds should be adequately protected from predators. Any methods used must conform to legal requirements. Pest control procedures should be operated to ensure the health and welfare of the birds* (section 3.8)
- *When birds are housed or penned, the accommodation should be well constructed and managed and of sufficient size to ensure good health and welfare. This is best achieved by: …. (ii) protection from adverse weather conditions, extremes of temperature and predators* (section 6.1)
- *Medicines for treatment should only be used when necessary or when prescribed by a veterinary surgeon. Preventative use of medicines should only be carried out where appropriate and in conjunction with good husbandry practices or when the birds are under the care of a veterinary surgeon who recommends a prescribed medicinal product* (section 7.2)
- *The siting of release pens should take into consideration the need to minimise the risk of subsequent harm or injury, for example by predators or vehicles* (section 9.2)

In their *Code of good shooting practice*, BASC [47,59] makes the following points that are particularly relevant to this review:-

- *One of the five golden rules is that it is fundamental to mark and retrieve all shot game*
- *Shooting should not be conducted where it will not be possible to retrieve shot game*
- *Shoot managers should locate, provide and manage adequate habitat and feed supplies to avoid boundary problems with neighbours*
- *The siting of release pens and feeding of game near highways should be avoided. Game managers should collect and dispose of road casualties where possible*
- *Guns must be competent at estimating range and shoot within the limitations of their equipment to kill cleanly and consistently*
- *Guns must satisfy themselves adequate provision is made for retrieval of the game they shoot*
- *Shoot managers must ensure that adequate provision is made to retrieve all shot game and dogs are an essential part of this process*
- *On driven days, any wounded game should be retrieved during drives whenever it is safe and practicable to do so*
- *Avoid birds and spent shot falling on to public places, roads and neighbouring property*
- *Shoot managers must ensure they have appropriate arrangements in place for the sale or consumption of the anticipated bag in advance of all shoot days*

Pheasants that have been 'breasted' and dumped on the side of a road

- *Sufficient feed for released birds remaining after the end of the shooting season should be provided until adequate natural food is available, normally to the end of May*
- *Shoot managers should be aware of SSSIs and other sensitive habitats on their ground, and should liaise with the landowner and the relevant statutory authorities to ensure they avoid potentially damaging activities*

There are strict rules on the disposal of fallen farm stock in England (the Animal By-Products (Enforcement) (England) Regulations 2013 and the Animal By-Products (Miscellaneous Amendments) (England) Order 2015); there is comparable legislation for other parts of Britain. While the carcasses of wild animals are exempt from these rules, wild game are not [60], although there has been confusion as to whether these guidelines apply to the disposal of surplus and/or dead gamebirds. For instance, the advice published by the National Gamekeepers' Organisation was that, while wild game species such as deer, grouse, partridges and pheasants are exempt if they were wild when killed and not intended for the human food chain, the eggs, chicks and gamebirds that died in captivity are covered by the Regulations [61].

Recently, the British Game Alliance *reviewed and clarified their standards* for the disposal of game that is unfit for human consumption [62]. They explained that *Most waste from shoots is classified at 'Cat 3' – products or food of animal origin originally meant for human consumption, but withdrawn for commercial reasons, not because it's unfit to eat. This could include badly shot birds, damaged by dogs, birds in bad condition and carcasses that have been processed, for example breasted.* Their standards require *that game that is unfit for consumption and processed game carcasses must only be disposed of by incineration, collection by a member of the National Fallen Stock Company, or via an Approved and Registered Animal By-Products Premises*. They also stated that *Game awaiting incineration or disposal must be stored securely. Incinerators must be approved by the Animal and Plant Health Agency (APHA) and serviced at least annually. Ash from the incinerator must be disposed of either via licenced premises or, with a permit from the Environment Agency (EA) or SEPA mixed with manure and spread on agricultural land.*

So the landowner, shoot manager and/or gamekeeper is responsible for the safe and legal disposal of dead gamebirds and, while awaiting collection, that person must ensure that animals and birds cannot access the carcass(es). However, it appears that this guidance is only interpreted as applying to the carcasses of gamebirds that were shot and thereby potentially destined for human consumption. As I show below, most released gamebirds die of a variety of other causes, and there is no clear guidance on the disposal of these carcasses. So, while BASC's *Code of good shooting practice* [47,59] requires that game managers should collect and dispose of road casualties *where possible*, this requirement is vague at best, and there is no stipulated method of carcass disposal. Game managers should be required to dispose of the carcasses of all released gamebirds, however they die, *by incineration, collection by a member of the National Fallen Stock Company, or via an Approved and Registered Animal By-Products Premises*. As I show below, this would remove 21.5 million kg of carrion from the British ecosystem each year and would have substantial beneficial environmental, ecological and conservation benefits.

How many foxes are there in Britain?

When assessing the impact of the gamebird-shooting industry on populations of small and medium-sized predators, it is important to have an accurate estimate of the number of foxes in Britain. The only comprehensive estimate based on a stratified survey across the whole of Britain calculated that the total rural fox population was 225,000 ± 179,000-271,000 (95% CIs) adult foxes; including foxes in urban areas, the total national fox population was estimated to be 258,000 adults, i.e., foxes ≥ one year old [63]. It is important to note that these data were collected in February and early March 1999 and 2000, after the main dispersal period and just before the birth of the next cohort of cubs.

There were two previous population estimates, each produced by internationally-recognised experts, who used habitat data and known/published fox densities to estimate the total number of foxes in Britain. One produced an estimate of 252,000 adult foxes in spring, i.e., before the next cohort of cubs joins the population [64], and the other estimated a total population of 240,000 adult foxes in Britain at the end of winter, with 195,000 in England, 23,000 in Scotland and 22,000 in Wales [65].

Thus three population estimates published between 1981 and 2004 produced remarkably consistent results, both in terms of the number of adult foxes in Britain at the end of the winter, and in the relative distribution of foxes across Britain. All three studies suggested that, at the turn of the century, there were around 250,000 adult foxes in Britain at the end of winter, i.e., this was the population minimum.

A more recent estimate based on habitat suitability modelling produced an estimate of 357,000 ± 104,000-646,000 (95% CIs) foxes in Britain, including those in urban areas. It is, however, hard to compare this figure with previous estimates since it is unclear whether it only included adult foxes, i.e., those ≥ one year old, or whether, unlike earlier studies, it also included subadults (i.e., animals < one year old), the time of year it referred to, and the proportion of those foxes that were in urban areas. Furthermore, the wide confidence intervals reflect the variability of the data that were used in the analyses. It is particularly important to note that the authors of this modelling exercise also state that they overestimated the numbers of foxes in Britain because: (i) *no percentage occupancy data were available*, which was a key requirement for their analyses; (ii) they used a *high median population density across all rural areas, including upland Wales and Scotland where food is scarcer*; and (iii) the density estimates they used *for urban areas may be higher than typically found in Great Britain* [66]. Since several significant factors inflated this population estimate, it will not be considered further.

In the absence of a scientifically credible fox population estimate in the past 20 years, I have considered the evidence for any changes in fox numbers since the national survey undertaken at the turn of the century [63]. There are two potential sources of information. The first is from the British Trust for Ornithology (BTO); they have collected data on mammal population trends since 1995 as part of the Breeding Bird Survey. This is a highly structured survey that uses consistent recording techniques on randomly selected sites and a stratified sampling approach; all of these factors have huge advantages in terms of data quality [67]. The annual indices produced by the BTO show a high level of consistency between years and narrow confidence intervals. The BTO's data showed that, between 1996 and 2018, there was a 44% decline in fox numbers in the UK as a whole and 49% in England [68].

The other potential source of information on changes in fox numbers in the UK is the GWCT's National Gamebag Census (NGC), which was established in 1961; it superseded an earlier collection of game records set up by Oxford University in the 1930s and was specifically designed by the GWCT *to provide a central repository of bag records from UK shoots* [69]. The NGC bag data suggest that, over the periods 1966 to 2016, 1991 to 2016 and 2004 to 2016, the numbers of foxes killed on gamebird-shooting estates rose by 180 ± 124-243% (95% CIs), 24 ± 13-36% and 4 ± -4-11% respectively [33]. The NGC data show that fox bags on shooting estates are now 3.5 times higher than in 1961, although the rate of increase slowed considerably from the early 1990s onwards [70].

However, it must be remembered that the NGC data were only ever intended to be a record of the numbers of animals killed on a sample of UK shooting estates [69] and have several significant flaws as a potential monitoring scheme. These include: (i) the number of animals killed depends on harvesting effort, but there is no measure of 'effort', a key requirement for a monitoring scheme; (ii) developments in predator control practice have affected the seasonality of culling: in recent decades, there has been a shift from killing foxes in spring and summer using snares and terriers to killing foxes in autumn and winter by lamping with a rifle. This will affect the number of foxes killed because the techniques used to kill foxes have changed, as have the time of year over which they are killed. So at least some of the NGC trend may reflect changing control methods [70], and the peak period for killing foxes now is before high levels of natural mortality in the winter [71]; and (iii) the sites contributing to the NGC are self-selecting, so they cannot be assumed to be representative of UK shooting estates, let alone indicative of changes in the wider British countryside [72,73]. Since the NGC data do not conform to the basic requirements of a monitoring scheme using game-bag statistics [72,74], the NGC data can only be interpreted as showing changes in numbers of foxes that were killed on a non-random sample of shooting estates [75].

So, of the two sources of data that are available, those collected by the BTO are clearly far more credible in terms of monitoring long-term fox population trends in Britain, and will be used in this review. At the turn of the millennium there were around 250,000 adult foxes in Britain at the end of the winter: the BTO's data suggest that, since then, the number of foxes in Britain has declined by around 40%, and so the current pre-breeding number of adult foxes in Britain is likely to be nearer 150,000.

Assuming a mean litter size of five cubs, in the 1990s it was estimated that around 425,000 cubs were born each spring [65], and so the total fox population in April was 240,000 adults and 425,000 cubs, i.e., 665,000 foxes. Typically 60 to 70% of the fox population dies each year [76]. While long-term data from urban fox populations showed some seasonal variation in mortality rates between different age and sex classes [77], there are limited data on monthly mortality rates for different rural fox populations. So, to illustrate the magnitude of monthly changes in fox numbers in Britain at the turn of the century, I have assumed that the population was stable and that mortality rates did not vary across the year. This would mean that there were around 665,000 foxes (adults and young of the year) in April, 625,000 in May, 590,000 in June, 550,000 in July, 510,000 in August, 470,000 in September, 430,000 in October, 390,000 in November, 355,000 in December, 315,000 in January, 280,000 in February and 240,000 in March. These are of course approximations, but they illustrate the annual rate of change in fox numbers in Britain, and hence the importance of specifying the exact time of year when comparing different fox population estimates, e.g. see [66].

Two factors could explain the decline in fox numbers in Britain over the past 25 years. The BTO's monitoring data show that rabbit numbers declined by 64% over the same period as fox numbers declined by 44% [68]. The NGC data suggest a similar pattern and magnitude of decline [70]: rabbit numbers were low in the 1960s, following the first outbreak of myxomatosis in 1953. As resistance to the Myxoma virus developed, the numbers of rabbits killed on shooting estates increased by 16-fold to a peak in the mid-1990s [70]. However, whether this was an accurate measure of the changes in the rabbit population in the wider countryside is unclear, since an experimental study suggested that myxomatosis was still having a considerable impact on rabbit numbers [78]. Over the next 15 years, the NGC data show that the numbers of rabbits killed on shooting estates declined by about a third, then appeared to stabilise, followed by a decline of another third during the past ten years. The first decline corresponded to the emergence of rabbit haemorrhagic disease (RHD1), which reached southern England in 1992 and spread north to Scotland by 1995. A more pathogenic variant of the same disease (RHD2) reached Britain around 2010, tying in with the second decline phase in the NGC bag data [70], although it should be remembered that the rabbit bags reported by the GWCT also reflect the extent to which shoots included rabbits in their records [79].

In the second half of the 20th century, rabbits were a key component of the diet of foxes in all rural landscapes in Britain. An analysis that collated data from 36 studies covering the half-century 1951 to 1997 estimated that a typical fox family group of 1 adult male, 1.5 adult females and 4 cubs consumed between 190 and 1160 rabbits (depending on their age) each year in arable landscapes, 91 to 355 rabbits in pastoral landscapes, 4 to 387 rabbits in marginal uplands, and 24 to 1067 rabbits in the uplands [80]. Based on the fox densities estimated in these landscapes at the turn of the century [63], foxes in Britain were consuming between 11 million and 64 million rabbits each year.

During this period British foxes were so heavily dependent on rabbits as a food source that levels of fox predation could regulate rabbit numbers [81]; similar impacts of fox predation on rabbit numbers were reported in Australia [82], although rabbit populations can escape predator regulation once they exceed a critical density [81,83-85]. Some studies found that red fox numbers were not affected by a large reduction in the abundance of rabbits, or *vice versa* [86], and high rabbit consumption by foxes in Spain occurred independently of rabbit abundance, suggesting that, in that study, foxes could still specialize on rabbits under the densities being considered [87].

Conversely, rabbit numbers can also limit fox numbers: in Australia, the size of the fox population in summer was dependent on the availability of rabbits in their preceding breeding season [83], and red fox numbers declined in Mediterranean Spain following the arrival of rabbit haemorrhagic disease (RHD) in the late 1980s and the collapse of the rabbit population [88], although foxes do not always show a numerical response to changes in rabbit numbers [89]. Since the impacts of foxes on rabbit populations and *vice versa* have been documented in several countries, and the benefits of foxes in controlling the impacts of rabbits on agricultural crops in Britain are well established [81,90], the parallel decline in fox and rabbit numbers in the UK is perhaps to be expected.

The second factor that is likely to have depressed red fox numbers in Britain over the past twenty-five years is a large-scale outbreak of sarcoptic mange that started in the 1990s: this spread from south-east England, and subjective estimates suggested that local fox populations took 15 to 20 years to recover after the arrival of the disease [91]. Independent evidence that fox numbers started to decline around this time came from a study which showed that there was a significant decrease in fox numbers in southern England during the 2001 foot-and-mouth epidemic [92].

Moreover, changes in fox numbers cannot be considered in isolation. There is a complex interplay between populations of small to medium-sized predators and, while this will have a significant impact on conservation interests in Britain, there is very little quantified information on how changes in the numbers of one species of predator affects other members of the guild. So, for instance, while it has been argued that culling badgers (*Meles meles*) was associated with increases in red fox numbers [93], the mechanisms for such an interaction remain unclear. It was suggested that this could be due to exploitative competition for key resources, leading to increased adult fox survival and immigration, or increased fox cub production and survival in the absence of badgers [93]. However, there is little evidence of food competition between foxes and badgers, e.g. [94,95], or for negative impacts of badgers on fox cub production or survival. In Bristol, for instance, foxes preferred to rest and breed in badger setts when fox numbers were lower and there was less intra-specific competition for the best breeding sites [96] and, in Poland, foxes actually raised more cubs in setts that they cohabited with badgers [97].

A number of studies have shown that pine marten (*Martes martes*) numbers are lower where red foxes are more abundant, e.g. [98,99], and foxes depress populations of stoats (*Mustela erminea*) and other mustelids [100,101]. Likewise, weasels (*Mustela nivalis*) are scarce or absent in habitats occupied by stoats, but weasels may soon move into areas vacated by stoats [102]. The effects of these intra-guild interactions on predation pressure on ground-nesting birds and other species of conservation concern are unknown, and assessments of the possible impacts of population changes of one species in isolation are simplistic [103]. While the potential impacts of anthropogenic perturbations on predator and/or prey abundance are of particular importance when considering the environmental, ecological and conservation impacts of the gamebird-shooting industry in Britain, very little information is currently available.

Long-term monitoring by the BTO has shown that fox numbers have declined by 44% over the past 25 years

How much food do foxes require?

A fox requires about 180 kg of meat per annum

When looking at the ecological impact of foxes, it is important to understand how much food is required to support the British fox population. Early estimates of the energy requirements of foxes were undertaken on fur farms [104-106] and are not applicable to free-living foxes because they were based on animals held in small cages with limited opportunities for movement. The amount of food required to meet a fox's energetic needs depends on its body weight, the type of food eaten, climate, and the distance travelled each day. Studies on free-living foxes in suburban Bristol and Australia suggest that the daily energy requirement of a red fox is about 330 kJ/kg/day [107,108], and that a free-living fox with a mean body mass of 5.7 kg eating birds/mammals would have a mean daily food intake of 520 g [109]. This seems to be a reasonable estimate. While a fox's stomach can hold up to 1 kg of meat [76], this would be exceptional; over 23 days two captive foxes consumed on average 1.0 kg of a sheep carcass each day [110].

There is, however, a substantial turnover in the fox population each year [76], and the majority of the population will be young of the year (see the preceding population estimates). Mean body mass of male and female juvenile foxes is similar to that of adults by around August or September [76,111]. Fox cubs undergo a rapid period of growth up to about three months of age, with slower rates of growth thereafter [112], and during this period their energy requirements are not dissimilar to that of adult foxes.

So I have assumed that all foxes, irrespective of their age, have the same energy requirements through the year, and that the food intake of a fox eating just gamebirds would be 500 g per day or 180 kg per annum.

The number of pheasants and red-legged partridges reared, released and shot in Britain

Although Defra commissioned a research project investigating the provision of bag statistics for huntable birds in the UK as part of an EU requirement to estimate the numbers of birds harvested in each member state [113], some 20 years later there is still no system for recording the numbers of gamebirds reared, released and shot each year in the UK [114]. In the absence of a standardised recording system, it is unsurprising that the available studies have produced a range of estimates.

In Britain there are believed to be around 300 game farms, mostly rearing pheasants and partridges. Some shoots retain a breeding flock to produce their own eggs, and others buy eggs or day-old chicks and rear them on [115]. In 2005 the Game Farmers' Association (GFA) suggested that 20 to 30 million gamebirds were reared for release [46], of which 80% were pheasants and 16-17% were red-legged partridges; the rest were grey partridges and mallards (*Anas platyrhynchos*). The GFA suggested that around 50% of the birds were reared on recognised game farms and the other 50% by gamekeepers for their own estates; an estimated 1000 to 2000 gamekeepers operated on this basis, buying day-old chicks from a hatchery.

In the mid-2000s, around 40% of the pheasants reared in Britain came from France, either as eggs or day-old chicks. There was also a small trade in 6- to 8-week-old poults from France, although this was probably less than 2% of the pheasants reared. Of the eggs laid in the UK, around 50% came from over-wintered stocks of parent birds and 50% from parent birds caught up from the wild. Around 90% of red-legged partridges were imported from France, with some from Spain and Portugal, because stock from these countries can be produced earlier in the year: these imports were roughly equal numbers of eggs and day-old chicks, with around 3% as part-grown poults [46].

While the Avian Influenza (Preventive Measures) (England) Regulations 2006 made it a legal requirement for anyone with 50 or more gamebirds to register their flock [40], it appears that many flocks are not registered [28], and the number of shoots operating in Britain is unclear. Surprisingly, even organisations such as the GWCT do not know the total number of released gamebird shoots in the UK [116]: estimates range between 5000 [28] and 10,000 [117]. This is of particular concern in view of the cases of avian influenza recorded in gamebird flocks in 2020 and 2021 [118].

There is also considerable uncertainty about the numbers of non-native gamebirds released in Britain each year. The GWCT estimated that 35 ± 31-39 (95% CIs) million pheasants were released in 2004, 44 ± 37-53 million in 2012 and 47 ± 39-57 million in 2016 [33]. However, another analysis suggested that between 24.3 million and 25.3 million pheasants are released annually in the UK, although the data used to produce these estimates were highly variable [28]. The GWCT estimated that 15 ± 13-17 (95% CIs) million pheasants were shot in the 2004/2005 season, 13 ± 11.7-14.3 million in the 2012/2013 season and 15 ± 13-18 million in the 2016/2017 season [33].

The GWCT also estimated that 6.3 ± 5.7-7.0 (95% CIs) million red-legged partridges were released in 2004, 9.5 ± 7.4-12 million in 2012 and 10 ± 8.1-13 million in 2016 [33]. Another study suggested that between 4.2 million and 9.4 million red-legged partridges are released annually in the UK, although the data used to produce these estimates were highly variable [28]. The GWCT estimated that 2.5 ± 2.3-2.8 (95% CIs) million red-legged partridges were shot in the 2004/2005 season, 4.4 ± 3.9-4.8 million in the 2012/2013 season and 4.6 ± 3.6-5.9 million in the 2016/2017 season [33].

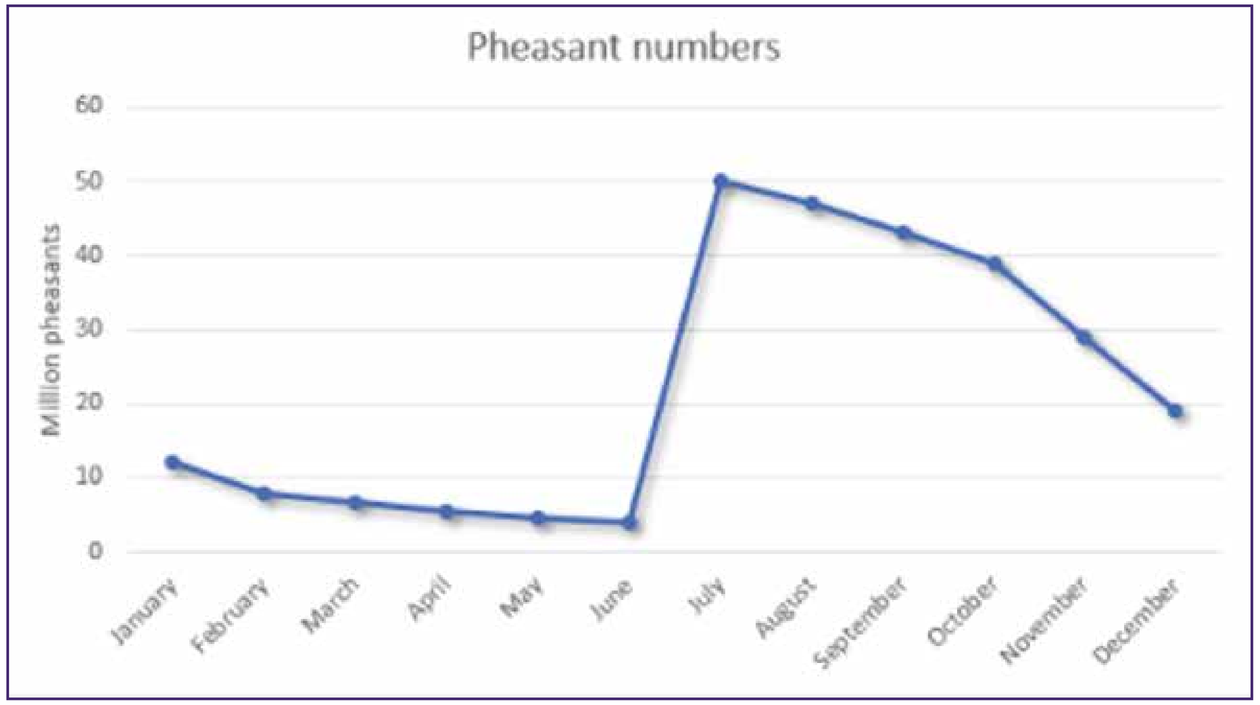


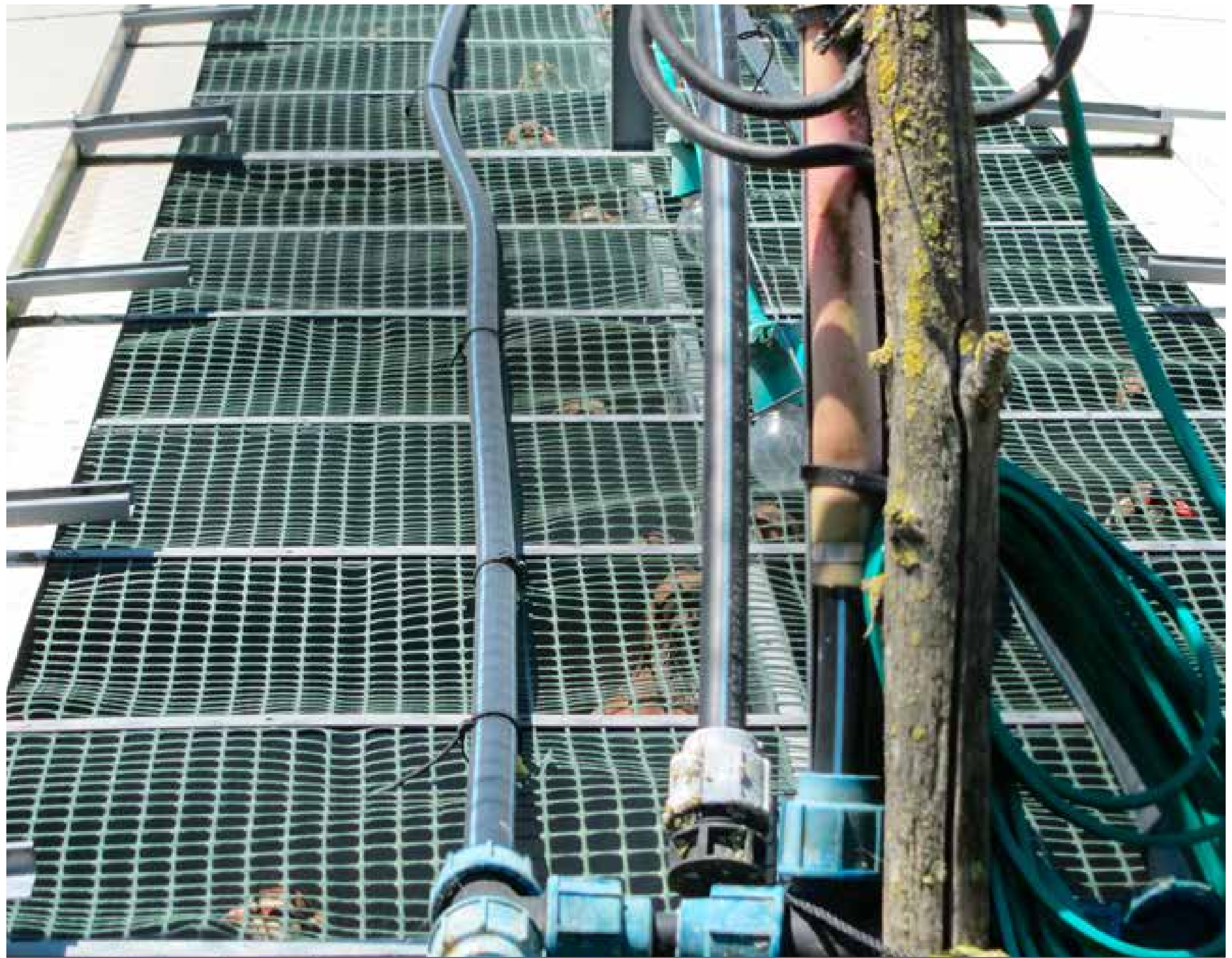
FIGURE 1. Estimated number of pheasants in the UK throughout the year, reflecting the numbers released and subsequent losses to predation, shooting and other processes [119]

There is a marked annual variation in the numbers of free-living pheasants in the UK countryside (Figure 1): numbers vary between a low of around 4.6 million in May (Figure 1), rising to around 50 million just after the releases in late summer. These consist of 47 million pheasants released ready for the start of the shooting season (based on data from 2016) [33], and 3 million survivors from the previous shooting season [119].

Releases of pheasants have increased nine-fold since 1961. Pheasant bags increased correspondingly up to 1990, but have increased little since then: the bag index is only 2.5 times as high as in 1961. Most noticeably, there was no rise during the 1990s despite the increase in releasing, and it was only from 2000 that the NGC bag index began a slow climb. One explanation for this may be that higher releases do not feed back into higher bags because many shoots now offer shoot days in January. Because of on-going losses of released birds from August to December, it may be that shoots need to release disproportionately more pheasants at the start of each season to achieve good late-season bags [120].

According to the GWCT, the average bag of pheasants in Britain in 1900 was approximately 25 per km^2, rising to almost 150 per km^2 in the 1980s [121]. For comparison, national statistics from hunters' associations show that harvest rates per km^2 in various European countries were: Austria 1.83, Belgium (Flandres) 7.61, Czech Republic 7.45, Denmark 17.35, Estonia 0.02, Finland 0.14, France 9.09, Germany 0.69, Hungary 5.02, Lithuania 0.01, Netherlands 2.21, Poland 0.64, Slovakia 2.76, Slovenia 1.77 and Spain 0.51 [122]. In central Europe (Austria, Czech Republic, Germany, Hungary, Poland, Slovakia, Slovenia, South Tyrol and Switzerland), 2,992,075 pheasants were shot in 1970 and 1,270,824 in 2014 in an area of 1.04 million km^2 [123], i.e., roughly four times the total land area of the UK.

These figures are not directly comparable because it is unclear how the GWCT's density estimates [121] were calculated, and biases in the way the NGC data are collected mean that great care has to be taken in interpretation [73]. However, as a rough estimate, since 15 million pheasants were shot in the UK in the 2016/2017 shooting season, and the



Many of the red-legged partridges released in Britain are reared on farms in France

total land area is around 240,000 km^2, 62.5 pheasants were shot and killed per km^2 across the whole of the UK. However, this includes large areas of uplands and urban areas, habitats where pheasant shooting does not take place. Since around 75% of pheasant shooting is believed to take place in rural England [4], approximately 11.25 million pheasants were shot in an area of roughly 91,000 km^2, i.e., around 125 pheasants were shot per km^2 of rural England. Locally, the number of pheasants shot will be significantly higher. Clearly, substantially more pheasants are shot per unit area of the UK, especially England, than in any other European country.

The lack of robust information on the numbers and types of premises used to rear gamebirds for shooting, the numbers of shoots that release gamebirds for shooting, the area of land used for shooting released gamebirds, and the numbers of non-native gamebirds reared, released and shot in Britain each year can only be described as surprising.

The biomass of the surplus non-native gamebirds released each year

The great majority of non-native gamebirds die within a few months of release, and there are several sources of information on the proportion of released gamebirds that are not shot and recovered; for convenience I refer to all these gamebirds, whatever their fate, as 'surplus'. Some are lost to predators as easy-to-catch or wounded prey. The rest die of diseases, parasites, accidents and similar causes. Since there is no evidence that these dead gamebirds are collected and disposed of by the shooting industry, it is reasonable to assume that they enter the ecosystem as carrion.

To put the biomass of released gamebirds into perspective, one analysis claimed that the biomass of pheasants released in Britain each year (red-legged partridges were not included) exceeded that of all native breeding birds [23]. A subsequent calculation by the same authors stated that annually around a quarter of the British bird biomass consists of pheasants and red-legged partridges and, at their peak in August, these two species constitute around half the total biomass of British birds [24].

There are marked annual changes in the biomass of pheasants in the countryside (Figure 2), and the GWCT estimated that the biomass of pheasants in July is about 24,600 tonnes, based on the 47 million released pheasant poults weighing about 0.45 kg each, and 3 million surviving adults from the previous year weighing an average of 1.15 kg. Thereafter, the number of pheasants falls rapidly: around 25% of the released birds die before shooting begins in October or November, most of them being predated by foxes. In September, there are 41 million young pheasants that weigh 0.95 kg, and 2 million adult pheasants from the previous year that weigh 1.15 kg. This is the peak biomass of pheasants in the UK, and amounts to 41,250 tonnes. About 39 million pheasants survive to the start of the shooting season (37 million released that year, 2 million from the previous summer), when the biomass of pheasants in Britain has declined to 41,000 tonnes [119].

In September the total biomass of released pheasants is 41,250 tonnes

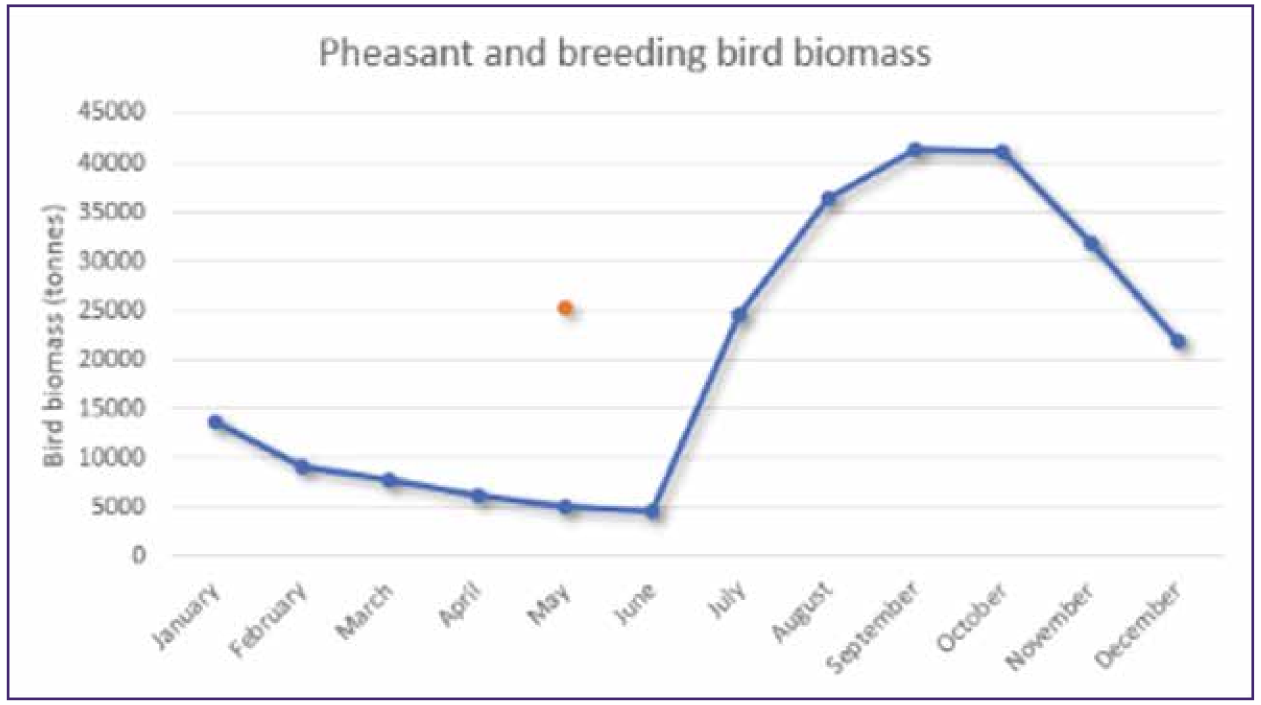


FIGURE 2. Annual changes in the biomassof released pheasants in the UK, taking account of shooting, other losses, and the changing weight of a pheasant after release. The numbers are calculated for the beginning of each month. The orange dot represents the total biomass of all Britain's birds in spring [119]

The GWCT estimates that 16.5% of the pheasants released are still alive after shooting finishes at the start of February, i.e., 7.75 million of the pheasants released the previous summer, with a few older birds, and so the total surviving pheasant population is around 8 million. Studies on hen pheasants from early February until June found that around half die, mostly from predation [124,125]. Around 4.5 million pheasants, with a biomass of 5175 tonnes, remain in May. Breeding success of these pheasants is particularly low, probably on average around 10%, and the survival of hatched chicks is also low [124,125], well below 0.5 million, and has a negligible effect on the overall population and biomass estimates [119].

These data provide an estimate of the biomass of the surplus pheasants that are produced by the gamebird-shooting industry each year. In the months between being released (taken for convenience to be July) and the start of the shooting season (1st October), 1 million adult pheasants from the previous year (averaging 1.1 kg) and 10 million poults released that year (average weight across the period 0.9 kg) are surplus: this is a biomass of 10.1 million kg.

During the shooting season (1st October to 1st February), pheasant numbers decline from 39 million to 8 million: only 15 million are shot [33]. The other 16 million are surplus: assuming an average weight of 1.1 kg, this means that a further 17.6 million kg of dead pheasants enters the ecosystem. After the shooting season (February to July), pheasant numbers decline from 8 million to 3 million; with an average weight of 1.1 kg, this means that a further 5.5 million kg of dead pheasants enters the ecosystem.

So, of the 50 million free-living pheasants in Britain at the start of each season (47 million released, 3 million survivors from the previous year), 47 million do not survive to the next year: 15 million are shot [33] and therefore in theory are removed from the ecosystem. The other 32 million (a total biomass of 33.2 million kg) are surplus and are either predated or enter the ecosystem as carrion.

Data on red-legged partridges are less detailed, but show a similar rapid pattern of decline. Of the estimated 10 million released in 2016, 4.6 million were shot in the 2016/2017 season [33]. A study following the fates of 274 radio-tagged red-legged partridges found that 38% were shot, 34% died of other causes (mainly predation by foxes), the fate of 13% was unknown (usually radio-tag failure) and 15% survived beyond the end of the shooting season [126].

As with pheasants, the long-term survival and breeding success of released red-legged partridges is low: there were only 73,000 breeding territories in 2016, mainly in England [127]. So the breeding population of red-legged partridges is negligible in relation to the 10 million that are released each year. Since 4.6 million are shot [33], I have assumed that the other 5.4 million die from other causes before the start of the next shooting season. Red-legged partridges start to be released when they are 10 to 12 weeks old [43], and they are fully mature when 15 to 16 weeks old [47], so I have assumed that all the surplus birds were adult and weighed 490 g [127]. Thus, the biomass of surplus red-legged partridges that enters the ecosystem each year via predation or as carrion is 2.6 million kg.

There are very few breeding pairs of red-legged partridges in Britain

The impact of predation on gamebirds on fox numbers in Britain

Levels of raptor predation on pheasant poults in and around release pens are low [128,129]. For 90% of shoots, ≤1% of birds released into pens were lost to raptors; losses were estimated to be >5% at one in 30 estates, and >10% at some estates [130]. Raptor predation declined with increasing age of the poults: losses were twice as frequent in poults released in June and July compared to those released in August and September, and predation by raptors increased in sites where more poults were released and so densities were high [130]. In contrast, industry figures show that 25 to 30% of poults are routinely lost from the time they have free access to and from their release pen to the start of the shooting season, mostly due to predation by foxes [44].

A study of radio-tagged pheasants on six estates in south-east England found that 29.8% were shot on-site and 5.0% off-site, i.e., 34.8% were shot, 35.1% were lost to predators, 13.3% died from accidents and unknown causes, and 16.6% were still alive at the end of the shooting season [124]. Foxes were the major predator: 19.2 ± 4.0% were lost to foxes before the start of the shooting season, and a further 15.9 ± 1.9% were killed during the shooting season, with predation levels significantly higher on estates with low-level predator control (59 ± 4.7%) compared to estates with high levels of predator control (30 ± 5.3%). During the shooting season, many of the pheasants believed to have been predated had probably been shot and either killed or wounded but not recovered by the shoot [125].

There are also high levels of fox predation after the shooting season, i.e., during their breeding season. Between 1992 and 2013, 811 hen pheasants at seven different sites were caught and radio-tagged in February or March. Of these, between 20% and 71% (average 46 ± 6.6%) were predated between mid-March and mid-July before, during and after nesting. Foxes were estimated to take over 95% of all predated adult hens [125]. There were similar high levels of predation on nests: of 534 monitored nests, incubation success was 34 ± 3.7%: the most common outcome was predation (40 ± 4.1%) [125]. Of 450 monitored nests, 43% were lost to predators: foxes accounted for 33% and corvids 20% of predated nests, but these were conservative estimates because the species of predator could not be confirmed in a third of cases [131].

Thus a number of studies have shown that, across the year, around 40% of released gamebirds, and possibly more, are predated by foxes. As a conservative estimate, I have assumed that, of the 35.8 million kg of surplus non-native gamebirds (33.2 million kg of pheasants, 2.6 million kg of red-legged partridges) released in Britain each year but not shot, 14.3 million kg (40%) is lost to predation by foxes. With each fox requiring around 180 kg of meat each year, this level of predation will support 80,000 foxes for a whole year eating nothing but surplus gamebirds.

It has long been known that released gamebirds are an important food source for foxes. An analysis that collated data from 36 food studies undertaken between 1951 and 1997 estimated that a typical fox family group of 1 adult male, 1.5 adult females and 4 cubs consumed between 49 and 248 adult pheasants in arable landscapes, 82 to 169 adult pheasants in pastoral landscapes, 12 to 34 adult pheasants in marginal uplands, and 4 to 126 adult pheasants in the uplands [80]. To put this into perspective, based on the range of estimates of the number of pheasants consumed by foxes, and assuming a weight of 1 kg per pheasant to allow for some birds that had not reached full weight [119], in the second half of the last century, between 8200 and 23,300 adult foxes were supported for the whole year eating nothing but pheasants. Since this calculation was based on food studies, it was not possible to determine the proportion that was predated and the proportion scavenged.

While these two estimates of the importance of pheasants in the diet of British foxes used data from different sources, i.e., a food study that included gamebirds that were both predated and scavenged versus estimates of predation rates, the differences are dramatic. There has been a marked increase in the contribution of predation on released gamebirds to supporting the British fox population over the past 20 years. Two factors are likely to have led to this change: the significant decline in rabbit numbers [68], and the substantial increase in the numbers of non-native gamebirds released each year [33,120]. It is hardly surprising therefore that the GWCT concluded that *the large-scale rearing and releasing of gamebirds has probably improved their* [foxes] *food supply*, and that this increased food availability has affected levels of culling on shooting estates [132].

The current welfare standards practiced by the gamebird-shooting industry, whereby between a quarter and a third of all poults are routinely killed by predators (mostly foxes) soon after they moved to a release pen, are unacceptable. It is hard to see how failing to address this issue fulfils a landowner's, shoot manager's and/or gamekeeper's responsibilities to meet the welfare requirements of the animals for which he/she is responsible. All gamebird release pens should be secured to prevent access by avian and mammalian predators and scavengers. Similarly, once poults are allowed to explore the area outside their release pen, they need to be contained within a predator-proof area until they are able to fly and roost. As well as addressing a significant animal welfare issue, this would reduce the amount of supplementary food that the gamebird-shooting industry provides for predators and scavengers.

Predation on released gamebirds supports 80,000 foxes a year

The impact of scavenging on fox numbers

What happens to the 60% (21.5 million kg) of the surplus gamebirds released into the British countryside each year that dies of causes other than predation by foxes is less clear. Vertebrate scavenging ecology remains an understudied area of science, especially with respect to how biotic and abiotic factors influence scavenger community composition. Carrion is particularly important as a food source for medium-sized predators in winter [133]: the availability of a small carcass has a significant impact on the nutritive status of a carnivore the size of a fox, so much so that a substantial fraction of the energy sequestered by mesopredators (i.e., medium-sized predators which typically hunt smaller prey items) may originate from carrion [134].

While there is no simple means to quantify the biomass of carrion available in most ecosystems [135], it has been recognised for some 60 years that the availability of sheep carrion in the British uplands plays a major role in supporting foxes and other predators through the winter, and that reducing the availability of sheep carrion through better management would be an effective, and long-lasting, method to reduce the number of foxes [136]. In mid-Wales, in the 1970s the minimum quantity of carrion available from sheep and lambs (excluding afterbirths) was at least 40 kg per fox per annum, and it could have been considerably higher [76]; this single source of food constituted a minimum of 25% of the fox population's annual food requirements.

This is comparable to an estimate of the amount of sheep carrion available on two estates in west Scotland [137]. Sheep carcasses were scavenged more quickly from November to March, and in July, when carrion was scarce, than in April to June, when carrion was abundant and much of it was not eaten. Assuming that the foxes ate nothing but sheep carrion, the available carrion could have sustained the entire local fox population except in years when foxes were particularly abundant [137]. A more recent study estimated that there was 0.61 kg of sheep carrion per hectare of Scottish grouse moors, although it was not possible to estimate the time period over which this carrion became available [38].

21.5 million kg of gamebird carrion enters the ecosystem each year and could support up to 120,000 foxes

Some of the 21.5 million kg of non-native gamebirds that I have categorised as 'carrion' may actually be predated by species other than foxes. However, the vast majority will be gamebirds that die from a variety of causes, including disease, road deaths and shooting injuries. There are no suitable data to quantify the different causes of mortality. Pheasants are disproportionately likely to be reported killed on UK roads. There are peaks in road deaths from September to November as the birds disperse from their release pens and in February, when supplementary feeding ceases on many estates [138].

A number of studies have shown that habitat affects which species are most likely to access carrion; carcasses in cover favour mammalian scavengers, whereas carcasses in the open are more likely to be accessed by avian scavengers [139]. So it is likely that avian scavengers will remove more carcasses from roads and similar open habitats, especially since pheasants roost at night and most road deaths occur during the day.

Another major source of carrion is birds shot and wounded but not recovered. A shot bird may fly 200 to 300 m before landing, especially on shoots that provide high-flying birds [140], i.e., birds that pass over the guns at a height of around 40 yards (36.6 metres) or more [141]. Wounded pheasants may also run up to 100 m after hitting the ground [142]. So, if birds are shot close to the boundary of another property, a substantial number are likely to fall on a neighbour's land and not be recovered by the shoot. This is a significant welfare issue: there should be a ban on shooting gamebirds within 300 m of the boundary of a property to ensure that all wounded birds are recovered as quickly as possible. This would have the added benefit of reducing the amount of carrion available to predators and scavengers.

Furthermore, there are no quantified data on wounding rates in pheasants and partridges. Since there is no requirement for a practical test before being allowed to shoot live quarry in Britain, and several shots are taken per pheasant killed, especially on 'high' bird shoots [142], wounding rates are likely to be high. This is a significant welfare issue.

Various studies have shown that carrion resources are used extensively by vertebrate scavengers: in one study, they removed 100% of small rodent carcasses left above ground in winter and spring, and 64% of carcasses placed in grassland and 90% of those in woodland habitats [143] in summer and autumn. This was because microbial decay in summer and autumn reduced the time available for scavengers to find the carcasses. Another study found that 76% of chicken heads placed on roads in an urban environment were removed by scavengers, mostly corvids, within 12 hours [144]. Since even such small carrion items are located and utilised, it is highly likely that most gamebird carcasses will be found and consumed to some extent by mesopredators, although at some times of the year it is also probable that so much gamebird carrion is available locally that not all of it will be consumed by scavengers.

The 21.5 million kg of gamebird carrion made available by the gamebird-shooting industry each year could support an additional 120,000 adult foxes for an entire year, i.e., the gamebird-shooting industry provides enough supplementary food to support between 80,000 (based solely on predation rates) and 200,000 foxes (assuming that all the gamebird carrion was also eaten by foxes). This is a ten-fold increase on the estimate of 8200 to 23,300 foxes (see above) that were supported by predating and scavenging pheasants in the second half of last century. Of course, not all of the available carrion will be consumed by foxes, but over the last 20 years there has clearly been a very substantial increase in the impact of the gamebird-shooting industry on the numbers of foxes in Britain.

This, and the other forms of supplementary feeding undertaken by the gamebird-shooting industry that I discuss below, have a variety of environmental, ecological and conservation impacts across much, if not all, of rural Britain. The impacts of supplementary feeding by the gamebird-shooting industry need to be quantified as a matter of urgency.

The impact of carrion on ecosystem functioning

An accumulating body of evidence shows that supplementary feeding of wildlife (such as by the provision of carrion) is a valuable nutrient resource used by a diversity of vertebrates across the globe [145], and population impacts are becoming widely recognised. For instance, supplementary feeding improves breeding success in the Spanish imperial eagle (*Aquila adalberti*) [146] and, in central England, each householder feeding red kites (*Milvus milvus*) in their garden provided enough food to support 0.12 to 0.26 individuals: this supplementary feeding played a major role in the success of the reintroduction [147]. A recent study has highlighted the positive associations between large-scale gamebird releases and populations of generalist avian predators, and that game management could have an indirect negative impact on some prey species [8].

Hitherto, the functional role of carrion in sculpturing ecological processes has been under-appreciated [148]. Carrion generates top-down and bottom-up feedbacks: predators, live prey, scavengers and carrion are more strongly interconnected dynamically than previously recognised, and these interactions play a central role in governing several key ecological properties of communities and ecosystems. A number of recent studies have shown that providing carrion and other sources of supplementary food alters predator-prey relationships [148,149]. Several studies have demonstrated that there is a higher predation risk associated with high carrion availability because of the increased numbers of predators that it attracts. On Fuerteventura Island (Canary Archipelago, Spain), for example, ground-nest predation occurred more frequently near carrion sources [150]. Similarly, baiting sites used by hunters are hotspots for ground-nest predation [151], and supplementary feeding stations for scavengers such as vultures can disrupt intraguild processes and favour the congregation of predators, increasing predation risk on small- and medium-sized vertebrates [133,152]. So both direct and indirect interactions involving carrion, mediated by mammalian carnivores, are key food-web components that have altered our perspectives of how natural communities are structured and function [149]. Future studies are likely to highlight further the importance of scavenging in under-standing community structure and dynamics [153].

Because carrion and other sources of supplementary food may have substantial impacts on the population dynamics of predators and their prey [148], there is growing concern about the use of supplementary feeding for wildlife conservation and similar purposes. Where this is essential, there are a number of principles that should be followed to minimise the risks of adverse environmental impacts. For instance, supplementary food should only be provided in small amounts for short periods at unpredictable times and places to prevent the aggregation of mesopredators, and supplementary feeding should be avoided during periods of migration and pulses of new recruits [154]. The provision of predictable sources of substantial amounts of supplementary food in the form of easy-to-catch and dead gamebirds contravenes these basic principles.

The role of scavenging in the spread of disease

There is growing concern about the role that scavenging plays in disease dynamics; scavengers, both specialist and facultative, are vectors of a number of diseases of agricultural importance because they ingest infected material and shed the disease agent in new areas [155]. Recent work has shown that increased scavenging activity can also increase the risks of disease transmission between mesopredators and other species [156-158].

Abundant resources such as carrion support higher densities of predators and scavengers, and can cause wildlife to move into less favourable environments, both of which can amplify host stressors [159]. Even moderate levels of provisioning can lead to significantly different outcomes in terms of pathogen extinction or maximizing prevalence [160]. A classic example of the risks of disease transmission associated with supplementary feeding is the role of garden bird feeders in the emergence of virulent pathogens such as *Mycoplasma gallisepticum* and *Trichomonas gallinae* in songbirds [161-164].

It is important, therefore, that the amount of carrion generated by the gamebird-shooting industry is reduced dramatically to reduce the associated risks of disease spread [158,165,166]. Furthermore, as I discuss below, surplus gamebirds pose a particularly high risk of disseminating antibiotics and other pharmaceuticals in the environment, and thereby enhancing levels of antibiotic resistance in wildlife [167].

Predating and scavenging sick and dead gamebirds increases the risks of disease transmission

The use of antibiotics by the gamebird industry

Although BASC promotes the sale and consumption of game meat *Because it is raised in the wild, without any drugs or chemicals* [42], this is far from the truth: weight for weight, more antibiotics are used to raise gamebirds than in any other UK terrestrial food-production system. I have not included aquaculture in the following analyses, since the levels of antibiotic use are very low compared to other forms of large-scale protein production [117].

Rearing and releasing large numbers of non-native gamebirds in close proximity outside their native range poses a high risk of disease transmission, and managing this risk requires extensive medication: this includes a variety of antibiotics, wormers and anticoccidial products [168,169]. The extent to which gamebirds are medicated prior to release is unregulated [19], although there is now a voluntary code whereby the gamebird rearing and shooting industries are trying to reduce their heavy reliance on antibiotics.

Antibiotics are used to treat sick gamebirds and prevent the spread of disease while they are being captive-reared, and in the period following their release. As two examples of the use of antibiotics to treat gamebirds, hexamitiasis is caused by the protozoan *Hexamita meleagridis* (*Spironucleus*) and is probably the major cause of mortality in gamebirds during the rearing and immediate post-release period [170]; it has been treated with antibiotics administered in liquid form and in feed [168]. One of the most challenging diseases of gamebirds is mycoplasmosis, caused by *Mycoplasma gallisepticum*. While this disease is seen most often in adult birds, it can cause significant levels of mortality in chicks from 7 to 14-days old onwards [171]. Strains specific to gamebirds can cause blindness and ultimately death, and some strains do not respond to antibiotics: culling rather than treating birds is now being promoted to reduce the unnecessary use of antibiotics [172]. Various antibiotics have been used to try to treat mycoplasmosis in pheasants and, with increasing rearing densities and associated health problems, the use of antibiotics in gamebirds increased through the 1990s and 2000s, exacerbated by the near absence of more specific medications for treating common and debilitating diseases of gamebirds [173].

To put the gamebird industry's reliance on antibiotics into perspective, in 2019 the UK produced 307,500 tonnes of sheep meat, 914,400 tonnes of beef [174], 960,100 tonnes of pig meat [175] and 1,937,000 tonnes of poultry meat [176], a total of 4,119,000 tonnes. To produce a rough estimate of how much meat is produced by the gamebird-shooting industry, I used the GWCT's estimate that 15 million pheasants (16,500 tonnes, assuming an average adult weight 1.1 kg) and 4.6 million red-legged partridges (2,254 tonnes, assuming an average adult weight of 490 g) were shot in the 2016/2017 season [33].

However, not all the gamebirds that are shot enter the food chain. In 2005 BASC stated that 80% of birds shot in the UK were exported to the continent [46]; this is no longer the case. In the 2017/2018 season, a survey of 566 shoots found that game dealers only took 48% of a shoot's game; the prices being received for pheasant carcasses had fallen by 60% over the preceding six years, and they fell by a further 35-38% between the 2016/2017 and 2017/2018 seasons. The lack of demand for shot gamebirds was such that, in the 2017/2018 season, 46% of the surveyed shoots were supplying their game dealer free of charge, and 12% were paying their game dealer (typically 20 to 30p per bird) to collect the carcasses [177]. Many other carcasses were given away. The ultimate fate of the gamebirds that are shot but do not enter the human food chain is unclear, but there are regular reports of surplus pheasants being dumped, e.g. [178], or being used as bait in 'stink pits' [38].

Even if all the gamebirds that are shot and recovered (a total weight of 18,754 tonnes) enter the human food chain (and this would appear to be an over-estimate, possibly a substantial overestimate), this would still only constitute <0.5% of the total meat production in the UK. Yet the gamebird industry uses a disproportionate amount of antibiotics: in 2016, the industry used 22.3 tonnes of active antibiotic, which was 7.5% of that used in all the UK livestock sectors combined [179]. Of this 22.3 tonnes of antibiotics, 75% was administered to gamebirds through feed and 25% via drinking water [117].

Because of the extremely high reliance on antibiotics by the gamebird industry, targets were set to reduce their use by 25% in 2017 [117]. Antibiotic use was reduced by 36% between 2016 and 2017, and a further 24% between 2017 and 2018 [172]. By 2019 the total antibiotic use in the UK gamebird sector had been reduced to 10.4 tonnes of active ingredient, and a target was set for the industry to reduce its use of antibiotics in 2024 to 6.24 tonnes [180]. It is unclear whether this target is achievable with the current high numbers of gamebirds being reared and released, the unsuitable climatic conditions for pheasants and red-legged partridges in Britain, and current production systems. In 2019, for instance, antibiotic use by the gamebird sector showed a 7% increase compared with 2018; this was associated with the very wet weather during the rearing season and an unprecedented need to treat sick gamebirds infected with *Mycoplasma* [181]. Despite current targets to reduce their reliance on antibiotics, the gamebird industry remains a disproportionately high user of antibiotics because gamebirds are reared outside at high densities where they are exposed to environmental challenges from both disease and the weather [172].

Gamebirds are also highly vulnerable to disease after they are moved to their release pens [173], and this is also a period of extremely high predation risk. This is of particular concern because various studies have shown that 25 to 30% of poults are lost from the time they have free access to and from their release pen to the start of the shooting season, mostly due to predation by foxes [44]. So medication of gamebirds during rearing and the period after release will affect the disease susceptibility of the non-game species that consume gamebird carcasses containing veterinary antibiotics. Antimicrobial resistant bacteria (ARB) have been detected in buzzard and other raptor faeces and foxes, both common predators/scavengers of gamebirds [29].

Unfortunately, there are no data on the seasonal use of antibiotics by the gamebird rearing and shooting industry, which complicates efforts to calculate the extent of the environmental risk. There is a 28-day withdrawal period after the last treatment with antibiotics before gamebirds destined for human consumption can be shot [182], so the use of antibiotics by the gamebird industry is largely finished by late summer [117]. However, there are no withdrawal periods to prevent the environmental dissemination of antibiotics, even though this is a significant risk to both wildlife and human health. The gamebird industry has been highly reliant on the use of antibiotics for at least the last 20 years, and the large annual losses of poults to foxes and other predators means that a significant proportion of the many tonnes of antibiotics administered to gamebirds each year will have been consumed by predators and scavengers.

Many shot gamebirds are either dumped or, as here, used a bait in 'stink pits'

It is easy for antimicrobial resistance (AMR) to spread between animals in intensive livestock production systems, and this can be exacerbated if biosecurity is inadequate. AMR to a range of commensal microorganisms has been detected in samples from pheasants long after medication [29]. This poses a significant risk because most other animals destined to enter the human food chain are confined and under close veterinary supervision when being medicated. This is not the case for gamebirds once they are allowed free access to and from their release pen. So even if the gamebird rearing and shooting industry achieves its targets to reduce its reliance on antibiotics, substantial amounts of antibiotics will still enter the environment each year under the low levels of biosecurity inherent in current rearing and releasing practices.

Increased AMR in human and veterinary medicine has reached alarming levels in most parts of the world and is a significant emerging threat to global public health and food security [183]. The agricultural use of antibiotics has been a major factor contributing to the development of resistant organisms that result in life-threatening human infections [184], and wild animals foraging in the human-influenced environment are colonised by ARB. The spread of ARB in wildlife is a major concern [185], not least because this impairs the immunity of wildlife [159]. There are frequent reports of multidrug-resistant bacteria in wild birds and mammals [186]; this is likely to have unpredictable and wide-reaching consequences for human and wildlife health [187], and the disruption of host-associated microbial diversity is a serious threat to wildlife populations [188].

Given the current crisis with AMR, any non-essential use of antibiotics should be halted as a matter of urgency to prevent the dissemination of resistance genes [189]. The lack of control over, or information about, the ultimate fate of the antibiotics used in the gamebird industry is therefore of great concern.

Feeders and drinkers used to administer antibiotics to gamebirds are accessible to wildlife

The use of other pharmaceuticals by the gamebird industry

A variety of other diseases affect gamebirds, and many of the issues relating to the use of antibiotics also apply to the other pharmaceuticals used by the gamebird industry. Some of the more common parasites and diseases of concern include coccidiosis, rotavirus, trichomaniasis, and a variety of parasitic worms [46]. The National Animal Disease Information Service (NADIS) has produced an extensive list of the parasites and diseases that can infect gamebirds immediately before and after release, and the pharmaceuticals that are used to treat these infections [168,169].

For example, Avatec is usually included as a standard in all feed for gamebirds up to 13 weeks old to treat coccidiosis, and Flubenvet is incorporated in feed to treat gapeworm *(Syngamus trachea)* infections [168]. Gapeworm is a highly pathogenic parasite of many species of birds, and is arguably the most economically important parasite of released pheasants. Infection can occur either directly by ingesting worm eggs or infective larvae, or indirectly by ingesting an infected invertebrate, most commonly an earthworm. In pheasant release pens, gapeworm eggs reach high concentrations around feed hoppers, and infected birds lose on average 26% of their body condition, which may have implications for the survival and reproductive success of released pheasants [190].

From July to September, when gamebird poults are in and around their release pen, they are treated with in-feed medication or medication in the drinkers that are placed in the pen and the surrounding area [19,191]. So these pharmaceuticals can be consumed by a variety of wildlife, either directly by accessing feed or drinkers, or indirectly through predation and scavenging on carcasses. This poses a significant environmental risk: while pharmaceuticals are now widespread in the environment, the exposure risk to higher vertebrates is largely unknown [187]. It is important to achieve ecosystem-level risk assessments that include evaluations of direct, e.g., toxic, and indirect, e.g., via trophic webs, effects associated with pharmaceutical exposure [192]. Currently there are no quantified data on pharmaceutical use by the gamebird rearing and shooting industry, the amount of these pharmaceuticals that enters the environment, or the impact on wildlife. The heavy use of antibiotics and pharmaceuticals by the gamebird industry involves public, animal and environmental health [192], i.e., it is a 'One Health' issue.

Where it is necessary to administer antibiotics and/or other pharmaceuticals for the welfare of released gamebirds, the birds requiring medication should be moved to a secure pen where they cannot be accessed by predators, and where other wildlife cannot access any food or water supplies used to administer the antibiotics or other pharmaceuticals. Gamebirds should only be returned to a release pen after the appropriate withdrawal period for the antibiotics or pharmaceuticals that were administered.

Free-living pheasants also suffer high levels of mortality due to disease. Of 50 hen pheasants radio-tagged early in 2011, 33 died between 1st March and 31st August. All were in good condition when tagged in March, but the majority (63%) of the birds subsequently found dead were in an extremely poor or emaciated condition due to a high incidence of parasites and kidney damage, probably caused by a coronavirus [193]. Coronavirus damages the kidneys of birds and can result in the sudden mortality of adult birds. It is typically seen when the birds are under stress, such as during bad weather and during the breeding season. Infected birds can shed the virus for several months, usually through the respiratory tract and infected droppings [194]. The role of non-native gamebirds in spreading parasites and diseases to native wildlife is unknown: coronaviruses are widespread in a variety of wild birds and *interspecies transmission poses a great risk of spreading, mutation and the emergence of new strains of the viruses* [195].

The environmental impact of the lead shot used by the gamebird-shooting industry

As well as depositing billions of lead pellets in the environment each year, the gamebird-shooting industry leaves vast amounts of plastic waste

Lead-based ammunition is one of the most prominent and controllable, but largely unregulated, sources of lead exposure in wild animals [196]. Even though the use of lead-based ammunition is another One Heath issue [197], there are no published data on the amount of lead shot used by the gamebird-shooting industry, or the associated ecological and environmental impacts. However, it is possible to provide an approximate estimate of the amount of lead shot used to kill pheasants and red-legged partridges each year, although the calculations I present below only apply to the lead shot used specifically to kill these two species of non-native gamebirds. It is not possible to estimate the amount of lead shot that is used by gamekeepers and others involved in the industry to kill the wildlife that they deem to be detrimental to their interests. Thus the estimate below should be viewed as a minimum.

The number of pellets in a shotgun cartridge depends on the size of the shot and the chamber length of the shotgun. For 12-bore shotguns (which are generally used for shooting gamebirds), the most common loads (i.e., the total weight of the shot) are 28 g, 30 g, 32 g or 36 g; the preferred shot load may vary [198]. For convenience, I have assumed that a 32 g shot load was used throughout the season.

For this calculation I have used the GWCT's estimates of the numbers of pheasants and red-legged partridges shot in the 2016/2017 season, i.e., 15 million pheasants and 4.6 million red-legged partridges [33]. On most shoots the acceptable shot-to-kill ratio is three to one [199]. However, this may be substantially higher on some shoots, particularly those that provide high-flying pheasants [200], where the shot-to-kill ratio may be as high as eight to one [201]. Also,

Each year around 78.4 million 12-bore shotgun cartridges are used to kill 19.6 million pheasants and red-legged partridges

since there is no test of shooting proficiency in Britain, shooting ability varies greatly. So, for this calculation, I have assumed there were four shots per kill, i.e., 78.4 million 12-bore shotgun cartridges were fired to kill 19.6 million non-native gamebirds. At 32 g of lead shot per cartridge, this would mean that circa 2500 tonnes of lead shot is expended each year to kill pheasants and red-legged partridges. Most lead from a shotgun cartridge falls into the environment, even if the target is hit [202] and, since most driven pheasant and red-legged partridge shoots have a limited number of shooting stands that are reused both within and between years, some tens of billions of individual lead pellets are deposited over limited areas each year [202].

It should also be remembered that this is a conservative estimate. One study suggested that between 2500 and 6700 tonnes of lead are fired at gamebirds annually [202] (i.e., not just pheasants and red-legged partridges), and ammunition-derived lead is the major source of widespread lead exposure. The risk this poses to wildlife has long been recognised: the first UK record of a bird poisoned following lead gunshot ingestion was a pheasant, in 1876 [202]. This is hardly surprising: each year, gamebird shoots are depositing billions of lead pellets in the same areas where pheasants are being reared for shooting, and hence as a potential source of human food. Pheasants are vulnerable to shot ingestion [203]; about 600,000 terrestrial gamebirds are likely to have ingested gunshot at any one time, and many more throughout the shooting season [202].

One of the main routes by which many wild birds and mammals are exposed to lead ammunition is by ingesting gunshot from gamebirds that were not recovered by the pickers-up [202]; the high levels of lead in the tissues of gamebirds are a significant risk to predators and scavengers [204]. It is likely that hundreds of thousands of gamebirds potentially contaminated with ammunition-derived lead enter the food supply of wild predators and scavengers each year [202]: gamebirds that die of lead poisoning pose a particular risk to other wildlife. Foxes can be used as sentinels of lead-levels in the environment [205,206], and a variety of corvids and raptors have been affected by lead intoxication from scavenging carcasses shot with lead-based ammunition [197,207-211]. This can have population-level effects [212].

Nine UK shooting and rural organisations issued a joint statement on 24th February 2020 saying that they wished to see an end to both lead and single-use plastics in ammunition used to shoot all live quarry with shotguns within five years [213]. However, 179 out of 180 pheasants for which shotgun pellets were recovered in the 2020/2021 shooting season were killed using lead ammunition [214]. The position statement issued in 2020 has no legal force, and evidence from shooting wildfowl suggests that it is unlikely to be effective. Even though it has not been legal to use lead ammunition to kill wildfowl in England since 1999, compliance studies found that between 68% and 77% of wild duck carcasses bought from game dealers up to fifteen years later had been shot using lead ammunition [215,216].

The same shooting stands are used repeatedly, leading to high levels of lead pollution

The impact on wildlife posed by the use of dogs on gamebird-shooting estates

The other issue relevant to this review is the heavy use of dogs on game-shooting estates, and the adverse impact this has on wildlife.

Since the impacts of dogs on the behaviour of wildlife has been reviewed already [217], this will not be discussed further.

The frequent use of dogs on shooting estates has a range of negative impacts on wildlife

Dispersal and food requirements of released gamebirds

When assessing the impact of released non-native gamebirds on or near EPSs and SSSIs, Defra's focus appears to have been on how far pheasants and red-legged partridges disperse from their release sites [1]. A study in Devon from mid-August to 1st October found that pheasants slowly moved away from their release pen, but almost all the birds remained within an area of 500 m radius up to 1st October. At the end of the shooting season (1st February), only a small proportion of the rapidly dwindling pheasant population were reported to make any further significant dispersal movements [29]. Similar dispersal distances were recorded in red-legged partridges studied at three sites in East Anglia and three in southern England: the average final per-bird dispersal distance from their release pen was 408 m [126]. These data probably explain why Defra concluded that the negative effects of gamebird releases on EPSs and SSSIs tend to be localised, with minimal or no effects beyond 500 m from the point of release [1].

There are a number of problems with this conclusion. Although 500 m seems to be a reasonable estimate of the dispersal distances of recently-released pheasants and red-legged partridges up to and during the shooting season, when they are still being fed by the shoot to deter them from straying, there are few data on dispersal distances, and the ecological impacts, of released gamebirds after the shooting season. This is when many shoots, especially non-commercial shoots, stop feeding their released birds, and pheasants start to disperse more widely, as indicated by an increase in pheasant road deaths in February [138]. Late winter and spring is the critical period for pheasants and red-legged partridges: by January most natural food supplies have been exhausted [218], and a considerable part of this resource depletion is likely to be due to the biomass of non-native gamebird releases. It is also why, after the end of the shooting season, farmers can experience significant losses of recently-drilled crops such as oil seed rape and wheat to both pheasants and red-legged partridges [219].

Game-cover crops are widely used to feed released gamebirds and help stop them from straying: the gamebird-shooting industry claims that these are also an important resource for wild bird populations [220]. However, this is another form of supplementary feeding that does not conform to the basic guidelines for conservation feeding [154], and hence will have a number of ecological effects that extend beyond the 500 m 'impact zone' considered by Defra [1]. It is also unclear whether it would be necessary to plant supplementary food crops for the benefit of Britain's wild birds without the ecological pressures posed by the release of some 47 million pheasants and 10 million red-legged partridges each year. *As a rule of thumb, eight tons of feed are required per 1000 pheasants released to support them from release to the end of the shooting season. This is about 2 tons of grower's pellets to take the birds to 12 to 14 weeks of age, followed by about 6 tons of wheat* [221]. So the estimated 47 million pheasants released each year require 376,000 tons of feed, of which 94,000 tons is grower's pellets and 282,000 tons is wheat. To put this into perspective, the five-year average for wheat production in the UK is 15.1 million tonnes [222], and so the shooting industry uses nearly 2% of the UK's annual wheat production to rear pheasants for shooting.

While there are no quantified data on the amount of wild seed eaten by released gamebirds each year, pheasants were estimated to eat up to 12,000 kg of invertebrates each day in early summer, when free-living pheasant numbers are at their lowest, and up to 150,000 kg of invertebrates each day in September [223]. This is a substantial pressure on rural invertebrate populations, and it is hard to believe that this is not having a significant negative impact on food webs. Furthermore, these estimates of the daily invertebrate consumption were made when the number of pheasants released each year was believed to be around 10 million lower than the GWCT's most recent estimate [33].

Industry advice is that, for welfare reasons, sufficient feed should be provided for released birds that have survived to the end of the shooting season until *adequate natural food* is available [47]: this recommendation does not define what constitutes adequate natural food, nor does it consider the continuing impact of competition from released gamebirds on native wildlife. Industry advice is that pheasants should be fed with wheat supplied in hoppers until at least mid-June and, for partridges, hoppers should be kept full until the end of May and then slowly allowed to run out [218]. Feeding after the end of the shooting season is also said to reduce the dispersal of gamebirds in late winter [224]. The relative proportion of seeds, green plant material and invertebrates in a pheasant's diet will vary with habitat and season. An adult pheasant requires the equivalent of about 500 g of dry food per week [225], and so the 8 million pheasants that survive the shooting season [119] will consume substantial amounts of natural food, especially on those shoots that do not continue supplementary feeding. This is another significant ecological pressure.

However, feeding released gamebirds after the shooting season also has several adverse ecological impacts. It maintains a population of easy-to-catch prey through the winter and spring [226,227], and predation rates remain high, e.g. [125,131]. This will attract a range of predators and scavengers, as does the carrion from gamebird carcasses. This is the bird breeding season and, as I have shown above, is likely to have an impact on ground-nesting birds and other species on EPAs and SSSIs over a substantial area beyond the 500 m 'impact zone' considered by Defra [1].

Winter feeding of gamebirds has other negative ecological impacts. A study on three farms in southern England, where supplementary feeding of gamebirds was part of the existing management programme, recorded non-target species (i.e., not gamebirds or song birds) in 54% of over 160,000 photos taken at game feeders in early and later winter 2012 and 2013. Non-target species consumed 67% of the grain provided for gamebirds [224], and rats were a major beneficiary of this supplementary feeding [228].

Providing supplementary food for brown rats through the winter and spring is a significant conservation issue because rats are also major predators of ground-nesting birds, e.g. [229], and brown rats in agricultural landscapes routinely travel distances greater than Defra's 500 m 'impact zone'. The mean range length for hedgerow-dwelling male rats was 660 m, for females 340 m [230], although substantially longer movements can occur [231]. Supporting large numbers of brown rats through supplementary feeding of gamebirds is likely to have a significant impact on EPSs and SSSIs outside Defra's 500 m 'impact zone'. Supplementary feeding of brown rats on gamebird shooting estates is also likely to provide another source of food that attracts predators.

At the moment there are not enough data to evaluate the positive and negative environmental, ecological and conservation effects of supplementary feeding gamebirds at the end of the shooting season. However, it is likely to have a wide range of impacts and, from a scientific perspective, it is naïve to consider EPSs and SSSIs in isolation from the surrounding countryside. EPSs and SSSIs are not habitat islands, and conservationists have long recognised that the viability of conservation areas is heavily influenced by, if not dependent on, surrounding land-management practices over a range of spatial scales, e.g. [232,233]. Rearing, releasing and shooting non-native gamebirds affects ecological processes over much of lowland Britain, and managing the impacts of gamebird releases on EPSs and SSSIs cannot focus solely on the proximity of releases. The gamebird-shooting industry should be required to develop means to support non-native gamebirds that do not affect other wildlife populations, or to remove their surplus gamebirds at the end of the shooting season.

Food availability, red fox population dynamics and the conservation benefits of widespread predator control

Surplus feeding by the gamebird-shooting industry has a wide range of ecological impacts

There is a lot of information on the impact of supplementary feeding and food availability on fox numbers and population dynamics, and how this varies depending on the local predator community. In Spain, for instance, red foxes consumed more carrion, and their abundance was higher, where vultures were absent, i.e., being facultative scavengers, red foxes were able to take advantage of the increased availability of carrion [234]. In Białowieża Primeval Forest in Poland, foxes were present at 86% of wolf kills and followed wolves to feed on their prey remains, and ravens and foxes were generally the first to arrive at a carcass [151,235]. Similarly, supplementary feeding can have a significant impact on red fox population dynamics in British urban areas [236].

Some of the most comprehensive data on the impacts of food availability on fox numbers comes from long-term studies in Scotland: see [237] for a summary. Changes in fox numbers in west Scotland were driven by changes in field vole (*Microtus agrestis*) populations, and over-winter survival of foxes was better in years with high field vole numbers [238]. There was also a decline in the ratio of cubs to adults killed, suggesting that fewer cubs were born, possibly because the fox population had reached its carrying capacity [239,240]. Similar data exist from other parts of the red fox's range. In Sweden, vole abundance positively affected red fox litter size and negatively affected fox survival [241], and fox mortality rates increased in response to food reduction [242]. A number of other studies on various species of foxes and wild canids have shown this close relationship between food availability and reproductive success [243], and is a major factor influencing the numbers of red foxes and other generalist predators.

There has been long-term resistance by the shooting industry to end their large-scale killing of native predators

There has been long-term resistance by the shooting industry to suggestions that they should end their large-scale killing of predators and scavengers. According to the GWCT, *Culling foxes to protect vulnerable species from predation on shooting estates is a defensible practice if it works* [244]. This assertion raises several important issues. First, the gamebird-shooting industry is currently providing enough supplementary food to support between 80,000 and 200,000 foxes a year. Having reinforced the British fox population to such a substantial extent (see the monthly estimates of the number of foxes in Britain), the gamebird-shooting industry then killed an estimated 120,000 ± 110,000-130,000 (95% CIs) foxes in 2004, 66,000 ± 59,000-73,000 foxes in 2012 and 89,000 ± 76,000-100,000 foxes in 2016 [33]. At best, it is incongruous for the gamebird-shooting industry to provide enough supplementary food to support a minimum of 80,000 foxes a year, and for the GWCT to then argue that killing the same number of foxes each year is a defensible practice.

Second, the ecological consequences and benefits of widespread predator killing have not been proven: to be *a defensible practice*, there needs to be clear evidence that the large-scale killing of (predominantly) native species has significant conservation and/or management benefits. I have already shown that killing large numbers of predators and scavengers each year is not preventing significant losses of released gamebirds to predators, so on this basis alone it is hard to see how the gamebird-shooting industry can justify continuing to kill large numbers of native birds and mammals each year.

The key measure of success for any predator removal operation should be whether widespread predator control helps to reduce or reverse population declines, or lead to population increases, of potentially vulnerable species. Yet despite killing large numbers of avian and mammalian predators in the UK each year, between 1995 and 2018 the numbers of curlews (*Numenius arquata*) and lapwings (*Vanellus vanellus*), both ground-nesting birds, declined by 48% and 43% respectively [245]. Similarly, birds feature in 7.8% of badger faeces and stomachs [246], and badgers are regularly recorded in camera-trapping studies of predation on ground-nesting passerines and waders [247-249].

Over a period of up to five years, badgers were killed in an area of 567 km^2 of south-west Britain: the badger population was reduced to less than half pre-cull numbers [250]. However, there was little evidence for positive or negative effects of badger removal on the growth rates of ground-nesting birds [250]. This could, in part, have been because badger removal is reported to lead to increases in the numbers of foxes [97] and hedgehogs (*Erinaceus europaeus*) [251], both of which are also predators of ground-nesting birds [103,252-254].

Attempts to control fox numbers to reduce predation on capercaillie (*Tetrao urogallus*) and black grouse (*Tetrao tetrix*) in Abernethy Forest, Scotland were also unsuccessful, but pine marten numbers increased and they became important predators of artificial nests [255]. There is a significant amount of data on such mesopredator release effects on conservation, e.g. [256], and on the complexity of trophic changes in predator populations following fluctuations in food abundance, e.g. [257]. A variety of studies have shown that the factors influencing populations of ground-nesting birds are complex and are unlikely to be attributable to a single species of predator [258]. There is also growing evidence that the perturbation effects associated with culling predators (see below) can enhance, rather than reduce, their impacts, e.g. [259].

Millions of birds and mammals are killed each year by the gamebird-shooting industry

The effects of alternative prey species and carrion also have a significant influence on the impacts of red fox predation on ground-nesting birds. For instance, lesser white-fronted goose (*Anser erythropus*) reproductive performance was influenced by fluctuating offspring predation by red foxes, mediated by mainly natural (rodents) and partly anthropogenic (semi-domesticated reindeer) carrion and their effects on food-web dynamics [260]: such factors are likely to be site-specific. There was no clear effect of a decade-long red fox culling programme on lesser white-fronted goose reproductive success and survival [260].

Numerous studies have also shown that the widespread killing of foxes has, at best, a temporary effect on fox numbers. A study in Welsh woodlands found that the number of foxes killed was large relative to the estimated resident population, but losses appeared to be negated by immigration, that the more foxes that were killed in winter, the higher the spring population, i.e., it was counter-productive, and that there was no evidence to suggest that culling reduced fox numbers [261]. In central Europe, restricted-area culling was not an effective conservation measure to reduce red fox predation on ground-nesting birds [262]. Even with the levels of culling undertaken for conservation purposes, the red fox can exhibit population increases due to its high intrinsic growth and immigration rates, which produces an unbalanced predator community biased towards red foxes [263].

It is well established that a strong compensatory density feedback acting through immigration allows red fox populations to resist high culling rates [264], and restricted-area culling fails to have an effect on fox abundance through the period that is most relevant to conservation [265,266]. A modelling study concluded that effective control of fox populations at landscape scales is neither feasible nor practical unless immigration from outside populations is low or can be controlled [267]. A recent modelling study by the GWCT reiterated that variation in the immigration rate broadly matched differences in fox density at the regional level and that the rate of fox removal

There are significant welfare issues associated with the techniques used to kill predators on shooting estates

must exceed the immigration rate for control to have any effect on local density [268]. The GWCT study also reinforced earlier studies that showed that the number of foxes killed is a poor indicator of effectiveness, e.g. [261], and rapid replacement of the foxes that were killed means that intensive culling efforts are required to maintain low fox densities [268,269].

It should also be remembered that the GWCT's analyses were based on just 22 shooting estates, with a minimum size of 2 km^2, even though various studies have shown that predator control only leads to increases in pheasant numbers when practiced at very large scales [270]. Furthermore, the GWCT measured control success against *carrying capacity*, which is a much misused concept [271]: the carrying capacity for a species is determined by a myriad of interrelated, ever-changing biotic and abiotic factors and it *must not be assumed constant, if we are to derive more effective and realistic management schemes* [272]. Crude measures of carrying capacity that fail to consider all the site-specific factors that affect fox numbers cannot provide a realistic measure of the effectiveness of fox population control.

The widespread killing of foxes also has a range of impacts on fox population dynamics that will have adverse environmental, ecological and conservation impacts. Culling promotes long-distance dispersal movements [76], and rural fox populations disperse earlier than those in urban areas [76,273], probably because of the disturbance of fox family groups. A study in the USA, for instance, found that juvenile foxes in rural areas dispersed 23 days earlier than those in urban habitats [274].

When foxes disperse, it is usually a protracted process: early studies used straight-line distances to measure dispersal, whereas recent studies using GPS collars have shown that the actual distances travelled by dispersing foxes are up to five times longer than straight-line distances. This is because foxes explore different areas and often return to settle in a place they explored earlier [275,276]. Killing foxes is likely to enhance dispersal distances, since the distance moved by dispersing foxes is negatively associated with population density [277]. Also, juvenile foxes emigrated more frequently from food-rich than food-limited habitats, whereas adult foxes showed the opposite trend. So extensive supplementary feeding by the gamebird-shooting industry will have a significant effect on the movement patterns of the British fox population [242].

There are a number of other population consequences of killing foxes. It has long been established that, the higher the culling pressure, the bigger the litter size and the greater the proportion of vixens that breed [77,278]. However, while killing foxes increases cub production, it does not have a permanent effect on population size [279], and a whole range of changes in fox movement and territorial behaviour are associated with population declines [280,281]. Such perturbation effects have long been recognised as having significant effects on the social group structure, movement patterns, population dynamics and disease spread in badgers, e.g. [282-286]. Foxes can also live in complex social groups comparable in structure to those of badgers [287-290], so it is hardly surprising that similar perturbation effects are associated with killing foxes. Furthermore, there is a cumulative effect: the 35.8 million kg of surplus gamebirds released by the gamebird-shooting industry each year, and the population vacuum following predator-killing operations, both attract predators and scavengers to an area.

Conclusions

Britain is the only European country that offers the opportunity to kill large numbers of released gamebirds in a single day. About a third of all the clients paying for released gamebird shooting in Britain come from overseas [4], and the number of birds killed per day/per gun is a major factor underpinning the scale of the industry. The environmental, ecological and conservation impacts of shooting non-native gamebirds extend across most, if not all, of lowland Britain, with the greatest impact in England. The scale of the industry has wide-ranging environmental, ecological, conservation, public health and animal welfare impacts. Rather than focus solely on effects within a limited 'impact zone', Defra needs to quantify the different spatial scales at which the gamebird-shooting industry affects EPSs, SSSIs, functionally linked land and other conservation interests in England.

The apparent lack of quantified data on the scale of the gamebird rearing and shooting industry in Britain is, at best, surprising. This poses a number of concerns, not least when dealing with potential outbreaks of bird flu and other infectious diseases in gamebirds [291], and the low levels of biosecurity associated with rearing and releasing non-native gamebirds, especially the range of species of wild birds and mammals that have free access to pheasant and red-legged partridge pens.

Both ring-necked pheasants and red-legged partridges are introduced species. Although the shooting industry portrays them as being naturalised, there is no evidence to support this assertion. A naturalised species *is an introduced species that does not need human help to reproduce and maintain its population in an area outside its native range*. In Britain, both ring-necked pheasants and red-legged partridges are supported by large-scale releases each summer, extensive habitat management (an issue not covered in this review), the killing of large numbers (possibly/probably millions) of (predominantly) native species of birds and mammals, large-scale supplementary feeding throughout much of the year, and extensive reliance on medication. Yet despite these support measures, there has been a long-term decline in 'wild' pheasant populations over the last 30 years [292], and only a small proportion of released gamebirds survive for more than a few months. It is doubtful whether ring-necked pheasant or red-legged partridge populations would persist in Britain in the absence of human support measures.

The game laws are outdated and need to be revised to reflect the status of ring-necked pheasants and red-legged partridges as non-native species. Since neither species is naturalised in Britain, it is unclear why they are not included in Schedule 9 of the Wildlife & Countryside Act 1981.

It is surprising that people who release large numbers of non-native gamebirds have limited responsibility for the consequences of their actions, and why landowners who do not want non-native gamebirds on their land do not have the freedom to remove them whenever they so wish. This problem is becoming more significant because the proportion of 'surplus' gamebirds is rising: from 1960 to 1990, 50% of released pheasants were shot. This declined to 35% by 2015, and has continued to decline thereafter [292]. A fundamental change is long overdue: people who release non-native gamebirds should be required to contain them on their own land. This would have a significant benefit in reducing the widespread environmental, ecological and conservation impacts of large-scale releases of non-native gamebirds.

In September 2019 the Secretary of State for Environment, Food and Rural Affairs decided to undertake a review of the way in which releases of pheasants and red-legged partridges on or near EPSs and SSSIs in England are managed [293]. This was a rapid evidence assessment: according to the government [294], *Rapid evidence assessments provide a more structured and rigorous search and quality assessment of the evidence than a literature review but are not as exhaustive as a systematic review. They can be used to:-*

- *gain an overview of the density and quality of evidence on a particular issue*
- *support programming decisions by providing evidence on key topics*
- *support the commissioning of further research by identifying evidence gaps*

Since the rapid evidence assessment commissioned by Defra was meant to provide *a structured and rigorous search and quality assessment of the evidence*, it is remarkable that the review *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* [29]. There is a substantial volume of peer-reviewed research on the impacts of supplementary feeding and carrion availability on the numbers of mesopredators worldwide. In Britain, industry data show that between 80,000 and 200,000 foxes a year (a substantial proportion of the British fox population) are supported solely by predating and/or scavenging non-native gamebirds. This has a significant impact on predator population and community structure, and food webs more generally, and ecological perturbations such as those caused by the gamebird-shooting industry are associated with a significant increase in the predation pressure on ground-nesting birds.

It is also remarkable that Defra concluded that *The negative effects of gamebird releases on protected sites tend to be localised with minimal or no effects beyond 500 m from the point of release* [1]. Had a *structured and rigorous search and quality assessment of the evidence* assessed all of the relevant scientific literature, it would have shown that the environmental, ecological and conservation impacts of the gamebird-shooting industry are wide-ranging, and that these affect EPSs, SSSIs and functionally linked land at a range of spatial scales. While the available data suggest that 500 m is a reasonable estimate of the pre-shooting season dispersal distances of released gamebirds, i.e., when they are still being supported by the shoot, it does not address the large-scale environmental and ecological processes that are affected by the gamebird-shooting industry, and the consequential impacts on EPSs, SSSIs, and conservation more generally. Altering resource availability, either directly or indirectly, increases predator densities above levels that would exist without the subsidy, and subsidized predator populations can have a drastic impact on their prey because food subsidies insulate the predator from declines in prey numbers [295]. Since shooting providers have management responsibilities for about two-thirds of the UK's rural land area, with active shoot management undertaken on about 12% [6], the ecological perturbations caused by the gamebird-shooting industry affect most, if not all, of England, to varying degrees.

Releasing pheasants near to roads, and failing to continue to feed them after the end of the shooting season, causes a number of serious injuries and deaths to motorists each year. This is a significant risk to public health and safety. The carcasses of pheasants killed on roads are a major source of carrion for a wide range of predators and scavengers, particularly avian species. The role of surplus food in influencing corvid population dynamics has been known for over thirty years, e.g. [296], and a recent analysis has demonstrated the population effects on generalist avian predators at a national scale [8].

Even though the BTO's long-term monitoring data show that the British fox population has been undergoing a steady decline for the last quarter of a century, a recent study has claimed that the British fox population has increased during recent decades, and that fox densities in Britain are high compared to other European countries [103]. Unfortunately, these authors failed to identify that a properly structured estimate of the number of foxes in Britain had been published in a leading ecological journal [297]. Instead, they relied on a preliminary report to Defra that claimed there were around 430,000 foxes in Britain; the final version of this analysis (which was published after their paper) reduced this estimate to 357,000 [297], and this revised estimate still had extremely wide confidence intervals. Furthermore, despite reducing their population estimate by some 20% [66], the authors included three important caveats explaining why their revised figure was still a significant overestimate. Conservation policy in Britain should be based on sound science. The best available scientific evidence suggests that fox numbers in Britain are declining, that they are not higher than those in comparable habitats in other western European countries, and that the decline in fox numbers this century following the decline in rabbit numbers would have been more marked had it not been for all the supplementary food provided by the gamebird-shooting industry.

In addition to the environmental, ecological and conservation impacts associated with releasing large numbers of gamebirds each year, the extensive reliance on antibiotics and other pharmaceuticals to rear non-native gamebirds is a significant 'One Health' issue. It is surprising that the sale of the meat from gamebirds is promoted as having been raised in the wild without the use of any drugs or chemicals [42]. Because of their lifestyle, red foxes are a good sentinel for monitoring AMR in the environment and the risk this poses to human and wildlife health [298].

Lead pollution by the gamebird-shooting industry is another significant 'One Health' issue. To put the amount of lead used by the gamebird-shooting industry into perspective, between 1970 and 1985, transport in Britain used an average of 7300 tonnes of lead in petrol each year [299]; sales of leaded petrol declined thereafter, and the sale of leaded petrol was banned in the EU in 2000. Currently, up to 6700 tonnes of lead are fired at gamebirds annually in Britain [202], and billions of lead pellets are deposited in the environment. This figure does not include the amount of lead shot that is used to kill various species of birds and mammals that the gamebird-shooting industry considers to be detrimental to its interests. Despite a joint statement issued by nine UK shooting and rural organisations on 24th February 2020 intended to encourage a voluntary transition to non-lead shotgun ammunition within five years [213], there has been little evidence thus far that this voluntary code is being implemented [214], or that it will be effective. There is little incentive to market lead-free ammunition in the UK without government regulation [300].

Despite having been in place for a decade, current codes of practice have had little if any impact on the environmental and ecological impacts of releasing large numbers of non-native gamebirds on or near EPSs, SSSIs, or on conservation generally. For instance, Defra's *Code of practice for the welfare of gamebirds reared for sporting purposes* [41] states that *All birds should be adequately protected from predators …. When birds are housed or penned, the accommodation should be well constructed and managed and provide protection from …. predators* …. and *The siting of release pens should take into consideration the need to minimise the risk of subsequent harm or injury, for example by predators or vehicles*, yet industry data show that around 30% of pheasants in release pens are lost to predators, and pheasants are disproportionately likely to be killed on roads. These are significant welfare issues.

Various management improvements have been proposed that might reduce the need to release so many pheasants each year. Further research should be undertaken into the requirements for support and adaptation of gamebirds during and after release, as recommended by the Farm Animal Welfare Council in 2008 [39]; progress in this area is long overdue. Recent research on 'enhanced' poults [18,270,301-303] suggests that it may be possible to increase the survival of released pheasants. While this in itself is not a solution to the ecological impacts of rearing, releasing and shooting large numbers of gamebirds in Britain, it may be a useful adjunct to developing predator-proof release techniques. However, any adverse ecological impacts of rearing and releasing 'enhanced' poults need to be fully assessed [270,303]. There are also significant welfare issues associated with rearing large numbers of gamebirds prior to their transfer to release pens: since these have already been described [304], I have not discussed them in this review.

Brown rats are a major beneficiary of supplementary feeding by the gamebird-shooting industry

BASC's *Code of good shooting practice* [47,59] states that *it is fundamental to mark and retrieve all shot game …. Shooting should not be conducted where it will not be possible to retrieve shot game …. Guns must be competent at estimating range and shoot within the limitations of their equipment to kill cleanly and consistently …. Guns must satisfy themselves adequate provision is made for retrieval of the game they shoot …. Shoot managers must ensure that adequate provision is made to retrieve all shot game …. The siting of release pens and feeding of game near highways should be avoided*, and *Game managers should collect and dispose of road casualties where possible*. Yet despite the code of practice, 21.5 million kg of non-native gamebird carrion enters the ecosystem each year, and this supports a significant number of predators and scavengers at densities above those that would occur in the absence of this food subsidy. Furthermore, wounding rates in shooting non-native gamebirds appear to be high, and this is another significant welfare issue.

BASC's *Code of good shooting practice* [47,59] also states that *Shoot managers must ensure they have appropriate arrangements in place for the sale or consumption of the anticipated bag in advance of all shoot days*, yet industry surveys show that there are problems disposing of the large numbers of released gamebirds shot in Britain, to the extent that they are either given away, dumped, or used as bait. The code also says that *Sufficient feed for released birds remaining after the end of the shooting season should be provided until adequate natural food is available, normally to the end of May*, yet many shoots do not provision released gamebirds after the end of the shooting season because it is expensive and the surviving gamebirds will, at best, only make a marginal contribution to the following season's shooting.

BASC's *Code of good shooting practice* [59] goes on to say that *Shoot managers should be aware of SSSIs and other sensitive habitats on their ground, and should liaise with the landowner and the relevant statutory authorities to ensure they avoid potentially damaging activities*. As I have shown in this review, released gamebird shooting has a wide range of damaging impacts on most if not all EPSs and SSSIs in England and conservation more generally. Clearly, codes of practice have been ineffective, and enforcement measures are needed to manage these impacts. It is surprising that one review concluded that self-regulation of the gamebird-shooting industry has been successful [305].

While wild game are covered by the Animal By-Products (Enforcement) (England) Regulations 2013 and the Animal By-Products (Miscellaneous Amendments) (England) Order 2015, the gamebird-shooting industry still deposits 21.5 million kg of pheasant and red-legged partridge carcasses in the environment each year. This has a dramatic impact on the British ecosystem. The British Game Alliance's standards require that *game that is unfit for consumption and processed game carcasses must only be disposed of by incineration, collection by a member of the National Fallen Stock Company, or via an Approved and Registered Animal By-Products Premises* [62,306]. These rules should be enforced to include the carcasses of all the non-native gamebirds that are reared for shooting but do not enter the human food chain. The removal of 21.5 million kg of carrion from the British ecosystem each year would have a dramatic impact on the numbers of predators and scavengers, and their potential impact on ground-nesting birds.

The gamebird-shooting industry currently supports anything between 80,000 and 200,000 foxes a year in Britain by providing a widespread supply of easy-to-catch prey and carrion. In the second half of last century, between 8200 and 23,300 foxes were supported for the whole year eating nothing but pheasants, i.e., in the last twenty years there has been a ten-fold increase in the number of foxes supported by the supplementary food provided by the gamebird-shooting industry. This has a major impact on the management of EPSs, SSSIs, and ecosystem function generally. Having played a significant role in supplementing fox numbers in Britain, it is extraordinary that the gamebird-shooting industry argues that the widespread killing of foxes is necessary to support their commercial interests.

Furthermore, the GWCT states that focussing fox control in the spring and summer may be essential to achieve their management goals [269], even though maintaining low fox densities through spring and early summer is clearly very challenging [244]. Advocating the extensive killing of foxes during their breeding season to protect a remnant population of non-native gamebirds that survived the shooting season [119,193], have a low breeding and survival success [125,130], and at best make a marginal contribution to the next season's shooting [119,307], is ethically highly questionable. Indeed, the UK is one of a minority of European countries that still does not have a close season for red foxes. Recognising predators and scavengers as sentient species is long overdue.

The widespread killing of foxes continues to be promoted as a means to reduce losses of ground-nesting birds, even though this policy has not had any obvious long-term benefits in terms of enhancing the populations of vulnerable ground-nesting birds or reducing the numbers of foxes in Britain. In Australia, over the last thirty years the approach to managing red foxes has moved from the short-term mind-set of killing as many animals as possible in the immediate area to a more carefully planned and coordinated landscape approach designed to reduce the impacts of foxes [308]. Such a change in mind-set is long overdue in Britain.

References

All websites were accessed between 13th March and 22nd May 2021

1. Defra (2020) *Defra concludes its review into releasing gamebirds on and around protected sites*. https://www.gov.uk/government/news/defra-concludes-its-review-into-releasing-gamebirds-on-and-around-protected-sites/.
2. Defra (2021) *Consultation launches on interim licences for releasing gamebirds*. https://www.gov. uk/government/news/consultation-launches-on-interim-licences-for-releasing-gamebirds.
3. Tapper S (1992) *Game heritage – an ecological review from shooting and gamekeeping records*. Game Conservancy Ltd, Fordingbridge.
4. Cobham Resource Consultants (1997) *Countryside sports – their economic, social and conservation significance*. Standing Conference on Countryside Sports, Reading.
5. PACEC (2006) *The economic and environmental impact of sporting shooting*. Public and Corporate Economic Consultants, Cambridge.
6. BASC (2014) *The value of shooting – the economic, environmental and social contribution of shooting sports to the UK by Public and Corporate Economic Consultants (PACEC)*. British Association for Shooting and Conservation, Wrexham. http://www.shootingfacts.co.uk/pdf/The-Value-of-Shooting-2014.pdf.
7. Lees AC, Newton I, Balmford A (2013) Pheasants, buzzards, and trophic cascades. *Conservation Letters*, **6**, 141-144.
8. Pringle H, Wilson M, Calladine J, Siriwardena G (2019) Associations between gamebird releases and generalist predators. *Journal of Applied Ecology*, **56**, 2102-2113.
9. Di Minin E, Clements HS, Correia RA, Cortés-Capano G, Fink C, Haukka A, Hausmann A, Kulkarni R, Bradshaw CJA (2021) Consequences of recreational hunting for biodiversity conservation and livelihoods. *One Earth*, **4**, 238-253.
10. Arroyo B, Beja P (2002) Impact of hunting management practices on biodiversity. www.uclm.es/irec/Reghab/inicio.html.
11. Stoate C (2002) Multifuctional use of a natural resource on farmland: wild pheasant (*Phasianus colchicus*) management and the conservation of farmland passerines. *Biodiversity and Conservation*, **11**, 561-573.
12. Duckworth JC, Firbank LG, Stuart RC, 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.
13. Oldfield TEE, Smith RJ, Harrop SR, Leader-Williams N (2003) Field sports and conservation in the United Kingdom. *Nature*, **423**, 531-533.
14. Sage RB, Ludolf C, Robertson PA (2005) The ground flora of ancient semi-natural woodlands in pheasant release pens in England. *Biological Conservation*, **122**, 243-252.
15. Sage RB, Woodburn MIA, Draycott RAH, Hoodless AN, 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.
16. Sage RB, Hoodless AN, Woodburn MIA, Draycott RAH, Madden JR, Sotherton NW (2020) Summary review and synthesis: effects on habitats and wildlife of the release and management of pheasants and red-legged partridges on UK lowland shoots. *Wildlife Biology*, **4**, wlb.00766.

17. Draycott RAH, Hoodless AN, Sage RB (2008) Effects of pheasant management on vegetation and birds in lowland woodlands. *Journal of Applied Ecology*, **45**, 334-341.

18. Draycott RAH, Hoodless AN, Cooke M, Sage RB (2012) The influence of pheasant releasing and associated management on farmland hedgerows and birds in England. *European Journal of Wildlife Research*, **58**, 227-234.

19. Bicknell J, Smart J, Hoccom D, Amar A, Evans A, Walton P, Knott J (2010) *Impacts of non-native gamebird release in the UK: a review*. Research Report No. 40. RSPB, Sandy.

20. Mustin K, Newey S, Irvine J, Arroyo B, Redpath S (2011) *Biodiversity impacts of game bird hunting and associated management practices in Europe and North America*. The James Hutton Institute, Aberdeen.

21. Mustin K, Arroyo B, Beja P, Newey S, Irivine RJ, Kestler J, Redpath SM (2018) Consequences of game bird management for non-game species in Europe. *Journal of Applied Ecology*, **55**, 2285-2295.

22. Neumann JL, Holloway GJ, Sage RB, Hoodless AN (2015) Releasing of pheasants for shooting in the UK alters woodland invertebrate communities. *Biological Conservation*, **191**, 50-59.

23. Blackburn TM, Gaston KJ (2018) Abundance, biomass and energy use of native and alien breeding birds in Britain. *Biological Invasions*, **20**, 3563-3573.

24. Blackburn TM, Gaston KJ (2021) Contribution of non-native galliforms to annual variation in biomass of British birds. *Biological Invasions*, https://doi.org/10.1007/s10530-021-02458-y.

25. Avery M (2019) The common pheasant: its status in the UK and the potential impacts of an abundant non-native. *British Birds*, **112**, 371-389.

26. Capstick LA, Draycott RAH, Wheelwright CM, Ling DE, Sage RB, Hoodless AN (2019) The effect of game management on the conservation value of woodland rides. *Forest Ecology and Management*, **454**, 117242.

27. Feber RE, Johnson PJ, Macdonald DW (2020) Shooting pheasants for sport: what does the death of Cecil tell us? *People and Nature*, **2**, 82-95.

28. Madden JR (2020) How many gamebirds are released in the UK each year? https://www.biorxiv. org/content/10.1101/2020.10.22.350603v1.full.pdf.

29. Madden JR, Sage RB (2020) *Ecological consequences of gamebird releasing and management on lowland shoots in England: a review by rapid evidence assessment for Natural England and the British Association of* [sic] *Shooting and Conservation*. Natural England, Peterborough.

30. Mason LR, Bicknell JE, Smart J, Peach WJ (2020) *The impacts of non-native gamebird release in the UK: an updated evidence review*. Research Report No. 66. RSPB, Sandy.

31. Massimino D, Woodward ID, Hammond MJ, Harris SJ, Leech DI, Noble DG, Walker RH, Barimore C, Dadam D, Eglington SM, Marchant JH, Sullivan MJP, Baillie SR, Robinson RA (2017) *Bird trends 2017: trends in numbers, breeding success and survival for UK breeding birds*. Research Report 704. British Trust for Ornithology, Thetford.

32. GWCT (2020) Grey partridge *Perdix perdix*. https://www.gwct.org.uk/game/research/species/grey-partridge/.

33. Aebischer NJ (2019) Fifty-year trends in UK 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.

34. Parish DMB, Sotherton NW (2007) The fate of released captive-reared grey partridges *Perdix perdix*: implications for reintroduction programmes. *Wildlife Biology*, **13**, 140-149.

35. Tapper S (ed.) (1999) *A question of balance – game animals and their role in the British Countryside*. Game Conservancy Trust, Fordingbridge.

36. Richardson DM, Pyšek P, Rejmánek M, Barbour MG, Panetta FD, West CJ (2000) Naturalization and invasion of alien plants: concepts and definitions. *Diversity and Distributions*, **6**, 93-107.

37. Robinson P (2002) Gamebirds in the wider environment. Unpublished report.

38. Harris S, Thain B (2020) *Hanged by the feet until dead – an analysis of snaring and trapping on Scottish grouse moors*. League Against Cruel Sports Scotland, Glasgow.

39. Farm Animal Welfare Council (2008) *Opinion on the welfare of farmed gamebirds*. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/325047/FAWC_opinion_on_the_welfare_of_farmed_gamebirds.pdf.

40. Animal & Plant Health Agency (2015) Gamebird keeping in Great Britain: general guidance. http://apha.defra.gov.uk/documents/surveillance/diseases/gamebirds-guidance.pdf.

41. Defra (2010) *Code of practice for the welfare of gamebirds reared for sporting purposes*. London: Department for Environment, Food and Rural Affairs. https://www.gov.uk/government/publications/code-of-practice-for-the-welfare-of-gamebirds-reared-for-sporting-purposes.

42. BASC (2011) *Aim of the game – driven game shooting in Britain today.* British Association for Shooting and Conservation, Wrexham.

43. GWCT (2020) *Releasing for shooting in lowland habitats*. https://www.gwct.org.uk/game/ research/species/pheasant/releasing-for-shooting-in-lowland-habitats/.

44. Turner C, Sage R (2004) Fate of released pheasants. *Game Conservancy Trust Review*, **35**, 74-75.

45. Natural England (2012) *Catching and releasing wild game birds: a legal summary*. Natural England Technical Information Note TIN104. http://www.adlib.ac.uk/resources/000/267/468/ TIN104.pdf.

46. Canning P (2005) *The UK game bird industry – a short study*. ADAS, Lincoln.

47. BASC (2012) *The code of good shooting practice*. British Association for Shooting and Conservation, Wrexham.

48. Langbein J (2007) *National deer-vehicle collisions project England (2003–2005)*. The Deer Initiative, Wrexham. http://www.deercollisions.co.uk/web-content/ftp/DVC_England_FinalAs.pdf.

49. Roberts G (2020) Warning as half of drivers report incidents involving animals. *Fleet Industry News*, 13th January 2020. https://www.fleetnews.co.uk/news/fleet-industry-news/2020/01/13/warning-as-drivers-report-incidents-on-the-road-involving-animals.

50. Anon. (2004) Biker dies after bird hits helmet. *Adventure Rider*, 24th November 2004. https:// advrider.com/f/threads/biker-dies-after-bird-hits-helmet.60588/.

51. Anon. (2008) Tragedy of biker, 52, killed by a pheasant. *Express*, 4th November 2008. https://www.express.co.uk/news/uk/69408/Tragedy-of-biker-52-killed-by-a-pheasant.

52. Anon. (2012) Killed by a pheasant: motorcyclist dies after bird flies into him at 60mph. *MailOnline*, 27th July 2012. https://www.dailymail.co.uk/news/article-2179414/Killed-pheasant-Motorcyclist-killed-bird-flies-60mph.

53. Anon. (2013) Essex motorcyclist killed after pheasant collision. *BBC News*, 15th October 2013. https://www.bbc.co.uk/news/uk-england-24539868.

54. Anon. (2016) Motorcyclist killed after pheasant flew into his helmet causing him to leave the A606 and crash – inquest hears. *Melton Times*, 17th March 2016. https://www.meltontimes.co.uk/ news/motorcyclist-killed-after-pheasant-flew-his-helmet-causing-him-leave-a606-and-crash-inquest-hears-2169717.

55. Anon. (2017) Motorcyclist Stewart Mackenzie died after hitting pheasant in Blythburgh. *East Anglian Daily Times*, 8th March 2017. https://www.eadt.co.uk/news/motorcyclist-stewart-mackenzie-died-after-hitting-pheasant-in-blythburgh-2327700.

56. Anon. (2018) Motorcyclist killed by pheasant that struck his helmet at 60mph. *Telegraph*, 21st February 2018. https://www.telegraph.co.uk/news/2018/02/21/motorcyclist-killed-pheasant-struck-helmet-60mph/.

57. Anon. (2019) The tragic death toll on Lincolnshire's roads in 2019. *LincolnshireLive*, 31st December 2019, https://www.lincolnshirelive.co.uk/news/local-news/tragic-death-toll-lincolnshires-roads-3653145.

58. Anon. (2019) Tragic uni student, 19, dies in crash 'after swerving to avoid pheasant in road'. *Mirror*, 28th May 2019, https://www.mirror.co.uk/news/uk-news/tragic-uni-student-19-dies-16212752.

59. BASC (2017) *The code of good shooting practice*. British Association for Shooting and Conservation, Wrexham. http://www.codeofgoodshootingpractice.org.uk/.

60. Defra (2015) *Fallen stock and safe disposal of dead animals*. London: Department for Environment, Food and Rural Affairs. https://www.gov.uk/guidance/fallen-stock.

61. Knight M (2014) Bring out your dead. *Keeping the Balance*, **Spring 2014**, 46-47. https://www. national-gamekeepers.org.uk/media/91/Carcass%20Disposal.pdf.

62. British Game Alliance (2021) Clarifying the correct game disposal – trying to clear up any confusion regarding the disposal of game. https://www.britishgamealliance.co.uk/clarifying-the-correct-game-disposal/.

63. Webbon CC, Baker PJ, Harris S (2004) Faecal density counts for monitoring changes in red fox numbers in rural Britain. *Journal of Applied Ecology*, **41**, 768-779.

64. Macdonald DW, Bunce RGH, Bacon PJ (1981) Fox populations, habitat characterization and rabies. *Journal of Biogeography*, **8**, 145-151.

65. Harris S, Morris P, Wray S, Yalden D (1995) *A review of British mammals: population estimates and conservation status of British mammals other than cetaceans*. Joint Nature Conservation Committee, Peterborough.

66. Mathews F, Kubasiewicz LM, Gurnell J, Harrower CA, McDonald RA, Shore RF (2018) *A review of the population and conservation status of British mammals: technical summary*. Natural England Joint Publication JP025. Natural England, Peterborough. http://nora.nerc.ac.uk/id/ eprint/520322/1/N520322CR.pdf.

67. Wright LJ, Newson SE, Noble DG (2014) The value of a random sampling design for annual monitoring of national populations of larger British terrestrial mammals. *European Journal of Wildlife Research*, **60**, 213-221.

68. Harris SJ, Massimino D, Balmer DE, Eaton MA, Noble DG, Pearce-Higgins JW, Woodcock P, Gillings S (2020) *The Breeding Bird Survey 2019*. BTO Research Report 726. British Trust for Ornithology, Thetford.

69. Gooderham G (2011) *Hunting records track UK game populations over centuries*. https://blog. national-geographic.org/2011/05/21/hunting-records-track-uk-game-populations-over-centuries/.

70. Aebischer N (2020) Rabbits, foxes and mustelids 1961-2018. *Game & Wildlife Conservation Trust Review*, **51**, 58-61.

71. Reynolds JC (2000) *Fox control in the countryside*. The Game Conservancy Trust, Fordingbridge.

72. Aubry P, Guillemain M, Sorrenti M (2020) Increasing the trust in hunting bag statistics: why random selection of hunters is so important. *Ecological Indicators*, **117**, 106522.

73. GWCT (2020) *General considerations for the interpretation of NGC trends*. https://www.gwct.org. uk/research/long-term-monitoring/national-gamebag-census/interpretational-considerations/.

74. Landry P (1983) Preliminary report on methods for collecting game bag statistics in European countries. In: Leeuwenberg F, Hepburn I (eds) *Working group on game statistics*, pp. 25-46. International Union of Game Biologists, Doorwerth.

75. GWCT (2020) *National gamebag census*. https://www.gwct.org.uk/research/long-term-monitoring/national-gamebag-census/.

76. Lloyd HG (1980) *The red fox*. Batsford, London.

77. Harris S, Smith GC (1987) Demography of two urban fox (*Vulpes vulpes*) populations. *Journal of Applied Ecology*, **24**, 75-86.

78. Trout RC, Ross J, Tittensor AM, Fox AP (1992) The effect on a British wild rabbit population (*Oryctolagus cuniculus*) of manipulating myxomatosis. *Journal of Applied Ecology*, **29**, 679-686.

79. Trout RC, Tapper SC, Harradine J (1986) Recent trends in the rabbit population in Britain. *Mammal Review*, **16**, 117-123.

80. Baker PJ, Harris S (2003) A review of the diet of foxes in rural Britain and a preliminary assessment of their impact as a predator. In: Tattersall F, Manley W (eds) *Conservation and conflict – mammals and farming in Britain*, pp. 120-140. Westbury Publishing, Otley.

81. Trout RC, Tittensor AM (1989) Can predators regulate wild rabbit *Oryctolagus cuniculus* population density in England and Wales? *Mammal Review*, **19**, 153-173.

82. Banks PB, Dickman CR, Newsome AE (1998) Ecological costs of feral predator control: foxes and rabbits. *Journal of Wildlife Management*, **62**, 766-772.

83. Pech RP, Sinclair ARE, Newsome AE, Catling PC (1992) Limits to predator regulation of rabbits in Australia: evidence from predator-removal experiments. *Oecologia*, **89**, 102-112.

84. Banks PB (2000) Can foxes regulate rabbit populations? *Journal of Wildlife Management*, **64**, 401-406.

85. Trout RC, Langton S, Smith GC, Haines-Young RH (2000) Factors affecting the abundance of rabbits (*Oryctolagus cuniculus*) in England and Wales. *Journal of Zoology*, **252**, 227-238.

86. Davey C, Sinclair ARE, Pech RP, Arthur AD, Krebs CJ, Newsome AE, Hik D, Molsher R, Allcock K (2006) Do exotic vertebrates structure the biota of Australia? An experimental test in New South Wales. *Ecosystems*, **9**, 992–1008.

87. Fernandez-de-Simon J, Díaz-Ruiz F, Rodríguez-de la Cruz M, Delibes-Mateos M, Villafuerte R, Ferreras P (2015) Can widespread generalist predators affect keystone prey? A case study with red foxes and European rabbits in their native range. *Population Ecology*, **57**, 591-599.

88. Ferreras P, Travaini A, Zapata SC, Delibes M (2011) Short-term responses of mammalian carnivores to a sudden collapse of rabbits in Mediterranean Spain. *Basic and Applied Ecology*, **12**, 116-124.

89. Scroggie MP, Forsyth DM, McPhee SR, Matthews J, Stuart IG, Stamation KA, Lindeman M, Ramsey DSL (2018) Invasive prey controlling invasive predators? European rabbit abundance does not determine red fox population dynamics. *Journal of Applied Ecology*, **55**, 2621-2631.

90. Macdonald DW, Reynolds JC, Carbone C, Mathews F, Johnson PJ (2003) The bio-economics of fox control. In: Tattersall F, Manley W (eds) *Conservation and conflict – mammals and farming in Britain*, pp. 220-236. Westbury Publishing, Otley.

91. Soulsbury CD, Iossa G, Baker PJ, Cole NC, Funk SM, Harris S (2007) The impact of sarcoptic mange *Sarcoptes scabiei* on the British fox *Vulpes vulpes* population. *Mammal Review*, **37**, 278-296.

92. Baker PJ, Harris S, Webbon CC (2002) Effect of British hunting ban on fox numbers. *Nature*, **419**, 34.

93. Trewby ID, Wilson GJ, Delahay RJ, Walker N, Young R, Davison J, Cheeseman C, Robertson PA, Gorman ML, McDonald RA (2008) Experimental evidence of competitive release in sympatric carnivores. *Biology Letters*, **4**, 170-172.

94. Kauhala K, Laukkanen P, von Rége I (1998) Summer food composition and food niche overlap of the raccon dog, red fox and badger in Finland. *Ecography*, **21**, 457-463.

95. Elmeros M, Mikkelsen DMG, Nørgaard LS, Pertoldi C, Jensen TH, Chriél M (2018) The diet of feral raccoon dog (*Nyctereutes procyonoides*) and native badger (*Meles meles*) and red fox (*Vulpes vulpes*) in Denmark. *Mammal Research*, **63**, 405-413.

96. Newman TJ, Baker PJ, Simcock E, Saunders G, White PCL, Harris S (2003) Changes in red fox habitat preference and rest site fidelity following a disease-induced population decline. *Acta Theriologica*, **48**, 79-91.

97. Nowakowski K, Ważna A, Kurek P, Cichocki J, Gabryś G (2020) Reproduction success in European badgers, red foxes and raccoon dogs in relation to sett cohabitation. *PLoS One*, **15(8)**, e0237642.

98. Lindström ER, Andrén H, Angelstam P, Cederlund G, Hörnfeldt B, Jäderberg L, Lemnell P-A, Martinsson B, Sköld K, Swenson JE (1994) Disease reveals the predator: sarcoptic mange, red fox predation, and prey populations. *Ecology*, **75**, 1042-1049.

99. Lindström ER, Brainerd SM, Helldin JO, Overskaug K (1995) Pine marten - red fox interactions: a case of intraguild predation? *Annales Zoologici Fennici*, **32**, 123-130.

100. Latham RM (1952) The fox as a factor in the control of weasel populations. *Journal of Wildlife Management*, **16**, 516-517.

101. Mulder JL (1990) The stoat *Mustela erminea* in the Dutch dune region, its local extinction, and a possible cause: the arrival of the fox *Vulpes vulpes*. *Lutra*, **33**, 1-21.

102. Erlinge S, Sandell M (1988) Coexistence of stoat, *Mustela erminea*, and weasel, *M. nivalis*: social dominance, scent communication, and reciprocal distribution. *Oikos*, **53**, 242-246.

103. Roos S, Smart J, Gibbons DW, Wilson JD (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**, 1915-1937.

104. Palmer LS (1927) Dietetics and its relationship to fur production. *Fox and Fur Farmer*, **7(2)**, 22-24.

105. Hodson AZ, Smith SE (1942) Estimated maintenance energy requirements of foxes and mink. *Fur Trade Journal of Canada*, **19(6)**, 12-15.

106. Leoschke WL (2011) *Nutrition and nutritional physiology of the fox – a historical perspective*. Trafford Publishing, Bloomington, Indiana.

107. Saunders G, White PCL, Harris S, Rayner JMV (1993) Urban foxes (*Vulpes vulpes*): food acquisition, time and energy budgeting of a generalized predator. *Symposia of the Zoological Society of London*, **65**, 215-234.

108. Winstanley RK, Buttemer WA, Saunders G (2003) Field metabolic rate and body water turnover of the red fox *Vulpes vulpes* in Australia. *Mammal Review*, **33**, 295-301.

109. Crocker D, Hart A, Gurney J, McCoy C (2002) *Project PN0908: methods for estimating daily food intake of wild birds and mammals*. https://www.hse.gov.uk/pesticides/resources/R/Research_ PN0908.pdf.

110. Hewson R, Kolb HH (1986) Scavenging on sheep carcases by foxes (*Vulpes vulpes*) and badgers (*Meles meles*). *Journal of Zoology*, **180**, 496-498.

111. Winstanley RK (2000) Morphological change and fat accumulation in juvenile red foxes (*Vulpes vulpes*) in the Central Tablelands of New South Wales. *Wildlife Research*, **27**, 525-529.

112. Vogtsberger LM, Barrett GW (1973) Bioenergetics of captive red foxes. *Journal of Wildlife Management*, **37**, 495-500.

113. Parrott D, Moore N, Browne S, Aebischer N (2003) *Provision of bag statistics for huntable birds*. Defra Project CRO281. http://randd.defra.gov.uk/Document.aspx?Document=WC04001_1244_ FRP.doc.

114. Aubry P, Guillemain M, Jensen GH, Sorrenti M, Scallan D (2020) Moving from intentions to actions for collecting hunting bag statistics at the European scale: some methodological insights. *European Journal of Wildlife Research*, **66**, 70.

115. Game Farmers' Association (2021) *Game farming in the UK*. https://www.gfa.org.uk/game-farming-in-the-uk.html/.

116. Aebischer N (2018) How many birds are shot in the UK? *Game and Wildlife Conservation Trust Review*, **49**, 42-43.

117. RUMA (2017) *Targets task force report 2017*. https://www.ruma.org.uk/wp-content/uploads/2017/10/RUMA-Targets-Task-Force-Report-2017-FINAL.pdf.

118. gov.uk (2021) *Avian influenza (bird flu)*. https://www.gov.uk/guidance/avian-influenza-bird-flu.

119. GWCT (2021) *Estimating the number and biomass of pheasants in Britain*. https://www.whatthesciencesays.org/estimating-the-number-and-biomass-of-pheasants-in-britain/.

120. Aebischer N (2013) National gamebag census: released game species. Game and *Wildlife Conservation Trust Review*, **44**, 34-37.

121. GWCT (2020) *Common pheasant*. https://www.gwct.org.uk/research/species/birds/common-pheasant/.

122. Schulp CJE, Thuiller W, Verburg PH (2014) Wild food in Europe: a synthesis of knowledge and data of terrestrial wild food as an ecosystem service. *Ecological Economics*, **105**, 292-305.

123. Reimoser F, Reimoser S (2016) Long-term trends of hunting bags and wildlife populations in central Europe. *Beiträge zur Jagd- und Wildforschun*g, **41**, 29-43.

124. Sage R (2018) Non-shooting losses of released pheasants. *Game & Wildlife Conservation Trust Review*, **49**, 26-27.

125. Sage RB, Turner CV, Woodburn MIA, Hoodless AN, Draycott RAH, Sotherton NW (2018) Predation of released pheasants *Phasianus colchicus* on lowland farmland in the UK and the effect of predator control. *European Journal of Wildlife Research*, **64**, 14.

126. Hesford NJ (2012) *Fate and survival of hand-reared red-legged partridges released for sport on farmland in the UK*. Unpublished MSc thesis, University of Cardiff.

127. BTO (undated) *Red-legged partridge* Alectoris rufa *Linnaeus*, *1758*. https://app.bto.org/birdfacts/results/bob3580.htm.

128. Park KJ, Graham KE, Calladine J, Wernham CW (2008) Impacts of birds of prey on gamebirds in the UK: a review. *Ibis*, **150 (Suppl. 1)**, 9-26.

129. 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.

130. 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*. Defra Project WM0415. http://randd.defra.gov.uk/Document.aspx?Document=10025_WM0415_ birdofprey-conflicts_finalversion_12June2012.pdf.

131. Draycott R (2009) Pheasant nest predation. *Game & Wildlife Conservation Trust Review*, **40**, 16-17.

132. Aebischer N (2004) Recent trends in predator culls. *Game Conservancy Review*, **35**, 92-95.

133. Beasley JC, Olson ZH, Selva N, DeVault TL (2019) Ecological functions of vertebrate scavenging. In: Olea PP, Mateo-Tomás P, Sánchez-Zapata AJ (eds) *Carrion ecology and management*, pp. 125-157. Springer Nature, Cham, Switzerland.

134. DeVault TL, Olson ZH, Beasley JC, Rhodes OE (2011) Mesopredators dominate competition for carrion in an agricultural landscape. *Basic and Applied Ecology*, **12**, 268-274.

135. Barton PS, Evans MJ, Foster CN, Pechal JL, Bump JK, Quaggiotto M-M, Benbow ME (2019) Towards quantifying carrion biomass in ecosystems. *Trends in Ecology and Evolution*, **34**, 950-961.

136. Lockie JD (1963) Eagles, foxes and their food supply in Wester Ross. *Scottish Agriculture*, **42**, 186-189.

137. Hewson R (1984) Scavenging and predation upon sheep and lambs in west Scotland. *Journal of Applied Ecology*, **21**, 843-868.

138. Madden JR, Perkins SE (2017) Why did the pheasant cross the road? Long-term road mortality patterns in relation to management changes. *Royal Society Open Science*, **4**, 170617.

139. Pardo-Barquín F, Mateo-Tomás P, Olea PP (2019) Habitat characteristics from local to landscape scales combine to shape vertebrate scavenging communities. *Basic and Applied Ecology*, **34**, 126-139.

140. Paul Tillsley, personal communication, 2nd March 2021.

141. Welch I (9 January 2018) Hitting high pheasants with shooting expert Ed Solomons. https://www.farlows.co.uk/blog/2018/01/09/hitting-high-pheasants/#.

142. Negus R (2019) How to mark pricked birds. *Shooting Times*, 11th November 2019. https://www.shooting.co.uk/shooting/marking-pricked-birds-40935.

143. Putman RJ (1983) *Carrion and dung: the decomposition of animal wastes*. Edward Arnold, London.

144. Schwartz ALW, Williams HF, Chadwick E, Thomas RJ, Perkins SE (2018) Roadkill scavenging behaviour in an urban environment. *Journal of Urban Ecology*, **4**, 1-7.

145. Turner KL, Abernethy EF, Conner LM, Rhodes OE, Beasley JC (2017) Abiotic and biotic factors modulate carrion fate and vertebrate scavenging communities. *Ecology*, **98**, 2413-2424.

146. González LM, Margalida A, Sánchez R, Oria J (2006) Supplementary feeding as an effective tool for improving breeding success in the Spanish imperial eagle (*Aquila adalberti*). *Biological Conservation*, **129**, 477-486.

147. Orros ME, Fellowes MDE (2014) Supplementary feeding of the reintroduced red kite *Milvus milvus* in UK gardens. Bird Study, **61**, 260-263.

148. DeVault TL, Rhodes OE, Shivik JA (2003) Scavenging by vertebrates: behavioral, ecological, and evolutionary perspectives on an important energy transfer pathway in terrestrial ecosystems. *Oikos*, **102**, 225-234.

149. Moleón M, Sánchez-Zapata JA, Selva N, Donázar JA, Owen-Smith N (2014) Inter-specific interactions linking predation and scavenging in terrestrial vertebrate assemblages. *Biological Reviews*, **89**, 1042-1054.

150. Cortés-Avizanda A, Carrete M, Serrano D, Donázar JA (2009) Carcasses increase the probability of predation of ground-nesting birds: a caveat regarding the conservation value of vulture restaurants. *Animal Conservation*, **12**, 85-88.

151. Selva N, Berezowska-Cnota T, Elguero-Claramunt I (2014) Unforeseen effects of supplementary feeding: ungulate baiting sites as hotspots for ground-nest predation. *PLoS One*, **9(3)**, e90740.

152. Cortés-Avizanda A, Blanco G, DeVault TL, Markandya A, Virani MZ, Brandt J, Donázar JA (2016) Supplementary feeding and endangered avian scavengers: benefits, caveats, and controversies. *Frontiers in Ecology and the Environment*, **14**, 191-199.

153. Wilson EE, Wolkovich EM (2011) Scavenging: how carnivores and carrion structure communities. *Trends in Ecology and Evolution*, **26**, 129-135.

154. Murray MH, Becker DJ, Hall RJ, Hernandez SM (2016) Wildlife health and supplementary feeding: a review and management recommendations. *Biological Conservation*, **204**, 163-174.

155. Harris S, Dorning J (2017) *Hunting with hounds and the spread of disease*. League Against Cruel Sports, London.

156. Ogada DL, Torchin ME, Kinnaird MF, Ezenwa VO (2012) Effects of vulture declines on facultative scavengers and potential implications for mammalian disease transmission. *Conservation Biology*, **26**, 453-460.

157. Carrasco-Garcia R, Barroso P, Perez-Olivares J, Montoro V, Vicente J (2018) Consumption of big game remains by scavengers: a potential risk as regards disease transmission in central Spain. *Frontiers in Veterinary Science*, **5**, 4.

158. Vicente J, VerCauteren K (2019) The role of scavenging in disease dynamics. In: Olea PP, Mateo-Tomás P, Sánchez-Zapata AJ (eds) *Carrion ecology and management*, pp. 161-182. Springer Nature, Cham, Switzerland.

159. Strandin T, Babayan SA, Forbes KM (2018) Reviewing the effects of food provisioning on wildlife immunity. *Philosophical Transactions of the Royal Society B*, **373**, 20170088.

160. Becker DJ, Hall RJ (2014) Too much of a good thing: resource provisioning alters infectious disease dynamics in wildlife. *Biology Letters*, **10**, 20140309.

161. Fischer JR, Stallknecht DE, Luttrell P, Dhondt AA, Converse KA (1997) Mycoplasmal conjunctivitis in wild songbirds: the spread of a new contagious disease in a mobile host population. *Emerging Infectious Diseases*, **3**, 69-72.

162. Lawson B, Robinson RA, Colvile KM, Peck KM, Chantrey J, Pennycott TW, Simpson VR, Toms MP, Cunningham AA (2012) The emergence and spread of finch trichomonosis in the British Isles. *Philosophical Transactions of the Royal Society B*, **367**, 2852-2863.

163. Adelman JS, Moyers SC, Farine DR, Hawley DM (2015) Feeder use predicts both acquisition and transmission of a contagious pathogen in a North American songbird. *Proceedings of the Royal Society B*, **282**, 20151429.

164. Becker DJ, Hall RJ, Forbes KM, Plowright RK, Altizer S (2018) Anthropogenic resource subsidies and host–parasite dynamics in wildlife. *Philosophical Transactions of the Royal Society B*, **373**, 20170086.

165. Sorensen A, van Beest FM, Brook RK (2014) Impacts of wildlife baiting and supplementary feeding on infectious disease transmission: a synthesis of knowledge. *Preventative Veterinary Medicine*, **113**, 356-363.

166. Pitarch A, Gil C, Blanco G (2017) Oral mycoses in avian scavengers exposed to antibiotics from livestock farming. *Science of the Total Environment*, **605-606**, 139-146.

167. van den Honert MS, Gouws PA, Hoffman LC (2020) A preliminary study: antibiotic resistance patterns of *Escherichia* coli and *Enterococcus* species from wildlife species subjected to supplementary feeding on various South African farms. *Animals*, **10(3)**, 396.

168. NADIS (2021) *Diseases of game birds Part 4 – diseases from 10 days to 7 weeks of age*. https://www.nadis.org.uk/disease-a-z/game-birds/diseases-of-game-birds/part-4-diseases-from-10-days-to-7-weeks-of-age/.

169. NADIS (2021) *Diseases of game birds Part 5 – the release pen*. https://www.nadis.org.uk/disease -a-z/game-birds/diseases-of-game-birds/part-5-the-release-pen/.

170. Davis C (2004) Glucose key to understanding *Hexamita*. *Game Conservancy Review*, **35**, 82-83.

171. Rogers S (2010) *Controlling mycoplasmosis in pheasants*. https://www.wattagnet.com/articles/6393-controlling-mycoplasmosis-in-pheasants.

172. RUMA (2018) *Targets task force: one year on*. https://www.ruma.org.uk/wp-content/uploads/2018/11/RUMA-TTF-1-year-on-Full-Report-FINAL.pdf.

173. RUMA (2019) *Targets task force: two years on*. https://www.ruma.org.uk/wp-content/uploads/2019/10/RUMA-TTF-update-2019-two-years-on-FULL-REPORT.pdf.

174. FarmingUK (2020) 'Efficiency, weather and politics' causes bumper year for UK red meat. https://www.farminguk.com/news/-efficiency-weather-and-politics-causes-bumper-year-for-uk-red-meat_54837.html.

175. Agriculture and Horticulture Development Board (2021) *UK pig facts and figures*. https://ahdb. org.uk/uk-pig-facts-and-figures.

176. AVEC (2020) *Annual report 2020*. https://www.avec-poultry.eu/wp-content/uploads/2020/09/05691-AVEC-annual-report-2020.pdf.

177. Savills (2018) *Spotlight: shoot benchmarking survey 2017/18 season*. https://www.savills.co.uk/research_articles/229130/249248-0.

178. Raptor Persecution UK (2019) *Dead pheasants dumped in Lincolnshire, presumed shot*. https:// raptor-persecutionscotland.wordpress.com/2019/01/30/dead-pheasants-dumped-in-lincolnshire-presumed-shot/.

179. gov.uk (2016) *Erratum: UK veterinary antibiotic resistance and sales surveillance report UK-VARSS 2016*. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/ attachment_data/file/915727/_1681622-v2-Erratum_2016_UK_VARSS-accessible.pdf.

180. RUMA (2020) *Targets task force report 2020*. https://www.ruma.org.uk/wp-content/uploads/2020/11/RUMA-Targets-Task-Force-Report-2020_download.pdf.

181. Anon. (2020) *Gamebirds*. https://www.farmantibiotics.org/progress-updates/progress-by-sector/game-bird-rearing/.

182. Anon. (2018) *Reducing antibiotic use in gamebirds*. http://www.gfa.org.uk/user_files/uploads/2018%20Joint%20Communication%20on%20AB%20Reduction.pdf.

183. Wall BA, Mateus A, Marshall L, Pfeiffer DU, Lubroth J, Ormel HJ, Otto P, Patriarchi A (2016) *Drivers, dynamics and epidemiology of antimicrobial resistance in animal production*. Food and Agriculture Organization of the United Nations, Rome.

184. Landers TF, Cohen B, Wittum TE, Larson EL (2012) A review of antibiotic use in food animals: perspective, policy, and potential. *Public Health Reports*, **127**, 4-22.

185. Dolejska M, Literak I (2019) Wildlife is overlooked in the epidemiology of medically important antibiotic-resistant bacteria. *Antimicrobial Agents and Chemotherapy*, **63**, e01167-19.

186. Silva V, Carvalho I, Igrejas G, Poeta P (2017) Soil antibiotics and transfer of antibiotic resistance genes affecting wildlife. In: Hashmi MZ, Strezov V, Varma A (eds) Antibiotics and antibiotics resistence genes in soils: monitoring, toxicity, risk assessment and management, pp. 307-319. Springer Nature, Cham, Switzerland.

187. Shore RF, Taggart MA, Smits J, Mateo R, Richards NL, Fryday S (2014) Detection and drivers of exposure and effects of pharmaceuticals in higher vertebrates. *Philosophical Transactions of the Royal Society B*, **369**, 20130570.

188. Trevelline BK, Fontaine SS, Hartup BK, Kohl KD (2019) Conservation biology needs a microbial renaissance: a call for the consideration of host-associated microbiota in wildlife management practices. *Proceedings of the Royal Society B*, **286**, 20182448.

189. Meek RW, Vyas H, Piddock LJV (2015) Nonmedical uses of antibiotics: time to restrict their use? *PLoS Biology*, **13(10)**, e1002266.

190. Gethings O, Sage R (2017) Gapeworm in pheasants. *Game & Wildlife Conservation Trust Review*, **48**, 18-19.

191. Pearson A (2014) *Game bird diseases – a gamekeeper's guide*. Blaze Publishing, Leamington Spa.

192. Arnold KE, Brown AR, Ankley GT, Sumpter JP (2014) Medicating the environment: assessing risks of pharmaceuticals to wildlife and ecosystems. *Philosophical Transactions of the Royal Society B*, **369**, 20130569.

193. Draycott R, Blamey R (2012) Disease and mortality in wild pheasants. *Game & Wildlife Conservation Trust Review*, **43**, 20-21.

194. NADIS (2021) *Coronavirus*. https://www.nadis.org.uk/disease-a-z/game-birds/coronavirus/.

195. Rahman MM, Talukder A, Chowdhury MMH, Talukder R, Akter R (2021) Coronaviruses in wild birds – a potential and suitable vector for global distribution. *Veterinary Medicine and Science*, **7**, 264-272.

196. Golden NH, Warner SE, Coffey MJ (2016) A review and assessment of spent lead ammunition and its exposure and effects to scavenging birds in the United States. *Reviews of Environmental Contamination and Toxicology*, **237**, 123-191.

197. Hampton JO, Laidlaw M, Buenz E, Arnemo JM (2018) Heads in the sand: public health and ecological risks of lead-based bullets for wildlife shooting in Australia. *Wildlife Research*, **45**, 287-306.

198. Anon. (2020) *How to choose the right cartridge for your shotgun*. https://www.shootinguk.co.uk/guns/ammunition/how-to-choose-the-right-cartridge-81517.

199. Reinhold S (2019) *The golden ratio*. https://simonreinhold.co.uk/partridge-shooting/2019/1/17/the-golden-ratio.

200. Anon. (2012) *What is the hardest shot?* https://www.thefield.co.uk/features/what-is-the-hardest-shot-21622.

201. Anon. (2019) *British country sports – bringing your field sports dreams to life*. https://www. british-countrysports.co.uk/shooting-glossary.

202. Pain DJ, Cromie R, Green RE (2015) Poisoning of birds and other wildlife from ammunition-derived lead in the UK. In: Delahay RJ, Spray CJ (eds) *Lead ammunition: understanding and minimising the risks to human and environmental health*, pp. 58-84. Edward Grey Institute, University of Oxford, Oxford. http://oxfordleadsymposium.info/proceedings/.

203. Butler DA, Sage RB, Draycott RAH, Carroll JP, Potts D (2010) Lead exposure in ring-necked pheasants on shooting estates in Great Britain. *Wildlife Society Bulletin*, **33**, 583-589.

204. Stamberov P, Zhelev C, Todorov T, Ivanova S, Mehmedov T, Manev I, Taneva E (2018) Epidemiological data on lead tissue concentration in game birds induced by lead pellets. *Agriculture for Life, Life for Agriculture*, 1, 479-484. https://doi.org/10.2478/alife-2018-0075.

205. Dip R, Stieger C, Deplazes P, Hegglin D, Müller U, Dafflon O, Koch H, Naegeli H (2001) Comparison of heavy metal concentrations in tissues of red foxes from adjacent urban, suburban, and rural areas. *Archives of Environmental Contamination and Toxicology*, **40**, 551-556.

206. Naccaria C, Giangrosso G, Macaluso A, Billone E, Cicero A, D'Ascenzi C, Ferrantelli V (2013) Red foxes (*Vulpes vulpes*) bioindicator of lead and copper pollution in Sicily (Italy). *Ecotoxicology and Environmental Safety*, **90**, 41-45.

207. Pain DJ, Meharg AA, Ferrer M, Taggart M, Penteriani V (2005) Lead concentrations in bones and feathers of the globally threatened Spanish imperial eagle. *Biological Conservation*, **121**, 603-610.

208. Nadjafzadeh M, Hofer H, Krone O (2013) The link between feeding ecology and lead poisoning in white-tailed eagles. *Journal of Wildlife Management*, **77**, 48-57.

209. Molenaar FM, Jaffe JE, Carter I, Barnett EA, Shore RF, Rowcliffe JM, Sainsbury AW (2017) Poisoning of reintroduced red kites (*Milvus milvus*) in England. *European Journal of Wildlife Research*, **63**, 94.

210. Pain DJ, Mateo R, Green RE (2019) Effects of lead from ammunition on birds and other wildlife: a review and update. *Ambio*, **48**, 935-953.

211. Taggart MA, Shore RF, Pain DJ, Peniche G, Martinez-Haro M, Mateo R, Homann J, Raab A, Feldmann J, Lawlor AJ, Potter ED, Walker LA, Braidwood DW, French AS, Parry-Jones J, Swift JA, Green RE (2020) Concentration and origin of lead (Pb) in liver and bone of Eurasian buzzards (*Buteo buteo*) in the United Kingdom. *Environmental Pollution*, **267**, 115629.

212. Meyer CB, Meyer JS, Francisco AB, Holder J, Verdonck F (2016) Can ingestion of lead shot and poisons change population trends of three European birds: grey partridge, common buzzard, and red kite? *PLoS One*, **11**, e0147189.

213. BASC (2020) A joint statement on the future of shotgun ammunition for live quarry shooting. https://basc.org.uk/a-joint-statement-on-the-future-of-shotgun-ammunition-for-live-quarry-shooting/.

214. Green RE, Taggart MA, Pain DJ, Clark NA, Clewley L, Cromie R, Elliot B, Green RMW, Huntley B, Huntley J, Leslie R, Porter R, Robinson JA, Smith KW, Smith L, Spencer J, Stroud D (2021) Effect of a joint policy statement by nine UK shooting and rural organisations on the use of lead shotgun ammunition for hunting common pheasants *Phasianus colchicus* in Britain. *Conservation Evidence Journal*, **18**, 1-9.

215. Cromie RL, Loram A, Hurst L, O'Brien M, Newth J, Brown MJ, Harradine JP (2010) *Compliance with the Environmental Protection (Restrictions on Use of Lead Shot) (England) Regulations 1999*. Report to Defra, Bristol. http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More & Location=None&ProjectID=16075.

216. Cromie RL, Newth JL, Reeves JP, O'Brien MF, Beckmann KM, Brown MJ (2015) The sociological and political aspects of reducing lead poisoning from ammunition in the UK: why the transition to non-toxic ammunition is so difficult. In: Delahay RJ, Spray CJ (eds) *Lead ammunition: understanding and minimising the risks to human and environmental health*, pp. 104-124. Edward Grey Institute, University of Oxford, Oxford. http://oxfordleadsymposium. info/proceedings/.

217. Harris S (2017) *The impact of hunting with dogs on wildlife and conservation: a review with particular reference to the National Trust*. https://www.researchgate.net/publication/320621956_The_impact_of_hunting_with_dogs_on_wildlife_and_conservation_A_review_with_particular_reference_to_the_National_Trust.

218. GWCT (2009) *Late winter and spring feeding of pheasants and partridges*. Game & Wildlife Conservation Trust, Fordingbridge.

219. Anon. (2016) Pheasant damage. https://thefarmingforum.co.uk/index.php?threads/pheasant-damage.133378/.

220. GWCT (2007) *Guidelines for sustainable gamebird releasing*. Fordingbridge, Hampshire: Game & Wildlife Conservation Trust.

221. GWCT (undated) *Winter feeding*. https://www.gwct.org.uk/advisory/faqs/winter-feeding/#faq-1.

222. Farmers Weekly (2020) *Defra's estimate 2020 UK wheat crop is lowest since 1981*. https://www.fwi.co.uk/business/markets-and-trends/defras-estimates-2020-uk-wheat-crop-is-lowest-since-1981.

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

224. GWCT (2017) *Guidelines for successful gamebird and songbird feeding*. Game & Wildlife Conservation Trust, Fordingbridge.

225. New South Wales Government (undated) *Nutritional requirements of pheasants*. https://www.dpi.nsw.gov.au/animals-and-livestock/poultry-and-birds/species/pheasant-raising/feeding-pheasants.

226. Draycott RAH, Woodburn MIA, Carroll JP, Sage RB (2005) Effects of spring supplementary feeding on population density and breeding success of released pheasants *Phasianus colchicus* in Britain. *Wildlife Biology*, **11**, 177-182.

227. Hoodless AN, Draycott RAH, Ludiman MN, Robertson PA (1999) Effects of supplementary feeding on territoriality, breeding success and survival in pheasants. *Journal of Applied Ecology*, **36**, 147-156.

228. GWCT (2021) *How to build the ultimate rat-proof gamebird feeder*. https://www.gwctlearning. com/p/how-to-build-the-ultimate-rat-proof-gamebird-feeder.

229. Jones HP, Tershy BR, Zavaleta ES, Croll DA, Keitt BS, Finkelstein ME, Howald GR (2008) Severity of the effects of invasive rats on seabirds: a global review. *Conservation Biology*, **22**, 16-26.

230. Taylor KD (1978) Range of movement and activity of common rats (*Rattus norvegicus*) on agricultural land. *Journal of Applied Ecology*, **15**, 663-677.

231. Taylor KD, Quy RJ (1978) Long-distance movements of a common rat (*Rattus norvegicus*) revealed by radio-tracking. *Mammalia*, **42**, 63-71.

232. Sodhi NS, Ehrlich PR (2010) Conservation biology for all. Oxford University Press, Oxford.

233. Primack R (2014) *Essential of conservation biology*, 6th edition. Sinauer Associates, Sunderland, Massachusetts.

234. Morales-Reyes Z, Sánchez-Zapata JA, Sebastián-González E, Botella F, Carrete M, Moleón M (2017) Scavenging efficiency and red fox abundance in Mediterranean mountains with and without vultures. *Acta Oecologica*, **79**, 81-88.

235. Selva N, Jędrzejewska B, Jędrzejewski W, Wajrak A (2005) Factors affecting carcass use by a guild of scavengers in European temperate woodland. *Canadian Journal of Zoology*, **83**, 1590-1601.

236. Baker P, Funk S, Harris S, Newman T, Saunders G, White P (2004) The impact of human attitudes on the social and spatial organization of urban foxes (*Vulpes vulpes*) before and after an outbreak of sarcoptic mange. In: Shaw WW, Harris LK, Vandruff L (eds) *Proceedings of the 4th International Symposium on Urban Wildlife Conservation, Tucson, May 1999*, pp. 153-163. School of Natural Resources, College of Agriculture and Life Sciences, University of Arizona, Tucson.

237. Harris S (2015) *The utility of killing foxes in Scotland*. League Against Cruel Sports Scotland, Glasgow.

238. Kolb HH, Hewson R (1980) A study of fox populations in Scotland from 1971 to 1976. *Journal of Applied Ecology*, **17**, 7-19.

239. Hewson R, Kolb HH (1973) Changes in the numbers and distribution of foxes (*Vulpes vulpes*) killed in Scotland from 1948-1970. *Journal of Zoology*, **171**, 345-365.

240. Hewson R (1984) Changes in the numbers of foxes (*Vulpes vulpes*) in Scotland. *Journal of Zoology*, **204**, 561-569.

241. Lindström ER (1989) Food limitation and social regulation in a red fox population. *Holarctic Ecology*, **12**, 70-79.

242. Kapota D, Dolev A, Bino G, Yosha D, Guter A, King R, Saltz D (2016) Determinants of emigration and their impact on survival during dispersal in fox and jackal populations. *Scientific Reports*, **6**, 24021.

243. Angerbjörn A, Arvidson B, Norén E, Strömgren L (1991) The effect of winter food on reproduction in the arctic fox, *Alopex lagopus*: a field experiment. *Journal of Animal Ecology*, **60**, 705-714.

244. Porteus T, Reynolds J (2020) Killing foxes and controlling fox density: when are they the same thing? *Game & Wildlife Conservation Trust Review*, **51**, 32-35.

245. British Trust for Ornithology (2021) BBS bird population trends. https://www.bto.org/our-science/projects/bbs/latest-results/population-trends.

246. Hounsome T, Delahay R (2005) Birds in the diet of the Eurasian badger *Meles meles*: a review and meta-analysis. *Mammal Review*, **35**, 199-209.

247. Bolton M, Butcher N, Sharpe F, Stevens D, Fisher G (2007) Remote monitoring of nests using digital camera technology. *Journal of Field Ornithology*, **78**, 213-220.

248. Macdonald MA, Bolton M (2008) Predation on wader nests in Europe. *Ibis*, **150**, 54-73.

249. Morris AJ, Gilroy JJ (2008) Close to the edge: predation risks for two declining farmland passerines. *Ibis*, **150 (Suppl. 1)**, 168-177.

250. Kettel EF, Lakin I, Heydon MJ, Siriwardena GM (2021) A comparison of breeding bird populations inside and outside of European badger *Meles meles* control areas. *Bird Study*, **67**, 279-291.

251. Trewby ID, Young R, McDonald RA, Wilson GJ, Davison J, Walker N, Robertson A, Doncaster CP, Delahay RJ (2014) Impacts of removing badgers on localised counts of hedgehogs. *PLoS One*, **9(4)**, e95477.

252. Jackson DB, Green RE (2000) The importance of the introduced hedgehog (*Erinaceus europaeus*) as a predator of the eggs of waders (Charadrii) on machair in South Uist, Scotland. *Biological Conservation*, **93**, 333-348.

253. Jackson DB (2001) Experimental removal of introduced hedgehogs improves wader nest success in the Western Isles, Scotland. *Journal of Applied Ecology*, **38**, 802-812.

254. Jackson DB, Fuller RJ, Campbell ST (2004) Long-term population changes among breeding shorebirds in the Outer Hebrides, Scotland, in relation to introduced hedgehogs (*Erinaceus europaeus*). *Biological Conservation*, **117**, 151-166.

255. Summers RW, Green RE, Proctor R, Dugan D, Lambie D, Moncrieff R, Moss R, Baines D (2004) An experimental study of the effects of predation on the breeding productivity of capercaillie and black grouse. *Journal of Applied Ecology*, **41**, 513-525.

256. Ritchie EG, Johnson CN (2009) Predator interactions, mesopredator release and biodiversity conservation. *Ecology Letters*, **12**, 982-998.

257. Schmidt NM, Ims RA, Høye TT, Gilg O, Hansen LH, Hansen J, Lund M, Fuglei E, Forchhammer MC, Sittler B (2012) Response of an arctic predator guild to collapsing lemming cycles. *Proceedings of the Royal Society of London B*, **279**, 4417-4422.

258. Calladine J, Humphreys EM, Gilbert L, Furness RW, Robinson RA, Fuller RJ, Littlewood NA, Pakeman RJ, Ferguson J, Thompson C (2017) Continuing influences of introduced hedgehogs *Erinaceus europaeus* as a predator of wader (Charadrii) eggs four decades after their release on the Outer Hebrides, Scotland. *Biological Invasions*, **19**, 1981-1987.

259. Wielgus RB, Peebles KA (2014) Effects of wolf mortality on livestock depredations. *PLoS One*, **9(12)**, e113505.

260. Marolla F, Aarvak T, Øien IJ, Mellard JP, Henden J-A, Hamel S, Stien A, Tveraa T, Yoccoz NG, Ims RA (2019) Assessing the effect of predator control on an endangered goose population subjected to predator-mediated food web dynamics. *Journal of Applied Ecology*, **56**, 1245-1255.

261. Baker PJ, Harris S (2005) Does culling reduce fox (*Vulpes vulpes*) density in commercial forests in Wales, UK? *European Journal of Wildlife Research*, **52**, 99-108.

262. Kämmerle J-L, Niekrenz S, Storch I (2019) No evidence for spatial variation in predation risk following restricted-area fox culling. *BMC Ecology*, **19**, 17.

263. Curveira-Santos G, Pedroso NM, Barros AL, Santos-Reis M (2019) Mesocarnivore community structure under predator control: unintended patterns in a conservation context. *PLoS One*, **14(1)**, e0210661.

264. Harding EK, Doak DF, Albertson JD (2001) Evaluating the effectiveness of predator control: the non-native red fox as a case study. *Conservation Biology*, **15**, 1114-1122.

265. Lieury N, Ruette S, Devillard S, Albaret M, Drouyer F, Baudoux B, Millon A (2015) Compensatory immigration challenges predator control: an experimental evidence-based approach improves management. *Journal of Wildlife Management*, **79**, 425-434.

266. Kämmerle J-L, Ritchie EG, Storch I (2019) Restricted area culls and red fox abundance: are effects a matter of time and place? No evidence for spatial variation in predation risk following restricted-area fox culling. *Conservation Science and Practice*, **1**, e115.

267. Rushton SP, Shirley MDF, Macdonald DW, Reynolds JC (2006) Effects of culling fox populations at the landscape scale: a spatially explicit population modeling approach. *Journal of Wildlife Management*, **70**, 1102-1110.

268. Porteus TA, Reynolds JC, McAllister MK (2018) Quantifying the rate of replacement by immigration during restricted-area control of red fox in different landscapes. *Wildlife Biology*, wlb.00416.

269. Porteus TA, Reynolds JC, McAllister MK (2019) Population dynamics of foxes during restricted-area culling in Britain: advancing understanding through state-space modelling of culling records. *PLoS One*, **14 (11)**, e0225201.

270. Madden JR, Hall A, Whiteside MA (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.

271. Chapman EJ, Byron CJ (2018) The flexible application of carrying capacity in ecology. *Global Ecology and Conservation*, **13**, e00365.

272. del Monte-Luna P, Brook BW, Zetina-Rejón MJ, Cruz-Escalona VH (2004) The carrying capacity of ecosystems. *Global Ecology and Biogeography*, **13**, 485-495.

273. Harris S, Trewhella WJ (1988) An analysis of some of the factors affecting dispersal in an urban fox (*Vulpes vulpes*) population. *Journal of Applied Ecology*, **25**, 409-422.

274. Gosselink TE, Piccolo KA, van Deelen TR, Warner RE, Mankin PC (2010) Natal dispersal and philopatry of red foxes in urban and agricultural areas of Illinois. *Journal of Wildlife Management*, **74**, 1204-1217.

275. Walton Z, Samelius G, Odden M, Willebrand T (2018) Long-distance dispersal in red foxes *Vulpes vulpes* revealed by GPS tracking. *European Journal of Wildlife Research*, **64**, 64.

276. Woollard T, Harris S (1990) Comparison of dispersing and non-dispersing foxes (*Vulpes vulpes*) and an evaluation of some dispersal hypotheses. *Journal of Animal Ecology*, **59**, 709-722.

277. Trewhella WJ, Harris S, McAllister FE (1988) Dispersal distance, home-range size and population density in the red fox *(Vulpes vulpes)*: a quantitative analysis. *Journal of Applied Ecology*, **25**, 423-434.

278. Harris S (1977) Distribution, habitat utilization and age structure of a suburban fox (*Vulpes vulpes*) population. *Mammal Review*, **7**, 25-39.

279. Pagh S, Chriél M, Madsen AB, Jensen T-LW, Elmeros M, Asferg T, Hansen MS (2018) Increased reproductive output of Danish red fox females following an outbreak of canine distemper. *Canid Biology & Conservation*, **21**, 12-20.

280. Baker PJ, Funk SM, Harris S, White PCL (2000) Flexible spatial organization of urban foxes, *Vulpes vulpes*, before and during an outbreak of sarcoptic mange. *Animal Behaviour*, **59**, 127-146.

281. Iossa G, Soulsbury CD, Baker PJ, Edwards KJ, Harris S (2009) Behavioral changes associated with a population density decline in the facultatively social red fox. *Behavioral Ecology*, **20**, 385-395.

282. Tuyttens FAM, Delahay RJ, Macdonald DW, Cheeseman CL, Long B, Donnelly CA (2000) Spatial perturbation caused by a badger (*Meles meles*) culling operation: implications for the function of territoriality and the control of bovine tuberculosis (*Mycobacterium bovis*). *Journal of Animal Ecology*, **69**, 815-828.

283. Delahay RJ, Walker NJ, Forrester GJ, Harmsen B, Riordan P, Macdonald DW, Newman C, Cheeseman CL (2006) Demographic correlates of bite wounding in Eurasian badgers, *Meles meles* L., in stable and perturbed populations. *Animal Behaviour*, **71**, 1047-1055.

284. Carter SP, Delahay RJ, Smith GC, Macdonald DW, Riordan P, Etherington TR, Pimley ER, Walker NJ, Cheeseman CL (2007) Culling-induced social perturbation in Eurasian badgers *Meles meles* and the management of TB in cattle: an analysis of a critical problem in applied ecology. *Proceedings of the Royal Society B*, **274**, 2769-2777.

285. Wilkinson D, Bennett R, McFarlane I, Rushton S, Shirley M, Smith GC (2009) Cost-benefit analysis model of badger (*Meles meles*) culling to reduce cattle herd tuberculosis breakdowns in Britain, with particular reference to badger perturbation. *Journal of Wildlife Diseases*, **45**, 1062-1088.

286. Riordan P, Delahay RJ, Cheeseman C, Johnson PJ, Macdonald DW (2011) Culling-induced changes in badger (*Meles meles*) behaviour, social organisation and the epidemiology of bovine tuberculosis. *PLoS One*, **6(12)**, e28904.

287. Dorning J, Harris S (2017) Dominance, gender, and season influence food patch use in a group-living, solitary foraging canid. *Behavioral Ecology*, **28**, 1302-1313.

288. Dorning J, Harris S (2019) Understanding the intricacy of canid social systems: structure and temporal stability of red fox (*Vulpes vulpes*) groups. *PLoS One*, **14(9)**, e0220792.

289. Dorning J, Harris S (2019) Individual and seasonal variation in contact rate, connectivity and centrality in red fox (*Vulpes vulpes*) social groups. *Scientific Reports*, **9**, 20095.

290. Dorning J, Harris S (2019) Quantifying group size in the red fox: impacts of definition, season and intrusion by non-residents. *Journal of Zoology*, **308**, 37-46.

291. gov.scot (2021) Avian influenza (bird flu): how to spot and report the disease. https://www.gov.scot/publications/avian-influenza-bird-flu/.

292. Robertson PA, Mill AC, Rushton SP, McKenzie AJ, Sage RB, Aebischer NJ (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.

293. gov.uk (2020) *Review of gamebird releases on and around European protected sites*. https: //www.gov.uk/government/publications/review-of-gamebird-releases-on-and-around-european-protected-sites.

294. gov.uk (2017) *Rapid evidence assessments*. https://www.gov.uk/government/collections/rapid-evidence-assessments.

295. Gompper ME, Vanak AT (2008) Subsidized predators, landscapes of fear and disarticulated carnivore communities. *Animal Conservation*, **11**, 13-14.

296. Hogstedt G (1981) Effect of additional food on reproductive success in the magpie (*Pica pica*). *Journal of Animal Ecology*, **50**, 219-229.

297. Jeremy Wilson, personal communication, 7th February 2021.

298. Mo SS, Urdahl AM, Madslien K, Sunde M, Nesse LL, Slettemeås JS, Norström M (2018) What does the fox say? Monitoring antimicrobial resistance in the environment using wild red foxes as an indicator. *PLoS One*, **13(5)**, e0198019.

299. National Atmospheric Emissions Inventory (2021) Pollutant information: lead. https://naei.beis.gov.uk/overview/pollutants?pollutant_id=17.

300. Thomas VG (2015) Availability and use of lead-free shotgun and rifle cartridges in the UK, with reference to regulations in other jurisdictions. In: Delahay RJ, Spray CJ (eds) *Lead ammunition: understanding and minimising the risks to human and environmental health*, pp. 85-97. Edward Grey Institute, University of Oxford, Oxford. http://oxfordleadsymposium.info/proceedings/.

301. Draycott R (2018) Improving survival & performance of released pheasants. https://www. pheasant.com/Portals/0/documents/Proceedings/Seminar2018.

302. Hall A (2019) *Improving sustainability and monitoring within the UK pheasant release system*. PhD thesis, University of Exeter.

303. Hall A, Sage R, Madden J (2019) The use of enhanced released pheasants. *Game & Wildlife Conservation Trust Review*, **50**, 38-39.

304. 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.

305. Tyler A (2002) Feathering their nests - the pheasant industry and the missing tax millions. Animal Aid, Tonbridge, Kent. https://www.animalaid.org.uk › uploads › 2017/01.

306. British Game Alliance (2020) The gamebird producer assurance scheme – rearing. https://www.britishgamealliance.co.uk/wp-content/uploads/BGA-REARING-STANDARDS-V10-08.01.20.pdf.

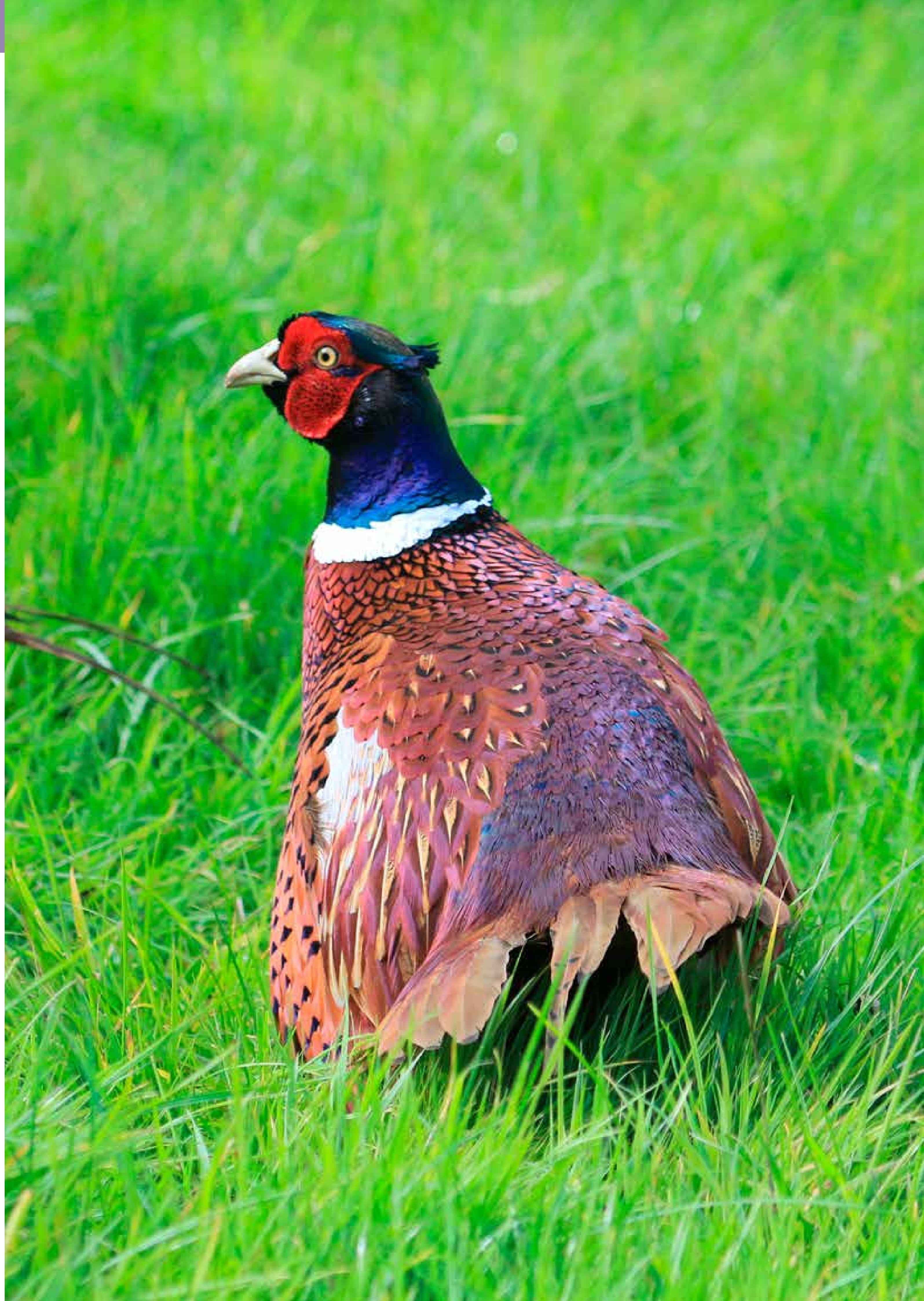
307. Baker P, Furlong M, Southern S, Harris S (2006) The potential impact of red fox *Vulpes vulpes* predation in agricultural landscapes in lowland Britain. *Wildlife Biology*, **12**, 39-50.

308. McLeod LJ, Saunders GR, Miners A (2011) Can shooting be an effective management tool for foxes? Preliminary insights from a management programme. *Ecological Management & Restoration*, **12**, 224-226.

Legend TBC

Legend TBC

Legend TBC



Published by

Labour Animal Welfare Society
30 Station Road, Carlton,
Nottingham, NG4 3AX.

www.labouranimalwelfaresociety.org.uk
/LabourAnimalWelfareSociety/