The impacts of non-native gamebird release in the UK: an updated evidence review
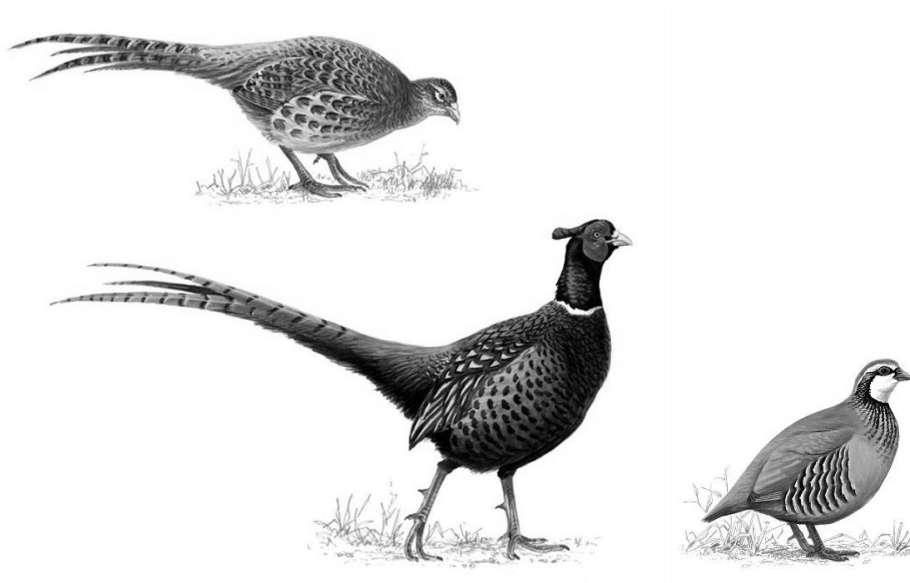
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Contents

LIST OF TABLES AND FIGURES

List of Tables

List of Figures

COMMON ABBREVIATIONS

AES	Agri-Environment Scheme(s)
AIC	Akaike Information Criterion
APHA	Animal & Plant Health Agency
BAP	Biodiversity Action Plan
BASC	The British Association for Shooting and Conservation
BBS	Breeding Birds Survey
BoCC	Birds of Conservation Concern
BTO	British Trust for Ornithology
CITES	Convention on International Trade in Endangered Species
CL	Confidence Limit
DEFRA	Department for Environment, Food and Rural Affairs (UK Government)
EICAT	Environmental Impact Classification for Alien Taxa
FAO/WHO	Food & Agriculture Organisation of the United Nations/ World Health Organisation
FBI	Farmland Bird Index
FTE	Full-time Equivalent
GDP	Gross Domestic Product
GO	Gross Output
GVA	Gross Value Added
GWCT	Game & Wildlife Conservation Trust
IQ	Intelligence Quotient
IQR	Inter-Quartile Range
IUCN	International Union for Conservation of Nature
LR	Likelihood Ratio
MCZ	Marine Conservation Zone
NCMPA	Nature Conservation Marine Protected Area
NGC	National Game Census
NNR	National Nature Reserve
OLM	Ordinal Logistic Model
OLMM	Ordinal Logistic Mixed Model
PACEC	Public and Corporate Economic Consultants
PTWI	Provisional Tolerable Weekly Intake
RSPB	The Royal Society for the Protection of Birds
SAC	Special Area of Conservation
SPA	Special Protection Area
SSSI	Site of Special Scientific Interest
UK	United Kingdom of Great Britain and Northern Ireland
USA	United States of America

SCIENTIFIC NAMES OF SPECIES MENTIONED IN THE TEXT

Birds

Black grouse	*Tetrao tetrix*	Marsh harrier	*Circus aeruginosus*
Blackbird	*Turdus merula*	Marsh tit	*Poecile palustris*
Blackcap	*Sylvia atricapilla*	Meadow pipit	*Anthus pratensis*
Blue tit	*Cyanistes caeruleus*	Merlin	*Falco columbarius*
Bullfinch	*Pyrrhula pyrrhula*	Mistle thrush	*Turdus viscivorus*
Buzzard	*Buteo buteo*	Moorhen	*Gallinula chloropus*
Capercaillie	*Tetrao urogallus*	Northern shoveler	*Spatula clypeata*
Carrion crow	*Corvus corone*	Nuthatch	*Sitta europea*
Chaffinch	*Fringilla coelebs*	Peregrine falcon	*Falco peregrinus*
Chiffchaff	*Phylloscopus collybita*	Pheasant	*Phasianus colchicus*
Coal tit	*Periparus ater*	Pied flycatcher	*Ficedula hypoleuca*
Collared dove	*Streptopelia Decaocto*	Raven	*Corvus corax*
Coot	*Fulica atra*	Red grouse	*Lagopus lagopus scotica*
Corn bunting	*Emberiza calandra*	Red kite	*Milvus milvus*
Corncrake	*Crex crex*	Red-legged partridge	*Alectoris Rufa*
Curlew	*Numenius arquata*	Redstart	*Phoenicurus phoenicurus*
Dunnock	*Prunella modularis*	Reed bunting	*Emberiza schoeniclus*
Fieldfare	*Turdus pilaris*	Robin	*Erithacus rubecula*
Gadwall	*Anas strepera*	Rook	*Corvus frugilegus*
Garden warbler	*Sylvia borin*	Short-eared owl	*Asio flammeus*
Goldcrest	*Regulus regulus*	Siskin	*Carduelis spinus*
Golden eagle	*Aquila chrysaetos*	Skylark	*Alauda arvensis*
Golden plover	*Pluvialis apricaria*	Song thrush	*Turdus philomelos*
Goldfinch	*Carduelis carduelis*	Sparrowhawk	*Accipiter nisus*
Goshawk	*Accipiter gentilis*	Spotted flycatcher	*Muscicapa striata*
Great spotted woodpecker	*Dendrocopos major*	Starling	*Sturnus vulgaris*
Great tit	*Parus major*	Stock dove	*Columba oenas*
Green woodpecker	*Certhia familiaris*	Tawny owl	*Strix aluco*
Greenfinch	*Chloris chloris*	Tree pipit	*Anthus trivialis*
Grey heron	*Ardea cinerea*	Tree sparrow	*Passer montanus*
Grey partridge	*Perdix perdix*	Treecreeper	*Certhia familiaris*
Hawfinch	*Coccothraustes coccothraustes*	Turtle dove	*Streptopelia turtur*
Hen harrier	*Circus cyaneus*	White-tailed eagle	*Haliaeetus albicilla*
Herring gull	*Larus argentatus*	Whitethroat	*Sylvia communis*
Hobby	*Falco subbuteo*	Willow tit	*Poecile montanus*
Hooded crow	*Corvus Cornix*	Willow warbler	*Phylloscopus trochilus*
House sparrow	*Passa domesticus*	Wood warbler	*Phylloscopus sibilatrix*
Jackdaw	*Corvus monedula*	Woodcock	*Scolopax rusticola*
Jay	*Garrulus glandarius*	Woodpigeon	*Columba palumbus*
Kestrel	*Falco tinnunculus*	Wren	*Troglodytes troglodytes*
Lapwing	*Vanellus vanellus*	Yellow wagtail	*Motacilla flava*
Lesser redpoll	*Acanthis cabaret*	Yellowhammer	*Emberiza citrinella*
Linnet	*Sylvia communis*		
Little owl	*Athene noctua*		
Long-eared Owl	*Asio otus*		
Long-tailed tit	*Aegithalos caudatus*		
Magpie	*Pica pica*		
Mallard	*Anas platyrhynchos*		

Invertebrates

Adonis blue	*Polyommatus bellargus*
Brown hairstreak	*Thecla betulae*
Chequered skipper	*Carterocephalus palaemon*
Dark-green fritillary	*Speyeria aglaja*
Duke of Burgundy	*Hamearis lucina*
Essex skipper	*Thymelicus lineola*
High brown fritillary	*Fabriciana adippe*
Large tortoiseshell	*Nymphalis polychloros*
Marbled white	*Melanargia galathea*
Marsh fritillary	*Euphydryas aurinia*
Meadow brown	*Maniola jurtina*
Painted lady	*Vanessa cardui*
Pearl-bordered fritillary	*Boloria euphrosyne*
Purple emperor	*Apatura iris*
Ringlet	*Aphantopus hyperantus*
Silver-spotted skipper	*Hesperia comma*
Silver-studded blue	*Plebejus argus*
Silver-washed fritillary	*Argynnis paphia*
Small blue	*Cupido minimus*
Small pearl-bordered fritillary	*Boloria selene*
White admiral	*Limenitis camilla*
Wood white	*Leptidea sinapis*

Mammals

American mink	*Neovison vison*
Badger	*Meles meles*
Bank vole	*Myodes glareolus*
Common shrew	*Sorex araneus*
Feral cat	*Felis catus*
Field vole	*Microtus agrestis*
Fox (Red fox)	*Vulpes vulpes*
Grey squirrel	*Sciurus carolinensis*
Mole	*Talpa europaea*
Rabbit	*Oryctolagus cuniculus*
Rat (Brown)	*Rattus norvegicus*
Stoat	*Mustela erminea*
Weasel	*Mustela nivalis*
Wood mouse	*Apodemus sylvaticus*

Reptiles

Adder	*Vipera berus*
Grass snake	*Natrix helvetica*
Sand lizard	*Lacerta agilis*
Slow-worm	*Anguis fragilis*
Common lizard	*Zootoca vivipara*

Plants

Annual meadow grass	*Poa annua*
Bracken	*Pteridium aquilinum*
Bramble	*Rubus fruticosus*
Broad-leaved dock	*Rumex obtusifolius*
Broadleaved spurge	*Euphorbia platyphyllos*
Chickweed	*Stellaria media*
Common honeysuckle	*Lonicera periclymenum*
Common nettle	*Urtica dioica*
Corn chamomile	*Anthemis arvensis*
Corn parsley	*Petroselinum segetum*
Creeping buttercup	*Ranunculus repens*
Dense silky bent	*Apera interrupta*
Dense-flowered fumitory	*Fumaria densiflora*
Dwarf spurge	*Euphorbia exigua*
Field gromwell	*Buglossoides arvensis*
Field woundwort	*Stachys arvensis*
Greater stitchwort	*Stellaria holostea*
Hawthorn	*Crataegus monogyna*
Herb robert	*Geranium robertianum*
Ivy	*Hedera helix*
Kale	*Brassica oleracea*
Meadow foxtail	*Alopecurus pratensis*
Millet	*Panicum miliaceum*
Narrow-fruited cornsalad	*Valerianella dentata*
Night-flowering catchfly	*Silene noctiflora*
Perennial ryegrass	*Lolium perenne*
Prickly poppy	*Papaver argemone*
Quinoa	*Chenopodium quinoa*
Red hemp-nettle	*Galeopsis angustifolia*
Rough poppy	*Argemone hispida*
Round-leaved fluellen	*Kickxia spuria*
Sharp-leaved fluellen	*Kickxia elatine*
Shepherd's needle	*Scandix pecten-veneris*
Spear thistle	*Cirsium vulgare*
Triticale	*X Triticosecale*
Tufted hair grass	*Deschampsia cespitosa*
Velvet feather-moss	*Brachythecium velutinum*
Venus's looking glass	*Legousia hybrida*
Wood avens	*Geum urbanum*
Wood sorrel	*Oxalis acetosella*
Yellow archangel	*Lamium galeobdolon*
Yellow pimpernel	*Lysimachia nemorum)*
Yorkshire fog	*Holcus lanatus*

SYNOPSIS

1.1   Executive Summary

Relevance to conservation
- Recreational shooting of non-native gamebirds (pheasants and red-legged partridges) in the UK is underpinned by a suite of management practices. A major aspect is the annual large-scale release of birds to be shot, which is steadily increasing with an estimated 57 million birds released in 2016; this scale of non-native gamebird release exceeds any similar activity in Europe or North America. Approximately one third of released birds are shot, while the majority of the remaining birds die from other causes during the year of release or survive in very low numbers to join the population of pheasants and partridges now residing in the wild in the UK.
- The number of gamebirds released has increased since the 1990s, while the number shot has remained relatively stable, and this reduction in releasing-efficiency is potentially driven by a reduction in gamebird survival and an increase in late winter shooting, meaning that more birds must be released during autumn to ensure enough survive to be shot in January.
- There is concern that this large and increasing release of gamebirds and associated shooting practices may be having negative impacts on the UK's native wildlife. However, there may also be positive ecological, economic and social impacts of gamebird release activities.

Review methods
- We conducted a review of published peer-reviewed and grey literature to assess the evidence for ecological (wildlife and habitat) and socio-economic (human) impacts of gamebird release in the UK.
- We scored ecological impacts on a scale from -2 to 2, where +/-2 indicated strong positive/negative evidence for a population-level impact on native UK wildlife, +/-1 indicated weaker evidence or a more local impact, and 0 indicated no evidence of any impact. Ecological impact scores were summarised within 19 secondary impact themes under six primary impact themes (game estate management, shooting practices, illegal persecution, direct impacts of gamebirds, disease transmission to wildlife and impacts on predators and predation).
- Socio-economic impacts were summarised using a 'vote-counting' method indicating the number of sources evidencing positive, benign or negative effects for six associated impact themes (economic value, employment, social cohesion and wellbeing for the shooting community, lead consumption in humans, disease transmission to humans and vehicle and aviation accidents caused by non-native gamebirds).

Key findings: Ecological impacts
- There was more evidence for negative ecological impacts of gamebird release than for positive impacts. The average impact scores for five of the six primary impact themes were negative, while one theme (game estate management) was associated with a positive average score. Of the secondary themes, 13 median scores were negative (68% of 19 themes), while 6 were positive or benign (32%).
- **POSITIVE** impacts were primarily associated with the secondary benefits of habitat management, supplementary feeding and legal lethal predator control on sites managed for gamebirds, affecting a wide range of taxonomic groups including plants, invertebrates, birds and mammals, often on a local (site) scale for groups other than birds.
- Most other impacts were **NEGATIVE**, affecting a wide range of taxa and with several having the potential to influence native wildlife populations beyond the boundaries of gamebird estates. The extent of the impacts was often dependent on the densities at which gamebirds were released, with higher density releases associated with more negative impacts. The strongest evidence for negative impacts was associated with direct impacts of gamebird release (browsing of vegetation

and predation of invertebrates by gamebirds) and current shooting practices (the use of lead ammunition).

Key findings: Socio-economic impacts
- There was evidence for positive, benign and negative socio-economic impacts of gamebird release, although a higher proportion of sources was associated with negative impacts.
- **POSITIVE** impacts include economic and employment benefits as well as social wellbeing, particularly for the shooting community and supporting industries.
- **NEGATIVE** impacts were associated with human health issues arising from the current use of lead ammunition, the potential for gamebirds to act as vectors of disease and human and economic costs associated with vehicle and aviation collisions.

Knowledge gaps and recommendations for further research
- A national database where the number and geographical distribution of all non-native gamebirds released annually is comprehensively and accurately recorded is urgently needed, as are further investigations to better understand the ecological and socio-economic drivers behind the rapid and steady increase in the numbers of gamebirds released since the 1960s.
- The direct impacts of released gamebirds themselves are potentially numerous and diverse but are particularly underrepresented in the peer-reviewed literature and require further study. Priority subject areas include impacts of predation by pheasants on amphibians and reptiles, and of resource competition with native birds.
- There is an urgent need for new replicated landscape scale field studies to test whether releases of non-native gamebirds enhance the local abundance of generalist predators (especially mammals), and consequentially increase predation-related pressure on ground-nesting birds.

1.2   Background

The number of non-native gamebirds (ring-necked pheasants and red-legged partridges) released on lowland shooting estates into the UK countryside has been increasing since 1960, with a 4.3% annual increase in the density of birds released per site between 1960 and 2014 (Robertson *et al.* 2017) and a 38% increase in the total number of birds released since 2004, with an estimated 57 million birds released in 2016 (Aebischer 2019b). The scale of these releases greatly exceeds any similar releasing of non-native birds for hunting elsewhere in Europe or North America (Arroyo and Beja 2002, Mustin *et al.* 2012).

Pheasants and red-legged partridges together comprise 82% of the 24 million birds of all species shot annually in the UK (Aebischer 2019b). The ratio of the number of gamebirds released to the number shot (hereafter efficiency of rear-releasing), which remained relatively stable from 1960 until 1990, has since suffered a steep decline (Robertson *et al.* 2017). This is because the number of birds shot has remained relatively constant since the 1990s while releases have increased, with only 34% of the gamebirds released (32% of released pheasants, 46% of released red-legged partridges) now shot during the autumn and winter (Aebischer 2019b). The increase in numbers of birds released despite the relative stability in the numbers shot may be linked to a reduction in pheasant survival combined with an increase in late winter shooting, which may have driven the need to release more gamebirds the preceding autumn to ensure enough survive to shoot in January (Robertson *et al.* 2017).

Pheasants also account for 82% of the gamebirds released, and these add to the 4.4 million pheasants which currently breed wild in the UK. At the point of release in autumn, released and naturalised pheasants and red-legged partridges together represent more than twice the spring biomass of all native UK breeding birds combined and also more than the post-breeding native bird biomass (estimated from Blackburn and Gaston 2018, and Aebischer 2019b). The potential negative impact that

this large and increasing influx of non-native birds may be having on the UK's native wildlife is of some concern, particularly because many of these potential impacts are poorly studied or not well understood. Some impacts of gamebird release may be positive, particularly those associated with management of some semi-natural habitats. Shooting is also considered a leisure activity which may provide positive social, economic and employment benefits to the shooting community and supporting industries.

Here we review the impacts of gamebird release in the UK, both ecologically (impacts on native UK wildlife, habitats and environment) and socio-economically (impacts on humans), building on and updating a previous review conducted by Bicknell *et al.* (2010).

1.3 Methods

We conducted a literature search for published peer-reviewed and grey-literature sources relating to impacts of gamebird release in the UK to update the literature database collated by Bicknell *et al.* in 2010. Studies relating to other forms of managed shooting (such as red grouse management, wildfowling or 'rough shooting' of e.g. snipe and woodcock) were excluded from this review unless they explicitly documented impacts relating to the release of pheasants or red-legged partridges.

The ecological impacts evidenced by sources were scored using a two-dimensional scoring system which incorporated both the strength of evidence for an effect on populations or communities of any native UK wildlife and the reliability of study design (scoring method details are described in section 3.3.1). Scores were measured on an ordinal scale from -2 to 2, with +/-2 indicating strong evidence for a population-level impact, +/-1 indicating weaker evidence or a more local impact, and 0 indicating no evidence of any impact. Sources that evidenced multiple impacts generated multiple scores, one for each impact. Scores were categorised into six primary impact themes (Table 1), and 19 secondary impact themes based on those presented by Bicknell *et al.* (2010).

Table 1. The six primary impact themes into which scores for the ecological impacts of gamebird release were categorised.

Primary theme	Description
Game estate management	Impacts relating to the management of habitats and gamebirds on lowland game sites, including farmland and woodland habitat management for gamebirds, supplementary feeding of gamebirds, legal control of predatory mammals, corvids and buzzards (under specific licenses) and rodent pest control.
Shooting practices	Impacts of accidental shooting of non-target species during gamebird shooting, and the use of lead ammunition.
Illegal persecution	Impacts relating to the illegal killing of protected species on lowland game shooting estates.
Direct impacts of gamebirds	Impacts of herbivory or predation by gamebirds on other wildlife, direct competition between gamebirds and wildlife, or secondary impacts of changes in soil chemistry linked to accumulation of gamebird faeces.
Disease transmission to wildlife	Gamebirds as vectors of disease to native UK species.
Impacts on predators and predation	Impacts of gamebirds as a supplementary prey source for predators and consequential impacts on predator abundance and predation pressure on other prey species.

Impacts of gamebird release on predators or predation were scored from the perspective of predation pressure on prey populations: negative impacts on predator abundance from gamebird management (e.g. legal lethal control) were considered likely to result in reductions in predation pressure (a positive impact) so were scored positively; in contrast, potential positive effects on predator abundance (e.g. from gamebirds as a supplementary food source) may result in increases in predation pressure (a negative impact for prey populations) so were scored negatively. This was to ease interpretation of score statistics and figures. Average (mean) scores for each primary impact theme were compared using an ordinal logistic regression model (OLM) which down-weighted grey-literature (relative to peer-reviewed literature) and indirect or potential impacts (relative to directly measured impacts). Small sample sizes precluded OLM analysis for secondary themes, so comparisons were made for these using median scores.

Socio-economic impacts were not scored in the same manner as it was difficult to define thresholds that usefully reflect the importance of different impacts. Socio-economic impacts were therefore summarised using a 'vote-counting' method similar to that used by Bicknell *et al.* (2010), indicating the number of sources evidencing positive, benign or negative effects for six associated subthemes.

1.4   Results overview

Ecological impacts

Overall there was a significant difference among primary impact themes in the evidence for impacts of gamebird release (OLM, LR χ^2 = 100.3, df = 5, P < 0.0001; Fig. 1). Game estate management was the only primary impact theme associated with a significantly positive average score; the average impact scores for the other five primary impact themes were all significantly negative (Fig. 1).

Table 2 summarises the scores and impacts associated with each primary and secondary impact theme. Of the secondary themes, 13 median scores were negative (68% of 19 themes), while five were positive (26%) and one was benign (5%). No secondary themes were associated with entirely positive scores: evidence for positive impacts always existed alongside evidence for either benign (one secondary theme, 5%), or both benign and negative impacts (six secondary themes, 32%). In contrast, evidence for negative impacts existed in the absence of benign or positive impacts for seven secondary themes (37%), or alongside evidence for benign impacts for five secondary themes (26%).

Positive impacts were mainly associated with habitat management on game shooting estates, with median impact scores of +1 for five of the six secondary impact themes associated with this primary theme (Table 2). These positive impacts were linked to the management of arable farmland and woodland to the benefit of taxa other than gamebirds, the retention of more semi-natural habitats on game estates, the provision of grain as supplementary gamebird feed and legal lethal predator control. Many of these positive impacts would have been restricted to the sites or estates on which the management was implemented.

Negative impacts were associated with current shooting practices (particularly the use of lead ammunition), illegal persecution of protected species, direct impacts of the gamebirds themselves (browsing and herbivory of vegetation, predation and competition with native wildlife), the potential for disease transmission and potential impacts on the abundance of generalist predators and subsequent predation pressures (Table 2). Several of these impacts were known, or had the potential, to affect native wildlife beyond the sites or estates on which gamebirds were released.

Impacts varied between ecological groups: native UK birds were most commonly associated with positive impacts of gamebird release, while the impact on predators and associated predation pressure

was usually negative. Birds, plants, invertebrates, mammals and predators all experienced both positive and negative impacts from gamebird releasing. However, the evidence relating to impacts on reptiles and amphibians was entirely negative.

	Game estate management	Shooting practices	Illegal persecution	Direct impacts of gamebirds	Disease transmission to wildlife	Impacts on predators and predation
No. of scores	125	16	18	42	13	19
No. of sources	59	12	13	20	13	17
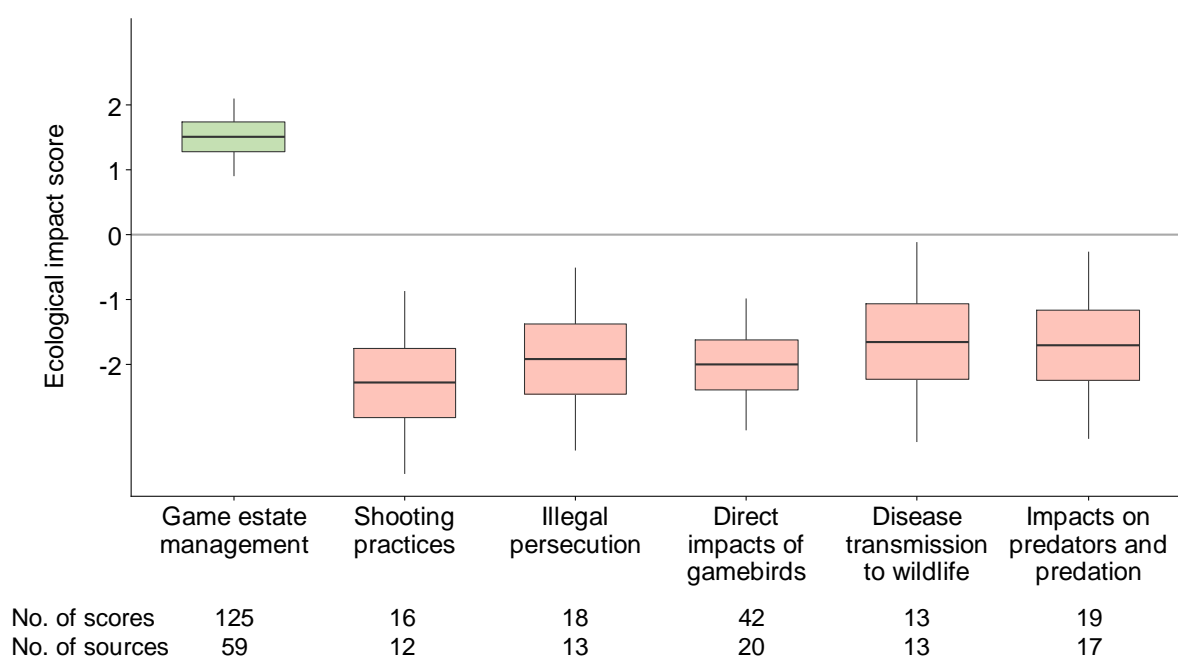
Fig. 1. The ecological impact of non-native gamebird releasing and shooting was mainly negative across the six primary impact themes studied (Table 1). Horizontal lines within boxes are estimated marginal means predicted by an ordinal logistic model (OLM), boxes are ±1 standard error and whiskers are 95% confidence intervals. There were no significant differences in mean score between any of the primary impact themes ($P \geq 0.05$ in all cases), with the exception of 'Game estate management' where the mean differed significantly from that of all other themes ($P < 0.05$ in all cases). All mean scores were significantly different from zero ($P < 0.05$). Any impacts on predators and predation (within 'Impacts on predators and predation', and legal control within 'Game estate management') were scored from the perspective of predation pressure, i.e. negative impacts on predator populations are likely to result in lower predation pressure and therefore positive impacts on prey species, and so have been scored positively (and *vice versa*). The number of scores and sources associated with each primary theme are provided below the plot; some sources were associated with scores for multiple themes.

Socio-economic impacts

The vote counting exercise for socio-economic impacts associated with gamebird release indicates that there is more evidence of negative socio-economic impacts than benefits: with 27 sources indicating negative effects, 1 source indicating a benign effect, and 10 sources indicating positive effects (Table 3). There is no doubt that gamebird release is valuable economically and socially, and provides employment opportunities to those supporting or directly involved in the shooting industry; however, the available published peer-reviewed and grey literature sources highlighting these benefits are relatively few in number. In contrast, the bulk of the published literature surrounding socio-economic impacts focuses on the impacts relating to lead consumption by humans, the potential for disease transmission and vehicle collisions, which are all inherently negative.

1.5   Conclusions

Ecological impacts

POSITIVE ecological impacts of gamebird release are largely restricted to the secondary benefits of gamebird management on the habitat quality of arable farmland and woodland, and are also associated with supplementary feeding and legal lethal predator control. These benefits affect a wide range of taxonomic groups including plants, invertebrates, birds and mammals, and tend to be local in scale for groups other than birds, largely influencing local populations on the sites on which they are implemented. A high proportion of the available literature and evidence is associated with these positive impacts.

Most other ecological impacts are **NEGATIVE**, with the evidence base strongest for direct impacts of gamebirds (browsing of vegetation, predation of invertebrates, competition) and current shooting practices (pollution and contamination from lead ammunition). Negative impacts affect a wide range of taxa and are often dependent on the densities at which gamebirds are released, with higher density releases associated with more negative effects. Negative impacts are often evident outside as well as inside the boundaries of gamebird estates, for example through direct impacts of gamebirds dispersing onto neighbouring land, the spread of disease, lead accumulation through food chains, or potential impacts on the abundance of wide-ranging generalist predators. The extent and strength of evidence underpinning most of these negative impacts is limited due to a lack of targeted studies, as reflected in the generally lower number of scores and sources (Fig. 1, Table 2), and further research is required to confirm the extent and magnitude of the effects summarised in this report (Table 2).

Socio-economic impacts

Vote counting of available evidence also indicates a mixture of positive and negative socio-economic impacts. Although sources evidencing negative impacts were more numerous, this probably reflects the topics where recent research has been concentrated, rather than a true indication of the relative importance of positive *versus* negative impacts.

POSTIVE IMPACTS are associated with economic benefits, employment and social wellbeing for UK shooting participants, other members of the shooting community and supporting industries.

NEGATIVE IMPACTS are associated with human health issues arising from lead contamination in gamebird meat (primarily impacting the shooting community), as well as the potential for disease transmission to humans and costs associated with vehicle and aviation collisions.

1.6   Key knowledge gaps requiring further study

The UK lacks a national database where the number and geographical distribution of all non-native gamebirds released annually is comprehensively and accurately recorded. Such data are fundamental for any assessment of the extent and magnitude of the ecological or socio-economic impacts of gamebird releasing, and are therefore urgently needed. Further investigations are also urgently required to better understand the ecological and socio-economic drivers behind the rapid and steady increase in the numbers of gamebirds released since the 1960s, without which improvements in the sustainability of gamebird releasing are unlikely to be achievable.

In addition, the ecological impact themes that are particularly under-represented in the peer-reviewed literature include:

Impacts on predators and predation
No studies have yet quantified the impacts of released gamebirds on the abundance of generalist mammalian predators, particularly whether the provision of released gamebirds as a supplementary food source may be increasing the environmental carrying capacity for mammalian predators such as foxes. There is an urgent need for new field studies to test whether releases of gamebirds (1) enhance the local abundance of generalist predators (birds and mammals), and (2) increase predation pressure on ground-nesting birds by increasing the abundance of generalist predators.

Direct impacts – predation, browsing and habitat modification by gamebirds
A large body of anecdotal evidence from individuals and organisations monitoring reptiles and amphibians suggests that pheasant releasing has contributed to the local decline and disappearance of some species. However, there are no conclusive or large-scale studies investigating the impact of pheasants on the populations of reptiles and amphibians. This topic would benefit from both a large-scale study (national or regional) examining the relationship between the trends in spatial distribution of reptile and amphibian species and gamebird releasing activities, and more intensive local or small-scale captive experimental studies examining changes in reptile and amphibian abundance in relation to pheasant release density or patterns of gamebird habitat use.

Direct impacts - resource competition
Recent studies have highlighted that the biomass of released and naturalised gamebirds is exceptionally high relative to that of other UK breeding birds, but no studies yet examine the impact this may be having on native UK bird populations through potential competition for resources such as food or space.

Illegal persecution
There is only one peer-reviewed UK study examining the impact of illegal raptor persecution linked with pheasant or red-legged partridge shooting activities on the population of a protected raptor (red kite). It would be useful to use collated records of illegal persecution of other protected species (particularly buzzard), identify any geographic or land use associations with this persecution, and whether this persecution may be impacting on populations of protected species at a local scale.

Impacts from the use of lead ammunition
The negative ecological and human-health impacts associated with the use of lead ammunition may reduce in future years if the voluntary call made by shooting organisations for the phasing out of lead ammunition by 2025 is acted upon by the shooting community. Further research is needed to understand the impacts of lead pollution on terrestrial UK wildlife and levels of lead contamination in gamebird meat for human consumption should be carefully monitored.

1.7   Impact theme summary tables

Table 2: Ecological impacts

Table 2. Summary of the ecological impacts of gamebird release associated 19 secondary impact themes under six primary impact themes. For each secondary impact theme, coloured circles indicate the number of impact scores associated with each score level (-2 to 2), with the circle diameter proportional to the number of scores (bigger circles therefore indicate stronger evidence for impacts at a given level). A score of +/-2 indicates strong evidence for a population-level impact on native UK wildlife, +/-1 indicates weaker evidence or a more local impact, and 0 indicates no evidence of any impact. Black-bordered circles and coloured shading indicate the average (median) score level for each secondary theme (two score levels are shaded if the median sits between them). Sources that evidenced multiple impacts generated multiple scores, one for each impact. Note that the ecological impacts for predators within both the *Impacts on predators and predation* primary theme and the *Game estate management: Legal predator control* secondary theme were scored from the perspective of predation pressures on prey populations, i.e. if a source suggested a positive impact on predator abundance then this was given a negative score, as increasing predator abundance is likely to negatively impact on prey populations through increasing predation pressure, and *vice versa*. For more detail of sample sizes, median scores and impacts see the relevant sections in the Main report.

Theme	-2	-1	0	1	2	Summary impact notes and species groups affected
GAME ESTATE MANAGEMENT						
Farmland management		●	●	◯	●	Positive effects of conservation headlands, cover crops, beetle banks, grassy margins and hedgerow management on abundance, breeding success, brood size and species richness of farmland passerine birds, grey partridge, invertebrates including butterflies and bumblebees, rare arable weeds and hedgerow plant communities. **32 impact scores (median = 1) evidenced by 20 sources**
Woodland management	●	●	◯	◯	●	Management of woodland to increase light levels, coppicing and maintaining open rides and glades leads to overall positive effects such as higher abundance and species richness of butterflies, woodland birds (e.g. nightingale, warblers), small mammals and detritivore invertebrates. Areas immediately surrounding gamebird release pens and woods with higher density releases are negatively affected however, with lower plant species richness, changes in plant community composition, lower tree seedling regeneration and lower carabid beetle abundance. **46 impact scores (median = 0.5) evidenced by 17 sources**

Theme	-2	-1	0	1	2	Summary impact notes and species groups affected
Woodland creation and retention			●	◯	●	A large proportion of gamebird estates plant new woodlands and manage existing woodlands to benefit gamebirds. Increases in bird abundance occur on farms which plant woodland in addition to other game-management activities. Other woodland wildlife likely to benefit from increased habitat provision. **8 impact scores (median = 1) evidenced by 6 sources**
Supplementary feeding		●	●	◯	●	Many species of farmland and woodland seed eating birds, thrushes and small mammals (rodents, lagomorphs) utilise supplementary feed for gamebirds in winter leading to local increases in abundance or national population increases (Columbids and corvids). Feeders may however represent a point of disease transfer (e.g. higher incidence of infection with Trichomonosis in Columbids at farms providing food for gamebirds), and may increase the abundance of non-native rodent predator species (e.g. brown rats, grey squirrels). **15 impact scores (median = 1) evidenced by 9 sources**
Legal predator control		●	●	◯	●	Legal lethal predator control suppresses local fox and corvid abundance which may reduce predation pressure on prey species. Such legal predator control of foxes and corvids, when conducted as part of a wider suite of game estate management, results in increases in hatching success, fledgling survival and/or breeding abundance for ground-nesting birds, particularly farmland passerine birds and waders. Also benefits to mammalian prey species such as brown hare. **22 impact scores (median = 1) evidenced by 18 sources**
Rodent pest control	◯	◯				Use of rodenticides to control rats at gamebird feeders is widespread (91% of gamekeepers) and leads to mortality and significant local population declines of wood mice, bank voles and field voles. Potential for residue accumulation through the food chain if contaminated small mammals are consumed by predators/scavengers. **2 impact scores (median = -1.5) evidence by 2 sources**

Theme	-2	-1	0	1	2	Summary impact notes and species groups affected
SHOOTING PRACTICES						
Accidental shooting of non-target species	●	○				Local population declines of grey partridge associated with large-scale releasing of red-legged partridge and pheasant, due to high unintentional shooting mortality during pheasant/red-legged partridge shoots. **2 impact scores (median = -1) evidenced by 3 sources**
Direct ingestion of lead by wildlife	●	○				Spent lead ammunition ingested by Galliformes (particularly grey partridge), pigeons, doves and waterfowl leads to poisoning and direct mortality as well as sub-lethal illness and welfare impacts. **5 impact scores (median = -1) evidenced by 5 sources**
Environmental lead concentrations	●	○				Estimated 1,760–4,700 tonnes of lead ammunition deposited over terrestrial habitats in the course of pheasant and red-legged partridge shooting annually. In areas of intensive shooting lead can be taken up by some plants and absorbed by earthworms, in turn ingested and absorbed by small mammals, which are in turn consumed by predators, enabling lead transfer up through the food chain. **3 impact scores (median = -1) evidenced by 2 sources**
Secondary poisoning of predators	●	○				Secondary poisoning of raptors, other generalist predators and scavengers feeding on prey which has been shot with or has ingested lead ammunition, often with lethal effects. **5 impact scores (median = -1) evidenced by 5 sources**
ILLEGAL PERSECUTION						
Illegal persecution	●	○				Some gamekeepers kill protected predators including buzzard, red kite, raven and badger by trapping, shooting or using poisoned bait. Most sources document anecdotal records, and population-level effects on the target species are largely unknown, but these activities are nationally widespread with multiple cases reported annually. **18 impact scores (median = -1) evidenced by 13 sources**

Theme	-2	-1	0	1	2	Summary impact notes and species groups affected
DIRECT IMPACTS OF GAMEBIRDS						
Browsing by gamebirds	●	◯	●	●		Pheasants are omnivorous: physical damage to plants from pecking can negatively influence sward height, botanical species richness and community composition, ground cover, hedge/shrub leaf density and cause the exclusion of some plant species, particularly in woodlands containing release pens, along hedgerows and on sensitive botanical sites close to release areas (e.g. chalk grasslands). Changes to hedge structure may modify nesting habitat for some birds and cause a reduction in host plants for some lepidopterans. Some positive impacts on cover of bramble and grasses in coniferous pheasant release woods. **12 impact scores (median = -1) evidenced by 9 sources**
Predation by gamebirds	●	◯	●			Predation of a wide range of ground-active invertebrates, amphibians and reptiles may cause local population declines and/or changes in community structure particularly close to high-density pheasant release sites. **14 impact scores (median = -1) evidenced by 10 sources**
Resource competition	●	◯	●			Biomass of released and naturalised gamebirds exceeds that of all other UK breeding birds, and pheasants account for a disproportionate amount of the total energy requirement of the breeding bird assemblage. Tree sparrow abundance is negatively correlated with pheasant release intensity during the previous year, possibly due to resource competition for winter seed resources. **4 impact scores (median = -1) evidenced by 4 sources**
Soil enrichment	●	◯	●	●	●	Gamebirds at high densities increase soil phosphate, potassium and nitrogen levels though droppings. This changes plant species composition and sward structure in woodlands and hedgerows, and threatens rare bryophyte species in moorland fringe habitats. Woodland bryophytes may respond negatively to increased nitrogen in the air, but soil enrichment increases detritivore invertebrate abundance. **12 impact scores (median = -1) evidenced by 8 sources**

20

Theme	-2	-1	0	1	2	Summary impact notes and species groups affected

DISEASE TRANSMISSION TO WILDLIFE

Disease transmission to wildlife — Studies that demonstrate or imply the transfer of pathogens from non-native gamebirds to native wildlife are few, mainly due to the difficulties associated with establishing the direction of infection. Pheasants act as carriers of Newcastle disease and Avian Influenza, with potential for transmission and mortality in wild native bird species. Pheasants are also important in maintaining the transmission of Lyme disease in lowland woods, and are carriers of pathogenic nematodes including *Heterakis gallinarum* which causes Histomonosis in grey partridges and may contribute to their population decline (although evidence for this is mixed). Gamebirds and supplementary feeding may also aid in the transmission of Trichomonosis to pigeons and doves.
13 impact scores (median = -1) evidenced by 13 sources

IMPACTS ON PREDATORS AND PREDATION

Food source for predators — Pheasants and red-legged partridges contribute to the diets of generalist predators and scavengers, particularly foxes which account for a substantial proportion of post-release pheasant mortality.
12 impact scores (median = -1) evidenced by 12 sources

Predator abundance — Abundance and biomass of released and free-roaming gamebirds is positively associated with the abundance and population increase of generalist avian predators and protected raptors including buzzard, crow, magpie and raven.
4 impact scores (median = -1) evidenced by 4 sources

Predation rates — Predation pressure from high and increasing abundance of generalist predators is limiting populations of ground-nesting birds. At the end of the shooting season, generalist predators may switch to other prey sources, coinciding with the nesting season of native birds (although we found no sources evidencing this effect). One study found no evidence that releasing pheasants increased predation pressure by foxes on grey partridge.
3 impact scores (median = 0) evidenced by 3 sources

Table 3: Socio-economic impacts

Table 3. A summary of the socio-economic impacts of gamebird release associated with six secondary impact themes. Coloured circles indicate the number of sources presenting evidence for negative, benign or positive effects ("vote counting"), with circle diameter proportional to the number of sources. Circle size does not indicate the relative importance of impact themes or effect direction. For more detail of methods, sample sizes and impacts see the relevant section in the Main report.

	Negative	Benign	Positive	Summary impact notes
SOCIO-ECONOMIC IMPACTS				
Economic value	●		●	Shooting activities (including non-game shooting in addition to pheasant and red-legged partridge shooting, and including all levels of the shooting supply chain) contributed approximately £2 billion to the UK economy in 2012/13 (Gross Value Added) although this value is disputed and may be considerably less (£267–£480 million). Surveys of shooting participants suggest that should all UK shooting activities cease, shooting participants would travel abroad to shoot and spend considerably less on alternative UK leisure activities, suggesting there would be an overall loss of leisure income to the UK economy. Some arable farmers report losing profit due to crop damage from pheasants. **# sources = 3**
Employment			●	Shooting (including other shooting types in addition to pheasant and red-legged partridge shooting) supports a total of 74,000 full-time equivalent jobs in the UK economy: 35,000 directly, 39,000 indirectly. **# sources = 2**
Social-cohesion and wellbeing for the shooting community			●	Estimated 430,000 people (0.7% of UK population of 64 million) participated in some form of driven or walked-up game shooting in 2012/13. 97% of shooting participants agree that shooting contributes to their wellbeing, and 87% agree that shooting contributes to the social fabric of the local area. We found no studies that assessed the impacts of shooting on wellbeing or other social factors from a representative sample of the population however (i.e. including both shooters and non-shooters). **# sources = 2**
Lead consumption in humans	●	●		There is a risk of human exposure to high lead concentrations when consuming gamebird meat, both from undetectable fragments of lead ammunition used to shoot the gamebird, and from lead accumulated in the meat and bones as a result of gunshot ingested during the gamebirds' lifetime. Lead levels associated with high-level consumption (>1–2 gamebird meals per week) have been linked to reductions in IQ and writing ability in children, and high systolic blood pressure, chronic kidney disease and high rates of spontaneous abortion in adults. **# sources = 12**

	Negative	Benign	Positive	Summary impact notes
Disease transmission to humans	●			Pheasants are reservoirs for tick-borne Lyme disease and may increase the risk of Lyme disease exposure in humans, especially in lowland woodlands. Gamebirds are also carriers of low-pathogenicity Avian Influenza, which could potentially mutate to a high-pathogenicity strain and pose a risk to humans. Low risk of human infection by *Campylobacter*, *Toxoplasma gondii*, *Salmonella* and *E. coli* from hunting, storage, preparation and consumption of gamebirds. Risks of antibiotic resistance developing through the indiscriminate use of antibiotics in gamebird rearing practices is starting to diminish due to veterinary, shooting industry and stakeholder initiatives to reduce their usage. **# sources = 8**
Vehicle and aviation accidents caused by non-native gamebirds	●			Approximately 2.4–6.1 million pheasants may be killed on the roads each year. Pheasants were reported as causing 87 road traffic accidents (12%) across 14 English counties 1999–2003, 5 (6%) of which resulted in human mortality or serious injury. Pheasants are also estimated to cost the British aviation industry at least £300,000 in damage annually; at least 1/3 of the total cost associated with non-native bird strikes. **# sources = 6**

MAIN REPORT

2   BACKGROUND

2.1.1   Annual release of non-native gamebirds

Non-native gamebird species

Two species of non-native gamebird are now commonly released for sport-shooting in the UK: ring-necked pheasant *Phasianus colchicus* (hereafter "pheasant") and red-legged partridge *Alectoris rufa* (hereafter "red-legged partridge"). Unless otherwise specified, the term "gamebird" is used in this report to refer to non-native pheasants and red-legged partridges.

The red-legged partridge has a small native range encompassing Spain, Portugal, Andorra, southern France and northern Italy (BirdLife International 2020a), and was first introduced into the UK around 1770 (Lever 1977). The native range of the pheasant and its ~30 subspecies extends throughout the central Asian continent from the Black Sea in the east to western China, but it is naturalised as an introduced species throughout much of Europe (Avery 2019, BirdLife International 2020b). By the 12th Century pheasants were already being hunted and eaten in England, largely through falconry rather than shooting, indicating that small scale releases and naturalisation had begun by this time (Lever 1977, Madden and Sage 2020). By the early 1800s pheasants had become an important quarry species for shooting, and were being introduced across much of western Europe including further releases in the British Isles (Hill and Robertson 1988). At least six pheasant subspecies are now found in the UK, with the breeding and released populations likely consisting of a conglomerate of numerous phenotypes (Hill and Robertson 1988). Further details about the history of pheasant and red-legged partridge releasing are provided in the recent review by Madden and Sage (2020).

Numbers released

The numbers of these gamebirds released on lowland shooting estates into the UK countryside has been increasing since 1960 (Fig. 2), with a 4.3% increase in the average annual density of birds released per site between 1960 and 2014 (Robertson *et al.* 2017) and an estimated 38% increase in the total number of birds released since 2004 (Aebischer 2019b). The number of pheasants now released annually is estimated to be 10 times higher than in 1961, and the number of red-legged partridges released is 220 times higher (Fig. 2; Aebischer 2019a). Madden and Sage (2020) suggest that the total number released annually is likely to be somewhere between 11–70 million birds (10–57 million pheasants and 1–13 million red-legged partridges) though closer to the upper end of that range, with the most recent published estimate being 57 million birds released in total in 2016, of which 47 million (82%) were pheasants and 10 million (18%) were red-legged partridges (Fig. 2; Aebischer 2019b).

The scale of these releases greatly exceeds any similar releasing of non-native or native gamebirds for hunting elsewhere in Europe or North America (Arroyo and Beja 2002, Mustin *et al.* 2012, Burden 2013). For European countries where these data exist, the second largest annual release of pheasants compared to the UK is 10–15 million birds in France (22–32% of UK released pheasant numbers; Arroyo and Beja 2002, ONCFS 2013), with around 10 million pheasants also reared in the United States (Burden 2013). Releasing also occurs at very low (although largely unrecorded) levels in much of the rest of Europe (Arroyo and Beja 2002, Mustin *et al.* 2012).
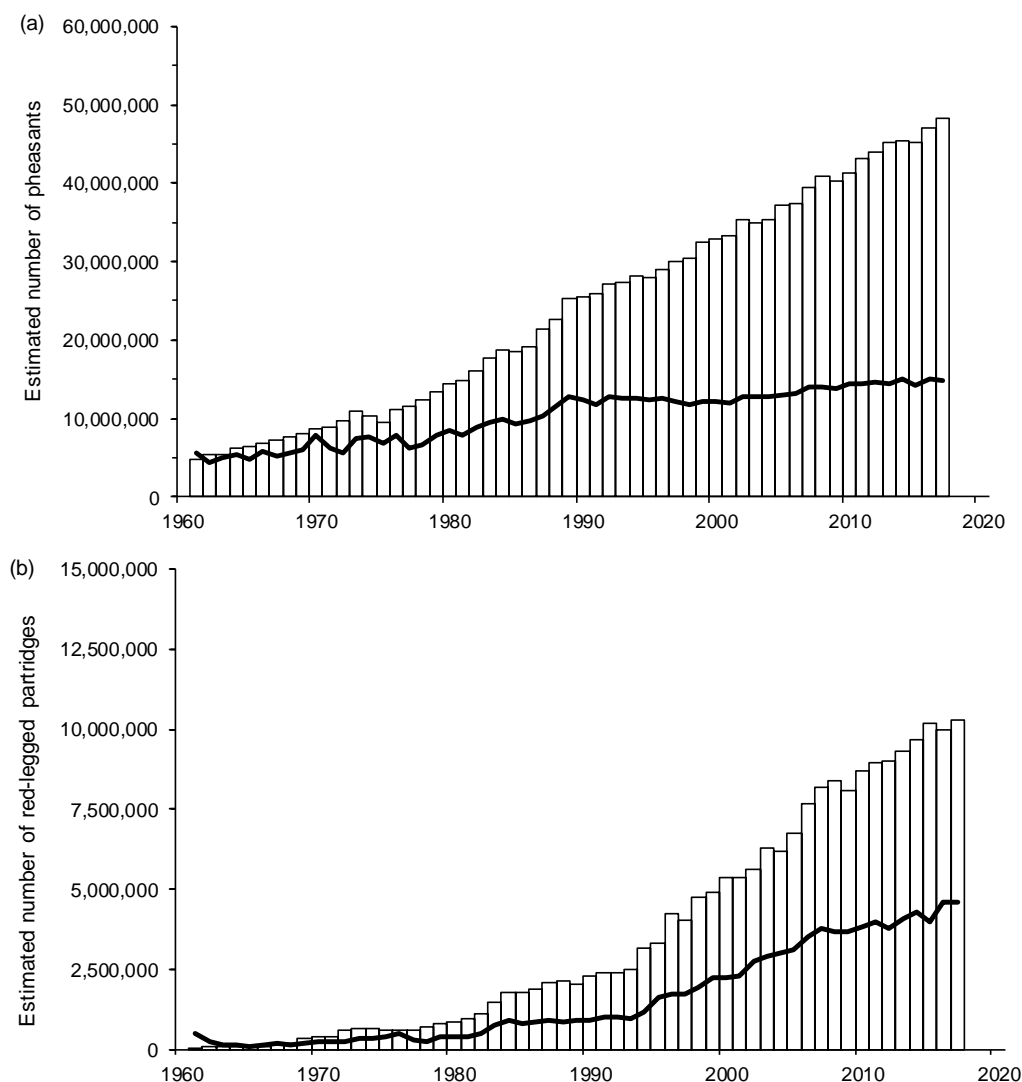
Fig. 2. Estimated relative numbers of a) pheasants and b) red-legged partridges released (bars) and shot (solid line) annually from 1961 to 2017 in the UK. These values were interpolated from indices of change in the numbers released and shot from the GWCT's National Gamebag Census (NGC) presented in Aebischer (2019a), using the calibration method and estimates of numbers released and shot from the NGC in 2016 by Aebischer (2019b): 47 million pheasants released, 15 million shot in 2016; 10 million red-legged partridges released, 4.6 million shot. The figures give an indication of change in the relative numbers of pheasants and red-legged partridges released and shot across the UK as a whole and may not represent the true numbers exactly. Note that scales differ.

Distribution of releasing in UK

Pheasants and red-legged partridges were reported as a sporting interest on 88% and 84% of estates respectively in a British national survey of gamekeepers in 2011 (National Gamekeepers' Organisation 2011), and Sage *et al.* (2005a) estimate that 1 in 12 of all woodlands in England contain a pheasant release pen. There is however no national database documenting the numbers of gamebirds released per estate per year in a comprehensive way.

The closest data is that held in the UK Government's Poultry Register by the Animal and Plant Health Agency (APHA[1]; Fig. 3). All poultry holdings with > 50 birds are legally required to register on this

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[1] https://www.gov.uk/government/publications/poultry-including-game-birds-registration-rules-and-forms

database ('poultry' including pheasants, red-legged and grey partridges) although Madden and Sage (2020) suggest that the level of compliance with this requirement to register is very low, with perhaps less than 40% of the red-legged partridges and less than 25% of the pheasants held in captivity prior to release recorded through this database. The APHA poultry register therefore only gives an indication of the locations and densities of captive-reared pheasants and partridges prior to release (Fig. 3). The latest published estimates indicate that the majority of land holdings rearing and/or releasing gamebirds are in England (85%), with approximately 13% of reared gamebirds being released in Scotland and 3% in Wales (APHA 2019).
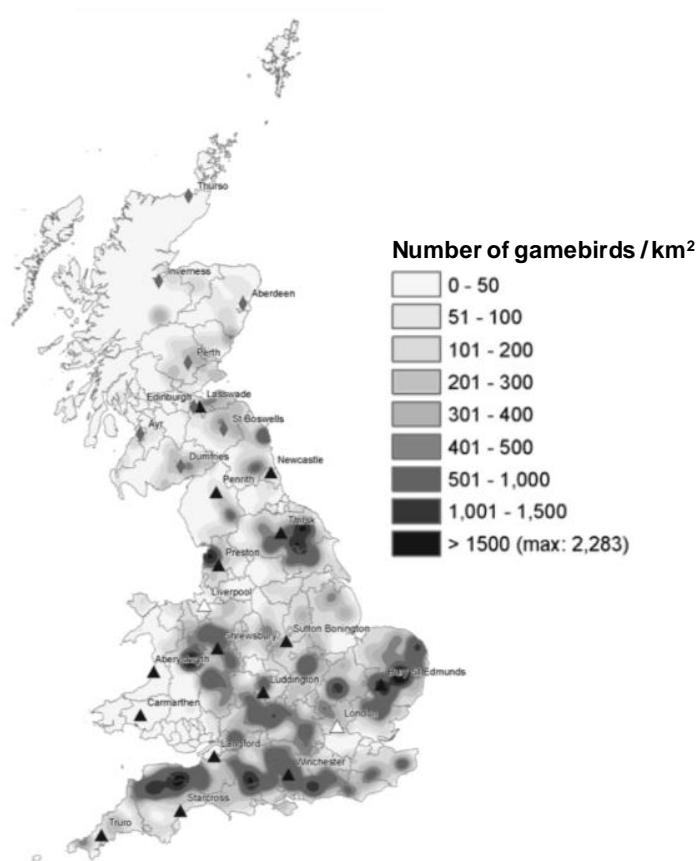
Fig. 3. The most recent published map of the number of pheasants and partridges (red-legged and grey) per km^2 held in captivity for releasing in the UK, as registered on the Animal & Plant Health Agency (APHA) Poultry Register in 2013. All holdings with >50 birds are required to register with APHA, including release estates and rearing farms, but compliance is thought to be low. No separate data exist for red-legged partridge or grey partridge although relatively few grey partridge are released. Reproduced from AHVLA (2013); contains public sector information licensed under the Open Government Licence v3.0).

In addition to the APHA Poultry Register data, Madden and Sage (2020) extracted spatial release data for pheasants and red-legged partridges in England from the commercial shooting website www.gunsonpegs.com, where larger shoots advertise their shoot days and specify location information, quarry species and the bag sizes offered. Lowland gamebird shooting was historically concentrated in the South and East of England where wild gamebirds were most abundant, but now both the APHA Poultry Register and 'Guns on Pegs' data show that the releasing of pheasants and red-legged partridges is more widespread and relatively evenly distributed throughout lowland UK (Fig. 3; Madden

and Sage 2020). Although generally concentrated in lowland habitats, the past few decades have also seen an increase in the releasing of red-legged partridges on the moorland fringe[2].

Despite this trend for more widespread gamebird releasing, there is little information on the numbers of birds or the areas involved. Up-to-date and comprehensive data on the annual numbers and distribution of released gamebirds, as well as further investigations into the ecological and socioeconomic drivers behind the rapid increase in the numbers released are therefore urgently needed.

Efficiency of releasing

Pheasants and red-legged partridges together comprise 82% of the 24 million birds of all species estimated to be shot annually in the UK (Aebischer 2019b). The annual proportion of pheasants shot compared to the number released (the "efficiency" of pheasant releasing), which remained relatively stable from 1960 until approximately 1990, has since suffered a steep decline (Fig. 4), with the proportion of released pheasants that were ultimately shot falling from around 50% to around 35% between 1990 and 2005 and remaining low thereafter (Robertson *et al.* 2017) This is because the number of birds shot has remained relatively constant since the 1990s while releases have increased (Fig. 2): only 34% of the gamebirds released (32% of released pheasants; 46% of released red-legged partridges) were shot during the autumn and winter in 2016 (Robertson *et al.* 2017, Aebischer 2019b). This means that although game managers are spending more time, effort and money releasing greater numbers of pheasants, their financial returns are likely to be declining.
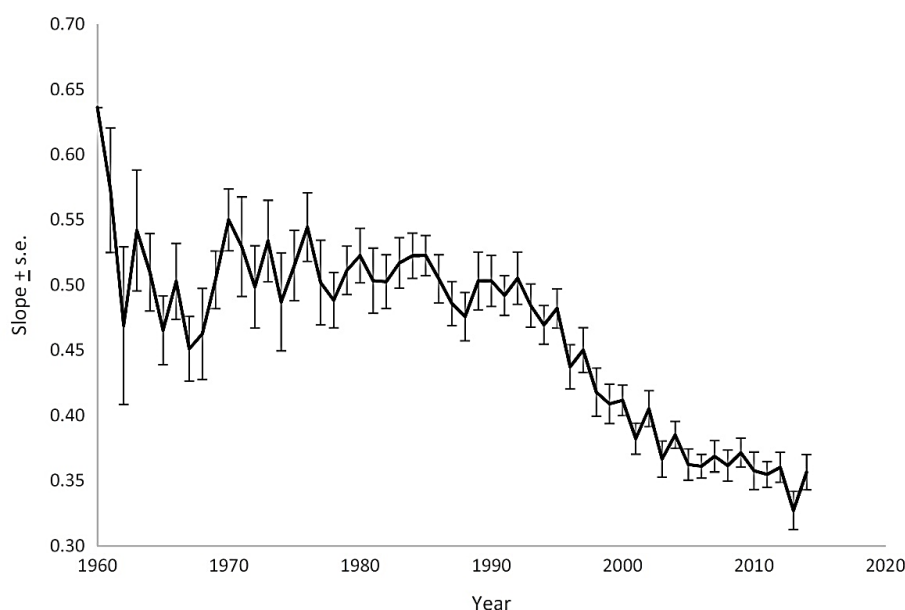
Fig. 4. The efficiency of pheasant releasing (the proportion of birds released that were shot) decreased from ca. 50% to ca. 35% from 1990 to 2005, stabilising thereafter. The graph shows the changes in the proportional efficiency (slope) of pheasant releasing between 1960 and 2014, indicated by annual estimates of the slope of the relationship between the numbers of pheasants released and numbers shot per unit area on sites contributing to the Game & Wildlife Conservation Trust (GWCT)'s National Gamebag Census (NGC) in Great Britain. A slope of 0.5 indicates that approximately 50% of the pheasants released were shot. Figure is an exact reproduction from Robertson *et al.* (2017)[3].

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[2] e.g. https://www.drumlanrigcastle.co.uk/field-sports/game-shooting/, and
https://www.shootinguk.co.uk/answers/shooting-answers/where-is-the-best-place-to-shoot-redleg-partridges-in-the-uk-40877.

[3] https://creativecommons.org/licenses/by/4.0/

The increase in numbers of gamebirds released despite relative constancy in the numbers shot may be linked to a reduction in post-release pheasant survival and an increase in shooting in late winter (due to a more even spread of shooting days throughout the season than was the case in the past, and January shooting perceived as more challenging[4]), which has driven the need to release more gamebirds during the preceding autumn to ensure enough survive to be shot in January (Robertson *et al.* 2017). There may be multiple potential drivers of the apparent reduction in post-release survival. Firstly, post-release predation of pheasants can be high (Table 4) and may have increased as a result of high or increasing predator abundance (see section 4.7). A change in the genetic composition (and therefore behavioural characteristics) of the released pheasants may also have had an impact on post-release survival, with annually-caught wild pheasants used for rearing stock prior to 1990, after which the releasing of centrally-reared captive stock became more common (Robertson *et al.* 1993a, Robertson *et al.* 2017). New smaller and "wilder" pheasant strains were also brought in to improve hunting experience during the 1990s, which may have influenced their survival and the numbers shot; "wilder" pheasants being generally shyer and better able to survive the shooting season (Robertson *et al.* 1993a, Madden and Whiteside 2014, Robertson *et al.* 2017).

Table 4. Estimated annual fate of the 47 million pheasants released in the UK compiled from multiple sources which specifically studied the survival of released pheasants post-release.

Fate	Number	%	Source
Shot	15 million	32	Aebischer (2019b)
Predated/scavenged	16.5 million	35	Sage *et al.* (2018)
Other (vehicle collision/disease)	6.1 million	13	Turner (2008)
Total non-shooting mortality	23 million	49	
Unknown	1.9 million	4	
Survive to breeding season	7.5 million	16	Turner (2008)
Total	47 million		Aebischer (2019b)

It is also possible that poisoning from agricultural seed-treatment pesticides such as neonicotinoids, which increased in usage throughout the 1990s (Lennon *et al.* 2019, Lennon *et al.* 2020a), may have had an impact on gamebird survival either directly (causing mortality) or indirectly (through effects on fitness, breeding success, reductions in invertebrate prey or increasing susceptibility to predation). Pheasants and red-legged partridges are subject to neonicotinoid (clothianidin) exposure from eating treated cereal seeds following normal winter arable crop sowing (Lennon *et al.* 2020a, Lennon *et al.* 2020b), and 89% of gamebirds (mostly red-legged partridges) contained clothianidin residues in their liver or blood after winter sowing in eastern England (Lennon *et al.* 2020b). Increases in potential dietary exposure to neonicotinoids has been linked in a correlational study to population declines in red-legged partridge (Lennon *et al.* 2019). Additionally, there has been a reduction in the level of compulsory set-aside, and a subsequent increase in the planting of game cover crops since 1990, which are often used to encourage pheasants away from release pens on game shooting estates. This may have unwittingly increased dispersal, or made the birds more difficult to flush and hence shoot (Bicknell *et al.* 2010). A small (though not unimportant) proportion of the birds released may also die from lead poisoning after ingesting lead gunshot pellets present in the environment on shooting sites (Pain *et al.* 2019a): see section 4.3.4 and 5.1.2. Finally, changes to the diet and conditions under which birds are reared can also have direct consequences for their post-release survival (Whiteside *et al.* 2015, Whiteside *et al.*

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[4] e.g. https://www.shootinguk.co.uk/shooting/shooting-in-january-109963

2016). This variety of potential mechanisms for apparent reductions in post-release gamebird survival highlight the urgent need for further research in this area.

The efficiency of releasing appears not to be linked to the density of birds released, as originally theorised by (Bicknell *et al.* 2010). It was suggested that increasing the density of birds released on a site would have a progressively smaller effect on the density shot, and that further releasing would eventually have no additional effect on the numbers shot (Bicknell *et al.* 2010). In examining the efficiency of large-scale pheasant releasing, Robertson *et al.* (2017) have shown this not to be the case however, instead finding that the efficiency of releasing declined equally over time on sites releasing both high and low densities of birds. The efficiency of releasing within a year across multiple sites was also not affected by the density of birds released (Robertson *et al.* 2017).

We found no single recent source which comprehensively estimated the fate of released pheasants. However, multiple sources combined suggest that, linked to the reductions in efficiency, approximately one third of released pheasants are now shot, while the majority die from other causes during the year of release and only a small proportion remain alive to join the wider population of gamebirds residing in the wild in the UK (Table 4 and references therein). Madden *et al.* (2018) suggest that the majority of reared pheasants released in the UK are likely to be dead within 15 months of release, with ~9% of released birds surviving to the start of the breeding season (Hoodless *et al.* 1999).

There is no published information on the efficiency of red-legged partridge shooting relative to the numbers released, although Fig. 2 suggests this may have remained relatively stable with approximately 50% of released red-legged partridges being shot annually, the numbers shot increasing broadly in line with the increase in numbers released (Aebischer 2019a). This may be because most red-legged partridge shooting is still conducted in the early part of the shooting season, so the impacts of lower survival of birds post-release are less pronounced on the shooting bag (Aebischer 2019a).

Biomass and changes to UK avifauna
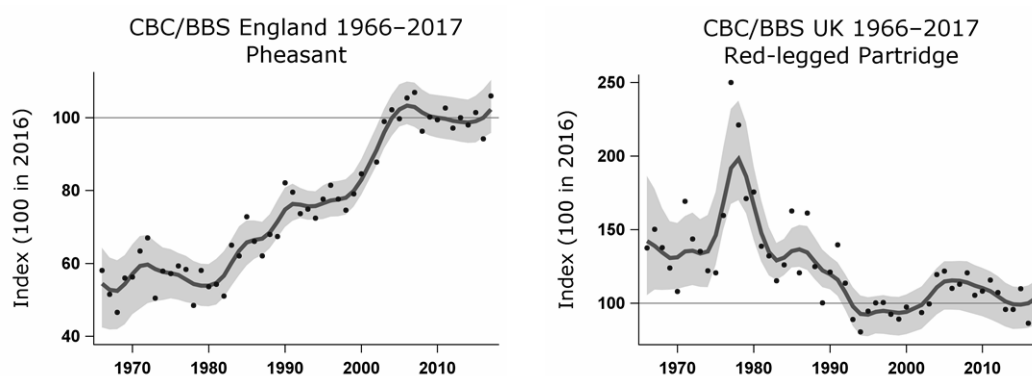
Fig. 5. Smoothed breeding population indices for pheasant and red-legged partridge, relative to an arbitrary index of 100 in 1966, with 85% confidence limits in grey. Graphs reproduced from Woodward *et al.* (2018).

Species that contribute a high proportion to the biomass of an ecological system can be key to shaping the animal and plant communities around them, and can have a substantial impact on the ecosystems into which they have been introduced (Gaston 2011, Eaton *et al.* 2012). The large-scale autumn releasing of pheasants and red-legged partridges has fundamentally changed the community structure of wild birds in the UK (Eaton *et al.* 2012, Blackburn and Gaston 2018). Pheasants contribute more biomass and use more energy than any other British breeding bird species, with the single exception of the woodpigeon (Blackburn and Gaston 2018). Non-native birds form only ~3% of the number of UK's breeding birds, but contribute at least 23% of the biomass of the UK's breeding birds (Eaton *et al.* 2012). The greatest contribution to the abundance, biomass and population energy use of non-native

bird species in Britain is made by pheasants, comprising 74% of all individual non-native birds, 74% of the total biomass and 77% of the energy use attributed to non-native birds in 2013 (Blackburn and Gaston 2018). The increase in pheasant biomass has also occurred during a period when the biomass of native bird species has been declining (e.g. Dolton and Brooke 1999).

The local abundance of breeding non-native gamebirds is highly dependent on the numbers released the previous year (Baker *et al.* 2006a, Pringle *et al.* 2019), so linked to the increase in large-scale releasing (Fig. 2a) there was a 70% increase in the number of UK breeding pheasant pairs between 1970 and 2015 (Hayhow *et al.* 2017), and a 32% increase since 1995 (Fig. 5; Woodward *et al.* 2018). The breeding population of red-legged partridge has fluctuated more, declining by 26% overall between 1970 and 2015 (Hayhow *et al.* 2017), but increasing by 10% since 1995 (Fig. 5; Woodward *et al.* 2018), again likely to be a result of the recent increases in large-scale red-legged partridge releasing (Fig. 2b). Pheasants and red-legged partridges are also widely distributed (Fig. 6), being recorded as breeding in 83% and 57% of 10x10 km quadrats respectively in Great Britain in 2008–2011 (Balmer *et al.* 2013).
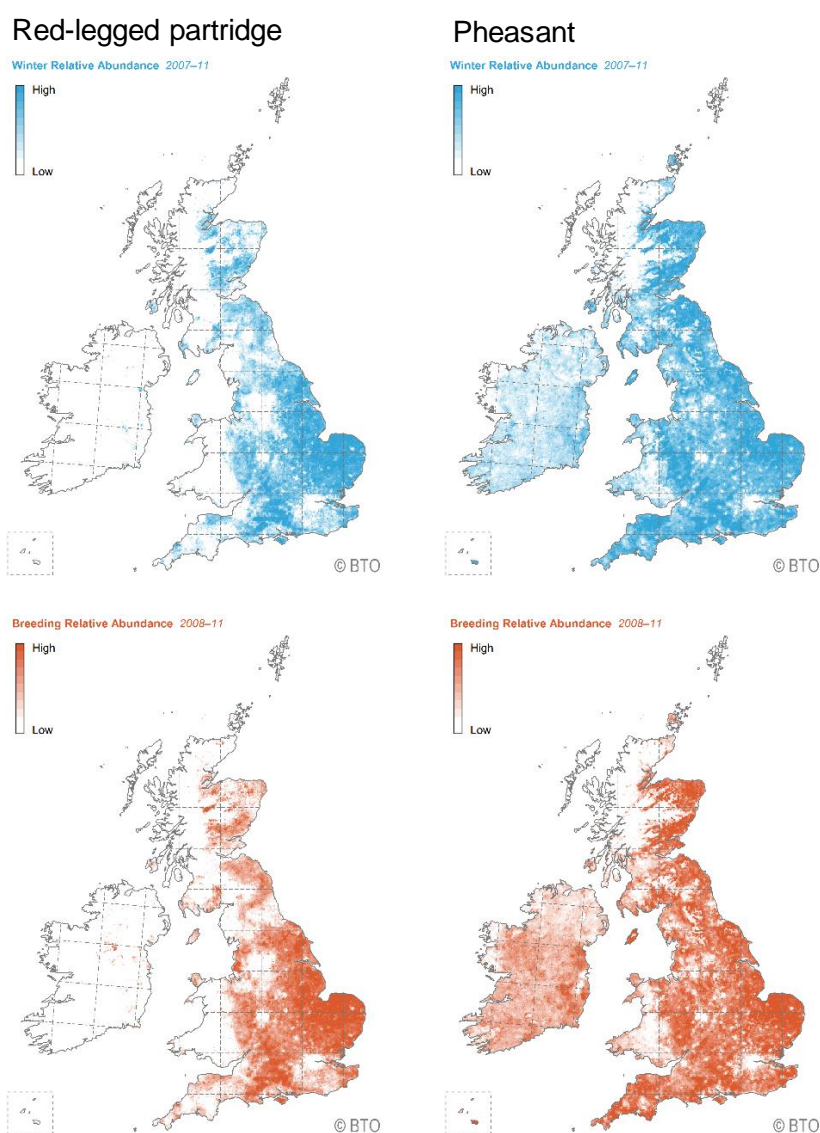
Fig. 6. Relative abundance of pheasant and red-legged partridge in Britain and Ireland in winter (top panel, blue; November–February) and the breeding season (bottom panel, orange; April–July). Darker and lighter squares represent higher and lower relative abundance respectively. Data do not represent release densities, but are strongly associated with captive gamebird densities prior to release (Pringle *et al.* 2019). The distribution maps are from Bird Atlas 2007-11 which is a joint project between BTO, BirdWatch Ireland and the Scottish Ornithologists' Club (Balmer *et al.* 2013). Maps reproduced with permission from the BTO.

Blackburn and Gaston (2018) estimated that the naturalised UK pheasant breeding population equated to a biomass of 3,470 tonnes in 2013 (using a population estimate of 4,400,000 individuals, assuming equal numbers of males and females from the estimate of 2,200,00 females in 2009 given by Musgrove *et al.* (2013), and assuming an average pheasant body mass of 850g). Updating this with the latest breeding population estimate of 2,350,000 females in the UK in 2016 (Woodward *et al.* 2020), using the same method this equates to approximately **3,995** tonnes of breeding pheasants. This is a conservative estimate because it does not account for non-breeding birds or the young raised by naturalised breeding birds (for which there are no published estimates).

Blackburn and Gaston (2018) also estimated that the naturalised UK red-legged partridge breeding population equalled a biomass of 71.996 tonnes in 2013 (using a population estimate of 164,000 individuals, assuming a pair on each territory from the estimate of 82,000 territories in 2009 given by Musgrove *et al.* (2013), and assuming an average red-legged partridge body mass of 439g). Updating this with the latest breeding population estimate of 72,500 territories in the UK in 2016 (Woodward *et al.* 2020), using the same method this equates to approximately **63.7** tonnes. Again, this is a conservative estimate because it does not account for non-breeding individuals or the young raised by naturalised breeding birds.

The combined biomass of breeding pheasants and red-legged partridges in the UK in 2016 is therefore estimated to have been:

3,995 tonnes (pheasant) + 63.7 tonnes (red-legged partridge) = **4,058.7** tonnes combined

Using the body mass estimates above (Blackburn and Gaston 2018) and numbers released (47 million pheasants, 10 million red-legged partridges) reported by Aebischer (2019b), the combined biomass of released gamebirds in 2016 is estimated to have been:

39,950 tonnes (pheasant) + 4,390 tonnes (red-legged partridge) = **44,340** tonnes combined

This means that, at the point of release in 2016 the combined biomass of breeding adult and released gamebirds equalled approximately **48,399** tonnes, which is more than twice the spring biomass of all native UK breeding birds, which was estimated as 23,964 tonnes by Blackburn and Gaston (2018).

It is difficult to compare gamebird biomass at the point of release in autumn with the autumn (i.e. post-breeding) biomass of native UK birds, for which there are no reliable estimates. However, Blackburn and Gaston (2018) suggest that non-breeding bird population size might be double that of the breeding total, so post-breeding native bird biomass could be crudely estimated at 23,964 x 2 = 47,928 tonnes, and post-breeding plus released gamebird biomass at 52,457 tonnes (post-breeding pheasants: 3,995 x 2 = 7,990 tonnes, post-breeding red-legged partridges: 63.7 x 2 = 127.4 tonnes, plus the total estimated 44,340 tonnes of gamebirds released). When comparing post-breeding populations in this way, gamebirds therefore account for 9% more biomass relative to that of native birds.

The UK pheasant breeding population of 2,350,000 females equates to at least 44% of the total population breeding in Europe (estimated at 4,140,000–5,370,000 pairs; Avery 2019, BirdLife International 2020b), while the population of red-legged partridges breeding in the UK accounts for only 1% of the European total (estimated at 5,060,000–7,080,000 pairs; BirdLife International 2020b).

Release of native UK gamebirds

Two native gamebird species are also reared and released for shooting in the UK: grey partridge and mallard (Aebischer 2019b). The extent of grey partridge releasing has fluctuated over time, increasing overall between 1966 and 2016, but decreasing since the 1990s (22% decrease in releasing 1991–

2016, 16% decrease 2004–2016; Aebischer 2019b). 190,000 grey partridges are estimated to have been released in 2016, at least some of which was for restocking purposes rather than directly for shooting (Aebischer 2019b). In contrast, there has been a sustained increase in mallard releasing across the same time period (+590% 1966–2016, +90% 1991–2016 and +34% 2004–2016; Aebischer 2019b), with the shooting bag of mallard largely sustained by this releasing (940,000 shot on average relative to ~1,000,000 released in 2016; Aebischer 2019b, Madden and Sage 2020). However, the releasing of these species is conducted on a considerably smaller scale compared to the release of pheasant and red-legged partridge (representing only 2% of the numbers of these gamebirds released), and as they are both native UK species with ecosystems and communities presumably already adapted to their presence, their impacts may be less pronounced than for the non-native pheasants and red-legged partridges. As such, we did not consider the impacts of releasing these species in this review. The potential impacts of mallard releasing are however considered by Madden and Sage (2020).

2.1.2   The lowland game shooting industry

The increase in the geographical extent and magnitude of commercial gamebird releasing may have developed through an increase in the popularity of pheasant shooting, the intensification of artificial rearing practices and a reduction in the profitability of farming and the need for landowners to generate alternative forms of income through the selling of 'large bag' shooting days (Greenall 2007, Martin 2011, Martin 2012, Robertson *et al.* 2017).

Rear and release practices

Many of the pheasants and red-legged partridges released in the UK originate from eggs and chicks bred in France and imported in large numbers (Green *et al.* 2020), with smaller proportions imported from Spain (pheasant and red-legged partridge), Poland, Portugal, Belgium, Ireland and the USA (pheasant; Canning 2005, Rutley 2019, Madden and Sage 2020). Between 2014 and 2018, gamebirds comprised ~92% of the 48.9 million individual live non-CITES wild terrestrial vertebrates declared as imported into the UK through APHA (Green *et al.* 2020). A small proportion of eggs and chicks are also bred within the UK, usually from captive breeding flocks with eggs incubated artificially (Pennycott *et al.* 2012). A common practice for the breeding of pheasants in the UK is to capture birds from the wild during the winter for use as next year's breeding stock. Large paddocks and other enclosed areas are sometimes used to overwinter birds chosen in this way (Canning 2005, FAWC 2008).

Breeding pheasants and red-legged partridges are often housed in raised outdoor cage systems, although pheasants may also be bred in outdoor grass floor pens of varying size (FAWC 2008, SAC Commercial Ltd 2012, Matheson *et al.* 2015). Eggs collected from breeding gamebirds are hatched in incubators, and the chicks grown on in brooder houses (Canning 2005). Hen pheasants are usually sold on to gamebird shoots after their first captive breeding period, while red-legged partridge hens are often retained for two years before being sold to shoots (Matheson *et al.* 2015).

At 6-8 weeks old, during July-August, pheasant poults are placed in woodland release pens which can range from 0.1 to 10 hectares in size[5]. Woodland release pens are opened-topped, fenced-off areas enclosing suitable natural cover where birds are slowly accustomed to natural surroundings, are supplied with food and water, and are protected from mammalian predators (JNCC 2000)[5]. As they mature, birds are encouraged to leave the pens, but usually return to roost at night (JNCC 2000). As the birds get older they spend more time foraging in woodlands and fields adjacent to release pens. Adult birds eat a higher proportion of natural foods and spend more time in cover crops, at woodland edges, and in fields. These rear and release methods are largely similar for red-legged partridges and pheasants, although red-legged partridges are usually released into a larger number of smaller, covered

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[5] https://www.gwct.org.uk/game/research/species/pheasant/releasing-for-shooting-in-lowland-habitats/

pens usually associated with game cover crops, stubble or grassland on farmland or the moorland fringe rather than in woodland (JNCC 2000)[5]. The aim is to keep small groups (coveys) of birds together once they are released, which is achieved by gradually releasing one or two individuals while providing food in the vicinity of the pen so that they stay close the remaining birds (JNCC 2000).

These breeding systems may have welfare implications if breeding cages are small, stocking densities are high, management devices are used to reduce injurious pecking or limit flight, and environmental enrichment is not provided (e.g. the provision of suitable materials for dustbathing, refugia, elevated perches and enrichment targeting foraging behaviours; Pennycott *et al.* 2012, Matheson *et al.* 2015, Madden *et al.* 2020). Voluntary Codes of Practice published by the UK government for the welfare of gamebirds reared for sporting purposes (Table 5) now provide guidelines on how welfare should be improved (DEFRA 2010b, The Scottish Government 2011, Welsh Assembly Government 2011), but despite compliance with many of these recommendations, there may still be areas where welfare improvements are required (Pennycott *et al.* 2012, Madden *et al.* 2020).

Regulation and Legislation

Gamebird shooting in the UK is covered by the Game Act 1831 in England and Wales, The Game (Scotland) Act 1832 in Scotland, or the Game Preservation Act (Northern Ireland) 1928 in Northern Ireland, supplemented by the Game Licenses Act 1860 and Game (Scotland) Act 1772, all of which have gone through several amendments and combine to protect gamebirds from shooting during the closed season (Canning 2005). This is any time outside of the shooting season which extends from 1st September for red-legged partridge or 1st October for pheasant, until 31st January (Northern Ireland, Isle of Man, Guernsey) or 1st February (England, Wales, Scotland) for both species (Canning 2005, Natural England 2012, BirdLife International 2016, Avery 2019).

The practice of releasing gamebirds is however largely exempt from statutory regulation. There is no legal requirement to record the numbers of gamebirds released or shot for example (although some shoots submit this data voluntarily to national monitoring schemes such as the GWCT's National Gamebag Census), and a license is not required to shoot game over much of the UK, only a gun license and permission from the landowner is needed (Madden *et al.* 2018, Avery 2019). The only exception is on Jersey, where a license is required both to release and shoot pheasants (Avery 2019). The industry therefore relies primarily on self-regulation. Sites holding more than 50 gamebirds in captivity prior to release are legally required to register these birds on the APHA poultry register[6] however, although compliance with this is very low (Madden and Sage 2020). During the production and breeding process, gamebirds are also protected by the Animal Welfare Act 2006 in England and Wales (DEFRA 2010b), Animal Health and Welfare (Scotland) Act 2006 (The Scottish Government 2011) and Welfare of Animals Act (Northern Ireland) 2011[7]. Gamebirds that are reared or kept in captivity fall within the legal definition of poultry (i.e. a form of livestock), but once released they are classed as wild birds (e.g. Avery 2019, Madden and Sage 2020). The 'catching up' of wild pheasants and partridges as captive breeding stock (after which they become classed as livestock again) is legally permitted under the Wildlife and Countryside Act 1981, as long as this is done during the shooting season (Canning 2005, Natural England 2012, BirdLife International 2016, Avery 2019).

Pheasant and red-legged partridge are not listed on schedule 9 of the Wildlife and Countryside Act of 1981 (as amended; England, Scotland or Wales) or the Wildlife (Northern Ireland) Order 1985, and as such are not considered as Invasive Non-Native Species (INNS) in a legal sense and therefore fall outside of regulations covering the release of INNS. The justification for this is not ecological, but economical, and is therefore counter-intuitive to the purpose of the Acts, thus raising questions about

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[6] https://www.gov.uk/government/publications/poultry-including-game-birds-registration-rules-and-forms

[7] https://www.legislation.gov.uk/nia/2011/16/contents

the validity of such an exemption (Bicknell *et al.* 2010). An analogous example of justified exemption is the agricultural livestock industry (Bicknell *et al.* 2010). However, livestock are carefully controlled, individually labelled and registered, and are not able to disperse freely as gamebirds are. Additionally, livestock represent essential components of the UK meat, dairy, egg and wool industries; whereas gamebird shooting is primarily a leisure activity, and does not supply a significant proportion of the UK food industry (PACEC 2006, PACEC 2014).

Shooting providers and participants

The majority of pheasant and red-legged partridge shooting in the UK, particularly that which is commercially sold, is 'driven shooting', where birds are driven towards a line of shooters ('guns') by a team of 'beaters' (Avery 2019). Large numbers of birds are shot in a day using this method, and for pheasants this can be as many as 200–800 per day (Avery 2019). 'Walked-up' shooting and shooting over dogs is also practised widely, but less commercially, and involves smaller numbers of birds (Avery 2019). A new form of shooting red-legged partridges on the moorland fringe involves 'high' or 'sky scraping' birds which are driven over the top of a deep valley where shooters are waiting[8].

Approximately 430,000 people participate in some form of driven or walked-up game shooting in the UK, including the shooting of waterfowl and red grouse in the uplands as well as the shooting of lowland pheasants and red-legged partridges (PACEC 2014). There are an estimated 7,000 shooting estates in the UK (FAWC 2008), and 23,000 providers (people occupying key roles in organisations which provide shooting sports opportunities) of driven game shooting (including pheasant, red-legged partridge, grey partridge, various ducks and red grouse as quarry species; PACEC 2014). 83% of shooting providers rely on released pheasants and/or red-legged partridges as their main quarry species (PACEC 2006). The number of birds that clients wish to shoot in a day varies, and 'large bag' days have recently come under some scrutiny within the industry (Greenall 2007).

The numbers of birds released varies between shoots, depending on the interest and business plan of the owner, and the size of the estate (Bicknell *et al.* 2010, Teanby *et al.* 2017). Teanby *et al.* (2017) provide crude definitions of what could be considered as 'small', 'medium' or 'large' shoots as those releasing <3,000, 3,000–10,000, and >10,000 birds respectively. Using these definitions and applying them to 155 UK shoots, Teanby *et al.* (2017) found that 'small' shoots released an average of 1,500 birds over 960 acres annually, while 'medium' shoots released 6,200 birds over 1,800 acres and 'large' shoots released 26,200 birds over 3,860 acres on average. These releasing densities refer to the size of the whole shooting estate and don't compare directly with the densities recommended for releasing pens within woodland (less than 700–1,000 pheasants per hectare of release pen; Sage 2007b); see section 4.5.7. Based on APHA poultry register data, Madden and Sage (2020) estimate that approximately 50% of English shoots could be classified as 'small', 25% classified as 'medium' and 25% as 'large', meaning that approximately half of shoots release >3000 birds, and a quarter release >10,000 annually.

Land under management for gamebird shooting

PACEC (2014) estimate that 75% of rural land (160,000 km[2]) is shot over in the UK, based on a total estimate of rural land coverage from Piddington (1980). This includes pheasant and red-legged partridge shooting, but also all other shooting types (such as grouse shooting in the uplands, deer stalking, wildfowling, mammalian or avian pest shooting, clay pigeon shooting and target shooting; PACEC 2014). A recent survey of British gamekeepers estimated that over 4,875,000 hectares may be under gamekeeper management for grouse, wildfowl, grey partridge and pheasant or red-legged partridge shooting combined, which equates to the area covered by 'on land' AONBs, National Scenic

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[8] e.g. https://www.shootinguk.co.uk/answers/shooting-answers/where-is-the-best-place-to-shoot-redleg-partridges-in-the-uk-40877

Areas and National Parks in England, Scotland and Wales, plus approximately 76% of the area covered by the terrestrial protected areas (SSSIs, MCZs, NCMPAs, NNRs, Ramsar sites, SACs and SPAs; (Ewald and Gibbs 2020). Approximately 10,000 hectares of woodland in the UK is enclosed within release pens (PACEC 2006).

2.1.3   Best-practice guidelines for sustainable gamebird releasing

Three best-practice guidelines relating to pheasant and red-legged partridge releasing have been published in recent years, all of which are summarised in Table 5. Of particular interest are the GWCT's "Guidelines for sustainable gamebird releasing" (Sage 2007b), and the Code of Good Shooting Practice[9], which suggest ways to minimise the direct impacts of released gamebirds and maximise the benefits from game estate management. These guidelines recommend that no more than 1,000 pheasants should be released per hectare of woodland release pen (400 per acre), and to no more than 700 birds per hectare of release pen (280 per acre) in sensitive ancient semi-natural woodlands (Sage 2007b), based largely on scientific research conducted prior to 2010 (e.g. Sage *et al.* 2005; see section 4.5.7). However, some release pens are still stocked at levels greater than 3,000 pheasants per hectare (e.g. Davey 2008, Pressland 2009, Neumann *et al.* 2015), with some reaching closer to 5,000 pheasants per hectare (Sage *et al.* 2005a, Davey 2008).

Table 5. Codes of practice or guidelines relating to the rearing, releasing and shooting of non-native gamebirds in the UK.

Citation and weblink	Focus
The Code of Good Shooting Practice (2012) BASC, CLA, Countryside Alliance, Game Farmers' Association, GWCT, Moorland Association, National Game Dealers' Association, National Gamekeepers' Organisation, Scottish Association for Country Sports, Scottish Gamekeepers' Association, Scottish Land and Estates. www.codeofgoodshootingpractice.org.uk/pdf/COGSP.pdf	Shooting behaviours, Shoot management, Rearing and releasing, Predator control, Legal requirements
Welsh Assembly Government. (2011) **Code of practice for the welfare of gamebirds reared for sporting purposes**. https://gov.wales/gamebirds-reared-sporting-purposes-welfare-code-practice.	Welfare of rearing and releasing
The Scottish Government. (2011) **Code of practice for the welfare of gamebirds reared for sporting purposes**. https://www.gov.scot/publications/code-practice-welfare-gamebirds-reared-sporting-purposes/.	Welfare of rearing and releasing
DEFRA. (2010) **Code of Practice for the welfare of gamebirds reared for sporting purposes**. Department for Environment, Food and Rural Affairs: London, UK. https://www.gov.uk/government/publications/code-of-practice-for-the-welfare-of-gamebirds-reared-for-sporting-purposes	Welfare of rearing and releasing
Sage, R. (2007) **Guidelines for sustainable gamebird releasing**. The Game and Wildlife Conservation Trust (GWCT): Fordingbridge, Hampshire. https://www.gwct.org.uk/advisory/guides/sustainable-gamebird-releasing/	Releasing, Habitat management
Sage, R. & Swan, M. (2003) **Woodland conservation and pheasants: A practical guide** produced by the Game Conservancy Trust for game managers and woodland owners, The Game Conservancy Trust (GWCT): Fordingbridge. https://www.gwct.org.uk/media/208626/woodland-conservation-and-pheasants.pdf	Releasing, Habitat management

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[9] http://www.codeofgoodshootingpractice.org.uk/pdf/COGSP.pdf

2.1.4   Impacts of gamebird releasing

The increasing numbers of gamebird released in the UK has triggered concerns about the ecological impacts of this widespread activity amongst conservationists, policymakers and within the shooting community itself. The effect of released gamebirds on other species was listed as one of the 100 most important ecological questions with policy relevance in the UK by Sutherland *et al.* (2006). Related to these growing concerns, the topic of gamebird releasing (as well as gamebird hunting more generally) has been covered in multiple different reviews in recent years (Table 6), including the precursor to this current review update (Bicknell *et al.* 2010). These reviews vary in their perspectives, geographical and taxonomic scope and the breadth of literature included, but all largely conclude that further research is required to determine the extent and direction of impacts of gamebirds after release.

Table 6. A list of review studies published in the last 25 years with relevance to the release of non-native gamebirds in the UK.

Madden, J. R. & Sage, R. B. (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 Shooting and Conservation. Natural England Evidence Review NEER016. Natural England: Peterborough.

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

Chapman, P. M. (2019) The direct ecological effects of non-native gamebird release in the lowland UK: an evidence synthesis. *Unpublished PhD Placement Report.* Natural England and Imperial College London.

NRW (2018) A review of the use of firearms on the land managed by Natural Resources Wales. Paper 2: Synthesis of Evidence. Natural Resources Wales.

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

Mustin, K., Newey, S., Irvine, J., Arroyo, B. & Redpath, S. (2012) Biodiversity impacts of game bird hunting and associated management practices in Europe and North America. *Contract report.* James Hutton Institute.

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. *RSPB Research Report 40.* RSPB: Sandy, Bedfordshire.

Fox, A. D. (2009) What makes a good alien? Dealing with the problems of non-native wildfowl. *British Birds,* **102:** 660-679.

Natural England. (2009) Chapter 10: Management for lowland gamebirds. *Environmental impacts of land management. Natural England Research Report NERR030.* (ed. C. Chesterton). Natural England: Sheffield.

Canning, P. (2005) The UK game bird industry - a short study. *Report to DEFRA.* ADAS: Lincoln, UK.

Arroyo, B. & Beja, P. (2002) Impact of hunting management practices on biodiversity (REGHAB). Instituto de Investigación en Recursos Cinegéticos (IREC).

Viñuela, J. & Arroyo, B. (2002) Gamebird hunting and biodiversity conservation: synthesis, recommendations and future research priorities. European Concerted Action within the 5th Framework Program: Reconciling Gamebird Hunting and Biodiversity (REGHAB).

Robertson, P. A. (1996) Naturalised introduced gamebirds in Britain. In *The Introduction and Naturalisation of Birds.* (eds. J. S. Holmes & J. R. Simons). HMSO: London.

PhD and Masters (MSc, MRes) theses by Callegari (2006b), Greenall (2007), Davey (2008), Turner (2008), Pressland (2009), Whiteside (2015), Rice (2016), Swan (2017), Gethings (2018) and Devlin (2019) also summarise aspects of the gamebird releasing or shooting literature in their introductory chapters.

Many of the potential impacts of gamebird releasing are poorly studied and understood, and are often under-represented in the peer-reviewed scientific literature. A common theme emerging from these reviews is that the ecological impacts of gamebird releasing appear to be strongly polarised, with potential negative effects associated with the released birds (e.g. enhanced predator abundance and predation, increased disease transmission, altered habitat structure, reduced invertebrate abundance; Table 6, Fig. 7); and positive effects associated with the management of semi-natural habitats on game shooting estates (conducted primarily for the benefit of released gamebirds, but which may also benefit wildlife; Table 6, Fig. 7). Gamebird releasing also has implications for human well-being with economic, employment and leisure benefits as well as potential health costs from zoonotic disease or the use of lead ammunition in gamebird shooting.
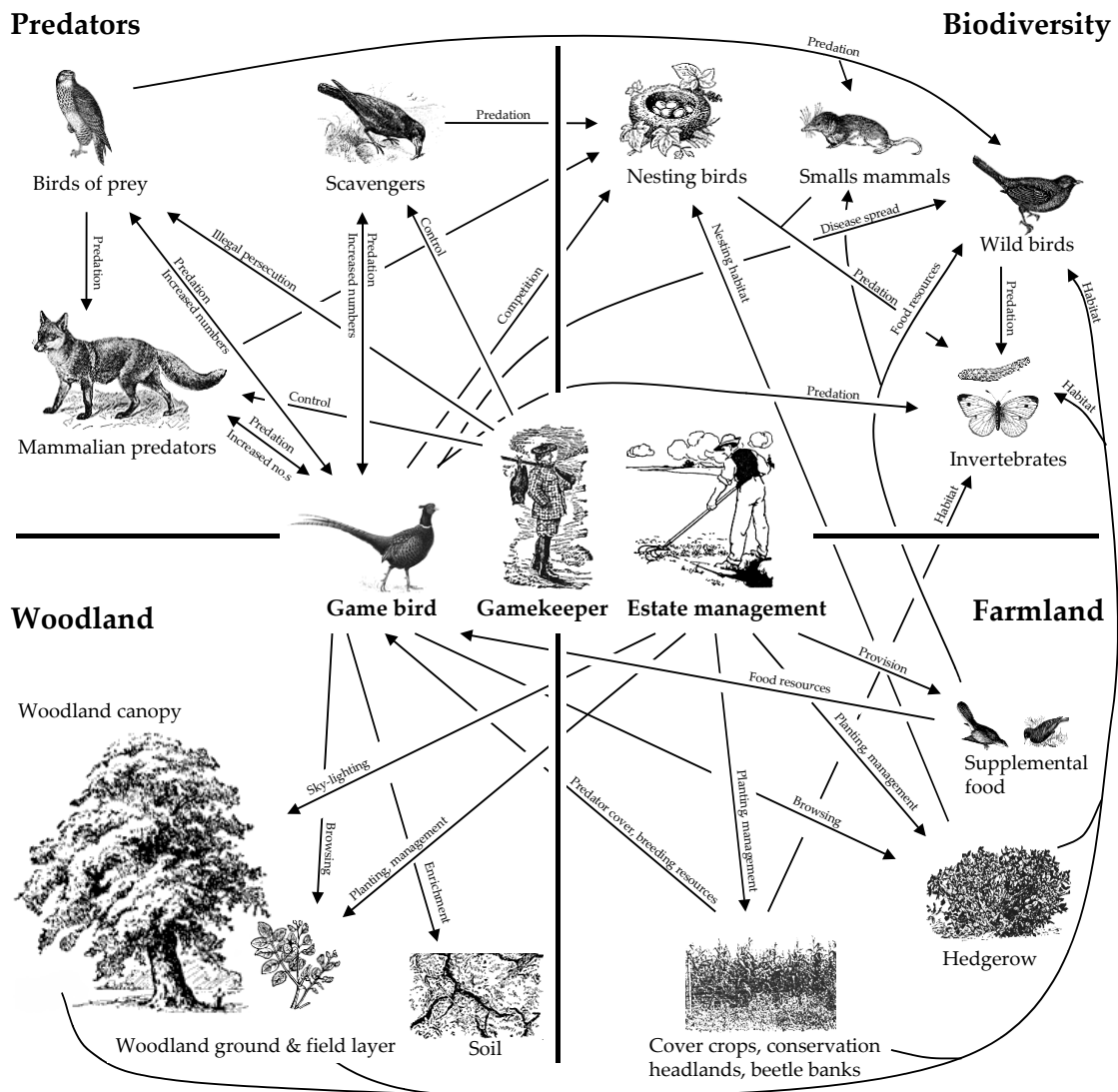
Fig. 7. Interactions between gamebird associated impacts (gamebirds, gamekeepers, game estate management), and the ecological features they may affect (divided into predators, biodiversity, woodland and farmland). Arrows point in the direction of the effect with short description of the interaction.

In this report, we review the impacts of gamebird release in the UK, both ecologically (impacts on UK wildlife, habitats and environment) and socio-economically (impacts on humans), building on and updating the previous review by Bicknell *et al.* (2010).

3   REVIEW METHODS

3.1   Aims and scope

In this review we assessed the evidence for ecological and socio-economic impacts of all aspects of gamebird release in the UK, including impacts from the presence of gamebirds in the environment, impacts associated with gamebird shooting practices, and impacts from management for gamebirds on game shooting sites, by updating a previous review published by Bicknell *et al.* (2010) with subsequent research and information published between 2010 and early-2020. Bicknell *et al.* (2010) and other contemporary reviews on the topic of gamebird releasing or hunting practices (see Table 6) focus on narrative review methods and the use of 'vote-counting' (tabulating the numbers of studies showing a particular effect) to determine the relative importance of different impacts. These reviews have generally found it difficult to draw overarching conclusions about the overall impact of gamebird release or hunting practices due to the diverse nature of the evidence and issues surrounding this topic. In this review, we therefore aimed for a more quantitative assessment of the ecological impacts of gamebird releasing, intended to provide a basis to draw more objective conclusions. Socio-economic impacts were more difficult to meaningfully assess in such a quantitative way, but we also aimed to update the evidence surrounding these impacts.

We focussed on research conducted in the United Kingdom or global or European studies which included data or examined studies from the UK. Bicknell *et al.* (2010) focussed on the UK but included example studies from elsewhere in Europe and the USA where UK examples were lacking, and we have retained these where still relevant. However, as previous reviews have highlighted that non-native gamebird release in the UK is conducted on a scale that greatly exceeds releasing practices elsewhere in Europe or the USA, and UK gamebird shooting focuses more on large-scale "driven" shooting rather than the "walked-up" shooting more common in other regions (Arroyo and Beja 2002, Mustin *et al.* 2012), we have not provided updated examples of impacts outside the UK, considering that only UK-based studies would provide the best evidence for impacts relating to UK gamebird releasing and shooting practices.

The majority of available impact studies were conducted on pheasants, so many of the discussions in this review are derived from information only available for this species. It is possible that with their lower numbers relative to pheasants (Fig. 2, section 2.1.1), and with an ecological niche potentially closer to that of the UK native grey partridge, which it has largely replaced in the British countryside following grey partridge declines (92% over the last 45 years; Hayhow *et al.* 2017), the impacts of red-legged partridge releasing on native wildlife may not be so pronounced. Pheasants in contrast represent a novel addition to the native UK avifauna in terms of ecological niche occupied, and are also released in considerably larger numbers.

3.2   Literature search

Bicknell *et al.* (2010) referred to 198 sources published between 1927 and 2010 in their bibliography, of which 75 were cited as evidencing an impact of non-native gamebirds. Their original reference database also contained an additional 127 sources, giving a combined total of 325 sources collated by their study (published between 1876 and 2010, with 99% from 1962 onwards). According to Bicknell *et al.* (2010) these sources were compiled through literature searches of scientific publications and grey literature, and through contacting stakeholder organisations including the Game and Wildlife Conservation Trust (GWCT), RSPB, BTO (British Trust for Ornithology), Avon Wildlife Trust, Buglife and Butterfly Conservation.

To update this list with sources published since 2010, a systematic literature search was conducted in Web of Science in early November 2019 with automatic weekly alerts enabled for any new sources added to the database using the same search criteria from November 2019 to April 2020. Initial test searches were run using different search terms to determine the most appropriate search string to use. The final search string focussed on general gamebird and geographical terms only; including search terms specific to individual aspects of gamebird release (e.g. habitat management or supplemental feeding) returned fewer results than more general terms and excluded some sources known to be relevant *a priori*. The final search string used in Web of Science was:

(gamebird* OR "game bird*" OR pheasant* OR "red* legged partridge*" OR "Alectoris rufa" OR "Phasianus colchicus") AND (UK OR "United Kingdom" OR Britain OR England OR Scotland OR Wales OR "Northern Ireland")

Quotation marks represent exact phrases and asterisks represent possible word truncations. The search was conducted using the TOPIC option in Web of Science, which searches the title, abstract and keywords of sources, and all available databases were included. Search results were refined to English as a language both pre- and post-searching, and the timespan set to 2010–2019 inclusive (though see above).

198 sources were identified by Web of Science from 2010-2019, plus an additional 2 sources from 2020. Search results were then exported into the reference manager EndNote X8.2 and a three-stage filtering process followed to select the most relevant sources:

1. Sources were excluded if their titles strongly suggested that they were not relevant for the current review (i.e. studies were not plausibly related to impacts associated with the release of pheasants or red-legged partridges).
2. Abstracts or summaries of the remaining publications were then read to deduct if the publication was broadly relevant for this review.
3. Electronic copies of the remaining publications were then obtained where possible and read in full, with only truly relevant papers retained.

The bibliographies of the retained papers were examined in detail to identify any further relevant sources missed by the Web of Science search (published 2010–2019) or not included in Bicknell *et al.* (2010) (i.e. published pre-2010), scrutinising particularly the references cited in other recent reviews on this topic (Arroyo and Beja 2002, Viñuela and Arroyo 2002, Mustin *et al.* 2012, Mustin *et al.* 2018, Avery 2019, Chapman 2019, Madden and Sage 2020). These were particularly important for identifying grey literature including reports, MSc and PhD theses, book chapters, conference proceedings, project summaries in organisational annual reviews and detailed anecdotal records. Relevant sources (both peer-reviewed and grey-literature) found on an ad-hoc basis were also included in the final reference list; these included recommendations by colleagues or additional sources found while searching for full texts of other references. Finally, we also searched for PhD theses relevant to the topic of gamebird release using EThOS (ethos.bl.uk: an electronic library of theses published by UK Universities) in November 2019. The search terms "pheasant", "red legged partridge", "gamebird" and "game bird" were searched for individually with no time period restriction (the website functionality precluded more complex search strings). An additional 3 PhD theses were identified through this route.

Our final reference database consisted of 651 sources, of which 326 were newly compiled through this review update (Table 7). A small proportion of these sources, particularly those originating from Bicknell *et al.* (2010), referred to studies conducted outside of the UK. These were retained for the purposes of continuity with the previous review, but were not assessed as part of the ecological impact scoring or socio-economic vote-counting processes (see below).

Table 7. The number of sources in the reference database considered as potential evidence for ecological or socio-economic impacts of non-native gamebird release in the UK. The majority of sources pre-2010 originated from the reference database compiled by Bicknell *et al.* (2010): 99% of these sources were from 1962 onwards. Additional sources identified during our new literature search included those returned by a Web of Science search (which was restricted to searching articles from 2010–2020), as well as additional sources identified from the bibliographies of other sources or on an ad-hoc basis.

Database	1876–2010	2010–2020	Total
Bicknell *et al.* (2010)	325		325
New literature search			326
Web of Science search (2010–2020)		200	
Bibliographic search/ ad-hoc (pre-2010)	58		
Bibliographic search/ ad-hoc (2010–2020)		68	
Total	383	268	651

3.3   Assessing the impacts of gamebird release

The compiled literature sources used a wide range of different study approaches and focussed on many different aspects of gamebird release or associated activities and their impacts on different species or functional guilds. As such, the evidence presented was highly heterogenous in terms of the type and scale of impact, and the reported metrics. Bicknell *et al.* (2010) summarised the evidence for impacts of gamebird release using narrative review methods and vote counting: reporting the number of sources evidencing a positive, negative or benign effect of gamebird release activities on UK biodiversity or socio-economics. However, vote counting accounts for neither the relative importance or magnitude of the reported impacts nor the level of certainty or generality associated with each source (Koricheva and Gurevitch 2013). For impacts on UK biodiversity, where the effects on populations or communities were more readily quantifiable, we therefore developed a two-dimensional scoring system to provide a comparative assessment of both the ecological impact of gamebird management and of the reliability of the study findings. Vote-counting was the only suitable method for assessing the socio-economic impacts of gamebird release as these were highly disparate in nature and therefore difficult to compare objectively or to distinguish in terms of relative importance.

3.3.1   Quantifying the evidence for ecological impacts

Ecological impacts were defined as those associated with impacts on native UK wildlife or habitats (not including any impacts on non-native species listed by the Non-native Species Secretariat of Great Britain[10]) and were grouped within the primary and secondary themes presented in Table 8. These themes largely matched those presented in the summary Table 1 of Bicknell *et al.* (2010).

Ecological impact scores

We developed a two-dimensional scoring system for the ecological impacts of gamebird release which attempted to distinguish sources that reliably identified important ecological impacts of gamebird management from those that identified less important impacts or whose findings were inconclusive for some reason (most commonly due to limited replication or the lack of a suitable control). Important ecological impacts included evidence of population-level demographic responses, while less important

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[10]http://www.nonnativespecies.org

impacts included changes in abundance that might have been a consequence of local re-distribution of individual animals or plants.

Table 8. Primary and secondary ecological impact themes associated with non-native gamebird release activities.

Primary theme	Secondary theme
Game estate management	Farmland management
	Woodland management
	Woodland creation and retention
	Supplementary feeding
	Rodent pest control
	Legal predator control
Shooting practices	Accidental shooting of non-target species
	Direct ingestion of lead ammunition by wildlife
	Environmental lead concentrations
	Secondary poisoning of predators
Illegal persecution	
Direct impacts of gamebirds	Browsing by gamebirds
	Predation by gamebirds
	Resource competition
	Soil enrichment
Disease transmission to wildlife	
Impacts on predators and predation	Food source for predators
	Predation rates
	Predator abundance

Many sources reported multiple different responses to gamebird management often involving multiple species or functional/ecological guilds, or responses that varied between geographical study regions. Where possible we summarised the impacts of gamebird release at the functional/ecological guild level (e.g. farmland birds, woodland birds, butterflies, tree seedlings, woodland ground flora, pollinating insects) by allocating a single overall impact score to each group, and in cases where responses varied between regions, to each region. We only categorised responses at the individual species level where a single species was the sole focus of a reported impact.

The reported responses of species or functional guilds ('cases') to gamebirds, gamebird management or gamebird release activities were categorised in two dimensions: the first reflecting the strength of evidence of an important ecological impact, and the second reflecting the quality of the study design and therefore the reliability of the findings.

Strength of evidence of ecological impacts:

Cases were assigned to one of three strength-of-evidence categories as follows:

High = Source demonstrates a substantial effect of gamebirds, gamebird management or gamebird release activities on demography or community structure of the focal species or functional guild. This includes effects on demographic parameters (e.g. survival, fitness, breeding success), community diversity, species richness or abundance/density measured at a sufficiently large spatial scale to infer a probable local or larger-scale population-level impact on the focal species or functional guild. A sufficiently large scale would include farm- or landscape-scale measures for birds, and field or management plot scale for invertebrates and plants.

Low = A change in abundance or resource usage detected at a scale too small, or using inappropriate field methods, to confidently infer genuine impacts on local population size or demography. The observed effect could reflect a local behavioural response rather than a genuine demographic or community-level response. This category might be assigned to measurements of bird abundance at the field scale (e.g. usage of supplementary feeders) or counts of moth abundance at light traps (which might attract animals from an unknown area and whose capture efficiency might interact with habitat quality or structure).

None = No effect of non-native gamebirds, gamebird management or gamebird release activities detected on the focal species or functional guild.

Quality of study design:

Cases were also assigned to one of two strength-of-evidence categories:

Fair = studies having no or limited replication in space or time, OR lacking suitable treatment vs. control comparisons, with or without evidence of statistical tests.

Good = studies having adequate replication in space and/or time (single subject compared multiple times, multiple subjects compared once, or multiple subjects compared multiple times; "multiple" being > 2) AND involving treatment vs. control or before vs. after comparisons, or exceptionally large-scale observational studies with a high degree of spatial and temporal replication (e.g. multiple subjects observed multiple times at a national scale) AND with evidence of statistical testing (either describing a statistical test and/or presenting a p-value or other test statistic and/or using the word "significant" to describe effects).

We combined the strength of evidence and study design categories into a single ecological impact score expressed on an ordinal scale between -2 and 2 using the matrix presented in Table 9, following the methods of Roos *et al.* (2018) and Buckingham *et al.* (2019). Restricting the quality of design assessment to two categories, rather than three categories used by other recent quantitative reviews (e.g. 'Fair', 'Good', 'Best': Roos *et al.* 2018), ensured that the combined ecological impact score remained easy to interpret: 0 indicating no impact, 1 indicating impacts with less support (either from a demographic/community or study design perspective), and 2 indicating impacts with a high level of support (both demographically and in terms of study design; Table 9).

Table 9. Ecological impact scores of the effect of non-native gamebirds, gamebird management or gamebird release activities on species or functional guilds, amalgamating the evidence for demographic/community-level effects and quality of study design.

Quality of study design	Support for demographic or community-level impacts		
	None	Low	High
Fair	0	1	1
Good	0	1	2

Due to the low number of empirical sources which directly studied any associated impacts of gamebird release activities for some themes (i.e. where the impact or association with gamebirds was measured directly), we also scored sources on the basis of indirect (i.e. potential) impacts of gamebird release where it seemed biologically and ecologically reasonable to do so. For example, we inferred that

evidence of substantial predation of pheasants by foxes suggests potential impacts of pheasant releases on fox fitness, survival and/or breeding success.

Eight sources demonstrated mixed response directions (+/-) for the same impact and impact theme for different members of a single functional guild in the same region or habitat. Rather than scoring each response direction individually, which would potentially have under- or over-represented impacts, we developed a method to amalgamate such mixed responses into a single score for the entire functional guild. First, we calculated the percentage ratio of significant positive to negative effects, after discounting any benign effects (i.e. where no effect existed). The functional guild was then allocated an overall score with a positive direction if ≥ 67% of the members were associated with positive responses, or negative if ≥ 67% of the members were associated with negative responses. If < 67% were associated with either positive or negative responses (i.e. a less than 2/3rd majority in either direction) then the functional guild was classed as showing no consistent response overall (score = 0). The mixed responses evidenced by the 8 sources were condensed into 14 scores using this method.

To aid interpretation of score statistics and figures, we reversed the sign (+/- to -/+) of scores for any impacts of legal gamebird release activities on generalist predators or scavengers (foxes, corvids or generalist raptors such as buzzard and red kite), so that these impacts could be viewed from the perspective of prey species and predation pressure. For example, if a source identified a positive effect on predator abundance this was scored as negative as it was likely to increase the predation pressure experienced by, and therefore have a negative effect on, prey species. Predation from generalist predators, particularly foxes and corvids, has been identified as limiting the populations of some ground-nesting birds in the UK (Roos *et al.* 2018); any increase in predator populations as a result of gamebird release is therefore likely to exacerbate this problem. The lethal control of buzzards for the purposes of protecting gamebirds under site-specific licenses was included as a legal activity (documented by 1 source), and we assumed such activities led to a reduction in buzzard abundance and therefore predation pressure. We did not alter the magnitude of predator impact scores, nor did we alter the categorisation of scores as directly or indirectly measured. By reversing the polarity of scores in this way, an overall negative impact of gamebird release on predators and predation could be interpreted as a potentially negative impact on sensitive prey species.

The direction of all other impacts on native UK wildlife or habitats was retained as presented in the original evidence sources. Evidence of enhanced native mammal (e.g. wood mouse, badger, brown hare) abundance as a result of gamebird release activities was scored positively for example. We did not score any ecological impacts of gamebird release activities on non-native species (for which there was very little evidence), unless those impacts could plausibly translate to an impact on native species. We found this to be relevant for 1 source, equating to 1 impact score, only: where a positive effect on the fitness and abundance of non-native brown rats and grey squirrels suggested by evidence of their frequent use of supplementary gamebird feed was inferred to lead to potential consequential negative impacts on nesting farmland or woodland native birds through increased rates of predation, and was therefore scored negatively.

We applied this scoring system to the sources cited in Bicknell *et al.* (2010) and those returned by the new literature search (Table 7). Evidence presented by a source was scored if the full text or abstract (where full text was not available) presented empirical results of primary research which indicated an impact on native UK wildlife or habitats from non-native gamebirds, gamebird releasing activities or management of shooting estates for the benefit of non-native gamebirds. Sources which presented opinions or policy perspectives relating to the topic were not scored, neither were reviews, unless these presented the results of empirical meta-analyses or of primary research which were not presented in any other source in sufficient detail to enable scoring. We attempted to reduce double-scoring of the same impacts reported in multiple outputs from the same study (identified based on authorship, study area, study period and design) by prioritising peer-reviewed literature where available, followed by grey

literature of decreasing detail and complexity. For example, where a peer-reviewed paper had been published from a PhD thesis which was also summarised as a project summary in an organisational annual review, we preferentially focussed on the peer-reviewed paper, then the thesis, then the project summary and only scored the latter two if they presented new evidence not covered by the peer-reviewed paper. Multiple sources from the same study were only scored individually if they presented evidence for different impacts or different functional guilds.

Where sources concluded or hypothesised that evidence for an impact on a functional guild was conceivably related to multiple primary and/or secondary themes (Table 8) then the score was allocated to each of the multiple themes. For example, if evidence of changes in woodland vegetation community structure surrounding gamebird release areas was hypothesised as related to browsing by gamebirds, soil enrichment from gamebird faeces and habitat management for the benefit of gamebirds, the same score would be assigned to all of 'direct impacts: browsing by gamebirds', 'direct impacts: soil enrichment' and 'game estate management: woodland management'.

In total, 233 impact scores evidenced by 122 sources (of which 75 were peer-reviewed and 47 were grey literature) were identified using the above scoring methods.

Statistical analysis

To analyse the ecological impact scores (an ordinal variable with levels -2, -1, 0, 1, 2) we used ordinal logistic models (OLMs), also known as ordinal logistic regression (OLR) or cumulative link models (CLM) using the '*ordinal*' package in R (R Core Team 2018, Christensen 2019). The ecological impact score was the response variable and we specified a logit link, Laplace likelihood approximation and equidistant ordinal category thresholds in all models (Christensen 2019). As we were primarily interested in the differences in average scores between the six primary themes, our main model incorporated a categorical explanatory variable (6 levels; Table 8) specifying the primary impact theme against which scores were associated. We also ran a model in which the two primary themes with the largest samples of case scores (Game estate management and Direct impacts of gamebirds) were split into their constituent secondary themes within the categorical explanatory variable. Under game estate management, rodent pest control (for which there were only two scores) was combined with supplementary feeding (14 scores) giving a total of 13 levels for OLM.

To control for single literature sources contributing multiple individual scores for multiple functional guilds and/or themes, our initial aim was to include a source identifier (122 levels) as a random intercept within the OLM (i.e. in an ordinal logistic mixed model, OLMM). Models including this random effect did not converge due to a lack of replication within random effect levels (e.g. just one score available for many studies). As a proxy for this random effect, we therefore tested an alternative random effect of first author (having 80 levels), assuming that studies authored by the same principal researcher may focus on similar topics or themes and could therefore evidence similar ecological impacts. This controlled for much of the pseudo-replication between sources/first authors, while still allowing comparisons between the primary themes without needing to amalgamate themes or scores. A likelihood ratio test indicated no significant improvement in model fit on inclusion of this random effect (variance of random effect = 0.0438, AIC OLMM including random effect of first author = 592.4, AIC OLM without random effect= 590.5, χ^2 = 0.07, df = 1, *P* = 0.7886; both models including primary theme as the explanatory variable), so we excluded it for all subsequent analyses.

The final OLMs also included a weighting term which down-weighted scores from grey-literature sources (which we considered to be less reliable) relative to peer-reviewed literature, and scores for indirect or potential impacts relative to directly measured impacts. These weights are summarised in Table 10. Removal of this weighting term from the model resulted in similar model outputs and patterns of predicted mean scores, although predicted error terms were marginally smaller.

Table 10. Weighting values applied to scores associated with directly measured or indirect (potential) impacts from grey or peer-reviewed literature in OLM analysis.

Impact type	Source type	
	Grey literature	Peer-reviewed
Indirect	0.25	0.5
Directly measured	0.5	1

We used Sidak post-hoc tests for multiple estimates to identify significant differences between estimated marginal means (also known as least-square means) of the impact theme scores using the '*emmeans*' and '*multcomp*' packages (Hothorn *et al.* 2008, Lenth 2020).

Narrative text accounts

In addition to the scoring and analysis described above, discursive text accounts for each primary theme with subsections for each secondary theme were also compiled, broadly based on text in Bicknell *et al.* (2010), but rearranged and extensively edited with the addition of more recent evidence. These text accounts refer to sources from which ecological impact scores were derived, but also include sources returned by the systematic literature search which were not suitable for scoring but which nevertheless provide useful evidence. We also draw on a wider body of literature to support specific discussion points.

3.3.2   Quantifying the evidence for socio-economic impacts

Socio-economic impacts were defined as those associated with impacts on humans or human society and were grouped within the primary and secondary themes presented in Table 11. These impacts were not scored in the same manner as ecological impacts because it was difficult to objectively define thresholds for what might be considered to be High or Low importance. These impacts were therefore summarised using a vote-counting method, indicating the number of sources evidencing positive, benign or negative impacts for relevant secondary themes. In total, 28 sources evidencing socio-economic impacts were identified. Narrative text accounts were then compiled for each impact theme as described for ecological impacts above.

Table 11. Primary and secondary socio-economic impact themes associated with non-native gamebird release activities.

Primary theme	Secondary theme
Socio-economic impacts	Economic value
	Employment
	Social cohesion and wellbeing for the shooting community
	Lead consumption in humans
	Disease transmission to humans
	Vehicle and aviation accidents caused by non-native gamebirds

4 ECOLOGICAL IMPACTS

4.1   Ecological impact score results overview

Source publication year and authorship

We identified 233 ecological impact scores evidenced by 122 sources published between 1962 and 2020, 91% of which were published in the last three decades (1990–2020; Fig. 8). Sources published after 2010 accounted for 32% of those scored (Fig. 8). Over half (52%) of the scored sources were authored by at least one author affiliated with an organisation with a stake in the shooting industry, and 41% had at least one author affiliated with an academic institution. Authors affiliated with conservation organisations with no stake in the shooting industry or government departments and advisory bodies collectively contributed to only 27% of sources (Fig. 9).
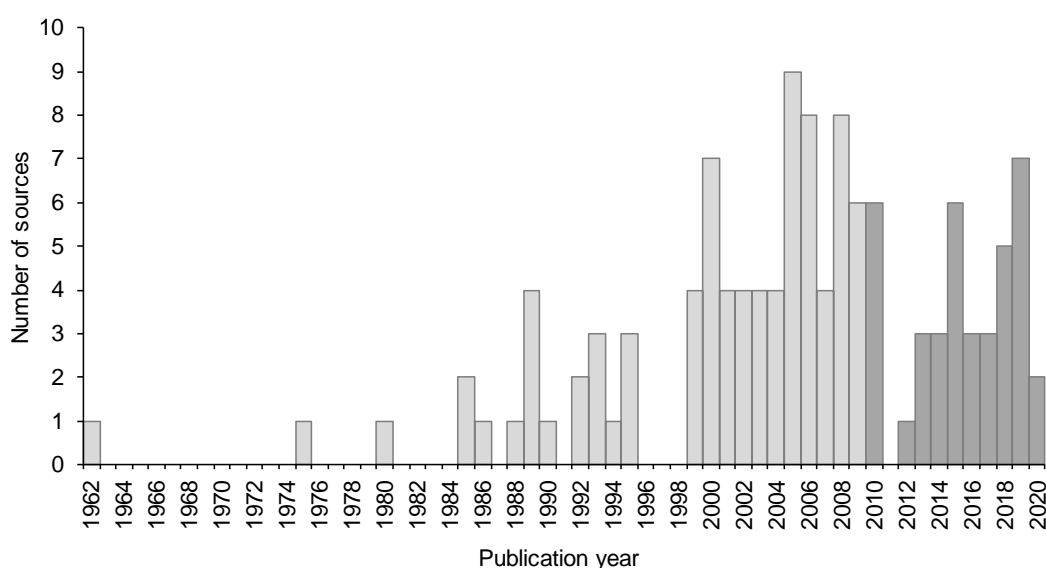
Fig. 8. The temporal distribution of sources (pre- and post-2010) from which ecological impacts of gamebird release in the UK were scored.
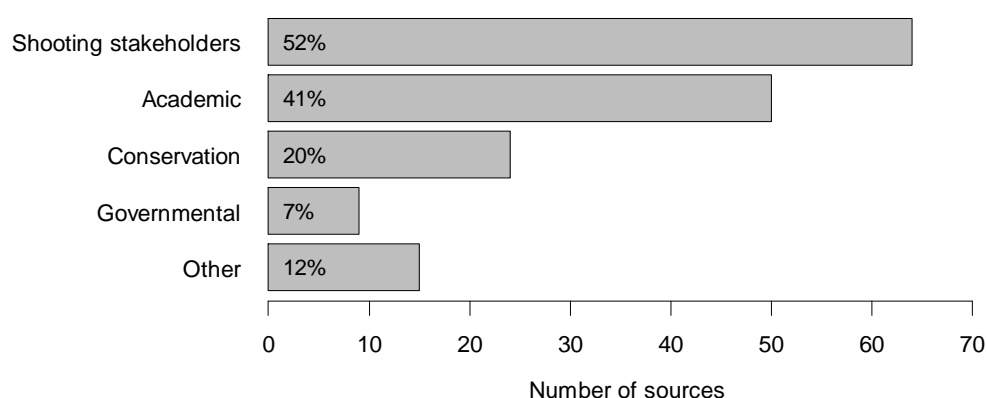
Fig. 9. Author affiliations for the 122 sources from which ecological impacts of gamebird release were scored. Bars indicate the number of sources authored by at least one author from an organisation with a stake in the shooting industry ('Shooting stakeholders', e.g. GWCT, BASC), an academic institution ('Academic', e.g. university or other research institute), conservation organisation with no stake in the shooting industry ('Conservation', e.g. BTO, Wildfowl & Wetlands Trust), governmental department or advisory body ('Governmental', e.g. DEFRA, Natural England, Scottish Natural Heritage), or other affiliation ('Other', e.g. consultancies, unaffiliated individuals). Percentages indicate the proportion of the 122 scored sources with at least one author from each affiliation type; these do not sum to 100% as many sources had multiple authors with different affiliations.

UK wildlife groups affected

Scores were associated with a wide range of taxonomic levels and functional guilds. Over half of the sources (55%) provided evidence of impacts for organisms at the taxonomic level of species, family or order (Fig. 10). 48% of the organisms with evidence of ecological impacts from gamebird release activities were native UK birds, 23% were plants and 14% were invertebrates. Collectively, mammals, predators/scavengers (undefined), reptiles and amphibians ('herptiles'), and 'wildlife' (undefined) accounted for only 15% of scores (Fig. 10).
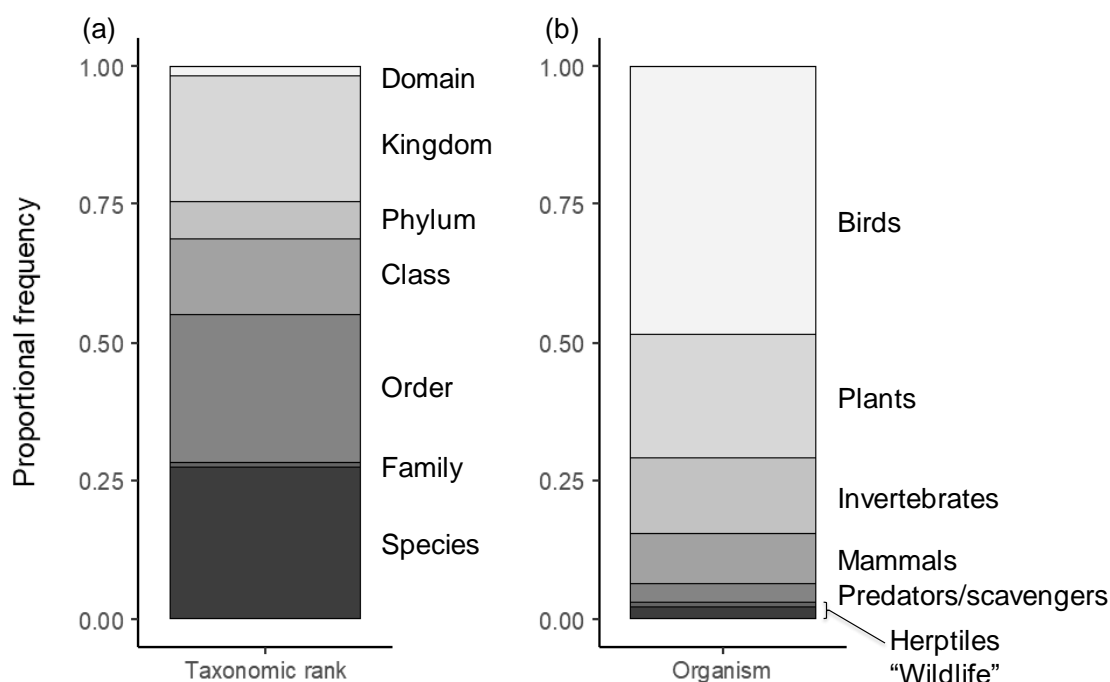
Fig. 10. The proportional frequency of ecological impacts scored at different taxonomic levels (a), and associated with broad organism types (b). In (a), *Kingdom* refers mainly to *Plantae* where plants were grouped by community which could not be easily split taxonomically (e.g. woodland ground flora); *Domain* refers to *Eukarya* incorporating "wildlife" as a functional guild which could not be summarised to any lower order taxonomic rank. In (b), *Predators/scavengers* and *Wildlife* include only those cases where individual organisms or taxonomic groups were not specified by the source.

Different groups of organisms experienced different proportions of positive and negative effects of gamebird release (Fig. 11). Native UK birds were affected most positively by gamebird release activities, particularly by the secondary impacts of habitat management on game shooting estates. Plants, invertebrates and mammals (other than generalist predators) also benefitted from this management, and there were some positive reductions in predation pressure (Fig. 11). Negative effects were however also documented for birds, plants, invertebrates and mammals (other than predators). There was a proportionally higher negative impact from gamebird release in relation to predation pressure, and the impacts on reptiles and amphibians ('herptiles') were exclusively negative (Fig. 11).
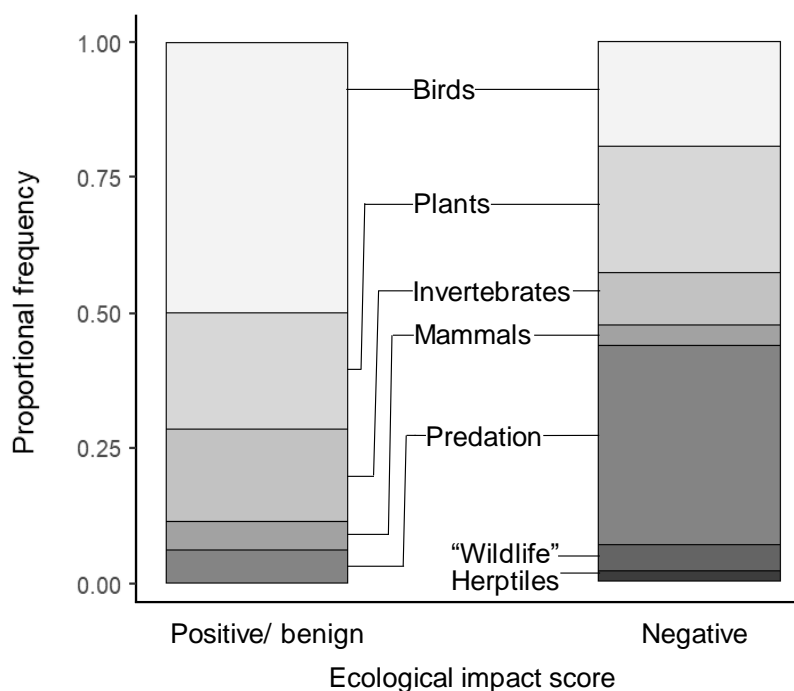
Fig. 11. The proportional frequency of positive or benign scores, and of negative scores, for the ecological impact of gamebird release on different broad organism types. *Predation* refers to scores associated with the impact on predation pressure or on any predator or scavenger species, such that a positive score indicates an improvement in predation pressure (i.e. a reduction in predation, which may be linked to a decrease in predator abundance), while a negative score indicates a deterioration in predation pressure (i.e. an increase in predation, which may be linked to an increase in predator abundance). *Wildlife* includes only those cases where individual organisms or taxonomic groups were not specified by the source. *Herptiles* are reptiles and amphibians collectively.

Ecological impact score analysis

Overall there was a significant difference among primary impact themes in the evidence for impacts of gamebird release (OLM, LR χ^2 = 100.3, df = 5, P < 0.0001; Fig. 12). Post-hoc Sidak tests indicated that there was a significant positive ecological effect of habitat and other management on gamebird rearing estates ('Game estate management': P < 0.0001), while the impacts associated with all other primary ecological impact themes were negative ('Shooting practices': P = 0.0001, 'Illegal persecution': P = 0.002, 'Direct impacts of gamebirds': P < 0.0001, 'Disease transmission to wildlife': P = 0.0267, 'Impacts on predators and predation': P = 0.0111; Fig. 12).

	Game estate management	Shooting practices	Illegal persecution	Direct impacts of gamebirds	Disease transmission to wildlife	Impacts on predators and predation
No. of scores	125	16	18	42	13	19
No. of sources	59	12	13	20	13	17

Fig. 12. Box plot showing variation in ecological impact scores across the six primary impact themes relating to non-native gamebird release. Horizontal lines within boxes are estimated marginal means predicted by an ordinal logistic model (OLM), boxes are ±1 standard error and whiskers are 95% confidence intervals. Lower-case letters (a, b) above box plots are used to indicate significant differences in mean score between themes: themes for which the letter differs had significantly different mean scores (P < 0.05). All mean scores were significantly different from zero (confidence intervals do not overlap zero, P < 0.05). Any impacts on predators and predation (within 'Impacts on predators and predation', and legal control within 'Game estate management') are scored from the perspective of predation pressure, i.e. negative impacts on predator populations are likely to result in lower predation pressure and therefore positive impacts on prey species, and so have been scored positively (and vice versa). The number of scores and sources associated with each primary theme are provided below the plot; some sources were associated with scores for multiple themes. [Note this is a repeat of Fig. 1 from the report synopsis]

The results from the main OLM (Fig. 12) were replicated when the two primary themes with the largest samples of case scores ('Game estate management' and 'Direct impacts of gamebirds') were split into their constituent secondary themes within the categorical explanatory variable. Within this expanded model, there was also a significant difference among themes (OLM, LR χ^2 = 112.9, df = 12, P < 0.0001; Fig. 13), with post-hoc Sidak tests indicating that the average scores for all secondary themes within 'Game estate management' were positive (two significantly so: 'Farmland management' and 'Legal predator control', P < 0.001 in both cases). In contrast, the average scores for all secondary themes within 'Direct impacts of gamebirds' were negative, one significantly ('Browsing by gamebirds', P = 0.009) and two at a marginal level of significance ('Predation by gamebirds', P = 0.042; 'Soil enrichment', P = 0.041). Impacts associated with the secondary theme of 'Resource competition' within 'Direct impacts of gamebirds' were negative on average but not significantly, and had a low sample size (4 scores) with very large error estimates (P = 0.557; Fig. 13).
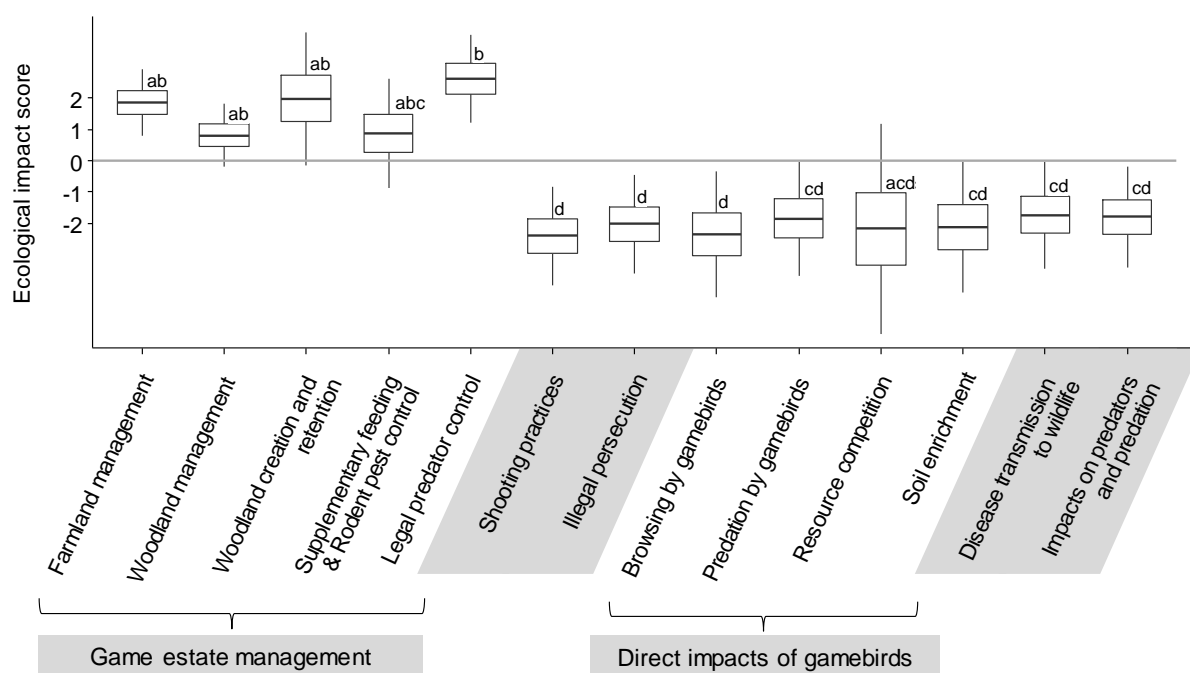
Fig. 13. Box plot showing variation in ecological impact scores across secondary impact themes for the two primary themes with the largest sample of scores ('Game estate management' and 'Direct impacts of gamebirds'), and across the remaining four primary themes (grey shaded; where small sample sizes prevented splitting by secondary theme). Horizontal lines within boxes are estimated marginal means predicted by an ordinal logistic model (OLM), boxes are ±1 standard error and whiskers are 95% confidence intervals. Lower-case letters (a, b, c, d) above box plots are used to indicate significant differences in mean score between themes: themes for which the letter differs had significantly different mean scores ($P < 0.05$). Mean scores which are significantly different from zero have confidence intervals (whiskers) which do not overlap zero ($P < 0.05$). Any impacts on predators and predation (within 'Impacts on predators and predation', and legal predator control within 'Game estate management') are scored from the perspective of predation pressure, i.e. negative impacts on predator populations are likely to result in lower predation pressure and therefore positive impacts on prey species, and so have been scored positively (and vice versa).

Results summary

The scores and impacts associated with each primary and secondary impact theme are further summarised together in Table 2 and separately in tables at the beginning of the primary theme text accounts:

Primary theme	Summary table	Page
Game estate management	Table 12	53
Shooting practices	Table 16	73
Illegal persecution	Table 17	85
Direct impacts of gamebirds	Table 19	91
Disease transmission to wildlife	Table 21	106
Impacts on predators and predation	Table 23	116

Of all the secondary themes summarised in Table 2, including those where small sample sizes prevented individual analysis within the OLM, 13 median scores were negative (68% of 19 themes), while five were positive (26%) and one was benign (5%). No secondary themes were associated with entirely positive scores: evidence for positive impacts always existed alongside evidence for either

benign (one secondary theme, 5%), or both benign and negative impacts (six secondary themes, 32%). In contrast, evidence for negative impacts existed in the absence of benign or positive impacts for seven secondary themes (37%), or alongside evidence for benign impacts for five secondary themes (26%). There is therefore more evidence for a negative ecological impact of gamebird release in the UK overall. This may in part reflect inherent bias in the impact themes studied. Some, such as 'Illegal persecution', 'Disease transmission to wildlife' and impacts associated with the use of lead ammunition under 'Shooting practices', may be expected to be more often associated with negative effects on wildlife, so it is perhaps unsurprising that they are scored negatively on average. Similarly, some aspects of 'Game estate management' such as 'woodland creation and retention' would be expected to have a more positive effect.

Positive ecological impacts of gamebird release are largely restricted to the secondary benefits of gamebird management on habitat quality on arable farmland and woodland, and associated with supplementary feeding and legal lethal predator control. These benefits affect a wide range of taxonomic groups including plants, invertebrates, birds and mammals, and tend to be local in scale for groups other than birds, largely influencing local populations on the sites on which they are implemented. A high proportion of the available literature and evidence is associated with these positive impacts.

Most other ecological impacts of gamebird release are negative, with the evidence base strongest for direct impacts of gamebirds themselves (browsing, predation, competition) and current shooting practices (particularly the use of lead ammunition). Negative impacts affect a wide range of taxa and are often dependent on the densities at which gamebirds are released, with higher density releases associated with more negative effects. They are often experienced outside as well as inside the boundaries of gamebird estates, for example through direct impacts of gamebirds dispersing onto neighbouring land, the spread of disease, lead accumulation up through food chains, or potential impacts on the abundance of wide-ranging generalist predators. The extent and strength of evidence underpinning most of these negative impacts is limited, as reflected in the generally lower number of scores and sources, and further research is required to confirm the extent and magnitude of the effects summarised in this report.

51

4.2   Game estate management

4.2.1   Impact summary

There was a relatively large amount of evidence (125 scores evidenced by 59 sources) for impacts of game estate management on native UK wildlife. The ecological impact scores associated with game estate management were significantly positive on average ($z = 6.64$, $P < 0.0001$; Fig. 12), with positive average impact scores associated with five secondary impact themes and a negative average impact score for one secondary theme (Table 2, Table 12, Fig. 13). These scores reflect largely positive, local, secondary benefits of habitat management, supplementary feeding and legal lethal predator control for gamebirds on game estates, which affect a wide range of taxonomic groups including plants, invertebrates, birds and mammals. Note that the scores associated with legal predator control largely reflect potential reductions in predation pressure through negative impacts on predator populations, the effects of which are likely to be positive for prey species and hence scored positively (see Methods). A high proportion (48%) of the total number of literature sources scored by this review was associated with game estate management (Fig. 12), providing direct evidence of an impact for 83% of the scores associated with this impact theme (Table 12).

On farmland, the creation and management of conservation headlands, cover crops, beetle banks, grassy margins and hedgerows all have positive effects on the abundance, breeding success, brood size and/or species richness of farmland passerine birds, grey partridge, invertebrates including butterflies and bumblebees, rare arable weeds and hedgerow plant communities. Management of woodland canopies to increase light levels, coppicing or maintaining open rides and glades leads to overall positive effects such as higher abundance and species richness of butterflies, woodland birds (e.g. nightingale, warblers), small mammals and detritivore invertebrates. Areas immediately surrounding gamebird release pens are negatively affected however, with lower plant species richness, changes in plant community composition, lower tree seedling regeneration and lower carabid beetle abundance. A large proportion of gamebird estates also retain existing woodlands and plant new ones to benefit gamebirds, with additional benefits for native birds and other woodland wildlife.

Many species of farmland and woodland seed-eating birds, thrushes and small mammals (rodents, lagomorphs) utilise supplementary feed provided for gamebirds in winter, leading to local increases in abundance. Feeders may however represent a point of disease transfer, and the use of rodenticides to control rats (often attracted by gamebird feeders) is widespread and leads to mortality and significant local population declines of wood mice, bank voles and field voles, with potential for residue accumulation through the food chain if contaminated small mammals are consumed by higher predators or scavengers. Legal lethal predator control suppresses local fox and corvid abundance which may reduce predation pressure on prey species. Such legal predator control of foxes and corvids, when conducted as part of a wider suite of game estate management, results in increases in hatching success, fledgling survival and/or breeding abundance for ground-nesting and hedge-nesting birds (particularly waders and farmland passerines), and there are some benefits to mammalian prey species such as brown hare. The effects of lethal control are local and short-lived however, and considerable immigration from predator populations in the wider countryside mean that high intensity control must be maintained, both within and between years, to maintain the suppression of predator numbers and therefore the positive benefits on prey species.

52

Table 12. Summary ecological impact scores associated with game estate management, including the distribution of scores at each score level, the estimated marginal mean score and 95% confidence limits (95% CL) returned by the Ordinal Logistic Model (OLM; only presented for the primary theme overall as sample sizes precluded OLM analysis for secondary themes), the median and interquartile range (IQR) for secondary themes, and the proportion of scores which originated from peer-reviewed research and for which there was direct evidence of an impact rather than a potential impact.

	Ecological impact score						
	-2	-1	0	1	2	Mean	95% CL
Game estate management	5	10	27	59	24	1.51	0.91 – 2.10
Secondary theme	-2	-1	0	1	2	Median	IQR
Farmland management	0	1	7	19	5	1	0.75 – 1
Woodland management	4	4	15	16	7	0.5	0 – 1
Woodland creation and retention	0	0	2	4	2	1	0.75 – 1.25
Supplementary feeding	0	3	1	9	2	1	0.5 – 1
Legal predator control	0	1	2	11	8	1	-0.75 – 2
Rodent pest control	1	1	0	0	0	-1.5	-1.75 – -1.25

125 scores evidenced by 59 sources*: 59% peer-reviewed, 83% direct evidence
* (Flowerdew 1972, Rands 1985, Robertson *et al.* 1988, Ludolf *et al.* 1989a, Ludolf *et al.* 1989b, Warren 1989, Dover *et al.* 1990, Robertson 1992, Thomas *et al.* 1992, Fuller and Warren 1993, Reynolds *et al.* 1993, Wilson 1994, Reynolds and Tapper 1995, Firbank 1999, Hinsley *et al.* 1999, Stark *et al.* 1999, Brickle *et al.* 2000, Heydon and Reynolds 2000a, Heydon and Reynolds 2000b, Mcdonald and Harris 2000, Stoate and Szczur 2001b, Thomas *et al.* 2001, Stoate 2002, Duckworth *et al.* 2003, Oldfield *et al.* 2003, Parish and Sotherton 2004a, Parish and Sotherton 2004b, Stoate 2004, Brakes and Smith 2005, Draycott 2005, Sage *et al.* 2005b, Stoate 2005, Woodburn and Sage 2005, Callegari 2006b, Hoodless and Draycott 2006, Hoodless *et al.* 2006, Stoate and Szczur 2006, Hoodless and Draycott 2007, Davey 2008, Draycott *et al.* 2008, Parish and Sotherton 2008, Siriwardena *et al.* 2008, Stoate *et al.* 2008, White *et al.* 2008, Aebischer and Ewald 2010, Smith *et al.* 2010, Draycott *et al.* 2012, Lennon *et al.* 2013, Pitches 2013, White *et al.* 2014, Larkman *et al.* 2015, Neumann *et al.* 2015, Sánchez-García *et al.* 2015, Aebischer *et al.* 2016, Franks *et al.* 2017, Swan 2017, Sage *et al.* 2018, Capstick *et al.* 2019a, Porteus *et al.* 2019)

Many of the positive impacts were identified by research conducted by or in collaboration with the GWCT for the benefit of naturalised breeding gamebirds, rather than for large-scale releasing (see author affiliations for the scored sources, Fig. 9 in section 4.1). Although many of the successful management methods are now recommended as good practice through the GWCT website[11], their implementation is likely to be variable, as with many land management programmes (Kleijn and Sutherland 2003, Batáry *et al.* 2015). Many of the beneficial impacts of game estate management will also be local in scale, although some are likely to have broader positive impacts (e.g. the provision of seed-rich habitats for granivorous birds during winter).

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[11] www.gwct.org.uk

4.2.2   Background

There are approximately 3,000 full-time gamekeepers in the UK and a similar number of part-time keepers (Ewald *et al.* 2010)[12], 83% of which are in England, 14% in Scotland and 3% in Wales and less than 1% in Northern Ireland (Ewald *et al.* 2010). Together they are responsible for managing a substantial proportion of the British countryside, estimated at over 4,875,000 hectares, which equates to 76% of all terrestrial protected areas (SSSI, MCZ, NCMPA, NNR, Ramsar, SAC and SPA site designations, Areas of Outstanding National Beauty, National Scenic Areas and National Parks in Great Britain; Ewald *et al.* 2010). This includes gamekeepers responsible for upland moorland and waterfowl shoots focusing on native quarry species, as well as those involved in pheasant and/or red-legged partridge releasing. Most game estate managers or gamekeepers actively manage their land in some way to benefit their released stock because gamebirds that survive from one shooting season to the next lower the number of birds that need to be released, and therefore reduce output costs; 'wilder', often stronger, fitter birds also make better sport for shooting (Robertson *et al.* 1993a, Madden and Whiteside 2014, Robertson *et al.* 2017).
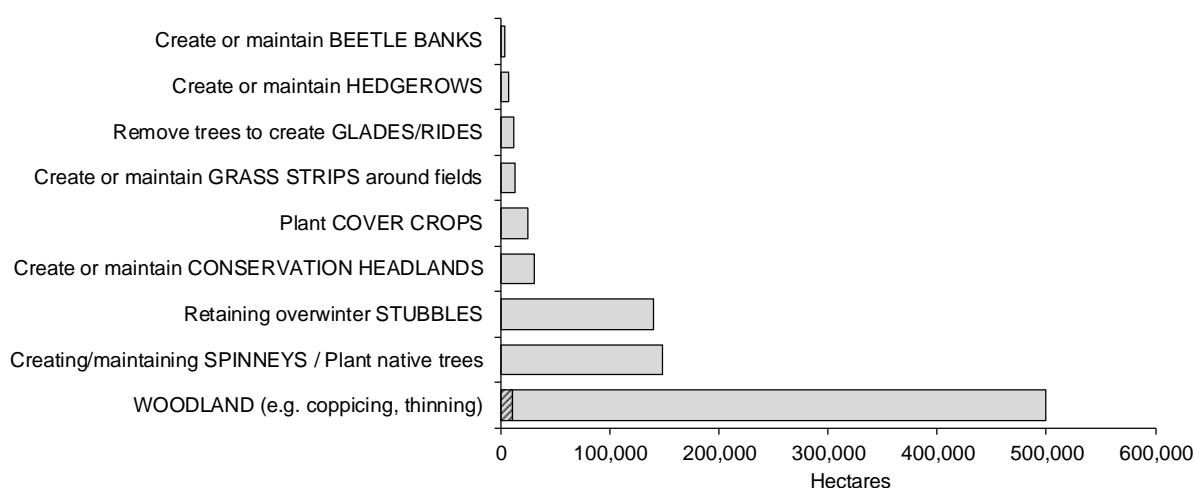
Fig. 14. Estimated area (ha) of habitat and wildlife management carried out on land managed for lowland sport shooting most likely to include pheasant and red-legged partridge shooting in the UK. Management of moorland (management or re-seeding of heather moorland, controlling bracken or re-wetting deep peat) and of wetlands (wetlands, flight ponds, stream or river banks) was not included as such management was unlikely to occur on pheasant and red-legged partridge releasing sites for the benefit of non-native gamebirds. The cross-hatched area within the Woodland management category indicates the area associated with pheasant release pens (~10,000 ha). Data after PACEC (2014).

An estimated 230,000 hectares of farmland habitat and wildlife management, 500,000 hectares of woodland management (including ~10,000 hectares of pheasant release pens) and 148,000 hectares of woodland creation is associated with UK pheasant and red-legged partridge shooting sites (Fig. 14). On farmland, game managers maintain hedgerows, plant cover crops, operate conservation headlands, install beetle banks and use several fallow types such as set aside and grassy field margins to create nesting and foraging habitat for breeding gamebirds, particularly red-legged partridge but also for pheasants (PACEC 2014, Teanby *et al.* 2019, Ewald and Gibbs 2020). Game managers also undertake woodland habitat management such as widening woodland rides, coppicing or sky-lighting to increase understorey light levels, particularly for the rearing and releasing of pheasants (PACEC 2014, Teanby *et al.* 2019, Ewald and Gibbs 2020). The importance of semi-natural habitats such as woodland as gamebird habitat means that many have been retained on game shooting estates whilst they have been

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[12] https://www.nationalgamekeepers.org.uk/about-gamekeeping

increasingly lost in the wider landscape, and game managers also plant new areas of woodland for the same reason (PACEC 2014, Teanby *et al.* 2019, Ewald and Gibbs 2020). Legal lethal predator control is also a very common activity undertaken by game managers, as is the provision of supplementary cereal grain as food to improve gamebird survival and fitness during and after the shooting season (PACEC 2014, Teanby *et al.* 2019, Ewald and Gibbs 2020). Many of these activities are likely to be beneficial to other wildlife but some may also have negative impacts.

These management practices are now widespread and conducted by the majority of game managers (e.g. PACEC 2014, Teanby *et al.* 2019, Ewald and Gibbs 2020), and this may be largely due to Government-funded agri-environment schemes (AES) which pay landowners to undertake land management practices that benefit biodiversity. In the 2018/19 season, 74% of gamebird shoots used land covered by an AES (Teanby *et al.* 2019), and much of the beneficial management above is likely to be funded by this route. Agri-environment incentives are available to any eligible UK landowner, so the motivation to undertake management beneficial to gamebirds and other wildlife is no longer exclusive to game estates, and confounds some of the benefits from game estate management. Ewald *et al.* (2010) and PACEC (2014) suggest that the uptake of wildlife-benefitting farmland management options could be low under agri-environment payments alone in the absence of additional shooting and gamebird release however, and game estates do fund some of the habitat management privately. 48% of lowland shoots reported paying for habitat improvement themselves in 2017/18, although the proportion of management paid for was not indicated (Steel *et al.* 2018). The value of self-funded non-maize wild bird cover crops reported by gamekeepers during a national survey in 2019 amounted to £1.53 million (estimated to cost over £2.25 million if funded through AES) however (Ewald and Gibbs 2020).

4.2.3   Farmland management

Many game estate managers operate various farmland habitat management practices to benefit released and sometimes breeding gamebird stock, primarily for red-legged partridges and native grey partridges, although pheasants may also benefit from this management (PACEC 2014, Teanby *et al.* 2019, Ewald and Gibbs 2020). There is no clear evidence that the farmland management conducted on sites managed for non-native gamebirds is any different to that conducted elsewhere in the UK countryside however (Arroyo and Beja 2002). Practices include the creation of uncut, unsprayed cereal field margins such as conservation headlands (Sotherton 1991) which are established by 47% of gamebird shoots (Teanby *et al.* 2019), buffer strips or fallow field corners (82% of shoots; Teanby *et al.* 2019), cover crops and wild bird seed mixes (86% of shoots), and the maintenance of hedgerows, grass margins and 'beetle banks' (Thomas *et al.* 1992, Tapper 1999, PACEC 2014, Teanby *et al.* 2019, Ewald and Gibbs 2020). Game estates are also more likely to plant hedgerows than non-game farms (Oldfield *et al.* 2003), and Firbank (1999) found that hedgerows on estates managed for gamebird shooting are more often connected to woodlands, making them more valuable as wildlife corridors (Davies and Pullin 2007). Game estates usually contain less grassland than non-game estates however, which may reflect a greater tendency towards arable conversion of grassland on these sites, or the greater frequency of game-releasing in historically more arable parts of the country (Stark *et al.* 1999).

Conservation headlands are strips of crop at field margins that are left unsprayed by herbicides and pesticides to encourage plants that support invertebrates beneficial to wildlife (Sotherton 1991). Grass margins and beetle banks are fallow strips of land that provide nesting and foraging cover as well as increased levels of arable weeds and invertebrates, and are often described as gamebird "chick food" habitats (Thomas *et al.* 2001). Game cover crops or wild bird cover crops are sown seed mixtures based largely on brassicas and cereals that provide food (seeds or associated invertebrates) and shelter for wildlife (Parish and Sotherton 2004a, Ewald and Gibbs 2020). The estimated area of these management options provided across UK farmland is provided in Fig. 14 using data from PACEC

(2014). Significantly more lowland gamekeepers planted game cover crops in 2019 (90%) compared to 2011 (Ewald and Gibbs 2020).

Impacts of farmland management on birds

Table 13 shows predictions that many farmland bird species may respond positively to the farmland habitat management intended to benefit gamebirds. These predictions are verified by findings at the GWCT's demonstration farm at Loddington (Fig. 15), where the majority of native farmland species have responded positively to farmland management deployed primarily for the benefit of gamebirds (Aebischer *et al.* 2016). Increases in abundance of several farmland bird groups on this site were higher in magnitude than at the RSPB's Hope Farm (albeit over a longer time period; Fig. 15), which is managed in a similar way but solely for native wildlife, although trends in farmland specialists and Farmland Bird Index (FBI) species were similar in magnitude between the two sites (Aebischer *et al.* 2016).

Table 13. Expected responses of farmland birds in terms of nesting resource, breeding season foraging, and winter foraging to habitat management practices typical of farmland managed for gamebirds (e.g. Ewald and Gibbs 2020) derived from the positive habitat associations given by Vickery *et al.* (2004). Black and grey cells indicate strong and weaker positive associations respectively. White cells indicate no expected response; 'na' means associations were not applicable, usually for migratory birds not present during winter in the UK. No information is available on negative habitat associations.

	Corn bunting	Grey partridge	Linnet	Reed bunting	Skylark	Starling	Tree sparrow	Turtle dove	Whitethroat	Yellowhammer	Yellow wagtail
Conservation headlands											
Nesting resources											
Breeding season foraging	+	+	+	+	+		+	+	+	+	+
Winter foraging								na	na		na
Beetle banks											
Nesting resources	+	+		+	+						
Breeding season foraging		+		+	+		+			+	+
Winter foraging								na	na		na
Grass margins											
Nesting resources	+	+		+					+	+	
Breeding season foraging	+	+	+	+		+	+		+	+	+
Winter foraging								na	na		
Game cover crops											
Nesting resources		+									
Breeding season foraging	+	+	+	+		+	+	+	+	+	+
Winter foraging	+	+	+	+	+	+	+	na	na		na
More hedgerow											
Nesting resources		+	+				+	+	+	+	
Breeding season foraging							+		+	+	
Winter foraging								na	na		na

56

Fig. 15. Maximum average annual rates of increase in abundance for farmland bird species groupings across five-year periods between 1992 and 2010 at Loddington Farm (a demonstration farm managed for breeding gamebirds and wildlife by the GWCT), and 2000–2010 at Hope Farm (a demonstration farm managed solely for native wildlife by the RSPB) both shown as dark bars, compared with rates of increase in their respective regions (UK East Midlands for Loddington, East of England for Hope Farm) shown as pale bars. BoCC: Birds of Conservation Concern; BAP: Biodiversity Action Plan species; FBI: Farmland Bird Indicator species. Data from Aebischer *et al.* (2016).

Conservation headlands contain broad-leaved weeds that support a rich invertebrate fauna, providing an important food source for the young of several wild bird species during the breeding season in spring and early summer (Vickery *et al.* 2004, Vickery *et al.* 2009). Conservation headlands may increase pheasant and red-legged partridge breeding productivity, and have been shown to increase grey partridge productivity through significantly larger average broods sizes (Rands 1985, Potts 1997).

Beetle banks, grass margins and cover crops provide foraging and breeding habitat for gamebirds, and these areas are important for other birds, particularly finches (Tapper 1999, Thompson and Sage 2000). Brickle *et al.* (2000) found that Corn buntings used grassy margins as a foraging habitat more than any other arable habitat relative to their availability in the landscape. The abundance of herbaceous vegetation in uncut field-boundaries also positively influences the abundance and breeding success of whitethroats and yellowhammers (Stoate and Szczur 2001a).

Significantly more (up to 80 times as many) songbirds including Hirundines, skylark, thrushes, finches, chats and warblers use game cover crops during the breeding season compared to nearby conventional crops (Parish and Sotherton 2004a). Cover crops also influence the breeding distribution of farmland songbirds, with hedges within 200 m of game cover crops on game estates supporting more breeding resident UK songbirds in spring than hedges further away (Sage 2018b). During winter, game cover crops and wild bird cover represent a major food source to native farmland birds (Stoate *et al.* 2003), supporting significantly more species (including chaffinch, greenfinch, blackbird, yellowhammer, dunnock, goldfinch and song thrush) and up to 100 times more birds than conventional crops or set aside and stubble fields (Parish and Sotherton 2004b). Sage *et al.* (2005b) found that game cover contained more than 10 songbirds per hectare during winter, many of which were nationally declining species, while adjacent arable fields contained less than 1 individual. Game cover crops may be particularly important as a habitat for feeding passerines in otherwise pastoral farmland regions, where the availability of other arable crops is low (Parish and Sotherton 2008). They are however only likely to support a wide range of non-game species if they include a diverse range and phenology of small

seed sources such as kale, quinoa, triticale and millet (Stoate *et al.* 2004); maize dominated cover crops designed primarily for gamebirds support very few non-game species (Stoate *et al.* 2004)

There is less consensus about the impact of hedgerow management on game estates. Sage (2018b) found that the number of breeding UK-resident birds was twice as high in game-managed hedgerows compared to hedgerows on non-game farms, while breeding migrant birds were no more numerous in hedgerows managed for the benefit of gamebirds. Draycott *et al.* (2012) also found no strong evidence that game management influenced hedgerow bird numbers when comparing farms with and without gamebird shoots, although this effect differed between regions, with more positive impacts on bird abundance and species richness in the east of England compared to the south.

When the suite of farmland gamebird habitat management measures are considered together, alongside additional measures such as legal predator control (see section 4.2.7) and supplementary feeding (see section 4.2.6), they collectively result in an increase in abundance of nationally declining songbird species relative to non-game farms (Stoate and Szczur 2001b). Increases in abundance over time for blackbird, song thrush, dunnock, whitethroat, skylark, willow warbler, linnet, bullfinch and yellowhammer at Loddington are also reported (Stoate and Szczur 2001b, Stoate 2002, Stoate 2004, Stoate 2005, Stoate 2006, Stoate and Szczur 2006, Stoate 2007, Stoate *et al.* 2008). Grey partridges also benefit from game crops and set-aside areas providing brood rearing, nest site and cover habitats on red-legged partridge releasing sites (Aebischer and Ewald 2010). Songbird species which are not nationally-declining did not differ in abundance between game and non-game farms however, and there was no temporal change in bird species diversity at Loddington (Stoate and Szczur 2001b, Stoate 2002). Callegari (2006b) also found no differences between sites releasing and not releasing gamebirds in total avian abundance, avian species richness or diversity.

A secondary benefit of farmland management aimed at non-native gamebirds, particularly when legal predator and pest control is undertaken, may be an increase in prey species abundance for some predators. Swan (2017) found that buzzards nested at higher densities in pheasant releasing areas due (in part) to the higher number of rabbits present on such sites, which could in turn be explained by the greater food availability and burrowing opportunities afforded by conservation headlands, cover crops, grassy margins and wider hedgebanks, as well as the legal predator control occurring on these sites (see section 4.2.7).

Impacts of farmland management on invertebrates

A reduction in invertebrates caused by agricultural intensification has been identified as a major cause of the decline of many farmland bird species (Fuller *et al.* 1995, Chamberlain *et al.* 2001, Fuller *et al.* 2005, Vickery *et al.* 2009, Hallmann *et al.* 2014, Hallmann *et al.* 2017), and field margins managed for biodiversity conservation have been considered to help reverse these declines (Vickery *et al.* 2009). Decreased use of pesticides at the edges of fields augment the abundance of invertebrate species essential to the rearing of gamebird and other bird nestlings. Conservation headlands contain more butterflies than normal fully-sprayed headlands (Dover *et al.* 1990, Tapper 1999), and significantly higher insect abundance than on plots with sprayed headlands (Rands 1985). Fallow strips such as beetle banks also reduce the need for pesticides throughout fields because they increase the abundance and dispersal of crop pest predators, such as beetles that feed on aphids (Tapper 1999). Thomas *et al.* (1992) found a large increase in abundance and diversity of invertebrate populations in arable fields where grass beetle banks had been added, particularly predatory invertebrates, after three years of the banks being in place, relative to fields without beetle banks. Beetle bank invertebrate communities are similar in structure (although not as diverse) as communities found in traditional grass field margins (Thomas *et al.* 2001), and Brickle *et al.* (2000) found invertebrate densities to be up to eight times higher in un-intensified grassland and grassy margins compared to other arable habitats. In

game cover crops, bumblebees are up to 40 times more abundant than in conventional crops, and butterflies are up to 15 times more abundant (Parish and Sotherton 2004a).

Impacts of farmland management on plants

The absence of broad-spectrum herbicide usage on conservation headlands leads to an increase in the numbers and frequency of occurrence of 17 rare arable weed species (dense silky bent, corn chamomile, dwarf spurge, broadleaved spurge, dense-flowered fumitory, red hemp-nettle, sharp-leaved fluellen, Venus's looking glass, round-leaved fluellen, field gromwell, prickly poppy, rough poppy, corn parsley, shepherd's needle, night-flowering catchfly, field woundwort, narrow-fruited cornsalad; Wilson 1994). Game cover crops contain on average 90% more weed species than conventional crops, particularly broad-leaved species that are important food plants for birds (Parish and Sotherton 2004a). Hedgerows on game estates can be 17–36% wider and 10–65% more numerous on gamebird shooting estates than on farms with no gamebird shoot (Draycott *et al.* 2012), and are more frequently buffered from the pesticide or fertiliser input from adjacent fields by grass margins or uncultivated or unsprayed strips, which has positive effects on hedgerow plant communities (Hoodless and Draycott 2007). However, game management does not influence the structure (height, number of gaps, basal vegetation cover and number of trees) or woody species richness of hedges (Hoodless and Draycott 2007, Draycott *et al.* 2012).

4.2.4   Woodland management

Many traditional woodland management techniques that have been largely phased-out elsewhere are retained within woodlands managed for gamebirds, particularly for pheasants which are usually released into woodland habitats (Arroyo and Beja 2002). Adult pheasants spend much of their time near woodland edges (Robertson 1997), and therefore estate managers often create large open rides thought to be more beneficial than farmland edge habitats by providing more cover from predators. Coppicing is also more commonplace in game managed woodlands to increase light levels for the same reason, as is the practice of "skylighting" where the canopy is opened up to increase light levels below (Arroyo and Beja 2002, Sage and Swan 2003).

Of 965 gamekeepers surveyed in 2019 (many of which were associated with pheasant or red-legged partridge shoots), 82% reported that they had woodland on their land and 96% managed this woodland, totalling 192,051 hectares (Ewald and Gibbs 2020). Similarly, Teanby *et al.* (2019) suggest that 86% of driven gamebird shoots have managed woodland in the last 10 years, averaging 125 hectares of woodland on each estate. PACEC (2014) estimate that up to 500,000 hectares of woodland may be managed by all lowland sports shooting providers nationally (Fig. 14), and that of the 560 shooting premises (of all types) they surveyed, 53% maintained woodlands through coppicing or thinning, 50% created or maintained pheasant or red-legged partridge release pens, 35% removed trees to create glades or rides and 37% created or maintained spinneys or copse woodlands. One in 12 woodlands in England are predicted to contain a pheasant release pen (Sage *et al.* 2005a), and woodlands across the UK are estimated to contain at least 10,000 hectares of release pens (PACEC 2014).

Impacts of woodland management on birds

A reduction in the amount of understorey vegetation in woodlands is considered a contributing factor to the decline in many woodland passerines, particularly migrant species (Fuller *et al.* 2005, Amar *et al.* 2006). Coppicing and increasing the light levels in woodland has been shown to benefit various wildlife, including a variety of birds particularly migrants (Fuller *et al.* 1989, Clarke and Robertson 1993, Fuller and Warren 1993, Fuller and Green 1998, Tapper 1999). Several bird species are therefore expected

to benefit from game estate management, particularly in broadleaved woodlands, where shrub layer and field layer vegetation is often encouraged (Table 14).

Table 14. Expected responses of breeding season woodland birds to habitat differences in game managed woodland, derived from the woodland habitat associations in Smart *et al.* (2007) and Carpenter *et al.* (2009). Habitat differences are defined as woodland characteristics that are significantly different in game-managed woodland compared with non-game woodlands from Draycott *et al.* (2008). Black and grey cells indicate strong and weaker associations respectively (+ = positive response, - = negative response, ∪ and ∩ represent the shape of non-linear relationships). White cells indicate no expected response. Species groupings are from Draycott *et al.* (2008) to enable direct comparison with Fig. 16.

	Finches				Ground feeders					Tits						Warblers						Others I				Others II					
	Bullfinch	Chaffinch	Hawfinch	Siskin	Blackbird	Dunnock	Robin	Song thrush	Wren	Blue tit	Coal tit	Great tit	Long-tailed tit	Marsh tit	Willow tit	Blackcap	Chiffchaff	Garden warbler	Goldcrest	Willow warbler	Wood warbler	Nuthatch	G. spotted woodpecker	Green woodpecker	Treecreeper	Jay	Lesser redpoll	Pied flycatcher	Spotted flycatcher	Redstart	Tree pipit
More grass		+			-	-	-	-	-	-		-		-		-	-		-	+		-	+	-	-	-		+			+
More herbs		+			+									-			+	-		-			+					-		+	
Less moss		-			-		+	+						+		+	+	-	+	-		+								-	-
Less canopy cover	+	+		+	∩	∩	∩	∩	∩	∪		+	+						∩	+	-	+	-		∩						+

Many studies have reported an overall increase in the abundance of woodland bird species on game estates compared with non-game estates (e.g. Hinsley *et al.* 1999, Draycott *et al.* 2008; Fig. 16), or in association with woodland pheasant release pens (e.g. Woodburn and Sage 2005). Hoodless *et al.* (2006) found higher bird abundance (1.5 times more) in broad-leaved game woods compared to non-game woods, and Sage (2018b) also found higher bird abundance in game-managed conifer woods relative to non-game woods. Stoate (2002) and Stoate and Szczur (2001b) found higher abundance, or increasing trends, of woodland breeding birds on the GWCT's demonstration farm at Loddington (where woodland habitat management is conducted alongside a suite of other, primarily farmland orientated, game management techniques) in comparison with nearby non-game farms, although this effect varied between species groups (Stoate 2002). The exact mechanisms explaining these differences cannot be identified with confidence but management of existing woodland (see section 4.2.5) combined with supplementary feeding (see section 4.2.6) and legal predator control (see section 4.2.7) are likely to have been important.

Fig. 16 and Table 15 summarise three studies (Hinsley *et al.* 1999, Davey 2008, Draycott *et al.* 2008) that provide detailed evidence for bird species responses to woodland gamebird management. In general, finch species and ground feeders are more abundant in woodland on game-managed estates (Fig. 16, Table 15). Finches are represented primarily by chaffinches (Table 15), which are widespread generalists, and being chiefly granivorous are likely to be benefitting more from supplementary feeding within the studied woodlands rather than effects of woodland habitat management itself (see section 4.2.6). Ground feeders may benefit from moderately less canopy in game-releasing woodlands (Table 14), but are also likely to benefit from supplementary feeding similar to finch species (Fig. 16, Table 15).
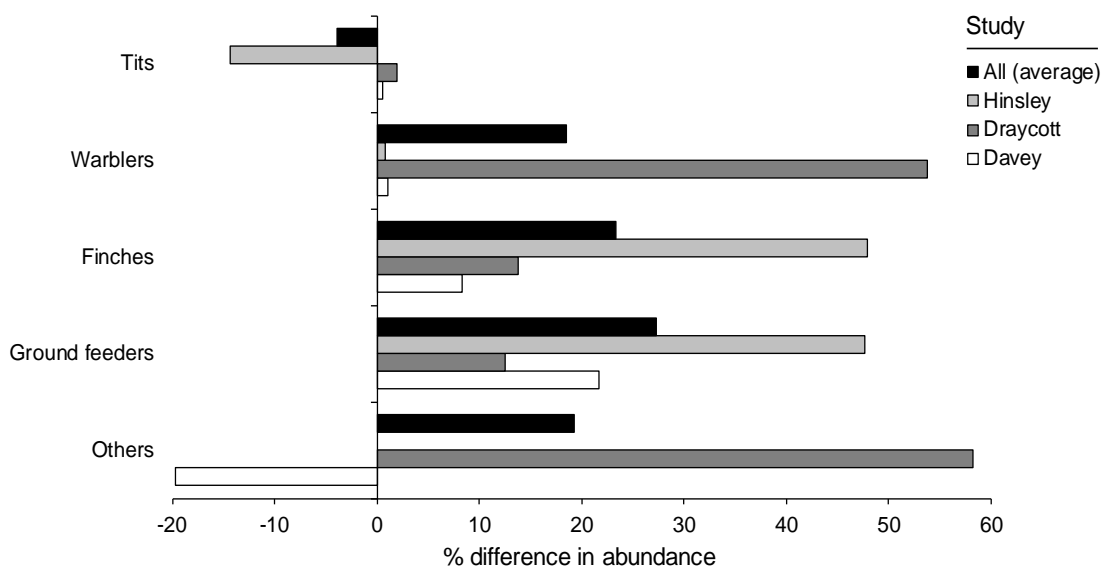
Fig. 16. Observed responses (pooled percentage difference in abundance) of bird groups between game managed and non-game managed estates from three studies testing the effects of woodland gamebird habitat management (Hinsley *et al.* 1999, Davey 2008, Draycott *et al.* 2008). Positive and negative figures represent greater and lower abundance on game managed sites respectively. Bird groupings as follows: Tits = Blue tit, great tit, marsh tit, long-tailed tit, coal tit; Warblers = Blackcap, willow warbler, chiffchaff, garden warbler, wood warbler, whitethroat, goldcrest; Finches = Chaffinch, greenfinch, bullfinch; Ground feeders = blackbird, song thrush, mistle thrush, robin, dunnock, wren; Others = Great spotted woodpecker, green woodpecker, nuthatch, treecreeper, spotted flycatcher. See Table 15 for study descriptions.

Warblers are variable in their responses to woodland management, with some species responding positively to the changes in woodland structure while others respond negatively (Table 15), although they are generally more abundant in gamebird releasing woodlands than those not releasing gamebirds (Fig. 16). These species may be responding differently to changes in ground flora based on differences in individual species ecology (expected associations: Table 14; observed associations: Fig. 16, Table 15). Willow warblers are associated with early successional woodland habitats and may therefore benefit from more grass and less canopy cover as provided through game management (Table 14). Chiffchaff however select older, more mature woods (Smart *et al.* 2007), so may be more negatively affected (Hinsley *et al.* 1999). In general, warblers require ground or understorey cover for nesting, which is more plentiful in game-managed woods, so this may explain their higher overall relative abundance (Fig. 16, Table 15) in such woods (Robertson 1992, Draycott *et al.* 2008).

Species characteristic of broadleaf woodland canopies such as tits, also seem to have few strong expected or observed associations with game management practices (expected: Table 14; observed: Fig. 16, Table 15). The specialist marsh tit may be most likely to respond to woodland game management, and is likely to respond positively to increases in shrub and field layer cover in game-managed woodlands (Carpenter *et al.* 2010). Great tit and blue tit are likely to be unaffected by management (Table 14) because they are habitat generalists (Smart *et al.* 2007).

Nuthatch, treecreeper, spotted flycatcher and woodpeckers ('Others I') may respond negatively to higher grass cover, and this may be due to foraging requirements (expected associations: Table 14, observed: Fig. 16, Table 15). Green woodpeckers for example, forage for ants on the woodland floor and are therefore positively associated with a less dense field layer (Carpenter *et al.* 2009). The evidence available in Table 15 indicates mixed responses across species but also across studies. Robertson (1992) for example, found spotted flycatchers were positively associated with open or coppiced shrubby woodland managed for pheasants, despite the negative associations expected in Table 14.

Table 15. Observed responses (percentage difference in mean bird abundance) between sites with and without woodland gamebird management reported by three studies: Davey (2008), Draycott *et al.* (2008) and Hinsley *et al.* (1999). Positive and negative figures represent greater and lower abundance in managed woodland on gamebird sites respectively; values close to zero show no difference from non-gamebird sites. Cells are left blank where the species is included in the study but pooled in analysis; "-" indicates species not studied. Species groupings are from Draycott *et al.* (2008), where species listed were pooled, except "Others II", which are miscellaneous, unrelated species, and are therefore not pooled in 'all' (differences averaged across species). * indicates significant differences at the *P* <0.05 level. Davey (2008) used the RSPB and BTO Repeat Woodland Bird Survey to compare abundance of birds in woods that contained a pheasant release pen with those that did not, but did not test for significant difference at the species or family level; Hinsley *et al.* (1999) tested bird abundance between paired game and non-game woods; Draycott *et al.* (2008) tested bird abundance between paired game and non-game woods, but did not report figures for individual species except woodpigeon.

Species	Study			Average
	Davey	Hinsley	Draycott	
Finches (all)	8	48	14	23
Bullfinch	-	-		-
Chaffinch	8	48		28
Greenfinch	-	-		-
Linnet	-	-		-
Ground feeders (all)	22	48	13	27
Blackbird	8	31		20
Dunnock	115	-27		44
Mistle thrush	-10	-		-10
Robin	1	6		4
Song thrush	10	150*		80
Wren	6	79		42
Tits (all)	1	-14	2	-4
Blue tit	-5	-14		-10
Coal tit	13	-		13
Great tit	-1	-14		-8
Long-tailed tit	-4	-		-4
Marsh tit	-	-		-
Warblers (all)	3	1	54*	19
Blackcap	-4	-14		-9
Chiffchaff	13	-50*		-19
Garden warbler	11	-		11
Goldcrest	-5	67		31
Whitethroat	-	-		-
Willow warbler	-	-		-
Wood warbler	3	-		3
Others I (all)	-20	-	58*	19
Great spotted woodpecker	-	-		-
Green woodpecker	-	-		-
Nuthatch	0	-		0
Spotted flycatcher	-	-		-
Treecreeper	-39	-		-39
Others II				
Jay	-26	-	-	-26
Redstart	-41	-	-	-41
Woodpigeon	-	-	223*	223

The effects on bird species richness to woodland management for gamebirds are more mixed. Draycott *et al.* (2008) found no difference in species richness between pheasant releasing and non-pheasant woods, while Hoodless *et al.* (2006) found higher bird species richness in pheasant woods compared to non-pheasant woods. Woodland bird species diversity has also been largely unaffected at Loddington (Stoate 2002). The community composition does seem to change as a result of game management however, with higher proportions of woodpigeon and finches in pheasant releasing woods, probably as a result of the addition of supplementary feeding rather than an effect of habitat management (e.g. Hoodless *et al.* 2006).

Impacts of woodland management on invertebrates

The management of woodland habitats for gamebirds has mixed impacts on invertebrates, but the overall pattern is for positive or no effects when game-releasing woodlands are compared to those not releasing or not managing for gamebirds. The impacts of the pheasants which are released into woodlands on invertebrates is more negative; these impacts are described in section 4.5.

Pheasant managed woodlands where pheasants are released often contain a higher abundance and species richness of butterflies than non-game woodlands (Robertson *et al.* 1988, Hoodless and Draycott 2006). Warren (1989) found that 67% of woodland sites considered to be highly important for rare woodland butterfly species, particularly fritillaries, had been used for intensive pheasant rearing or had specifically been managed for pheasants for many years. There are also small scale impacts on butterfly abundance within pheasant releasing woods, with higher occurrence of butterfly species in habitats managed (and more suitable) for pheasants, such as open rides, coppiced oak/hazel stands and sky-lit broadleaf woodland, compared to habitats not managed for pheasants such as unmanaged coppice, mature oak woodland and conifer plantation (Ludolf *et al.* 1989a, Robertson 1992).

The effects of pheasant management on butterflies may be regional however, and depend on the type of woodland within which pheasants are released (Hoodless and Draycott 2006). Broadleaved woodland which would be left unmanaged in the absence of pheasant releasing, is likely to experience the greatest positive effects (Hoodless and Draycott 2006). Capstick *et al.* (2019a) found that the abundance of butterflies was unaffected by gamebird management, with abundance being low in both game-managed and non-gamebird woods. There were more meadow browns in gamebird woods, but more silver washed fritillaries, ringlets, Essex skippers and painted ladies in non-gamebird woods, although overall abundance was so low that these differences were not significant in statistical models and differences in species richness could not be examined (Capstick *et al.* 2019a). Pressland (2009) also found that the releasing of pheasants had no impact on the abundance or species diversity of day flying woodland Lepidoptera when compared to woods where pheasants were not released, suggesting that any positive or negative effects of pheasant management on butterflies may be insubstantial.

The diversity and abundance of bumblebees is unaffected by distance from pheasant release pens (Woodburn and Sage 2005), and Neumann *et al.* (2015) found no significant differences in overall invertebrate abundance or beetle species richness (Carabidae and Staphylinidae) in spring or autumn between either pheasant release pens and non-release control areas within the same woods, or between woods with and without pheasant release. However, Neumann *et al.* (2015) also found that woodland pheasant release pens resulted in a large change in the Carabid beetle community with shifts towards species more typical of arable fields and grassland rather than woodland. There were also changes in the composition of the invertebrate assemblage associated with pheasant releasing, with an overall increase in the abundance of detritivores, including Diplopoda (millipedes), Oniscoidea (woodlice) and Gastropoda (slugs and snails; Neumann *et al.* 2015). This may partly be a consequence of the differences in habitat conditions as a result of woodland management around release pens, but is also suggests a negative effect of pheasants themselves (see section 4.5) through predation of invertebrates and faecal deposition (Neumann *et al.* 2015).

Impacts of woodland management on plants

The impacts of woodland management for pheasants on woodland plants are highly variable and differ between studies and geographical regions. In general, the presence of release pens, management to increase light levels, and the actions of the gamebirds themselves (herbivory, faecal build-up; see section 4.5) combine to significantly alter the community structure of the woodland ground flora.

In a study of 159 woodlands, Draycott *et al.* (2008) found that woodlands managed for pheasants had a more open vegetation structure with 2-7% less canopy cover, a denser herb layer with 5-58% more ground vegetation, more grass, and less moss (Table 14). They found no differences in the woodland shrub layer between game and non-game estates, and actually found a lower mean percentage cover of regenerating tree seedlings in pheasant woods, despite it being widely considered that game estates encourage the woodland shrub layer. Firbank (1999) found no differences in canopy cover, or shrub and field layers in a study of 26 sites, and Davey (2008) found that shrub cover was 30% lower on game sites. Hoodless and Draycott (2006) found that the edges of pheasant-releasing woods had a greater cover and density of shrubs however, and more shrub species, many of which were flowering and fruiting species compared to woods not managed for pheasants. This effect was only found in East Anglia and not in Hampshire, so may reflect woodland habitat type and traditional management differences as well as differences in game management (Hoodless and Draycott 2006).

Ancient woodland indicator plant species are expected to be more common in habitats favoured by pheasants, such as managed coppices, open rides, glades and sky-lit areas (Ludolf *et al.* 1989a) due to the higher light levels on the woodland floor, although Capstick *et al.* (2019a) found no significant effect on the number of ancient semi-natural woodland indicator plant species in woods where pheasants were released compared to non-game woods. On tree trunks, moss and liverwort species diversity, and liverwort abundance, are approximately half that in woods where pheasants are released compared to woods lacking releases (Sage 2018b). These effects on lower order plants may be related to increased nitrogen in the air in pheasant releasing woods (possibly from pheasant faecal decomposition and nitrogen accumulation in the soil; see section 4.5.6), and extend into wooded areas outside of pheasant release pens themselves (Sage 2018b).

Game management can be a significant factor explaining the composition of the ground flora vegetation community of woodland rides, with more fast growing ruderal species (herbs, grasses and other fast growing species indicative of high fertility) and less bare ground in pheasant-releasing woods compared to non-game woods, likely due to the rides in pheasant woods being wider with a more open canopy, more disturbance by vehicles and potentially increased soil nutrient input (see section 4.5) from pheasants (Capstick *et al.* 2019a). The species richness of shrub species along woodland rides is also affected by game management, but the direction of effect varies with region: in the East of England, Capstick *et al.* (2019a) found no clear difference between pheasant-releasing and non-game woods, but the number of shrub species was 41% higher in non-game woods compared to pheasant-releasing woods in the South. In conifer woods, herb abundance did not differ with pheasant-release status, but Sage (2018b) found 30% more bracken and a tendency towards more bramble and grasses in pheasant woods.

Pheasant release pens can have substantial impacts on the woodland ground flora. The presence of pheasant release pens (both when compared to control areas in the same wood, and to separate non-release woods) significantly altered the vegetation community composition of the woodland ground flora: ruderal and disturbance-tolerant perennials, including spear thistle, broad-leaved dock and common nettle, were more frequent at release pen sites, while archetypal woodland species and ancient semi-natural woodland indicator plants were more frequently identified at control sites (Neumann *et al.* 2015). Yellow archangel, greater stitchwort, tufted hair grass, wood sorrel, common honeysuckle, creeping buttercup, yellow pimpernel and herb robert were only present in the absence

of pheasant releasing for example (Neumann *et al.* 2015). Plant species richness was significantly lower in pheasant release pens compared to control sites in the same woodland, with archetypal woodland species and ancient semi-natural woodland indicator plants much less common inside release pens and in pheasant release woods (Neumann *et al.* 2015). There is also generally more bare ground (40%), fewer woodland herb species (15%) and a reduced fern community within pheasant release pens compared to outside (10% bare ground, 30% herbs; Sage 2018b). These effects within and around release pens are strongly influenced by pheasant densities, with more than 1,000 birds per hectare linked with a reduction in the cover of low vegetation and shade-tolerant perennial plants and in increase in the cover of bare ground and ruderal plant species, reducing overall plant species diversity compared to control sites and resulting in changes in species composition away from flora characteristic of ancient woodland (Sage *et al.* 2005a). The influence of pheasant releasing densities are further discussed in section 4.5.7.

Overall plant diversity can be similar inside and outside pheasant release pens (Sage 2018b), as can overall percentage cover and plant species richness (Neumann *et al.* 2015), but this is possibly due to ruderal, annual species displacing perennial ones. Ludolf *et al.* (1989b) found that seedling hawthorn, wood avens and other shade-tolerant perennial plant species were less abundant or absent from active woodland pheasant releasing pens, whereas nettle, chickweed and other fast-growing annuals were more abundant; once release pens were unused for at least 3 years, perennial species started to recover (Tapper 2005). Woodland edges closest to release pens have lower plant diversity than other woodland edges, although the mechanism for this is unknown (Woodburn and Sage 2005). Davey (2008) found no statistically significant evidence that the intensity of game management, such as the size of release pen or the density of pheasants released, had any detrimental impacts on woodland habitats however. The above impacts associated with release pens in woodland are likely to be the results of a combination of factors, including herbivorous browsing, physical damage and nutrient enrichment by pheasants, in addition to the changes in microclimate, particularly light levels, resulting from woodland habitat management for pheasants in these pens (see section 4.5).

Impacts of woodland management on small mammals

Small woodland mammals may be affected by woodland game management such as modifications to field layer floral characteristics as well as supplementary feeding (see section 4.2.6). In a study of 19 game-woods, wood mouse and bank vole distribution was positively correlated with release pens (Davey 2008). In autumn wood mice were more ubiquitous near pheasant release pens, but common shrew abundance was lower nearer release pens (Davey 2008). Shrews may be more sensitive to disturbance, and wood mice may be responding to increased resource levels in release pens during autumn, and protection from terrestrial predators. Despite their avoidance of release pens, Davey (2008) also found that common shrews were positively associated with pheasant density, and suggests that this may be related to increased invertebrate prey around pheasant carrion and/or pheasant faeces (also see 4.5). Coppicing has also been shown to benefit small mammals such as dormice (Fuller and Warren 1993).

4.2.5   Woodland creation and retention

Game estates usually contain more woodland than non-game estates, particularly older, broadleaved woodlands (Stark *et al.* 1999, Arroyo and Beja 2002), largely because game estate managers have retained more woodland on their land (Firbank 1999, Oldfield *et al.* 2003). Oldfield *et al.* (2003) found that game estates were ~6% covered by woodland compared with ~0.6% on non-game estates, and Firbank (1999) found that game-managed estates contained 1-4% more woodland than non-game estates.

The retention and creation of woodland on land managed for non-native gamebird shooting is likely to be of considerable importance to the cover of this important habitat within the UK landscape, with associated benefits to native woodland wildlife (e.g. small mammals: Fuentes-Montemayor *et al.* 2020), birds: (Stoate and Szczur 2001b, Fuller *et al.* 2005), and of particular importance against ongoing semi-natural habitat loss more widely (Ridding *et al.* 2020).

Between the 1960s and 1980s, the number of parcels of woodland in 1 km squares conducting management for gamebirds increased more than in non-game squares, indicating an increase in woodland planting over this period (Duckworth *et al.* 2003). Of 88 lowland driven gamebird shoots (primarily pheasant and red-legged partridge) surveyed in 2018/19, 52% had planted woodland in the preceding 10 years, with areas planted averaging 10 hectares per estate (Teanby *et al.* 2019). This may represent a decline in woodland planting, with an average of 64% of shoots surveyed in the same manner planting woodland between 1980–2018 (Ludolf *et al.* 1989b, Tapper 2005, Steel and Draycott 2014, Steel *et al.* 2018), although 71% of 965 UK gamekeepers surveyed in 2019 (associated with any kind of game shooting activity, not just pheasant or red-legged partridge shoots) reported having planted trees since 2011 (Ewald and Gibbs 2020).

However, since the 1980s both game-managed and non-game UK 1 km squares have seen increases in broadleaved woodland (Duckworth *et al.* 2003). This indicates that woodland creation has not been confined to game-releasing estates, which may in part be explained by the existence of woodland creation payments through AES. Only 45% of gamekeepers report shooting as a reason for planting trees, while 25% cite conservation as the reason for planting, and 19% cite commercial forestry (Ewald and Gibbs 2020).

4.2.6   Supplementary feeding

Supplemental feed in the form of cereal grains is provided on pheasant and red-legged partridge releasing sites to increase the survival of gamebirds and to ensure that any gamebirds surviving the shooting season enter the breeding season in good condition. Food is normally provided in feed hoppers placed throughout pheasant woods and farmland, or less commonly by spreading or broadcasting grain, for example along straw covered woodland rides (Ewald and Gibbs 2020).

Following release at around 6-8 weeks of age, a protein rich 'grow mix' is provided to gamebirds chicks for several weeks in and around the release pen. At around 12 weeks of age, grain (often wheat) is gradually introduced into the diet but the grow mix remains an important component until feathering is complete at around 4 months. After release, natural food inevitably becomes an increasingly important component of gamebird diet, but grain continues to be supplied at feed hoppers. This food improves the condition of adult pheasants, and yields higher winter densities (Robertson *et al.* 1993b, Draycott *et al.* 1998), but does not increase breeding productivity of naturalised birds because chicks require a protein rich diet consisting principally of invertebrates (Sage *et al.* 2002a, Draycott *et al.* 2005). Pheasants that are fed for too long on feed mixtures do not adapt well to being released, although if the feed mix is replaced too early, pheasants suffer deficiencies in condition and growth particularly where they are reared and released at high densities (Sage *et al.* 2002a).

Approximately 85% of pheasant and red-legged partridge shoots provide supplementary food (Ewald and Gibbs 2020), with the amount of food provided annually per site ranging from 0.5 to 3,000 tonnes, with an average of 50 tonnes per site (PACEC 2014, Ewald and Gibbs 2020). In total, at least 23,426 tonnes of supplementary food are provided during the shooting season across the UK, primarily for gamebirds, the majority of which is fed during the release and shooting seasons (August–January; (Ewald and Gibbs 2020). Up to 98% of lowland gamekeepers or gamebird shoots also continue to feed their gamebirds during the 'hungry gap' after the shooting season (February–May; Teanby *et al.* 2019,

Ewald and Gibbs 2020). Much less supplementary food is provided during this period (4,309 tonnes in total; Ewald and Gibbs 2020) and the proportion of estates feeding in this way may have declined since 2015 (Teanby *et al.* 2019). 23% of lowland shoots now also voluntarily feed songbirds, and 14% feed songbirds through supplementary feeding AES options (Steel and Draycott 2014, Steel *et al.* 2018, Teanby *et al.* 2019).

The planting of wild bird cover crops or other seed-bearing crops which may also provide a more natural supplementary food source to native wildlife (e.g. Ewald and Gibbs 2020), is covered in section 4.2.3 above.

Impacts of supplementary feeding on birds

The evidence for impacts of supplementary feeding on birds is largely positive, with supplemental food often considered beneficial to passerines (Hinsley *et al.* 1999, Stoate and Szczur 2001b), although evidence to support this at a more national scale is lacking (Arroyo and Beja 2002). Many previously common farmland birds have declined since the mid-1970s, and for many granivorous species this has often been attributed to a shortage of food during the breeding season or winter months (Fuller *et al.* 1995). Species such as corn bunting, yellowhammer, reed bunting and linnet may benefit from the provision of supplemental food at pheasant hoppers in arable landscapes (Brickle 1997, Siriwardena *et al.* 2008, Bicknell *et al.* 2010). Generalists and human-associated granivores use supplementary seed sources between early winter and January, with peak use by yellowhammer, reed bunting, chaffinch and dunnock occurring in February or later as naturally occurring and arable seeds are depleted in the environment (Siriwardena *et al.* 2008). In a study of 20 game-managed woodlands, the abundance of ground feeding birds such as blackbird, dunnock, robin and wren, as well as nuthatch and blue tit, increased with pheasant feed hopper density (Davey 2008). Another study of pheasant feed hopper use by wildlife in winter showed that 45% of the time spent feeding at hoppers was by passerines, 6% by corvids and 9% by columbids, while only 22% of the time spent feeding at hoppers was by gamebirds themselves (Draycott 2005). More recently, Sánchez-García *et al.* (2015) found that 29 species of native UK birds utilise gamebird feeders in early and late winter on three farms in southern England, including 15 songbird species (chaffinch, blue tit, nuthatch, goldfinch, greenfinch, fieldfare, great tit, dunnock, yellowhammer, house sparrow, linnet, song thrush, starling, blackbird and robin) as well as woodcock, grey partridge, woodpigeon, stock dove, crow, jay, magpie, rook, jackdaw, moorhen, mallard, buzzard, sparrowhawk and kestrel (the last three likely attracted by the presence of prey species). Pheasants and woodpigeons were the main avian consumers of grain during early winter (eating 39% and 29% respectively), whereas usage by songbirds increased during late winter (Sánchez-García *et al.* 2015). The method of providing feed also influences native bird usage, with chaffinch and woodpigeon numbers 2.5 and 3.5 times higher respectively in woods with concentrated winter pheasant feeding at hoppers compared to those where feed is broadcast (Hoodless *et al.* 2006).

There is clear evidence therefore that many farmland and woodland passerines feed on, and benefit from supplemental feed provided at pheasant hoppers. It is unknown whether this has wider demographic benefits for bird populations, although Larkman *et al.* (2015) suggest this may be the case for large seed eating species such as woodpigeon, stock dove, collared dove, jackdaw, crow and magpie, whose combined national annual breeding population size is positively correlated with the supplementary food provision for released pheasants (modelled as the density of pheasants released the previous year; see section 4.5.5).

However, negative impacts may also arise from supplementary feeding of non-native gamebirds. Song thrush and willow warbler abundance decreased as feed hopper density increased in woodlands (Davey 2008), and pheasant hoppers may also provide a point of parasite transfer to native wildlife, with a higher incidence of Trichomoniasis in columbids (woodpigeon, collared dove, stock dove, turtle dove) at farms providing food for gamebirds (Lennon *et al.* 2013). Birds congregating at predictable feeding sites may also increase the potential for predation (e.g. Siriwardena *et al.* 2007, Sánchez-García *et al.*

2015), and increased food availability attracts and may increase the abundance of non-native rodent species such as brown rat and grey squirrel (Sánchez-García *et al.* 2015). These have the potential to predate the eggs and chicks of ground-nesting and woodland birds, although they are often lethally controlled (see sections 4.2.7 and 4.2.8) and evidence of their impact is limited (e.g. Newson *et al.* 2010a, Newson *et al.* 2010c, Broughton 2020). Pheasant feed in some cases may also contain levels of chemicals and medications that may harm or have secondary impacts on wildlife (see section 4.6).

Impacts of supplementary feeding on mammals

Many native UK mammals utilise supplementary food or visit feeders provided for non-native gamebirds, including yellow-necked mouse, wood mouse, bank vole, hedgehog, badger, roe deer, brown hare, rabbit, fox and stoat (the last two species likely attracted by the prey species using the food; Sánchez-García *et al.* 2015), and 18% of time spent at pheasant feed hoppers in woodland was by mammals (Draycott and Hoodless 2005). Wood mouse and bank vole populations increase closer to feed hoppers during autumn and spring (Davey 2008), and experimental supplementary feeding with cereal grains increases juvenile survival and therefore population density in wood mice when food is otherwise scarce (Flowerdew 1972).

The level of excess grain at the end of the shooting season may have negative implications for small mammals however, with the mass and abundance of wood mice and bank voles declining as grain surplus increases (Davey 2008). This may reflect sub-optimal diet choice, increased competition or predation, or disease transmission at feed hoppers (Davey 2008). The use of rodenticides to control brown rats attracted to supplementary food also has negative effects on non-target rodents (see section 4.2.8 below).

4.2.7   Legal predator control

Predator control on game estates

Game estate managers regularly control predators with the aim of protecting the pheasants and red-legged partridges they release and those surviving to breed after the shooting season (Reynolds and Tapper 1996, PACEC 2014, Ewald and Gibbs 2020, Swan *et al.* 2020b). Other motivations for conducting lethal predator control exist among gamekeepers, including the maintenance of professional identity and personal norms, potential penalties from employers if gamebird numbers are not delivered, the perception that such activities impact positively on gamebird and wildlife, as well as personal enjoyment and perceived ease (Swan *et al.* 2020b).

Lethal control of predators on sites managed for gamebirds can be intensive and focus on removing (or reducing) a wide range of (legally-controllable) native and non-native predator species, including fox, stoat, weasel, grey squirrel, brown rat, American mink, feral cat, carrion and hooded crow, magpie, jackdaw, rook and jay. These species were controlled on 74–98% of lowland gamebird shoots where they occurred in 2019 (Ewald and Gibbs 2020). High intensity control, involving a full network of tunnel traps requiring daily checks to catch smaller mammalian predators, combined with fox and corvid control (shooting or snares set around bait stations, or nest destruction and trapping respectively; Tapper *et al.* 1996), is not conducted on all lowland shoots however (Sage *et al.* 2018). The timing and intensity of control on gamebird shooting sites is variable, and may be influenced by the type of shooting favoured: estates that rely on high density releases rather than wild gamebirds may be more likely to control predators during the gamebird rearing, releasing and shooting season only, rather than throughout the year (McDonald and Harris 1999).

A small number of pheasant shoots have been granted licenses by Natural England to legally destroy buzzard nests, and to capture or kill adult buzzards, to protect young pheasants (which are classed as livestock prior to release), despite the protected status of buzzards in the UK (9 licenses issued during 2016–2018; Pitches 2013, Arraut *et al.* 2015, Diamond 2019). Some gamekeepers in gamebird releasing areas also illegally kill protected species, including buzzards, the potential impacts of which are discussed in section 4.4.

Impacts of legal predator control on predators and scavengers

The legal, lethal control of foxes has an overall negative impact on fox populations on a local scale, restricted primarily to the sites undertaking control. Although there are fewer reported effects when culling effort is low, where fox productivity and culling mortality are high, lethal control can effectively suppress fox numbers (Reynolds *et al.* 1993, Reynolds and Tapper 1995, Heydon and Reynolds 2000b, Porteus *et al.* 2019). The numbers of foxes culled on lowland farms to protect game animals can be close to published estimates of annual productivity in British fox populations, meaning that deliberate culling is likely to be the chief cause of mortality limiting fox populations in these regions (Pye-Smith 1997, Heydon and Reynolds 2000a, Heydon and Reynolds 2000b, Heydon *et al.* 2000). Intensive culling efforts on gamebird-releasing sites in the spring and summer are however required to maintain low fox densities, with immigration from the surrounding countryside (where lethal control is less common; Baker *et al.* 2006b) rapidly replacing foxes removed through lethal control (Reynolds *et al.* 1993, Porteus and McAllister 2018, Porteus *et al.* 2019). This effect is similar to those observed in other systems (such as the conservation of ground-nesting waders), where the effectiveness of lethal control is rarely observed or maintained unless control efforts are considerable and maintained, due to the immigration of predator individuals which replace those removed (e.g. Bolton *et al.* 2007).

Interestingly, the control of mammalian predators may have a secondary impact on the abundance of avian predators: Swan (2017) found that buzzards nested at higher densities in pheasant release areas, an effect apparently driven by higher densities of rabbits, which could be linked to the extent of control of mammalian predators (which might otherwise suppress rabbit populations) on these sites, as well as beneficial farmland management (see section 4.2.3). However, the national abundance and spatial distribution of avian predators and scavengers (buzzard, carrion and hooded crow, magpie, raven, jay) appears unaffected by legal control activities on pheasant releasing sites, and indeed is positively associated with pheasant and red-legged partridge releasing, despite legal control of corvids occurring on a widespread basis (Pringle *et al.* 2019).

The nine licenses issued to control buzzards for the protection of young pheasants during 2016–2018 (4 in 2016, 4 in 2017, 1 in 2018) together permitted a maximum of 60 buzzards to be killed (Diamond 2019), although the number of individuals killed may be lower than the number permitted. This may have had an impact on the local buzzard population and their breeding success on the sites licensed to conduct these activities, although no published evidence demonstrating this is available.

Impacts of legal predator control on prey species

Ground-nesting birds are at particular risk of population limitation by predation from mammalian and avian predators such as foxes and corvids (Gibbons *et al.* 2007, Roos *et al.* 2018, McMahon *et al.* 2020). Impacts of predator species on passerines are however less clear (Roos *et al.* 2018), although predator removal may lead to increased hatching success, fledging success and breeding abundance, particularly when all predator species are removed (as is often the case with gamebird management) rather than just a subset (Smith *et al.* 2010).

When conducted alongside other farmland and woodland gamebird management, and over a number of years, intensive lethal predator control has been shown to increase the abundance and/or nest

survival of blackbird, skylark, spotted flycatcher, linnet, bullfinch, dunnock, song thrush, whitethroat, willow warbler and yellowhammer (evidence from Loddington, the GWCT's lowland demonstration farm which manages for, but does not release, gamebirds: Stoate and Szczur 2001b, Stoate 2002, Stoate 2004, Stoate 2005, Stoate 2006, Stoate and Szczur 2006, Stoate 2007, Stoate *et al.* 2008, White *et al.* 2008, Stoate *et al.* 2009, White *et al.* 2014, Aebischer *et al.* 2016). Benefits may also be apparent for mammalian prey species, with brown hare numbers increasing at Loddington during the predator control period, and decreasing again once control ceased (Stoate 2004). These increases in breeding bird and brown hare abundance are thought to have been sustained only by the addition of predator control (Aebischer *et al.* 2016). The negative effect that predators had on songbirds on this site may only occur on sites where generalist predator densities are high in the absence of predator control (as is the case at Loddington); at Hope Farm where predator densities are lower, similar levels of songbird recovery were delivered (see Fig. 15) without any predator control (Aebischer *et al.* 2016).

In other studies, predator control in the absence of beneficial habitat management is predicted to produce a 166% increase in the number of grey partridges a habitat can support (the equilibrium density; Aebischer and Ewald 2004), and increase grey partridge breeding densities almost threefold (2.8 times) when conducted in addition to supplementary food provision (Aebischer and Ewald 2010). Grey partridge productivity also increased by up to 3.5 times over three years as a result of lethal predator control, but did not increase on a nearby site with no predator control, indicating that seasonally targeted fox culling had a local impact only on the site where it is conducted (Reynolds *et al.* 1993, Tapper *et al.* 1996). Curlew abundance, density and population change is also strongly positively associated with pheasant abundance and gamekeeper density, a relationship which is likely to be driven by predator control reducing fox and crow abundance on pheasant releasing or gamekeeper-managed sites (Douglas *et al.* 2014, Franks *et al.* 2017). Predation of released pheasants themselves is also significantly lower at sites with high-intensity predator control (30% predated) relative to those where predator control is less intensive (59% predated; Sage *et al.* 2018), and these effects may extend to other ground-nesting birds or mammals.

The available evidence therefore suggests that predator control on game estates may be beneficial to prey species, particularly where background densities of generalist predators are high and where predator control effort is intensive and sustained throughout the year (Aebischer *et al.* 2016, Sage *et al.* 2018). Due to the short-term impact of control particularly on fox abundance (immigration from the surrounding countryside rapidly replacing individuals removed through lethal control), control effort focussed on the spring and summer period in addition to the autumn/winter (i.e. the gamebird releasing and shooting period) may be essential to achieve management goals for prey species however (Porteus *et al.* 2019). A similar effect is found in other habitats and predator-prey systems (e.g. predator control on lowland wet grasslands to protect breeding waders; Bolton *et al.* 2007).

4.2.8   Rodent pest control

Supplementary feeding of gamebirds and the provision of cover crops attract and provide food and shelter for brown rats, whose numbers can increase and become detrimental to gamebird releasing enterprises (Smith and Shore 2015). The use of anticoagulant rodenticides to control rats at gamebird feeders is widespread, with 91% of gamekeepers reporting their use (Mcdonald and Harris 2000). The use of rodenticides to control rats is ubiquitous across farmland throughout Great Britain, usually in association with farm buildings where grain is stored (Mcdonald and Harris 2000). On game shooting estates, rodenticides are also commonly used by 56% of gamekeepers well away from buildings during the shooting season when natural food sources are relatively scarce, which may increase the likelihood of their being consumed by non-target small mammals (Mcdonald and Harris 2000, Smith and Shore 2015).

Other small rodents are also attracted by supplementary feed and cover crops (Sánchez-García *et al.* 2015), and are able to consume rodenticide bait because they are too small to be excluded by bait boxes (Smith and Shore 2015). A small-mammal trapping study using Longworth live traps by Brakes and Smith (2005) found that 32–67% of wood mice, bank voles and field voles fed on anticoagulant rodenticide from bait boxes targeting rats around different feed hoppers for gamebirds, feeding on bait immediately once it was provided. Wood mice were the most common non-target rodent consumers of rodenticide bait (57% of individuals trapped on average were found to have eaten bait), followed by bank voles (31% of individuals) and field voles (20%; Brakes and Smith 2005). Clinical signs of poisoning were observed in the trapped non-target small mammals (bleeding from orifices), and changes in behaviour were observed in both wood mice and bank voles, which showed reduced escape responses, sometimes with uncoordinated movement and a staggering gait (Brakes and Smith 2005). Shrews were also found dead from rodenticide poisoning in small mammal traps (Brakes and Smith 2005). As a result of poisoning mortality, wood mouse and vole populations declined significantly by an average of 48% in areas surrounding the pheasant feeders following rodenticidal rat control (Brakes and Smith 2005). Populations recovered partially after three months, but the rate of recovery depended on the time of year relative to the small mammal breeding cycle, as recovery was largely dependent on immigration of dispersing juveniles after the breeding season (Brakes and Smith 2005). The timing of rodenticide application is therefore likely to be important in determining small-mammal population recovery, with winter applications which are common in relation to gamebird management likely to severely reduce local populations (Cox and Smith 1990, Cox 1991, Brakes and Smith 2005).

The consumption of rodenticide bait by non-target small mammals also provides the potential for non-target exposure and poisoning of generalist predators and scavengers through residue accumulation through the food chain (Mcdonald and Harris 2000). Anticoagulant rodenticides have a delayed action in rats and other small mammals, in that harmful effects and death may take several days to occur following ingestion of a lethal dose (Smith and Shore 2015). During this time, rodents are likely to continue with normal behaviours while representing a dose of rodenticide to any predator that was to predate them (Smith and Shore 2015). Rodent behaviour may also change in a way that may increase their susceptibility to predation once harmful effects begin to occur (e.g. Cox and Smith 1992, Brakes and Smith 2005, Smith and Shore 2015), and after death scavengers may consume their carcasses which will still contain large amounts of rodenticide (Mcdonald and Harris 2000). Red kites for example, preferentially consume viscera most likely to contain rodenticide (such as the small intestine which may contain undigested rodenticide bait, and liver where rodenticide becomes bound after absorption; (Brakes 2003). Poisoning through the misuse of anticoagulant rodenticides was diagnosed as the cause of death in 17% of 110 red kites found dead and subject to toxicological testing between 1989 and 2007 in England (Pain *et al.* 2007, Molenaar *et al.* 2017), and may be slowing their rate of population recovery and range expansion in England (Molenaar *et al.* 2017). Rodenticide poisoning was the cause of mortality for buzzards (43%), red kites (24%), peregrine falcons (4%), ravens (1%), tawny owls (1%) and unspecified birds of prey (27%) in 9% of 1225 cases of illegal persecution reported across the UK 2007–2018 by the RSPB Raptor Persecution Map Hub[13] (see section 4.4, Fig. 19).

As of 2016, training and certification is now legally required by gamekeepers for the use and purchase of professional-use rodenticides (certified for outdoor use and supplied in large quantities), and Codes of Practice exist to reduce their impacts on non-target species (e.g. CRRU 2015, Sánchez García and Buner 2017).

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[13] www.arcgis.com/apps/opsdashboard/index.html#/0f04dd3b78e544d9a6175b7435ba0f8c

4.2.9   Key knowledge gaps and recommendations

- This impact theme is one of the most commonly studied, with the highest quantity and most extensive bodies of associated peer-reviewed and grey literature. However, research examining the wider demographic effects of supplementary feeding on farmland and woodland birds at a national scale is lacking, as are studies into the impacts of misuse of rodenticides for populations of small mammals and generalist predators at a regional or national scale.

- Legal predator control on game-releasing estates may be beneficial to prey species where background densities of predators are high and predator control effort is intensive and sustained, but there is still a lack of understanding as to whether locally reduced predator populations result in consistent positive effects on the abundance of prey species, and whether this effect is consistent and maintained over time. The extent and intensity of legal predator control on the majority of non-native gamebird releasing estates is also largely unknown.

4.3   Shooting practices

4.3.1   Impact summary

There was a relatively small amount of evidence (16 scores evidenced by 12 sources) for impacts of gamebird shooting practices on native UK wildlife. The ecological impact scores associated with gamebird shooting practices were significantly negative on average (z = -4.27, P = 0.0001; Fig. 12), with negative average impact scores associated with all four of the secondary impact themes, and no benign or positive scores (Table 2, Table 16). These scores reflect the negative environmental impacts of the use of lead ammunition which results in poisoning and mortality where it is directly ingested by wildlife, and can indirectly affect other wildlife, particularly predators, as it accumulates up through food chains. There are also negative impacts associated with the unintentional shooting of sympatric species during pheasant and red-legged partridge shoots.

Table 16. Summary ecological impact scores associated with non-native gamebird shooting practices, including the distribution of scores at each score level, the estimated marginal mean score and 95% confidence limits (95% CL) returned by the Ordinal Logistic Model (OLM; only presented for the primary theme overall as sample sizes precluded OLM analysis for secondary themes), the median and interquartile range (IQR) for secondary themes, and the proportion of scores which originated from peer-reviewed research and for which there was direct evidence of an impact rather than a potential impact.

	Ecological impact score						
	-2	-1	0	1	2	Mean	95% CL
Shooting practices	4	12	0	0	0	-2.29	-3.70 – -0.88
Secondary theme	-2	-1	0	1	2	Median	IQR
Accidental shooting of non-target species	1	2	0	0	0	-1	-1.5 – -1
Direct ingestion of lead by wildlife	1	4	0	0	0	-1	-1 – -1
Environmental lead concentrations	1	2	0	0	0	-1	-1.5 – -1
Secondary poisoning of predators	1	4	0	0	0	-1	-1 – -1

16 scores evidenced by 12 sources*: 75% peer-reviewed, 67% direct evidence
* (Pain *et al.* 1995, Aebischer and Ewald 2004, Butler *et al.* 2005, Potts 2005, Fisher *et al.* 2006, Pain *et al.* 2007, Watson *et al.* 2007, Pain *et al.* 2009, Sneddon *et al.* 2009, Aebischer and Ewald 2010, Lead Ammunition Group 2015, Pain *et al.* 2015)

On shooting estates, unintentional shooting of wild grey partridge can severely impact breeding populations. Galliformes and other bird species ingest gunshot that they mistake for grit, and raptors and other predators and scavengers can suffer secondary poisoning following the consumption of lead-contaminated prey (which have either consumed lead themselves, or contain remains of lead ammunition from old shot wounds). Lead may also be degraded into the soil where it is absorbed by soil invertebrates and plants and can be subsequently passed on up the food chain. Lead poisoning can cause rapid or slow death, reduced breeding success, and have a range of negative sub-lethal fitness, behaviour and welfare effects. It is important to note that there has been a recent call by shooting stakeholders to phase out the use of lead and single-use plastics in ammunition within 5 years (by 2025) in the UK[14]; if complied with, this may dramatically reduce the negative impacts associated with shooting practices.

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[14] https://basc.org.uk/a-joint-statement-on-the-future-of-shotgun-ammunition-for-live-quarry-shooting/

4.3.2   Background

The shooting of released non-native gamebirds can have impacts on UK wildlife, both as a result of accidental shooting of non-target species while shooting pheasants and red-legged partridges, as well as through the use of lead ammunition which acts as an environmental pollutant. Although the use of lead gunshot can also have deleterious effects on human health when consumed (see section 5.1.5), in this section we deal only with the impacts of lead on wildlife, focusing specifically on impacts linked to shooting with lead gunshot over terrestrial rather than wetland habitats in the UK, and where the shooting of pheasants or red-legged partridges is stated or implied.

The use of lead ammunition

The majority of gamebird shooting is conducted using shotguns loaded with plastic cartridges typically containing 100–600 individual lead gunshot pellets depending on size, with a typical 30g load of "no. 6" gunshot (lead pellets ~2.6mm in diameter) for shooting gamebirds containing around 270 individual pellets[15,16] (De Francisco *et al.* 2003, Pain *et al.* 2015). As gunshot leave the barrel of the gun they spread out so only a small proportion hit the target, thus even when the target is hit a considerable proportion of the gunshot will miss and be distributed in the local environment (De Francisco *et al.* 2003, Cromie *et al.* 2010, Pain *et al.* 2010, Pain *et al.* 2015). Intense hunting pressure can result in high deposition rates of lead pellets, and up to tens of thousands of tonnes of lead is thought to be deposited in the environment from shooting activities across Europe and north America annually (Ferrandis *et al.* 2008, Quy 2010).

The tonnage of lead ammunition deposited annually into the UK environment is not precisely known, as there are no official estimates of the number of cartridges fired (Pain *et al.* 2015). The Lead Ammunition Group (2015) however estimates that some 6,000 tonnes of lead ammunition is dispersed irretrievably into the environment each year in UK, while Pain *et al.* (2015) suggest this figure may be closer to 8,000–13,000 tonnes (Fig. 17). At least 40% of this (2,400–5,200 tonnes) is likely to be linked to live quarry shooting (including red grouse shooting, deer stalking and pest control activities, as well as the shooting of non-native gamebirds), while the remaining 60% is linked to clay pigeon and other target shooting (Lead Ammunition Group 2015). Pain *et al.* (2015) also estimate that 2,500–6,700 tonnes of lead gunshot are fired at all types of gamebirds annually, most of which will again fall into the environment. Pain *et al.* (2015) based this estimate on an average of 3–8 shots using 30g lead cartridges made per gamebird killed (which they assessed from shooting web articles and social media); if these same estimates of shooting efficiency and cartridge size are applied to the 19,600,000 gamebirds estimated to have been shot in the UK in 2016 (Aebischer 2019b), this equates to 58,800,000–156,800,000 shots fired, and 1,764–4,704 tonnes of lead potentially deposited over terrestrial habitats in the course of pheasant and red-legged partridge shooting annually. This excludes the gunshot used in the process of legal pest control on game estates (see section 4.2.7), which is likely to increase this tonnage further (Pain *et al.* 2015).

In many soils the degradation of lead is slow, which means that gunshot remain as pellets for a considerable time (up to hundreds of years), and as lead gunshot has been used for over two centuries in the UK there may be a considerable historical legacy of accumulated gunshot in the soil where shooting is traditional (Pain *et al.* 2015, Pain *et al.* 2019b). The speed with which lead pellets sink down through the soil varies between soil types and management practices, but they usually remain available for potential direct consumption by wildlife for many years, although a high proportion of gunshot ingested will be those which were most recently deposited (Pain *et al.* 2015).

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[15] http://shotgunsuk.weebly.com/cartridges.html
[16] https://en.wikipedia.org/wiki/Shotgun_shell

Routes of lead exposure in wildlife
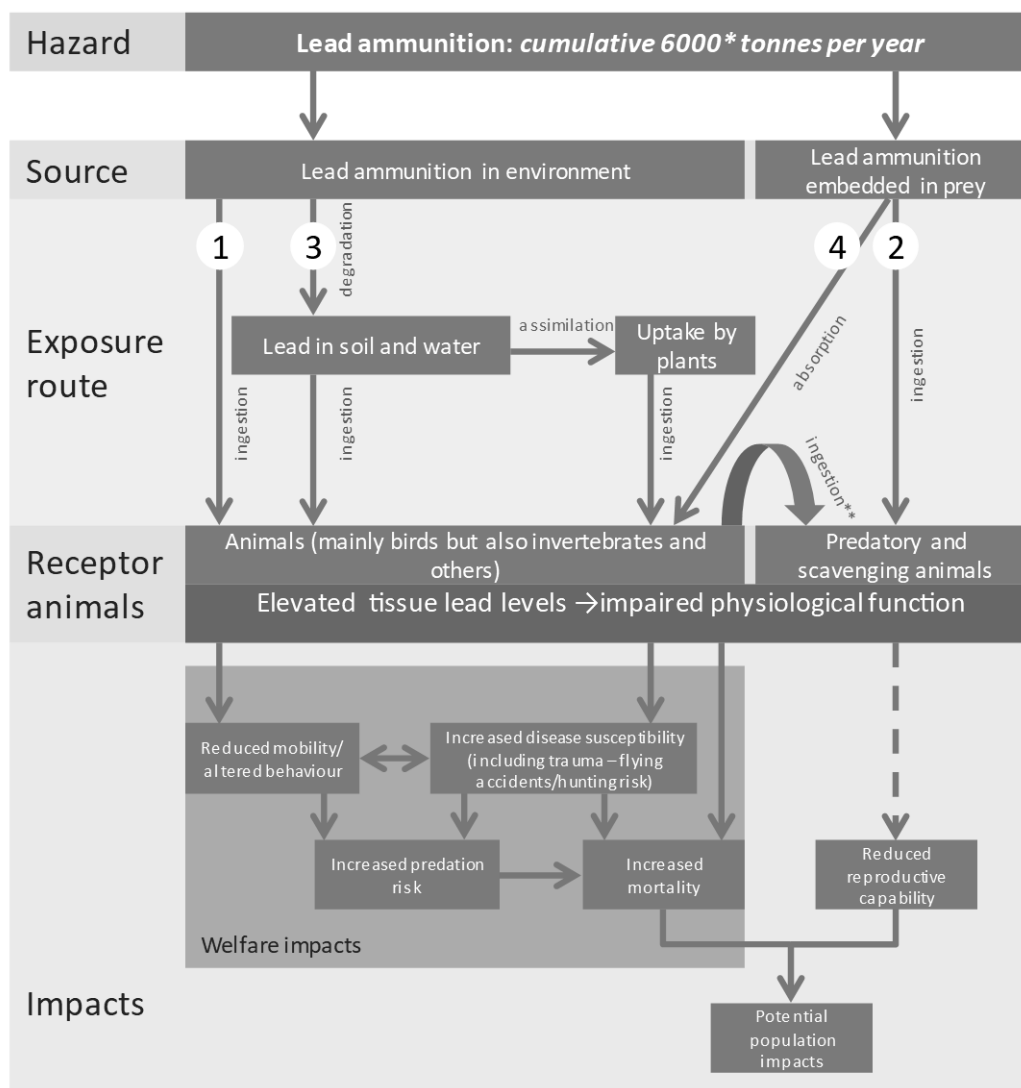
* Other calculations between 8,000-13,000 tonnes per year
** Ingestion of lead in tissues or particulate lead from intestines of prey/carrion

Fig. 17. Schematic illustration summarising the four exposure routes (1–4) and range of potential impacts on wildlife of poisoning from lead ammunition sources. Exposure routes: 1 = direct ingestion; 2 = secondary ingestion from lead contaminated prey; 3 = ingestion of soil, water or plants contaminated with lead that has entered the environment; 4 = absorption of lead into the tissues of animals that have been shot and wounded but survived. Figure is reproduced from Pain et al. (2015) with permission from D. Pain and R. Cromie.

There is strong scientific consensus that lead can, and does, enter wildlife (and humans) by several routes (Lead Ammunition Group 2015, Arnemo *et al.* 2016, Pain *et al.* 2019a, Pain *et al.* 2019b). These routes of exposure and the potential impacts of lead poisoning on wildlife from lead ammunition sources are summarised in Fig. 17, reproduced from Pain *et al.* (2015). Lead gunshot pellets or fragments may be directly ingested either accidentally in the course of feeding, or are mistaken for small stones or grit which are ingested by many bird species to aid digestion in their gizzards (Pain *et al.* 2009, Quy 2010, Pain *et al.* 2015, Pain *et al.* 2019b). Lead can also be absorbed directly into the tissues of animals that have survived being shot and wounded and which contain embedded gunshot fragments (Pain *et al.* 2009, Quy 2010, Pain *et al.* 2015, Pain *et al.* 2019b). Spent gunshot deposited in the environment can degrade into soils and leach into soil water to be taken up by plants and ingested by soil invertebrates,

which are in turn consumed by other animals further up the food chain (Pain *et al.* 2009, Quy 2010, Pain *et al.* 2015, Pain *et al.* 2019b). Finally, secondary lead poisoning occurs in many predatory or scavenging species which consume carcasses of prey with gunshot fragments or accumulated lead in their tissues, or gunshot in their digestive tracts, again providing a pathway for accumulation up through the food chain (Pain *et al.* 2009, Quy 2010, Pain *et al.* 2015, Pain *et al.* 2019b).

Mechanisms and effects of lead poisoning

Lead is a toxic non-essential metal that has no beneficial effects in living organisms, instead acting as metabolic poison which accumulates in living tissues with non-specific effects on a wide range of physiological and biochemical systems (Pain *et al.* 2019b). Lead poisoning (toxicosis) can begin at relatively low lead concentrations, and different levels of exposure can result in sub-clinical poisoning with no outward symptoms, clinical poisoning with outward fitness and behavioural symptoms, or severe poisoning resulting in death. Pain *et al.* (2019b) suggest that sub-clinical poisoning in waterbirds and raptors could occur at levels of lead as low as 20–50 µg/dl in blood, 2–6 mg/kg in liver tissues (wet weight), 2–6 mg/kg in kidney tissues (wet weight) and 10–20 mg/kg in bone (dry weight). After lead is ingested and absorbed by birds, it is transported in the bloodstream before being deposited into soft tissues such as the liver and kidneys where it may remain for several weeks or months (Pain *et al.* 2019b). Lead is also absorbed into bones, where it may be retained for up to tens of years and can accumulate during an animal's lifetime (Pain *et al.* 2019b), and can also be mobilised back into the bloodstream as bones change throughout life.

There is strong evidence that lead causes harm to wildlife, with up to hundreds of thousands of birds potentially killed through lead poisoning annually in the UK (e.g. Lead Ammunition Group 2015, Pain *et al.* 2019b). There is also a considerable body of evidence documenting negative effects on wildlife fitness, breeding success, welfare and behaviour, all of which combined have ultimately deleterious impacts on some populations (e.g. Pain *et al.* 2019b). Much of the published literature surrounding the impacts of lead on wildlife focusses on the negative impact of lead ingestion by waterbirds as a consequence of the use of lead ammunition during wildfowling activities over wetland and coastal habitats (e.g. Pain *et al.* 2019b). More recently, attention has turned to the impacts of lead in terrestrial (non-wetland) habitats and their associated species, with most of the published literature focussing on avian impacts (Fisher *et al.* 2006, Pain *et al.* 2007, Pain *et al.* 2009). Many terrestrial bird species have been found with high concentrations of ingested lead gunshot in their digestive systems, or elevated bone or organ lead levels, including numerous waterfowl and waders, birds of prey, pigeons and doves, and several terrestrial game species (Butler *et al.* 2005, Fisher *et al.* 2006, Mateo 2009, Pain *et al.* 2009). At least 63 bird species have been documented as ingesting lead or suffering lead poisoning from hunting ammunition in terrestrial habitats globally (primarily in Europe and North America), with gamebirds and raptors the most commonly reported (Fisher *et al.* 2006, Pain *et al.* 2009).

Regulation and voluntary use of alternatives to lead

A review of scientific literature detailing the environmental and health consequences of the use of lead in ammunition by Arnemo *et al.* (2016) concluded that the understanding of the widespread deleterious impacts of lead exposure from ammunition on wildlife and humans will change little with further scientific research, and that the scientific community has reached a consensus on the extent and negative severity of this issue[17]. Tackling the issue by implementing restrictions on the use of lead ammunition and moving to non-toxic alternatives is therefore now a question of socio-political will (Arnemo *et al.* 2016, Kanstrup and Thomas 2020).

By 2000, the use of lead ammunition over wetlands in Britain was banned (AEWA 1999), following abundant evidence that the use of lead gunshot causes high levels of lead poisoning in waterfowl (e.g.

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[17] http://www.europeanscientists.eu/open-letter-2020/

(Pain *et al.* 1992, Scheuhammer and Norris 1996). There is a high degree of non-compliance with this restriction in England however (Cromie *et al.* 2010), and it appears that the existing regulations are not yet addressing their principal objective in reducing the use of lead in wetland habitats for wildfowl (Lead Ammunition Group 2015). In Europe, only Denmark and the Netherlands already have a total ban on lead gunshot in all habitat types, while 21 other countries have only implemented partial bans in wetlands and/or for waterbird hunting (Mateo and Kanstrup 2019). Compliance is again variable, and regulation does not secure the protection of birds from lead poisoning unless there is strict law enforcement and continuous monitoring to confirm that lead poisoning is no longer an issue (Mateo and Kanstrup 2019). The European Commission has therefore made proposals to introduce legislation which would further restrict the use of lead ammunition around wetlands (Kanstrup and Thomas 2020)[18].

Globally, other nationally regulated bans of the use of lead shotgun and rifle ammunition are similarly few (Arnemo *et al.* 2016), although many have made calls for a global ban of lead ammunition in all habitats (e.g. Sonne *et al.* 2019). Thomas (2015), Kanstrup and Thomas (2019) and Kanstrup and Thomas (2020) suggest there is no financial or availability barrier to switching to alternatives to lead ammunition such as steel, tungsten, bismuth or copper. In the UK, the British Game Alliance of shooting industry organisations have issued a joint statement committing to the voluntary phasing out of the use of lead ammunition and single-use plastics in shotgun cartridges by 2025[19]. Considerable progress has also been made in raising awareness and encouraging a switch to non-toxic ammunition alternatives in recent years, despite a conspicuous absence of political and regulatory action (Pain *et al.* 2020). However, there is still resistance to change from parts of the shooting and ammunition-making communities (Newth *et al.* 2019, Kanstrup and Thomas 2020), with several sociological and political barriers combining to inhibit both compliance with existing regulations and a transition to wider use of non-toxic ammunition (Cromie *et al.* 2015, Newth *et al.* 2019, Kanstrup and Thomas 2020). Some members of the shooting industry still fight against restrictive bans of the use of lead gunshot for example[20], despite committing to the voluntary phasing out of lead ammunition[18]. Pain *et al.* (2020) suggest that the threat of lead poisoning to wildlife in the UK is unlikely to be fully resolved until Government action is taken to regulate or ban the sale and use of lead ammunition .

4.3.3   Accidental shooting of non-target species

The unintentional, accidental shooting and mortality of non-target species may occur during pheasant and red-legged partridge shooting activities. Published evidence is sparse and deals primarily with the unintentional shooting of sympatric Galliform species such as grey partridge.

Commercial shooting of released red-legged partridges may result in the accidental shooting of grey partridges because the two species can be difficult to differentiate from a distance in flight. Grey partridges are themselves a quarry species and are shot intentionally on some sites (e.g. Watson *et al.* 2007, Draycott 2012), but unlike red-legged partridges, grey partridges are native to the UK and have suffered population declines of 92% over the last 45 years, resulting in their classification as a Red Listed species (Hayhow *et al.* 2017), largely as a result of habitat deterioration through agricultural intensification and increased predation (Potts 1986). The GWCT therefore recommends that grey partridge are only shot where wild densities are greater than 20 per 100 hectares (0.2 per hectare) and where conservation measures (habitat creation, supplementary feeding and legal predator control) are undertaken to ensure a shootable surplus (GWCT 2013).

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[18] https://www.europarl.europa.eu/doceo/document/E-9-2020-000744-ASW_EN.html
[19] https://www.britishgamealliance.co.uk/moving-away-from-lead-shot/
[20] https://basc.org.uk/basc-fights-eu-lead-ammunition-restrictions/

Shooting based on large-scale releases of red-legged partridges acts in a density-independent manner and can lead to local grey partridge extinction (Watson *et al.* 2007). Levels of accidental shooting are strongly positively correlated with the intensity of red-legged partridge releasing (Aebischer and Ewald 2010)[21], and a substantially higher rate of decline in local grey partridge abundance may be observed on some sites (Fig. 18) as levels of unintentional grey partridge shooting increase during driven shooting of released red-legged partridges (Aebischer and Ewald 2004), although Aebischer and Ewald (2010) did not find a similar relationship when examining a larger number of sites. Where shooting takes place in the absence of red-legged partridge releasing, the average percentage of grey partridges shot is 3%, increasing to 16% at the highest level of red-legged partridge releasing (> 5 birds per hectare; Aebischer and Ewald 2010). Unintentional shooting losses across one Sussex study area amounted to 35–39% of autumn grey partridge density, with a predicted reduction of 68–85% in the density of spring pairs (Watson *et al.* 2007). Sustainable harvest models suggest that at low grey partridge density on modern farmland (below 0.2 per hectare; GWCT 2013), the optimum sustainable yield for shooting is 20% of autumn density with extinction occurring beyond 50% shooting (Watson *et al.* 2007). Where precautionary measures are put in place to avoid this accidental shooting of grey partridges, the percentage of autumn grey partridge numbers shot drops from an average of 64% over 3 years to 16% over 3 years (Aebischer and Ewald 2004), below the optimal sustainable shooting rate of 20% at low densities (Watson *et al.* 2007, Aebischer and Ewald 2010).
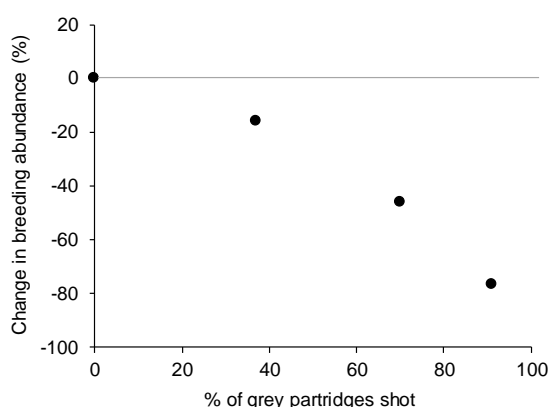
Fig. 18. Grey partridge experience more negative changes in breeding abundance (average numbers of breeding pairs) as the percentage of accidental shooting of the autumn grey partridge population increases during driven red-legged partridge shoots. Data is from Aebischer and Ewald (2004): three farms in Sussex where red-legged partridges were released and shot (grey partridges shot > 0%) and one farm where releasing and shooting did not take place (grey partridges shot = 0%) between 1995 and 2000. The "optimal sustainable shooting rate" for grey partridges is cited as 20% (Watson *et al.* 2007).

Accidental shooting of grey partridges may play a secondary role in driving their national population declines compared to farmland habitat deterioration however. Aebischer and Ewald (2004) found that the annual rate of decline in the numbers of grey partridges shot nationally (used as a proxy for national grey partridge population change) was only marginally (and not significantly) higher (-9.8%) on estates that released red-legged partridges and pheasants compared to sites with no non-native gamebird releasing (-7.9%), although declines were twice as high on estates releasing pheasants only (-16%). There is therefore no evidence that shoots based on red-legged partridge releasing have had an impact on national grey partridge populations beyond that of agricultural intensification (Aebischer and Ewald 2004). Aebischer and Ewald (2004) also suggest that the higher rates of decline observed on pheasant releasing sites may be more related to an increase in habitat types suboptimal for grey partridge on these estates (e.g. more woodland, see section 4.2.5) rather than accidental shooting mortality during

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[21] https://onlinelibrary.wiley.com/doi/epdf/10.1111/j.1474-919X.2010.01037.x

pheasant shoots. Red-legged partridge shooting sites generally increased their provision of beneficial partridge brood rearing and game cover habitats as the numbers of red-legged partridges released increased; the provision of which may counteract the shooting losses of grey partridges on red-legged partridge shoots when active measures keep those losses below 20% (Aebischer and Ewald 2010).

We found no other published evidence for the impacts of accidental shooting on other native UK species during pheasant and red-legged partridge shoots. It is unclear whether this represents a lack of an effect on other species or a lack of reporting of these events. The threat of increased incidences of accidental shooting is however mentioned in relation to black grouse by Cole *et al.* (2012), who suggest this may be an issue in areas where pheasants and/or red-legged partridges are released and shot in close proximity to habitats supporting black grouse.

4.3.4   Direct ingestion of lead by wildlife

The direct ingestion of mis-identified lead gunshot pellets or fragments (Fig. 17) is most common in bird species which ingest small stones and grit to aid digestion such as waterbirds (Anatidae), cranes (Gruiformes) and gamebirds (Galliformes; Fisher *et al.* 2006, Pain *et al.* 2009, Lead Ammunition Group 2015, Pain *et al.* 2015, Pain *et al.* 2019b). The impacts of lead ingestion have been most extensively studied in waterfowl in wetland habitats, while relatively few published studies have focussed on the impacts on terrestrial (non-wetland) animals, particularly in the UK (Fisher *et al.* 2006, Pain *et al.* 2009). Twenty-six terrestrial bird species across Europe and North America are recorded as having directly ingested lead gunshot with deleterious effects, including woodpigeon, moorhen, coot and herring gull which are native to the UK (Fisher *et al.* 2006, Pain *et al.* 2009). Estimates of mortality resulting from lead poisoning for terrestrial game birds in the UK are however less accurate and precise than those for wildfowl (Pain *et al.* 2015, Pain *et al.* 2019b).

Pain *et al.* (2015) and Pain *et al.* (2019b) suggest that hundreds of thousands of terrestrial gamebirds may die from lead poisoning annually, the majority of which are likely to be pheasants and red-legged partridges (see sections 2.1.1 and 5.1.2). Of 637 red-legged partridges examined by post-mortem 1955–1992 in Great Britain (of which 58% were released birds), Butler (2005) report that only one (0.16%) which was from a releasing estate contained lead gunshot in its gizzard. The gizzards of 1.4% (2) of 144 hunted red-legged partridges shot in 2001/02 contained ingested lead gunshot however (Butler 2005). A similar study of lead gunshot ingestion in 437 pheasants shot across 32 estates in Great Britain from 1996 to 2002 found that 13 individuals (3%) had ingested lead gunshot in their gizzard contents, and had correspondingly elevated bone lead levels (Butler *et al.* 2005). Pain *et al.* (2019a) estimate that 0.558% of pheasants and 0.323% of red-legged partridges die from lead poisoning annually based on studies by Butler *et al.* (2005) and Butler (2005), which, when applied to the most recent estimates of the number of gamebirds released from Aebischer (2019b), equates to a combined estimated total of up to 320,696 birds (see section 5.1.2).

Of all native UK Galliform species, the incidence of direct lead ingestion has been most commonly studied in grey partridge, which utilise much of the same habitat and geographical range within the UK as pheasants and red-legged partridges (Balmer *et al.* 2013). Of 1,318 grey partridge carcasses examined between 1947 and 1992 in southern England, 18 (1.4%) were killed by elevated lead exposure (Potts 2005). 76% of individuals that had ingested lead gunshot died as a result of lead poisoning, rising to 100% of individuals that contained three or more ingested gunshot (Potts 2005). The incidence of lead poisoning also increased between 1947–1958 and 1963–1992: prior to large-scale non-native gamebird releasing during 1947–1958 (see Fig. 2 in section 2.1.1), ingested lead was cited as a cause of death for 0.3% of grey partridges; this rose to 4.5% in adults and 6.9% in chicks during 1963–1992 when non-native gamebird release and shooting numbers were increasing (Potts 2005). Between 1968 and 1978, two chicks sampled from separate grey partridge broods in Sussex

had ingested 13 and 14 lead gunshot respectively within 3 weeks of hatching (Potts 2005). From 1963 to 1992 the overall incidence of ingested lead gunshot in UK grey partridge was 52% of that recorded for waterfowl (Anatidae excluding mute swan) however (Potts 2005).

These incidence rates of lead gunshot in the gizzard contents of non-native gamebirds and grey partridges are likely to underestimate the true annual exposure of these species because lead is retained in the gizzard for a relatively short period, being rapidly eroded through abrasion (Potts 2005, Mateo 2009). This means that even though the proportion of birds detected with ingested gunshot at a given time is low, the gizzards of some living birds that contain no lead gunshot at the time of sampling will probably have contained gunshot previously (Potts 2005). These studies also rely largely on the post-mortem of dead birds and may not sample individuals experiencing sub-clinical or sub-lethal effects which are also common at low levels of lead poisoning (Pain *et al.* 2015, Pain *et al.* 2019b). Several million undetected birds may therefore be likely to ingest lead gunshot during the course of a year in the UK (Pain *et al.* 2015, Pain *et al.* 2019b).

Effects of lead-related mortality on grey partridge (or pheasant and red-legged partridge) population trends are unknown, although simple modelled scenarios based on the European grey partridge population and estimates of mortality from lead ingestion given by Potts (2005) above, demonstrate that lead gunshot ingestion at observed UK rates could reduce the population size of grey partridges by 10% (Meyer *et al.* 2016). Considerably lower detected levels of lead gunshot ingestion (present in up to 0.4% of dead individuals) in eight duck species that winter in freshwater habitats in the UK were significantly negatively correlated with their population growth rates (Green and Pain 2016), suggesting that the higher levels of ingested lead gunshot found in both native and non-native gamebirds may limit their population growth in a similar way.

4.3.5   Environmental lead concentrations

Degradation of lead ammunition is caused by a combination of physical erosion and abrasion, which is accelerated in acidic, sandy soils and those with marked levels of movement and chemical activity (Ma 1987, Pain *et al.* 2015). Lead is relatively strongly absorbed by soil particles (Kabata-Pendias 2010, Pain *et al.* 2015), but the mobility of lead in soils is highly variable in relation to environmental conditions (Pain *et al.* 2015). In areas of intensive shooting, lead leaching from spent ammunition can be taken up by some plants and soil microfauna (Fig. 17), thereby entering the food chain (Pain *et al.* 2015). The number of research studies conducted on this route of exposure to lead are limited, and are particularly sparse in relation to the use of lead for the shooting of non-native gamebirds in terrestrial habitats (Quy 2010, Lead Ammunition Group 2015).

Sneddon *et al.* (2009) found that soil lead concentrations in a wood and field over which driven shooting of released pheasants was annually conducted were significantly higher (160 mg/kg and 68 mg/kg in wood and field respectively) than in a control non-shooting wood and field (60 mg/kg and 44 mg/kg respectively) due to lead gunshot deposition. The woodland lead concentrations were higher than the average soil lead concentration of 72 mg/kg in rural English topsoil (covering 94% of the area of England, although 'Normal Background Concentrations' of up to 180 mg/kg have been recorded in England previously; DEFRA 2012, Ander *et al.* 2013, Pain *et al.* 2015). The heightened soil concentrations associated with pheasant shooting corresponded to significantly higher lead concentrations in earthworm gut contents in the shooting field compared to the control field (shooting 36 mg/kg, control 16 mg/kg), and lead concentrations of 299 mg/kg were recorded in the gut contents of earthworms from the shooting woodland (Sneddon *et al.* 2009). Lead was also significantly more concentrated in the tissues of earthworms sampled from the shooting woodland relative to the control woodland by several orders of magnitude (shooting 112 mg/kg, control 5 mg/kg), but no difference was

found in earthworm tissue lead concentrations between the shooting and control fields (Sneddon *et al.* 2009).

Lead is relatively strongly absorbed by soil particles but not readily transported to the above-ground parts of plants (Kabata-Pendias 2010). Lead was however concentrated in the leaves of common plants such as perennial ryegrass (38 mg/kg) in the shooting field relative to the control field (0.89 mg/kg), and velvet feather-moss (19 mg/kg) in the shooting wood (Sneddon *et al.* 2009); these concentrations were above the range of plant lead concentrations considered as "normal" (5–10 mg/kg) and within the range considered as potentially excessive and toxic (30–300 mg/kg) in the case of ryegrass (Sneddon *et al.* 2009, Kabata-Pendias 2010). Lead concentrations in other plants (meadow foxtail, annual meadow grass, Yorkshire fog, common nettle and ivy) were all low irrespective of treatment however (0.9–1.9 mg/kg), though were not compared directly between shooting sites and control areas (Sneddon *et al.* 2009).

4.3.6   Secondary poisoning of predators

Predators and scavengers may ingest lead through gunshot pellets in un-retrieved game, injured game or animals that have themselves ingested gunshot (Fig. 17), and therefore experience secondary poisoning (Fisher *et al.* 2006, Pain *et al.* 2009). If lead concentrations in individual prey items are low, then the effects may be experienced sub-lethally as lead concentrations accumulate; alternatively, consumption of high lead concentrations, animals that have themselves been killed by lead poisoning for example, may result in direct mortality.

Raptors are commonly associated with impacts of secondary lead poisoning globally and are therefore of particular concern regarding the threat of lead poisoning in the UK. Evidence of secondary poisoning exists for at least 37 avian predator and scavenger species across Europe and North America, most of which are hawks, falcons, owls, vultures or eagles (Fisher *et al.* 2006, Mateo 2009, Pain *et al.* 2009, Pain *et al.* 2019b). Many of these species breed in the UK, including golden eagle, red kite, white-tailed eagle, common buzzard, peregrine falcon, marsh harrier, hen harrier, sparrowhawk, goshawk, long-eared owl, raven, rook, hooded crow and magpie, although the impacts of lead were usually studied in populations in other countries (Fisher *et al.* 2006, Pain *et al.* 2009, Pain *et al.* 2019b).

Predators and scavengers, particularly raptors, may be highly susceptible to poisoning from ingested lead in any form because their digestive tracts are highly acidic to help break-down the less-digestible parts of prey items and are therefore efficient at dissolving lead which is then rapidly absorbed instead of being passed through the gut (Pain *et al.* 1995, De Francisco *et al.* 2003, Pain *et al.* 2007). Regurgitation of pellets by raptors will often remove ingested lead fragments, but regurgitation can take several days, in which time a considerable amount of lead may be absorbed into the bloodstream (Pain *et al.* 2007).

Secondary lead poisoning of raptors in the UK

Lead concentrations in the livers of 424 carcasses of birds of prey from 16 species were tested by Pain *et al.* (1995). Elevated lead concentrations (>20 ppm dry weight) within the range associated with lead poisoning mortality in raptors were recorded for one peregrine falcon (4% of species sample) and one buzzard (2% of species sample; Pain *et al.* 1995). Lead concentrations of 15–20 ppm, reflecting unusually high lead absorption and acute exposure likely to result in clinical signs of poisoning, were detected in an additional peregrine and buzzard (Pain *et al.* 1995). No individuals of any other species had >15 ppm, although one short-eared owl (7% of species sample), three little owls (5%), one kestrel (3%), one sparrowhawk (0.7%), four peregrine falcons (15%), six merlins (9.5%) and two hobbys (29%)

had levels of 6–15 ppm, which was considered above background levels of exposure (Pain *et al.* 1995). Pain *et al.* (1995) suggest that these birds were likely to have ingested lead gunshot from their prey.

Taggart *et al.* (2020) examined the lead concentrations in liver and/or bone tissues of 220 wild buzzards found dead or dying in the UK between 2007 and 2018, two thirds of which were collected from lowland areas in which pheasant and red-legged partridge shooting is common. 2.7% of these buzzards had lead concentrations in their livers consistent with acute exposure (>20,000 µg/kg dry liver weight), and 4% had lead concentrations in their bones consistent with exposure to lethal lead poisoning (>20,000 µg/kg dry bone weight; Taggart *et al.* 2020). Stable isotope analysis suggested that buzzards with higher concentrations of lead in their liver tissues contained a markedly higher proportion of lead originating from gunshot: 89% of the mass of lead in acutely exposed individuals was derived from shotgun pellets commonly used to shoot gamebirds, while over half (57%) of the mass of lead in the livers of all the buzzards sampled was also likely to have been derived from this source (Taggart *et al.* 2020). This suggests that exposure to lead originating in gunshot may have caused buzzard deaths in the UK, although the proportion of those sampled could not be reliably estimated (Taggart *et al.* 2020). Lead concentrations were highest in buzzards found dead during the gamebird shooting season, which is consistent with a probable increase in the availability of carcasses of unrecovered shot gamebirds and those that died from other causes (e.g. predation, vehicle collisions) during the shooting season with embedded or ingested shot in their bodies (Madden *et al.* 2018, Taggart *et al.* 2020). Bone lead concentrations were also greater in older buzzards, which may be evidence of lead accumulation through repeated exposure over the birds' lifetime (Taggart *et al.* 2020).

Incidences of secondary lead poisoning have also been particularly well documented for red kites in the UK, probably as a result of the increased monitoring effort surrounding the reintroduction of this species into the British countryside throughout 1989–2006 (Smart *et al.* 2010, Molenaar *et al.* 2017). Red kites catch live prey and scavenge substantial amounts of carrion, so are particularly at risk of secondary lead poisoning if consuming carcasses of animals which themselves succumbed to lead poisoning or were shot with lead (Davis and Davis 1981, Wildman *et al.* 1998, Carter and Grice 2000, Murn and Hunt 2012). Considerable prevalence of lead-related mortality and sub-lethal lead poisoning in the reintroduced English population during 1995–2003 was identified by Pain *et al.* (2007) and Molenaar *et al.* (2017). Of 125 captive young red kites (originally taken from wild breeding pairs) monitored prior to release, 46 (37%) had elevated blood lead levels thought to originate from unknown small fragments of lead ammunition in the carcasses of birds or mammals either fed to the nestlings by their parents or subsequently whilst in captivity (Pain *et al.* 2007, Molenaar *et al.* 2017). After release, a minimum of 1.5–2.3% of 264 regurgitated pellets from wild-feeding birds contained lead gunshot, while of 44 red kites that died or were found dead, seven had elevated lead levels in their liver tissue; of these, six (14%) had concentrations compatible with fatal lead poisoning (>15 mg/kg dw) and secondary lead poisoning was concluded as their cause of death (Pain *et al.* 2007, Molenaar *et al.* 2017). Bone samples from 86 dead red kites also indicated elevated exposure to lead (>20 mg/kg dw) at some point in at least 18 of the birds' lives (21% of samples; Pain *et al.* 2007, Molenaar *et al.* 2017). The primary source of this lead when tested against different potential lead sources using isotopic signatures was found to be lead ammunition in their food (Pain *et al.* 2007).

UK red kite populations have all shown evidence of varying rates of recovery and growth following the reintroduction programme (Smart *et al.* 2010, Hayhow *et al.* 2017). However, simple modelled scenarios based on the recovering Welsh red kite population and the estimates of lead-related mortality above (for England; Pain *et al.* 2007, Molenaar *et al.* 2017), demonstrate that secondary lead poisoning at observed UK rates could reduce the annual growth rate of the red kite population from 6.5% to 4%, thereby slowing recovery (Meyer *et al.* 2016). The effects may be higher if ingestion of lead additionally causes sub-lethal reproductive impairment (e.g. see Pain *et al.* 2019b).

Secondary lead poisoning of other predatory and scavenging species

The concentrations of lead in earthworms, gained through ingestion of lead-containing soil particles and water on non-native gamebird shooting sites (e.g. see Sneddon *et al.* 2009) can accumulate up through the food chain in animals such as moles and shrews that consume them (Ma 1987, Ma *et al.* 1991). We found no UK studies investigating the potential for this effect, although in two nature reserves adjacent to industrial areas of heavy metal pollution in the Netherlands, Ma *et al.* (1991) and Ma (1987) found that common shrews, field voles and moles suffered high levels of lead exposure. Shrews and moles feeding on lead-polluted heathland sites exhibited particularly high lead levels, gained after consumption of lumbricid earthworms containing a high concentration of lead in their tissues acquired by consuming lead-polluted soil (Ma *et al.* 1991). On high intensity gamebird shooting sites, earthworms from soils contaminated with lead gunshot contain higher lead concentrations than uncontaminated sites (Sneddon *et al.* 2009), so represent a source of secondary lead consumption to animals that might predate them (for example moles, shrews, blackbirds, robins and thrushes, raptors such as buzzards and red kites, waders and badgers). This is likely to be most marked in areas with low soil pH, such as heathlands and conifer woodlands, where high acidity may facilitate more rapid decomposition of lead gunshot pellets into the soil and increase uptake by soil organisms (De Francisco *et al.* 2003).

Field voles in the Dutch study absorbed lead after consuming contaminated grasses and bryophytes (Ma *et al.* 1991). Sneddon *et al.* (2009) found that the bioavailability of lead to herbivorous small mammals on a single shooting site in England was however low, with lead concentrations in the unwashed hair of wood mice and field voles not differing significantly between woodlands where shooting was and was not conducted. Sneddon *et al.* (2009) concluded that their "results indicate that managed game shooting presents minimal risk in terms of element transfer to soils and their associated biota" due to this lack of effect on wood mice and field voles. However, although peer-reviewed, this study had a very low level of spatial and temporal replication, being conducted on an area of 5.2 hectares on just one UK shooting estate, with results based on very low numbers of soil (~8 in total), earthworm (~24 in total), plant (~9 in total) and small mammal (~63) samples. Both plant matter (grasses and mosses) and earthworms from shooting areas were more contaminated with lead than those where shooting did not take place (Sneddon *et al.* 2009).

Insufficient information exists to estimate the numbers of mammals exposed to lead shot poisoning in Europe and elsewhere, although studies of predatory mammals globally highlight the potential risk of exposure for many species (Pain *et al.* 2015, Pain *et al.* 2019b). The relatively high incidence of lead ingestion and associated poisoning in pheasants and red-legged partridges (Butler 2005, Butler *et al.* 2005), and their importance as a prey source for generalist predators such as foxes (see Table 24 in section 4.7.3), suggests that secondary lead poisoning may be also widespread in mammalian predators in the UK, although we found no literature examining this.

4.3.7   Key knowledge gaps and recommendations

- There is a considerable body of literature from Europe and North America documenting the negative environmental impacts of the use of lead ammunition to shoot waterbirds, and the impacts of secondary lead poisoning of raptors in terrestrial habitats through the consumption of lead-contaminated prey. Studies conducted in the UK, particularly on mammals and invertebrates, are relatively sparse however, and we found only one study which attempted to compare lead contamination and exposure in areas with and without non-native gamebird shooting.

- The contamination of UK terrestrial habitats with lead from non-native gamebird shooting has not been studied in a spatially replicated manner across regions, nor compared across a gradient of gamebird hunting intensity. The extent of lead contamination from non-native gamebird shooting, and the associated risk to UK wildlife, is therefore difficult to assess. Thorough testing of soil lead

contamination and associated exposure of plants and animals is therefore urgently required across a gradient of gamebird hunting intensity.

- The extent to which accidental shooting of native species other than grey partridge occurs during pheasant and red-legged partridge shooting is unknown, possibly due to a lack of reporting of these events.

4.4   Illegal persecution

4.4.1   Impact summary

There was a relatively small amount of evidence (18 scores evidenced by 13 sources) for the impacts of illegal persecution of protected species on gamebird-releasing estates. The ecological impact scores associated with illegal persecution were significantly negative on average ($z = -3.58$, $P = 0.002$; Fig. 12), with no positive or benign impact scores associated with this impact theme (Table 2, Table 17). These scores largely reflect evidence for mortality, and therefore potential effects on local abundance, of the protected species targeted, with only a single study (on red kites in Scotland) demonstrating clear negative effects of illegal persecution on a regional raptor population. Little peer-reviewed evidence exists for this topic in relation to pheasant and red-legged partridge releasing however, and most sources contributing scores to Table 17 document anecdotal records of illegal persecution. The available evidence suggests that these illegal activities are nationally widespread with multiple cases reported annually, although only on a small proportion of game-releasing sites. The overall impacts of persecution are not well understood, and the topic would benefit from further research, although conducting representative research into illicit illegal activity is likely to be difficult.

Table 17. Summary ecological impact scores associated with the impacts of illegal persecution of protected species, including the distribution of scores at each score level, the estimated marginal mean score and 95% confidence limits (95% CL) returned by the Ordinal Logistic Model (OLM), and the proportion of scores which originated from peer-reviewed research and for which there was direct evidence of an impact rather than a potential impact.

	Ecological impact score						
	-2	-1	0	1	2	Mean	95% CL
Illegal persecution	1	17	0	0	0	-1.92	-3.34 – -0.51

18 scores evidenced by 13 sources*: 31% peer-reviewed, 77% direct evidence
* (Swann and Etheridge 1995, Kenward *et al.* 2000, Kenward *et al.* 2001, RSPB 2003, RSPB 2005, RSPB 2007, RSPB 2009, RSPB 2010, Smart *et al.* 2010, RSPB 2013, RSPB 2014, RSPB 2015, RSPB 2018)

4.4.2   Background

The legal killing of many predator and scavenger species is conducted on most UK pheasant and red-legged partridge releasing sites: 96% of 770 gamekeepers across the UK reported controlling foxes where they were present on their shoot in 2019, 95% controlled carrion crows, 85% controlled jackdaws, 82% controlled jays, 84% controlled rooks and 97% controlled magpies (Ewald and Gibbs 2020; see section 4.2). For some of these species, legal killing was covered under the terms of separate general licensing arrangements in each UK country, the scope of which is currently under review. Temporary licenses in place which allow legal killing of different species and in different country-specific situations[22]. Predator species that have previously been included under these licenses include carrion crow, jackdaw, magpie, rook and jay for the purpose of conserving wild birds (not including gamebirds) or preventing serious damage to livestock (includes gamebirds prior to release; also see section 4.2).

Predator or scavenger species which are protected by law under the Wildlife and Countryside Act 1981 and Wildlife Order (Northern Ireland) 1985 are also illegally killed in the UK because they are perceived

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[22] https://basc.org.uk/gl/

to threaten gamebirds, including raptors such as common buzzard, red kite or goshawk, corvids such as raven, and mammals such as European badger, despite this illegal practice now being condemned within the shooting community[23]. Of a sample of gamekeepers who reported these species on the land they managed, 81%, 89% and 69% perceived that buzzards, goshawks and sparrowhawks had a negative effect on game respectively, and 70% also perceived that badgers were detrimental to gamebirds (Ewald and Gibbs 2020).

The illegal killing of protected predator species often involves the use of methods which are themselves illegal when used for the catching or killing of any animal. The use of poisoned bait is common for example, where carrion (often a pheasant or partridge carcass) is infused with one of numerous toxic insecticides or rodenticides, some of which are themselves illegal to possess or misuse in this manner (e.g. Kenward *et al.* 2000, Kenward *et al.* 2001, RSPB 2003, RSPB 2007, RSPB 2010, Smart *et al.* 2010, RSPB 2013, RSPB 2014, RSPB 2015). Live-baited cage traps may also be used to catch animals before dispatching them (RSPB 2005, RSPB 2014), and pole or spring traps may be placed around gamebird release pens (RSPB 2009). Animals may be shot (Kenward *et al.* 2000, Kenward *et al.* 2001), or their nests destroyed.

Illegal killing of raptors on lowland shooting estates is much less empirically studied than similar practices in the uplands. The illegal killing of hen harriers to protect red grouse on upland moorland is well documented for example (Murgatroyd *et al.* 2019), and hen harrier populations on upland moorland across the UK are substantially lower than the habitat could support as a result (Etheridge *et al.* 1997). In contrast, with the exception of four peer-reviewed studies (Swann and Etheridge 1995, Kenward *et al.* 2000, Kenward *et al.* 2001, Smart *et al.* 2010), much of the published evidence for illegal predator persecution associated with pheasant and red-legged partridge releasing originates from anecdotal records in the UK Government Wildlife Incident Investigation Scheme reports (HSE WIIS 2019), or the RSPB's annual *Birdcrime* report[24]. The RSPB also published an interactive online map in 2018 indicating existing records of such incidents and allowing new evidence to be added (Raptor Persecution Map Hub[25]; Fig. 19). Despite the existence of this database, no studies have yet attempted to compare the spatial and temporal distribution of illegal persecution records with the distribution of gamebird releasing, to determine how widespread these practices may be and to what extent they may be associated with pheasant and red-legged partridge releasing.

4.4.3   Impacts of illegal persecution on protected species

Persecution of many raptor species can limit their population growth (Newton 1979). Illegal killing of red kites is the likely cause of their poor population growth in north Scotland, with 40% of the 103 red kites found dead between 1989 and 2006 having been killed illegally, mainly by direct poisoning from ingesting poisoned baits thought to have been set to protect gamebird interests (Smart *et al.* 2010). Smart *et al.* (2010) further estimate that between 1999 and 2006, the period when the increase of the north Scotland red kite population halted, 166 individuals may have been killed illegally in this manner. This north Scotland population is established in a lowland area where many landowners release pheasants and red-legged partridges (Swann and Etheridge 1995), but is also surrounded by large areas of active grouse moor managed for red grouse shooting, so this persecution may conceivably have originated from either (or both) of these gamebird interests. The illegal poisoning of red kites, both intentionally and through the misuse of pesticides (see section 4.2.8), may also be slowing their rate of expansion in England, with poisoning diagnosed as the cause of death for 29% of dead red kites in England between 1989 and 2007 (Pain *et al.* 2007, Molenaar *et al.* 2017).

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[23] https://basc.org.uk/zero-tolerance-for-raptor-persecution-a-joint-statement/
[24] www.rspb.org.uk/birds-and-wildlife/advice/wildlife-and-the-law/wild-bird-crime/
[25] www.arcgis.com/apps/opsdashboard/index.html#/0f04dd3b78e544d9a6175b7435ba0f8c

A detailed survey in 1954 indicated that the distribution of buzzards and gamekeepers across the UK at that time were mutually exclusive as a result of persecution, with buzzards commonest in the west where gamekeepers were scarcest, and absent altogether from eastern areas where gamekeepers were numerous despite suitable habitat being available throughout (Moore 1957, Newton 1979). UK Buzzard abundance is now increasing (Hayhow *et al.* 2017), and their range is expanding from west to east (Balmer *et al.* 2013), so illegal persecution is perhaps likely to be having less of an impact on national populations now than historically. However, persecution may have local impacts, particularly in regions where gamebird releasing is common. In a pheasant releasing region of southern England for example, 32% of all radio-tagged buzzard mortality was caused by shooting or poisoning near pheasant release pens, with 10% of first-year buzzards killed illegally in this way (Kenward *et al.* 2000, Kenward *et al.* 2001). Swann and Etheridge (1995) also found that buzzards breeding in Moray (north Scotland) did not reach the breeding densities achieved in a nearby forested region, despite producing twice as many fledged young per nesting pair, suspecting that their low overall breeding density was due to an almost total destruction of the buzzard population by illegal persecution aimed at protecting gamebird interests. Moray is a lowland farmland region containing several shooting estates which release pheasants. On three of these estates no buzzard nests and only occasional sightings of adults were recorded during the study period, while elsewhere in the same region eight buzzards were found poisoned on occupied territories prior to egg laying on different occasions (Swann and Etheridge 1995). The results of these local studies however contrast with those of the more recent national study by Pringle *et al.* (2019), who found that buzzard abundance and growth may be more positive in areas with larger gamebird releases. Any negative effects of illegal persecution on buzzard abundance may therefore be restricted to local areas where persecution occurs and therefore not detectable at a national scale, or may have declined over time. It is also possible that the incidence of illegal persecution may be more common in areas releasing fewer gamebirds, where predators may be expected to have a relatively larger impact on pheasant numbers and therefore be viewed as a bigger threat and more likely to be persecuted, thus potentially restricting the increase in buzzard abundance at lower gamebird densities; this has not been tested however.

Additional evidence for prosecutions relating to the illegal killing or targeting of protected avian and mammalian species to protect the interests of gamebird releasing is available from the RSPB's annual *Birdcrime* reports, with poisoning cases related to the misuse and abuse of pesticides also reported by the UK Government Wildlife Incident Investigation Scheme (HSE WIIS 2019). Table 18 lists the mortality of protected species associated with a small subset of cases reported in sufficient detail to specifically link them to gamebird-rearing, pheasant release or red-legged partridge release sites (sufficient detail meant there was explicit mention of gamebirds, and/or gamebird rearing or release activities on the site where the persecution took place). There are likely to be more cases which were not reported in this way. Of 1,225 cases of illegal persecution recorded through the RSPB Raptor Persecution Map Hub[26] 2007–2018 (Fig. 19), buzzards were the most frequently affected species (listed in 34% of cases), followed by red kites (15%), peregrine falcons (11%) and sparrowhawks in (5%), with 16 other raptor species plus raven affected in the remaining 37% of cases. Where illegal persecution activities were undertaken, the numbers of individuals reported as affected are, in several cases, large enough to suggest possible impacts on local abundance (Fig. 19, Table 18).

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[26] www.arcgis.com/apps/opsdashboard/index.html#/0f04dd3b78e544d9a6175b7435ba0f8c

Table 18. Mortality of protected native UK species associated with prosecution cases since 2002 reported in enough detail by annual RSPB *Birdcrime* reports to specifically link them to non-native gamebird releasing sites. For each case, the number of each species illegally killed and the method used, as well as the type of non-native gamebird release with which the case was associated (pheasant, red-legged (RL) partridge or both), and the geographic location are given.

Source	Species	# killed	Method	Release type	Location
RSPB (2003)	Buzzard	1	Poisoned bait	Both	Norfolk
RSPB (2005)	Buzzard	1	Cage trapping	Both	Buckinghamshire
RSPB (2007)	Buzzard	2	Poisoned bait	Pheasant	Norfolk
RSPB (2009)	Buzzard	102	Spring traps	Pheasant	Shropshire
RSPB (2009)	Raven	40	Unknown	Pheasant	Shropshire
RSPB (2009)	Badger	37	Unknown	Pheasant	Shropshire
RSPB (2010)	Buzzard	1	Poisoned bait	RL partridge	Wales
RSPB (2010)	Buzzard	2	Poisoned bait	Pheasant	Herefordshire
RSPB (2010)	Raven	5	Poisoned bait	Pheasant	Herefordshire
RSPB (2013)	Buzzard	2	Poisoned bait	Pheasant	Lincolnshire
RSPB (2014)	Buzzard	1	Poisoned bait	Pheasant	Galloway
RSPB (2014)	Buzzard	7+	Cage trapping	Pheasant	Cumbria
RSPB (2015)	Buzzard	1	Poisoned bait	Pheasant	Galloway
RSPB (2015)	Buzzard	19	Poisoned bait	Pheasant	Norfolk
RSPB (2015)	Sparrowhawk	1	Poisoned bait	Pheasant	Norfolk
RSPB (2015)	Tawny owl	1	Poisoned bait	Pheasant	Norfolk
RSPB (2018)	Buzzard	1	Shooting	Pheasant	Inverness-shire

In addition to the anecdotal case records in Table 18, the RSPB has recently published data on confirmed UK raptor persecution incidents between 2007 and 2018 (Fig. 19). These represent substantiated evidence for shooting, poisoning, trapping or nest destruction of birds of prey, including species commonly associated with lowland habitats where non-native gamebirds are released (e.g. buzzard, red kite, sparrowhawk). The highest densities of illegal raptor persecution incidents are found in the uplands (Fig. 19), particularly in association with moorland managed for native red grouse shooting (Murgatroyd *et al.* 2019). However, incidents are also widespread throughout lowland UK and in regions where pheasant and red-legged partridge releasing is ubiquitous (e.g. eastern, southern and central England, Fig. 19; compare with Fig. 3 in section 2.1.1). Of 87 cases of shooting, trapping or poisoning of at least 94 individuals of 15 bird of prey species in 2018, 38% originated in lowland counties in the south and east of England where the density of gamebird releasing is highest (Fig. 3 in section 2.1.1), and which could not have been associated with other gamebird management (e.g. for red grouse on moorland; RSPB 2019). We stress that only persecution records with a demonstrable link to a pheasant or red-legged partridge release estate are included in our quantitative assessment (Table 1).
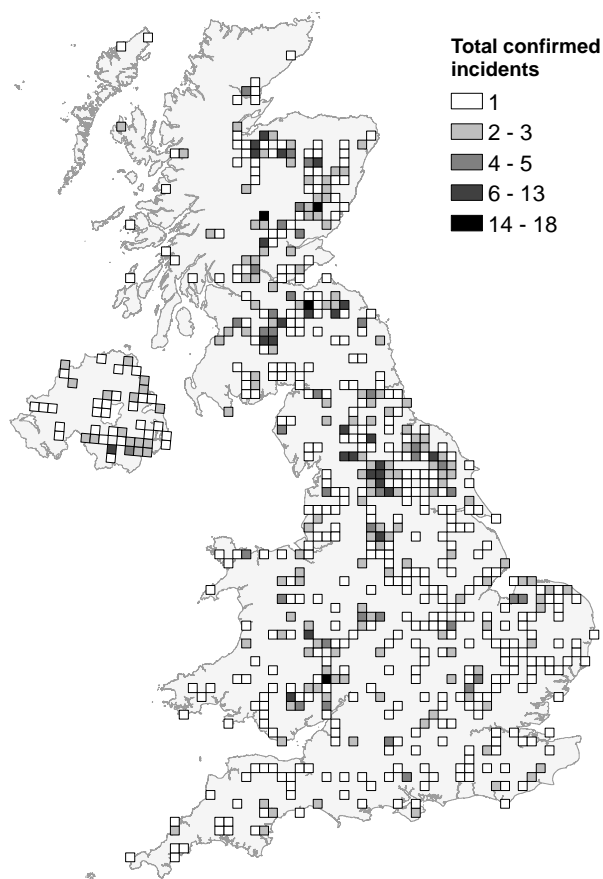
Fig. 19. Confirmed incidences of raptor persecution per 10 km square in the UK between 2007 and 2018. Confirmed incidences were of the illegal killing or targeting of birds of prey, most typically by shooting, poisoning, trapping or nest destruction, substantiated by evidence such as post-mortem or toxicological analysis, or reliable eyewitness evidence. Data are from the RSPB Raptor Persecution Hub[27].

4.4.4   Key knowledge gaps and recommendations

- There are very few peer-reviewed, empirical UK studies examining the impact of illegal raptor persecution linked with pheasant or red-legged partridge shooting activities on the populations of protected raptors. It would be useful to use collated records of illegal persecution of other protected species, identify any geographic or land use associations for these activities, and examine the population trends of the protected species affected in these regions to determine whether illegal activities are linked to gamebird releasing areas nationally, and whether these activities may be impacting on populations of protected species at a local scale.

- Pringle *et al.* (2019) found that buzzard abundance and growth may be more positive in areas with larger gamebird releases and more negative where fewer gamebirds are released; it would be useful to determine whether the density of gamebird release influences the likelihood of illegal persecution taking place, and thus whether such activities may be restricting the increase in buzzard abundance in these areas.

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[27] www.arcgis.com/apps/opsdashboard/index.html#/0f04dd3b78e544d9a6175b7435ba0f8c; Confirmed Raptor Persecution Incidents Heat Map Data downloaded from https://opendata-rspb.opendata.arcgis.com/datasets/confirmed-raptor-persecution-incidents-heat-map-data-public.

- Further tagging studies for raptor species other than buzzard, or buzzards in other regions of the UK (e.g. southern England) would be useful to determine the extent of mortality associated with illegal persecution in other areas.

4.5   Direct impacts of gamebirds

4.5.1   Impact summary

There was a reasonable amount of evidence (42 scores evidenced by 20 sources) for direct impacts of gamebirds on native UK wildlife. The ecological impact scores associated with the direct impacts of gamebirds were significantly negative on average ($z = -5.17$, $P < 0.0001$; Fig. 12), with negative average scores associated with all four secondary impact themes (Table 2, Table 19, Fig. 13). These scores reflect predominantly negative effects on plants, invertebrates and reptiles though the herbivory, predation or feeding activities (pecking, scratching) of gamebirds themselves, as well as competition for resources and effects on soil fertility. Many of these impacts are density dependent however, with lower densities of pheasants resulting in fewer negative effects that are more easily reversed once gamebirds are removed from the system. Impacts on plants and habitats in woodlands where pheasants are released dominate the literature, while peer-reviewed, empirical studies evidencing the impacts relating to the predation of invertebrates and herptiles and resource competition are generally lacking (Table 19). Evidence for these impacts is therefore mainly indirect (not directly studied; based on reasonable inference) or originates in anecdotal records.

Table 19. Summary ecological impact scores associated with direct impacts of gamebirds, including the distribution of scores at each score level, the estimated marginal mean score and 95% confidence limits (95% CL) a returned by the Ordinal Logistic Model (OLM; only presented for the primary theme overall as sample sizes precluded OLM analysis for secondary themes), the median and interquartile range (IQR) for secondary themes, and the proportion of scores which originated from peer-reviewed research and for which there was direct evidence of an impact rather than a potential impact.

	Ecological impact score						
	-2	-1	0	1	2	Mean	95% CL
Direct impacts	13	16	6	3	1	-2.01	-3.03 – -0.99
Secondary theme	-2	-1	0	1	2	Median	IQR
Browsing by gamebirds	4	4	2	2	0	-1	-2 – 0
Predation by gamebirds	4	6	4	0	0	-1	-1.75 – -0.25
Resource competition	1	2	1	0	0	-1	-1.25 – -0.75
Soil enrichment	4	4	2	1	1	-1	-2 – 0

42 scores evidenced by 20 sources*: 45% peer-reviewed, 80% direct evidence
* (Hill 1985, Corke 1989, Clarke and Robertson 1993, Low *et al.* 2003, Sage *et al.* 2005a, Callegari 2006a, Rothero 2006, Hoodless and Draycott 2007, Pressland 2009, Sage *et al.* 2009, Callegari *et al.* 2014, Larkman *et al.* 2015, Neumann *et al.* 2015, Evans *et al.* 2016, Rice 2016, Blackburn and Gaston 2018, Sage 2018b, Capstick *et al.* 2019b, Devlin 2019, Hand 2020)

Pheasants and red-legged partridges are omnivorous and opportunistic feeders, consuming a large amount of both plant and animal matter. Physical damage to plants from pecking can negatively influence sward height, botanical species richness and community composition, ground cover, hedge and shrub leaf density and cause the exclusion of some plant species, particularly in woodlands containing pheasant release pens, along hedgerows and on sensitive (and often designated) botanical sites close to release areas (e.g. chalk grasslands). Resulting changes to vegetation structure may modify nesting habitat for some birds and cause a reduction in host plants for some lepidopterans.

There are however some positive impacts on the cover of bramble and grasses in coniferous pheasant release woods. Predation of a wide range of ground-active invertebrates also drives changes in their community structure particularly close to high-density pheasant release sites, and predation of reptiles may contribute to local declines in abundance. Gamebirds at high densities also increase soil phosphate, potassium and nitrogen levels though their droppings, which changes plant and invertebrate species composition and sward structure in woodlands and hedgerows, and threatens rare bryophyte species on the moorland fringe. Woodland bryophytes may also respond negatively to increased nitrogen in the air as a result of increased emissions from enriched soil. The biomass of released and naturalised gamebirds exceeds twice that of all other UK breeding birds, and pheasants require a disproportionate amount of the total energy of the breeding bird assemblage. There is therefore the potential for competition with native wildlife, with some evidence that small seed-eating farmland birds and sympatric Galliform species may be impacted.

4.5.2   Background

Natural England cite "game management" or "pheasant rearing" as responsible for the adverse condition assessments of 608 hectares (0.6%) of 25 SSSI Units in England in 2020 (Natural England 2020). As the impacts of habitat management associated with gamebird rearing are mostly positive (see section 4.2), it is possible these effects are primarily due to the impacts on wildlife and habitats by gamebirds themselves, through browsing or predation, competition for resources and soil enrichment.

Pheasants are omnivorous and opportunistic feeders, consuming plants, invertebrates and occasionally small vertebrates (Collinge 1927, Hill and Robertson 1988, Hoodless *et al.* 2001). Plant matter is the most common component of their diets, with animal matter contributing a relatively small proportion, although this is highly variable both temporally and spatially (Collinge 1927, Hill and Robertson 1988, Hoodless *et al.* 2001). Some studies suggest that pheasants consume greater amounts of grains during the winter months, and switch to new shoots and buds as well as arthropods during the spring and summer months as these become more available (Pressland 2009). Reared pheasants are more reliant on grain feed provided at hoppers, and exhibit a less variable diet than wild pheasants (Whiteside *et al.* 2015). Breeding pheasants require large quantities of invertebrates to feed their young during the spring, therefore exerting a predatory pressure on invertebrate populations, and potentially competing with native breeding birds whose young also require invertebrate prey.

The high densities of pheasants and red-legged partridges contained within releasing pens prior to and during release means that browsing and predatory activities are likely to be concentrated here, with habitats further from release pens potentially experiencing less pronounced effects. Pheasants, when released into semi-natural woodland habitats where ecological communities are less resilient to disturbance, may have greater impacts than red-legged partridges which are more often released into arable environments. When pheasants are released they disperse and often make regular journeys to and from release pens, following linear boundaries such as hedgerows and ditches where they browse as they travel (Sage and Swan 2003, Sage *et al.* 2009).

4.5.3   Browsing by gamebirds

Gamebirds damage leaves, stems, flowers and seeds when feeding on plants and also cause damage to roots and disturb the soil by pecking, digging, scratching and trampling in the process of feeding or dust-bathing (e.g. Neumann *et al.* 2015). Here we used the term "browsing" to refer to all these actions. This browsing, combined with changes in soil nutrients as a result of faecal build-up (see section 4.5.6), as well as changes in microclimate due to gamebird-related habitat management (see section 4.2), combine to significantly alter botanical community structures. Many of the sources evidencing these

impacts attribute their causes to this combination of effects (e.g. Sage *et al.* 2005a); these impacts are therefore previously described within section 4.2, but are repeated here for completeness.

Plant matter in gamebird diets

Primarily seed-eaters (Larkman *et al.* 2015), pheasants will also feed on multiple plant parts including leaves, roots, flower buds, fruits, nuts, seeds, roots, bulbs, rhizomes and tubers (Collinge 1927, Dalke 1937, Hill and Robertson 1988). In America, average pheasant diets consist of 92% plant material, predominantly cereal grains (72.6%) as well as leaves, roots, other seeds and fruits (9.4%; Hoodless *et al.* 2001). In the UK these proportions are influenced by the availability of supplementary cereal grain feed, but still represent a predominance of plant matter in the diet (97.3% plant matter overall; Hoodless *et al.* 2001). The proportion of plant matter does vary however: Collinge (1927) recorded only 63% vegetable matter in a study of 183 pheasant stomachs over a 12-month period.

Impacts of browsing by gamebirds on plants

Ancient forest plants tend to be shade tolerant and prefer low fertility, meaning that where light levels are increased and soil chemistry is changed (as is often the case in pheasant-releasing woodlands), the habitat conditions become sub-optimum and these species are outcompeted by others which are faster-growing and can take advantage of high nutrient availability (ruderal species e.g. nettles and grasses; Hermy *et al.* 1999, Hill *et al.* 1999). Ancient forest plant species are also slow to recolonise areas from which they have been removed (Hermy *et al.* 1999)

Changes to ground flora communities are most pronounced in areas where gamebirds are held in high densities such as release pens. During the release period, and particularly towards the end of it, there are often visible changes to the woodland ground flora inside release pens, in the form of damage to plants and disturbance to the soil (Sage *et al.* 2005a). Sage *et al.* (2005a) found that the density of low vegetation cover (<0.5 m) was reduced in pheasant release pens compared to control (non-release) plots, and there was a reduction in cover of shade tolerant and winter-green perennials, with an increase in the cover of bare ground and species characteristic of disturbed, fertile soil. Subsequently, plant species diversity was lower in release pens compared with control sites, with a change in species composition away from flora characteristic of ancient woodland towards more undesirable grass species (Sage *et al.* 2005a). Pheasant stocking density had a strong effect on vegetation characteristics with negative impacts usually seen at densities greater than 1000 pheasants per hectare, and with older and smaller pens (Sage *et al.* 2005a).

Similar impacts were also detected in more recent studies by Neumann *et al.* (2015) and Sage (2018b). Neumann *et al.* (2015) found that the vegetation community composition of woodland ground flora was similarly significantly altered within pheasant release pens compared to control areas in the same wood, with ruderal and disturbance-tolerant perennials more frequent in release pens while archetypal woodland species and ancient semi-natural woodland indicator plants, which are less tolerant of disturbance and enriched soils, were much less abundant (Neumann *et al.* 2015). Sage (2018b) found more bare ground (40%), fewer woodland herb species (15%) and a reduced fern community within pheasant release pens compared to outside (10% bare ground, 30% herbs), although in conifer woodlands herb abundance did not differ between woods releasing and not releasing pheasants, and the presence of pheasants may have encouraged 30% more bracken and a tendency towards more bramble and grasses. Overall plant diversity can however be similar inside and outside pheasant release pens (Sage 2018b), as can overall percentage cover and plant species richness (Neumann *et al.* 2015), but this is possibly due to the displacement of perennial species by ruderal, annual species (Sage *et al.* 2005a). These effects are further discussed in section 4.2.4.

Differences in floral composition within pheasant release pens as a result of pheasant browsing and soil enrichment may still be evident several years after use by pheasants (Fig. 20), with slow recolonisation by woodland herb species (Low *et al.* 2003, Capstick *et al.* 2019b). Capstick *et al.* (2019b) found that soil fertility (see section 4.5.6) and cover of species preferring fertile soils (e.g. nettles) were still higher in disused pens even after 10 years of abandonment, whereas winter green perennials, richness of ancient semi-natural woodland plant species and overall species richness remained low. Total species richness and richness of ancient semi-natural woodland plants did show signs of recovery in pens that had been disused for longer than ten years, but this recovery only occurred in pens where ≤ 1,000 pheasants per hectare had been released (Capstick *et al.* 2019b).
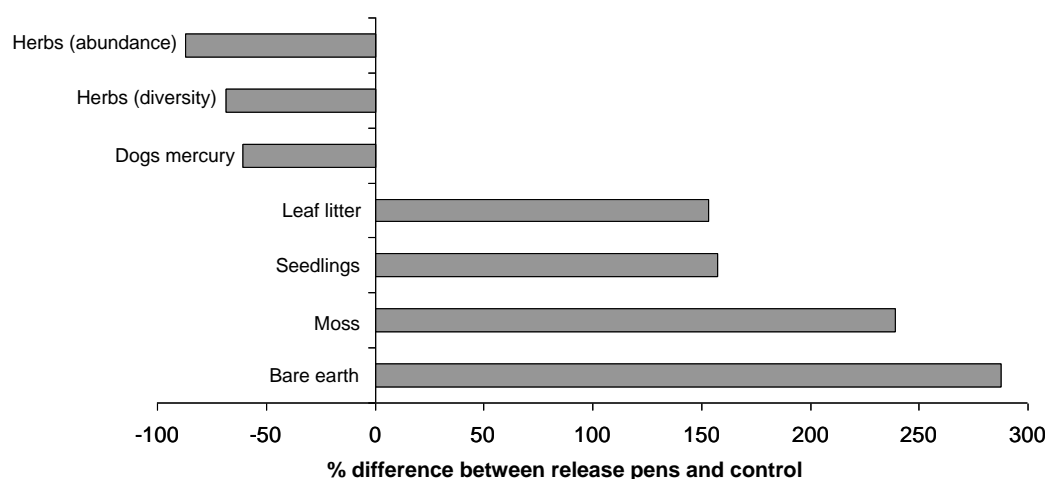
Fig. 20. Percentage difference in ground flora characteristics between a pheasant release pen that had not been used for four years, and control plots. Positive and negative values represent more and less in pheasant pens than control plots respectively. After Low *et al.* (2003).

Hedgerows connecting pheasant-releasing woodlands to game cover crops and other arable habitats managed for gamebirds are also impacted by pheasant browsing. Outside of release pens, pheasant browsing results in increased weed species richness and bare ground, and reduced numbers of stable perennials on hedgebanks close to release pens (Sage *et al.* 2009). Hedgebanks in areas where >1000 pheasants were released also contained fewer shrub and seedling species, and shrub leafiness was reduced in the lower portions (the first 20 cm) of hedges (Sage *et al.* 2009). These effects on weeds and stable perennials were consistent across arable and grassland farmland regions (Sage *et al.* 2009). Woodburn and Sage (2005) also found a strong negative correlation between pheasant release numbers and low (11–40 cm) herb cover of hedgerows adjacent to pens. However, Hoodless and Draycott (2007) found that the cover of woody hedge species at the base of hedges was 10% greater in hedges next to pheasant release woods than in those next to non-game woods, suggesting that pheasants do not damage woody shrubs.

All the impact evidence above deals with the release of pheasants into woodland; very few studies have focussed on the impacts of browsing by red-legged partridges on the habitats into which they are released or may disperse. One exception is a study by Callegari (2006b) who investigated the impact of both red-legged partridges and pheasants on vegetation in chalk grasslands in southern England (64,000 and 37,000 released across 4 sites respectively). They detected no significant differences in the percentage cover of bare ground or litter, or the number of positive and negative indicator plant species when comparing areas of high and low densities of gamebirds across three sites (high and low gamebird density being defined on a site-specific basis, with low density areas ranging from 0–3 gamebirds per hectare, and high density areas those where gamebirds were consistently present in densities of 2–16.1 per hectare; Callegari 2006b). Vegetation was however shorter where red-legged

partridges were released compared to chalk grassland sites where they were absent (Callegari *et al.* 2014).

In addition to the above impacts on native plants and habitats, pheasants may also damage crops, particularly wheat, oil seed rape, barley and potatoes when they disperse into adjacent arable farmland (Hoodless *et al.* 2001, Rice 2016). This is discussed in section 5.1.2.

Impacts of browsing by gamebirds on other wildlife

Browsing by gamebirds released at high densities may have indirect impacts on other wildlife. The modifications to hedgerows and their surrounding microhabitats partly as a result of pheasant browsing (e.g. Woodburn and Sage 2005, Sage *et al.* 2009), may affect nest site suitability for birds that nest on hedgebanks or in the lower portion of hedges. The abundance and diversity of bird groups such as finches, buntings, larks and tits is positively associated with hedgerows that have adjacent species rich verges (Parish *et al.* 1994, Parish *et al.* 1995, Hinsley and Bellamy 2000). Therefore, where gamebirds released at high densities reduce hedgerow and hedgebank species richness, there is the potential for indirect effects on a range of farmland birds. Yellowhammer for example regularly nest in the lower portion of hedgerows, or in the vegetation below, and are positively associated with wide grass margins and species-rich verges adjacent to hedges that provide good foraging habitat (Green *et al.* 1994, Bradbury *et al.* 2000b, Hinsley and Bellamy 2000). Yellowhammers are nationally declining in the UK (Baillie *et al.* 2009), with reductions in nesting success cited as one of the potential drivers (Leech and Barimore 2008). On a localised scale, the effects of pheasant browsing could influence nesting success by reducing the availability of suitable nesting sites, encouraging nesting in suboptimal habitats and thus increasing nest failure as a result of higher predation rates (nests in sparsely vegetated hedges are more easily detected and accessed by predators; Hinsley and Bellamy 2000). Yellowhammer nest predation associated with hedges is lower on sites where pheasants are not released (Bradbury *et al.* 2000b), compared to sites where pheasants are common (Stoate and Szczur 2001a), which could be due to changes in hedge structure caused by pheasant browsing, although differences between these two studies may also potentially be explained by site or regional effects.

Additionally, high densities of pheasants can result in the loss of larval food plants for Fritillary butterflies such as *Viola* species (Ludolf *et al.* 1989b, Clarke and Robertson 1993), although game estate habitat management may increase the numbers of butterflies overall (see section 4.2).

4.5.4   Predation by gamebirds

As opportunistic feeders, adult pheasants and red-legged partridges consume a wide variety of fauna, mostly invertebrates but also some vertebrates such as reptiles, amphibians, small mammals and juvenile birds. We refer to predation here as the direct effect of the consumption of these other animal species, but also include the effects of disturbance or attacks on (and therefore potential injury of) other animal species.

Invertebrates and vertebrates in non-native gamebird diet

Invertebrate prey of pheasants and red-legged partridges includes beetles, spiders, ants, caterpillars, slugs, snails, earthworms and flies among many others (Collinge 1927, Clarke and Robertson 1993, Callegari 2006a). These protein-rich foods form a varying proportion of adult diets, from 2.7% to 54% for pheasants (Collinge 1927, Hoodless *et al.* 2001, Callegari 2006b) and up to 44% for red-legged partridge (Callegari 2006b). The proportion of arthropod content in adult pheasant faeces is also higher during April to July when most supplementary feeding is stopped, suggesting that pheasants have an increased reliability on invertebrates after the end of the shooting season (Pressland 2009). In the

breeding season, pheasant and red-legged partridge chicks are also reliant on invertebrates in their diet (e.g. Robertson 1997, Callegari 2006b): Hill (1985) identified 22 insect taxa from pheasant chick faecal remains in arable habitats including Carabid (ground) beetles, chrysomelid (leaf) beetles, sawfly larvae, Lepidopteran (butterfly and moth) larvae, Staphylinid larvae (rove beetles), elaterid (click) beetles, Delphacidae (planthoppers) and Heteroptera (bugs; Table 20).

Impacts of predation by gamebirds on invertebrates

The evidence for impacts of predation by released pheasants and red-legged partridges on invertebrates is all either negative or benign (Table 19), and is most likely to affect species of lower mobility or those with larval stages during late summer or autumn (Neumann *et al.* 2015). Corke (1989) assessed 11 butterfly species which declined in range between 1969 and 1989 as being at "high-risk" of predation by pheasants based on their life-history information (chequered skipper*, wood white, brown hairstreak*, Duke of Burgundy, small pearl-bordered fritillary*, pearl-bordered fritillary*, high brown fritillary, dark-green fritillary*, silver-washed fritillary*, marsh fritillary*), with seven of these species (*) showing a significant negative spatial association between 10km map squares where the species were lost 1969–1989 and the density of pheasants in those squares. Eight butterfly species were classed as "low-risk" and none of these exhibited this negative association (silver-spotted skipper, small blue, silver-studded blue, Adonis blue, white admiral, purple emperor, large tortoiseshell, marbled white; Corke 1989).

When further examined in an experimental way, Clarke and Robertson (1993) found no evidence to support the hypothesis that pheasant predation leads to significantly raised rates of larval loss of pearl-bordered or small pearl-bordered fritillaries (although this experiment was conducted in only one woodland in association with one pheasant release pen of unknown size, containing an unknown number of pheasants). In a survey of 52 woodlands in southern England in 1990/91, Clarke and Robertson (1993) also found no evidence to support the hypothesis that butterfly colonies in woods used for pheasant release had been more prone to extinction. The numbers of pheasants released into the countryside has however almost doubled since this study was conducted in the 1990s (see Fig. 2 in section 2.1.1).

Gamebirds can however have impacts on invertebrate abundance and community structure. Pressland (2009) found that the overall biomass and diversity of ground-active invertebrates was lower in late spring (May–June) on grassland field edges adjacent to woodlands where pheasants were released compared to control sites without pheasants, and measured an increase in arthropod content in pheasant faecal samples during the same period. Specifically, Carabidae (ground beetles), Curculionidae (weevils), Isopoda (woodlice), Collembola (springtails) and Tetragnathidae (long-jawed spiders) exhibited significantly lower total biomass in spring at field edges where pheasants were released (Pressland 2009). In contrast, Chrysomelidae (leaf beetles) exhibited a greater overall biomass in areas where pheasants were released, possibly due to decreased competition from invertebrate groups that were less abundant where pheasants were released (Pressland 2009).

When comparing the invertebrate communities at two different scales (inside and outside woodland pheasant release pens, and between woods where pheasants were and were not released), Neumann *et al.* (2015) found no significant impact of pheasant releasing on overall invertebrate abundance, or Carabidae (ground beetle) or Staphylinidae (rove beetle) species richness in spring or autumn, although both Carabid and Staphylinid beetle abundance was considerably lower in release pens. However, the presence of pheasant release pens resulted in significant changes in the composition of Carabidae communities, with shifts away from species typical of woodland towards species typical of arable fields and grassland (Neumann *et al.* 2015). Carabid species active in spring and relatively large species (>17.0 mm) also declined at higher pheasant release densities, as did the abundance of spiders,

harvestmen and centipedes, with Neumann *et al.* (2015) concluding that these effects were strongly suggestive of a negative effect of pheasant predation at the time of pheasant release.

Pressland (2009) also studied pheasant impacts on the biomass of woodland Lepidoptera (butterflies and moths) and Symphyta (sawfly) caterpillars and found a significant negative relationship between pheasant release densities and caterpillar biomass when modelled as the only predictor. Pressland (2009) considered this to be "weak evidence" linking pheasants to larval declines because sampling success was low, and the effect disappeared when habitat variables were included in statistical models, but that the results suggested a "negative relationship warranting further investigation". No impact of releasing pheasants was detected on the abundance or diversity of day-flying adult Lepidoptera between pheasant-managed and non-pheasant woodlands (Pressland 2009).

Callegari (2006a) and Callegari (2006b) found no significant differences in the number of species, abundance, diversity or community composition of invertebrate species emerging from open exclosures (accessible to gamebirds) and closed exclosures (inaccessible to gamebirds) on chalk grasslands where large numbers of red-legged partridge and pheasants were released. However, there was a tendency for higher Adonis blue butterfly emergence when gamebirds were excluded or absent on the same sites, indicating a potential, though relatively small, negative effect of gamebird predation (this relationship was largely non-significant but also suffered from a small sample size (Callegari *et al.* 2014).

The impacts on invertebrates of pheasant predation may still be felt in habitats far from original release sites, due to the dispersal of released birds and the wide extent of the naturalised breeding population. Increasing pheasant densities on an upland grazed grassland site in Wales 2.43 km from the nearest small pheasant-releasing shoot had a negative impact on the diversity of grassland invertebrates sampled, despite pheasant densities being relatively low (average 164 individuals per km2; Devlin 2019). Impacts on invertebrate abundance were more variable, with no significant effects detected on either Coleoptera or Lepidoptera, although increasing pheasant density had a negative impact on the abundance of Orthoptera (Devlin 2019).

Impacts of predation by gamebirds on reptiles and amphibians

There is a large body of anecdotal and photographic evidence from individuals and organisations monitoring reptiles and amphibians suggesting that pheasants predate reptiles and amphibians in the UK (e.g. Baker *et al.* 2004, Gleed-Owen and Langham 2012, Rice 2016, Hand 2020, Madden and Sage 2020)[28]. Phelps (2004) reported that pheasants were among the most consistent predators of adders, and Rice (2016) reports that 4% of 548 people responding to a survey of pheasant impacts on Jersey had observed pheasants predating an amphibian or reptile (pheasants are no longer released on Jersey but are naturalised). In the "Reptile Habitat Management Handbook", pheasants and domestic cats are listed as the key non-native predators of herptiles, particularly adders and sand lizards, with releases of large numbers of pheasants a cause for concern which may result in a high rate of reptile mortality (Edgar *et al.* 2010).

The release of gamebirds in August and September coincides with the birth or hatching of young reptiles, which are small and therefore considerably more vulnerable to predation (e.g. Dimond *et al.* 2014). Releases of large numbers of pheasants close to key areas for reptiles, especially hibernation sites, breeding sites or favoured basking banks are therefore a cause for concern (Edgar *et al.* 2010), and the GWCT advises that release pens are not located on or close to these areas (Sage 2007a).

The extent of this predation, and its effects on larger-scale herptile population trends, are unknown, but when combined with habitat modification and gamebird disturbance at basking and breeding sites,

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[28] https://www.flickr.com/photos/61827574@N03/sets/72157629958660635/

predation may have contributed to the local decline and disappearance of some species (Baker *et al.* 2004, Worcestershire Biodiversity Partnership 2018, Hand 2020). Severe declines in local abundance of adders and other reptiles (including grass snake, common lizard and slow-worm) are attributed to the presence of pheasant and red-legged partridge releasing on at least two protected sites in the UK (Wyre Forest SSSI and Warburg Nature Reserve, Bix Bottom SSSI; Hand 2020).

However, much of the evidence on the impacts of pheasant predation remains anecdotal due to a lack of quantitative studies of sufficient size (Dimond *et al.* 2014, Gardner *et al.* 2019). Rice (2016) dissected the remains of 7 pheasants between 2014 and 2017 in Jersey but found no evidence of reptile or amphibian remains in their crops or gizzards, and the only empirical study which used DNA analysis to investigate the presence of slow worm, grass snake and adder DNA in pheasant faecal samples also failed to find any evidence of reptile predation (Dimond *et al.* 2013, Dimond *et al.* 2014). Extremely limited replication and study duration may however have contributed to these negative results: Rice (2016) dissecting only 7 carcasses collected outside the period of peak herptile activity[29], and Dimond *et al.* (2013) and Dimond *et al.* (2014) collecting 50 faecal samples from only one site with sample collection restricted to July.

4.5.5   Resource competition

Species which contribute a high proportion of the biomass of an ecosystem are likely to represent a substantial source of competition for resources such as food and space. At the point of release, the combined biomass of released and naturalised pheasants and red-legged partridges exceeds twice the spring biomass of all native UK breeding birds, and also more than the post-breeding native bird biomass; see section 2.1.1 (Blackburn and Gaston 2018, Aebischer 2019b, Woodward *et al.* 2020). Only 2.8–3.7% of British breeding bird individuals are non-native species, but these species co-opt a disproportionate amount (up to 13.8%) of the energy used by the UK breeding bird assemblage (Blackburn and Gaston 2018). Up to 81% by mass of these non-native species are pheasants, and therefore as part of the non-native species group (which also includes red-legged partridge) pheasants are expected to have an important role in structuring the communities in which it is embedded (Blackburn and Gaston 2018).

Under the IUCN Environmental Impact Classification for Alien Taxa (EICAT) developed by Hawkins *et al.* (2015) and Blackburn *et al.* (2014), Evans *et al.* (2016) have classified pheasants as having a globally moderate (MO) environmental impact outside of their native range due to "competition resulting in a decline of population size of at least one native species, but with no changes in community composition". This assessment was made with a high level of confidence (Evans *et al.* 2016), although the sources of evidence used to make this assessment were not recorded. Other impact categories defined under EICAT are minimal concern (MC), minor (MN), major (MO) or massive (MV) impacts (Evans *et al.* 2016). Red-legged partridges could not be assessed due to data deficiency (Evans *et al.* 2016).

In the UK there is mixed evidence, much of which is indirect, for competitive interactions between gamebirds and native wildlife, particularly in terms of population-level effects. For example, Callegari (2006b) found no differences in the total breeding season abundance or diversity of native bird species between chalk grassland sites releasing high and low densities of pheasants and red-legged partridges, potentially indicating a lack of population-level effects resulting from resource competition. However, Table 20 highlights the potential for considerable resource competition between pheasants and three native declining farmland bird species by demonstrating the large degree of overlap in the importance of various arthropod species in their chick diets (Green 1984, Hill and Robertson 1988, Stoate *et al.* 1998, Brickle and Harper 1999).

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[29] e.g. https://www.bsg-ecology.com/survey-calendar/

Naturalised pheasants and red-legged partridges breed during the same months as most breeding birds in the UK, and their chicks, similar to those of many native birds, require a protein rich, invertebrate diet. A reduction in invertebrate abundance caused by agricultural intensification is thought to be one of the drivers of recent population declines and range contractions affecting many farmland birds in the UK (Fuller *et al.* 1995, Vickery *et al.* 2009). Resource competition for invertebrates may therefore potentially lower the breeding productivity of native birds where gamebirds breed in abundance. In particular, yellowhammer and corn bunting may compete with pheasants for sawflies and lepidopteran caterpillars (Hill and Robertson 1988, Stoate *et al.* 1998, Brickle and Harper 1999), and grey partridges may compete with pheasants for aphids, bugs and flies (Green 1984, Hill and Robertson 1988); see Table 20. As adult pheasants are approximately 19 and 31 times the mass of corn buntings and yellowhammers respectively, and twice the mass of grey partridge (Blackburn and Gaston 2018), it is likely that pheasant chick arthropod consumption is correspondingly higher than in the other species. Furthermore, pheasant brood size is 3–4 times greater than most farmland passerines as pheasants lay an average of 11 eggs (Robertson 1997). The consumption of invertebrates by gamebird chicks could therefore play an important role in reducing food availability for other farmland birds.

Table 20. Chick dietary composition (% in diet) of arthropods for pheasant (Hill and Robertson 1988), grey partridge (Green 1984), yellowhammer (Stoate *et al.* 1998), and corn bunting (Brickle and Harper 1999). Farmland species were selected based on data availability. Sørenson similarity coefficient describes the similarity between the numbers of shared food items (1 = the same, 0 = no similarity).

Food item	Pheasant	Grey partridge	Yellowhammer	Corn bunting
Symphyta (sawflies)	18.6	-	8.4	15.3
Hemiptera (true bugs)	17.5	4.8	9.3	0.06
Lepidoptera larvae (butterflies, moths)	11.6	1.9	12.2	17.9
Delphacidae (planthoppers)	10.4	-	-	-
Aphididae (aphids)	8.1	32.4	6.4	-
Tipulidae (crane flies)	6.4	-	9.5	-
Other Diptera (flies)	5.7	8.9	-	0.2
Cicadellidae (leafhoppers)	4.1	-	-	-
Curculionidae (weevils)	3.7	-	3.8	-
Carabidae (ground beetles)	2.7	16.7	8.2	0.9
Hymenoptera (wasps, bees, ants)	2.6	6.4	3.4	-
Araneida (spiders)	2.3	0.6	10.4	14
Elateridae (click beetles)	2.2	0.2	0.9	0.7
Staphylinidae (rove beetles)	1.6	14.5	2.4	0.2
Other coleoptera (beetles)	1.6	4.5	-	0.2
Dermaptera (earwigs)	0.6	-	-	0.2
Chrysomelidae (leaf beetles)	0.1	9.1	-	-
Orthoptera (grasshoppers, crickets)	-	-	-	19
Opilionidae (harvestmen)	-	-	-	1
Chrysopidae (lacewings)	-	0.1	-	-
Sørenson similarity coefficient (with pheasant)	-	0.83	0.79	0.69

There is also evidence for potential competition between pheasants and small seed-eating farmland birds during winter, when reductions in food supplies (especially seeds) have been implicated as a further driver of population declines (Newton 2004, Siriwardena *et al.* 2008). Larkman *et al.* (2015) found that national breeding population of tree sparrow is strongly negatively correlated with the density of pheasants released the previous year, with the tree sparrow population halving with every increase of approximately 40 pheasants released per 100 hectares (Larkman *et al.* 2015). A similar albeit weaker

association is also found with the density of red-legged partridge released (Larkman *et al.* 2015). A strong negative correlation also exists between pheasant release density and the combined breeding populations of five other declining small (< 50g) seed-eating birds (linnet, reed bunting, yellowhammer, skylark and corn bunting; Larkman *et al.* 2015). In contrast, national populations of large (> 200g) seed-eating farmland birds which have undergone population increases (e.g. woodpigeon, jackdaw, stock dove), show a strong positive correlation with pheasant release densities (Larkman *et al.* 2015). For both these small and large farmland species, seeds are an important component of winter diet but their population trends all correspond poorly to indices of agricultural intensification (Larkman *et al.* 2015).

Larkman *et al.* (2015) suggest that these associations could be driven by competition-mediated exclusion of smaller seed-eating species, facilitated by an increase in the provision of high-density seed sources for released gamebirds (see sections 4.2.3 and 4.2.6) combined with a reduction in low-density seed sources (e.g. overwinter stubble, weedy habitats, set aside) as a result of agricultural intensification. Prior to agricultural intensification and the increase in large-scale pheasant and red-legged partridge releasing, small seed-eating species may have used low-density seed sources which larger species could not use effectively, leading to niche separation (Larkman *et al.* 2015). The reduction in these low-density sources combined with the increasing provision of high-density seed sources may have reduced the effectiveness of this separation, with small species now forced to use high-density seed sources where they may be outcompeted by larger seed-eating birds such as pheasants and red-legged partridges, potentially resulting in a lower intake of lower-quality food, thereby reducing the ability of small species to co-exist alongside larger seed-eating birds (Larkman *et al.* 2015). However, Bright *et al.* (2014) showed no effect of larger species on seed depletion rates at wild bird seed-bearing plots in farmland, and Sánchez-García *et al.* (2015) observed regular usage of gamebird feeders by small bird species, indicating that both small and large seed eating species may be able to exploit high density seed patches on farmland.

It is possible that competition may occur between non-native gamebirds and other sympatric species such as grey partridge, black grouse and red grouse (Chapman 2019). The GWCT suggest that releasing pheasants or red-legged partridges along the moorland fringe could displace black grouse (GWCT 2003), and although the mechanisms for this are not discussed, competition for food resources, disturbance at black grouse leks, the potential for disease transfer (section 4.6) and accidental shooting mortality (section 4.3.3) are all plausible (e.g. Cole *et al.* 2012). The GWCT also "strongly discourages" any releasing of pheasants and red-legged partridges in areas where grey partridge recovery is targeted (Aebischer 2009), again presumably due to the possibility for resource competition, disease transfer and accidental shooting mortality (see sections 4.6 and 4.3.3). Additionally, pheasants commonly practice brood parasitism, where the female lays her eggs in another nest to be cared for by another individual, and have been recorded parasitizing the nests of grey partridge, black grouse, red grouse and capercaillie as well as non-sympatric species including northern shoveler, mallard, gadwall, woodcock and corncrake (geographic range of records unknown, but likely not confined to the UK; Hill and Robertson 1988, Krakauer and Kimball 2009). It is unknown to what extent nest parasitism may impact on the breeding success of native UK species, although nest parasitism of grey partridge by pheasants has been reported as a cause of grey partridge nest failure, resulting in abandonment either during incubation or at the point of hatching (pheasant eggs hatch first and the adult grey partridges have been known to leave the nest with the young pheasants, deserting their own eggs at the point of hatching (Jenkins 1961).

4.5.6   Soil enrichment

Many of the impacts on woodland plant communities associated with browsing by gamebirds detailed above (section 4.5.3) could plausibly also be linked to changes in microclimate associated with woodland management for pheasants (see section 4.2.4), as well as with the enrichment of soil by the

gamebirds themselves. Where pheasants are present at high densities, particularly within woodland release pens, their faeces build up and inevitably lead to changes in soil chemistry. This enrichment can significantly increase the concentration of nutrients such as nitrogen, phosphorus and potassium in soils associated with pheasant releasing (Sage *et al.* 2005a, Sage 2018b), although soil pH and the concentrations of magnesium appear not to be affected (Sage *et al.* 2005a).

Impacts of soil enrichment on plants

Soil enrichment resulting from the presence of pheasants in woodland can have negative effects on archetypal and ancient woodland indicator plant species for which higher fertility soils are sub-optimal (Hermy *et al.* 1999, Hill *et al.* 1999, Sage *et al.* 2005a, Neumann *et al.* 2015, Sage 2018b, Capstick *et al.* 2019b), and which cannot compete with species associated with high nutrient availability, such as common nettle, bracken, bramble and annual grasses (Hill *et al.* 1999, Sage *et al.* 2005a, Sage 2018b). Overall plant species richness and vegetation cover may be similar (Neumann *et al.* 2015, Sage 2018b), but communities shift from those associated with low fertility to those associated with high fertility in pheasant-releasing woodlands (Sage *et al.* 2005a, Neumann *et al.* 2015, Capstick *et al.* 2019b).

The increase in soil nutrients may also have an impact on the concentrations of these nutrients in the air through emissions from the soil. Nitrogen can be emitted from soils in the form of nitrous oxide and ammonia and is commonly recorded from nitrogen-rich soils in agricultural settings (Air Quality Expert Group 2018). It is possible that the build-up and subsequent decomposition of pheasant faeces in pheasant-releasing woods results in increased soil emissions and therefore higher concentrations of nitrogen in the air (Sage 2018b). Indeed, Sage (2018b) found that in pheasant-releasing woodlands, the species diversity of mosses and liverworts on tree trunks was half that of woods where pheasants were not released, and liverworts were also half as abundant. These effects were not confined to pheasant release pens but extended into the surrounding woodland (Sage 2018b), suggesting that any effects of soil enrichment associated with faecal build-up within release pens may affect airborne nitrogen concentrations throughout the woodland as a whole. Other potential mechanisms exist, including browsing and mechanical damage from pecking by pheasants, although deleterious effects on woodland bryophyte communities from increased atmospheric nitrogen originating from soil emissions and other sources (but independent of gamebird release) have been documented elsewhere in the UK (e.g. Mitchell *et al.* 2004, Mitchell *et al.* 2005).

Outside of the immediate release area the effects of soil enrichment may become be less pronounced as gamebirds (and therefore their droppings) disperse throughout adjacent habitats. For example, hedgebank vegetation may be affected by proximity to release pens, with more annual, fast-growing weeds on hedgebanks near to pheasant release sites compared to those further way, an effect which may be explainable by a decreasing gradient of soil fertility linked to fewer pheasant droppings as well as less soil disturbance from browsing pheasants as they disperse (Sage *et al.* 2009).

Few sources have studied the potential for effects of soil enrichment in relation to red-legged partridge releasing. Callegari (2006b) however found no difference between areas of high and low density red-legged partridge and pheasant releasing in terms of soil pH or phosphorus and nitrogen content, and Callegari *et al.* (2014) found shorter vegetation and no significant differences in the number of positive and negative indicator plant species on chalk grassland sites experiencing considerable releases of red-legged partridges in addition to pheasants (64,000 and 37,000 released across 4 sites respectively). This may indicate few effects of soil enrichment in this habitat as a result of these releases (taller vegetation and increases in numbers of negative species would be expected), although the conditions within and immediately outside release pens were not examined, and livestock grazing was common across all sites studied (the considerable nutrient input from which may have rendered the impacts of gamebirds undetectable; Callegari 2006b)

In recent years red-legged partridges and pheasants have been released more frequently on the edge of upland moorland[30]. Craig Leek, a limestone crag outcrop SSSI in Scotland is home to an extremely rich bryophyte community with eight Nationally Rare species, and one Red Data Book species. A game estate adjacent to Craig Leek now release red-legged partridges which roost on the crags at Craig Leek, causing soil eutrophication from faecal build-up with detrimental effects to the fragile bryophyte community (Rothero 2006). Some of these rare bryophyte species are only represented by one known colony in the area, and just two or three populations in the entire UK. Therefore, soil enrichment by partridges severely threatens their existence in the UK (Rothero 2006), and demonstrates that releasing gamebirds in close proximity to sensitive areas with fragile species of high conservation importance can be extremely detrimental.

Impacts of soil enrichment on invertebrates

The impacts of soil enrichment linked to gamebird releasing on invertebrate communities has not been well studied, although Neumann *et al.* (2015) indicate that, when combined with invertebrate predation by pheasants, soil enrichment may act to change the invertebrate community composition towards a higher abundance of detritivores including Diplopoda (millipedes), Oniscoidea (woodlice) and Gastropoda (snails) as pheasant release densities increase. These invertebrate groups are commonly associated with higher organic matter and nutrient content of the soil, so are likely to have benefitted from high densities of pheasant droppings as well as physical damage to plants caused by browsing activities (see section 4.5.3).

4.5.7   Density dependent effects of pheasant releasing

The extent and longevity of many of the direct impacts of pheasants on plants, invertebrates and birds described above are density dependent, in that they are more pronounced at higher pheasant densities (whether that be releasing densities within release pens, densities within woodlands or the wider countryside, or the absolute number released). Multiple studies suggest that the intensity of gamebird release (the density of releasing, or birds per hectare) can influence the extent of associated impacts (e.g. Sage 2003, Sage *et al.* 2005a, Gortazar *et al.* 2006, Pressland 2009, Neumann *et al.* 2015, Capstick *et al.* 2019b, Madden and Sage 2020). It is unclear to what extent this may also be true for red-legged partridges whose impacts have been less commonly studied, although Callegari (2006b) suggests density dependant effects may be less pronounced. Sage *et al.* (2005a) demonstrate how negative impacts on woodland vegetation characteristics within release pens increase as pheasant stocking densities increase, with a reduction in species of shaded habitats and an increase in species of fertile and disturbed soil seen at densities greater than 1,000 pheasants per hectare (Sage *et al.* 2005a). Recovery of woodland vegetation is also influenced by gamebird stocking densities, with the total species richness and richness of ancient semi-natural woodland plants only showing signs of recovery after 10 years in release pens where ≤1,000 pheasants per hectare were released (Capstick *et al.* 2019b). Hedgebanks in areas where >1,000 pheasants are released also consistently contain fewer shrub and seedling species and hedgerows are less leafy (Sage *et al.* 2009), while the cover of shorter herbs along hedgerows adjacent to pens is influenced by absolute pheasant release numbers (Woodburn and Sage 2005).

Additionally, high densities of pheasants can result in the loss of larval food plants for Fritillary butterflies such as *Viola* species (Ludolf *et al.* 1989b, Clarke and Robertson 1993), and the larval biomass of woodland caterpillars decreases as pheasant release densities increase up to 300 pheasants per hectare (Pressland 2009). The abundance of Carabid beetles, spiders, harvestmen and centipedes also decline within release pens as stocking densities increase to 3,000 pheasants per hectare (Neumann

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et al. 2015). Increasing pheasant densities on an upland grazed grassland site in Wales 2.43 km from the nearest small pheasant-releasing shoot also had a negative impact on the diversity of grassland invertebrates (average 164,000 individuals per hectare; Devlin 2019).

It is also likely that other impacts may be more pronounced at higher gamebird densities (Gortazar *et al.* 2006, Pringle *et al.* 2019, Madden and Sage 2020), such as disease transmission to wildlife (section 4.6) or humans (section 5.1.6) and impacts on predators and predation (section 4.7). The GWCT (Sage 2007b) and The Code of Good Shooting practice[31] recommend that pheasants be released at densities not exceeding 1,000 birds per hectare of woodland release pen in general, and no more than 700 birds per hectare when pens are situated in ancient semi-natural woodland (see Table 5 in section 2.1.3). However, many release pens are still stocked at levels greater than 3,000 pheasants per hectare (e.g. Davey 2008, Pressland 2009, Neumann *et al.* 2015), with some reaching more than 5,000 pheasants per hectare: Fig. 21 (Sage *et al.* 2005a, Davey 2008). With the increasing numbers of pheasants and red-legged partridges being released (Fig. 2 in section 2.1.1), densities of these birds on shooting estates and in the wider UK countryside are also likely to be increasing. Dividing the 57 million pheasants and red-legged partridges released in 2016 (Aebischer 2019b) by the estimated area of release pens in 2012/13 (10,000 hectares; see Fig. 14 in section 4.2), gives an estimated density of 5,700 birds per hectare, considerably above the 1,000 per hectare recommended.
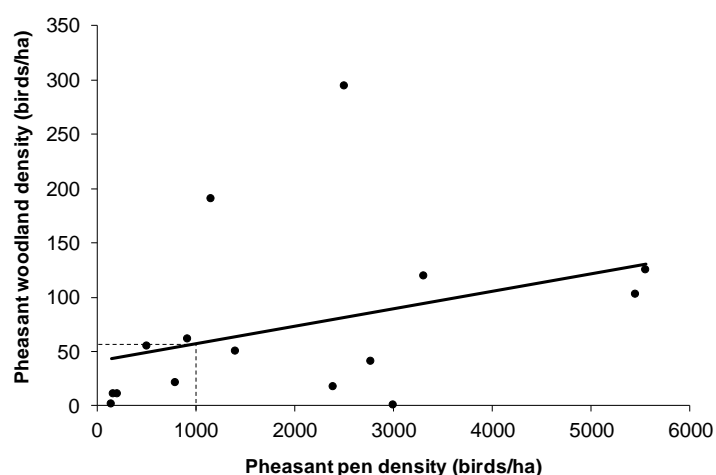
Fig. 21. Relationship between the estimated density of pheasants within woodland release pens and the density of pheasants within the surrounding woodland from 15 sites surveyed in 2007 by Davey (2008) where pheasant releasing occurred and the estimated pen density was <10,000 pheasants per hectare. The dashed line shows an approximate woodland density of 55 per hectare where pens are stocked at 1,000 per birds hectare. This subsequently enables the comparison of different measures of pheasant abundance. Data is from Davey (2008).

The density of pheasants within release pens can understandably directly affect the density of pheasants within the surrounding woodland post-release. Estimates of total woodland pheasant density also vary depending on the area of the woodland relative to the number and area of release pen(s). The GWCT recommend that on a 450 hectare estate with 30 hectare (6.7%) of woodland (typical for lowland England), the total area of release pens should not exceed one third of the woodland (10 hectares), into which they suggest it would be appropriate to release between 7,000 and 10,000 pheasants (700–1,000 per hectare of release pen; Sage 2007b). This would equate to 16–22 birds per hectare of the entire estate, or 233–333 birds per hectare of estate woodland (Sage 2007b). In practice, the total woodland density associated with a release pen density of 1,000 pheasants per hectare may be considerably lower. According to one available study from which a comparison of release pen and woodland density is possible (Davey 2008), at the recommended release pen stocking density of 1,000

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[31] http://www.codeofgoodshootingpractice.org.uk/pdf/COGSP.pdf

birds per hectare, the average post-release pheasant density within the surrounding woodland (assuming it contained only one release pen) was actually closer to 55 birds per hectare of woodland (Fig. 21); approximately five times lower than the GWCT recommendation (Sage 2007b). If a typical 450 hectare estate in Britain contains 30 hectares of woodland (Sage 2007b), 55 birds per hectare in that woodland (Fig. 21) would equate to 1,650 birds at a density of 3.7 birds per hectare of estate.

These calculations do however make broad assumptions about release estate characteristics, and do not account for the potentially aggregated and clumped distribution of birds within the estate. Furthermore, while the figure of 700–1,000 birds per hectare of release pen may be suitable to mitigate impacts within pens, it seems likely that the recommendation for pens to cover no more than one third of available woodland may be too high to mitigate impacts outside of pens, particularly if pens cover almost the entirety of small woodlands (currently acceptable if balanced by un-penned woodland elsewhere on the holding, according to the best practice guidance; Sage 2007b).

Fig. 22 attempts to theorise the overall impact of gamebird release on native wildlife at different release densities and management practices. When habitat management is optimal, any negative impacts on wildlife may be mitigated to a certain degree, and at low release densities habitat management may offset any negative effects, and the net impact may be positive. In circumstances where birds are stocked at more than 1,000 birds per hectare of release pen (equal to approximately 55 birds per hectare of woodland, or 3.7 birds per hectare of estate; Fig. 21), negative impacts on wildlife are likely despite beneficial habitat management. When no habitat management is implemented, negative impacts may occur at lower pheasant densities.
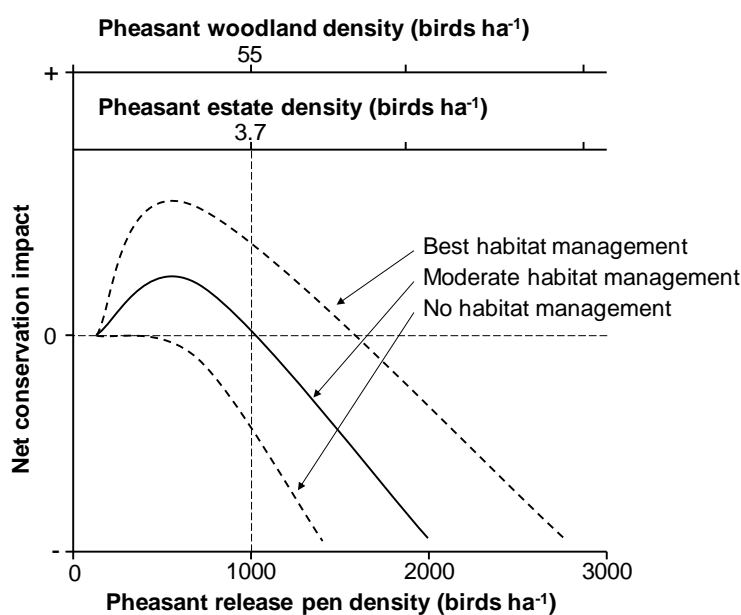
Fig. 22. Theoretical net conservation impacts of three gamebird habitat management scenarios at differing pheasant release densities.

4.5.8   Key knowledge gaps and recommendations

- A large body of anecdotal evidence from individuals and organisations monitoring reptiles and amphibians suggests that pheasant releasing has contributed to the local decline and disappearance of some species. However, there are no conclusive or large-scale studies demonstrating a clear impact of pheasants on the populations of reptiles and amphibians. This topic would benefit from both a larger scale study (national or regional) examining the

relationship between the trends in spatial distribution of reptile and amphibian species and gamebird releasing activities, and more intensive local or small-scale captive experimental studies examining changes in reptile and amphibian abundance in relation to pheasant release density or patterns of gamebird habitat use.

- Recent studies have highlighted that the biomass of released and naturalised gamebirds exceeds that of all other UK breeding birds, but few studies yet examine the impact this may be having on native UK bird populations through potential competition for resources such as food or space. Studies of native species diet and performance between sites with contrasting gamebird densities, or in relation to the manipulation of gamebird densities through time, would therefore be beneficial.

4.6 Disease transmission to wildlife

4.6.1 Impact summary

There was a relatively small amount of evidence (13 scores evidenced by 13 sources) for impacts of disease transmission from gamebirds to native UK wildlife. The ecological impact scores associated with disease transmission from gamebirds to wildlife were significantly negative on average (z = -2.83, P = 0.027; Fig. 12, Table 2, Table 21). Sources that demonstrate or even imply the transfer of parasites from gamebirds to native wildlife are few, mainly due to the difficulties associated with establishing the direction of infection of a shared parasite. Likewise, the effects on native wildlife from the various parasites and diseases for which gamebirds may act as vectors have not been extensively quantified, except where significant outbreaks have occurred, and even in these cases, the source of the outbreak is rarely well-understood.

Table 21. Summary ecological impact scores associated with disease transmission from gamebirds to wildlife, including the distribution of scores at each score level, the estimated marginal mean score and 95% confidence limits (95% CL) returned by the Ordinal Logistic Model (OLM), and the proportion of scores which originated from peer-reviewed research and for which there was direct evidence of an impact rather than a potential impact.

	Ecological impact score						
	-2	-1	0	1	2	Mean	95% CL
Disease transmission to wildlife	2	7	4	0	0	-1.66	-3.20 – -0.12

13 scores evidenced by 13 sources*: 92% peer-reviewed, 54% direct evidence
* (Keymer *et al.* 1962, Tompkins *et al.* 1999, Tompkins *et al.* 2000a, Tompkins *et al.* 2000b, Tompkins *et al.* 2001, Ewald and Touyeras 2002, Sage *et al.* 2002a, Sage *et al.* 2002b, Aldous and Alexander 2008, Potts 2009, Aldous *et al.* 2010, Potts 2010, Hillman *et al.* 2019)

Pheasants and red-legged partridges are subject to pathogen loads from numerous different groups, including protozoa, helminths (nematodes, trematodes, cestodes), fungi, viruses, bacteria and arthropods. While over 80 parasites have been identified in red-legged partridges (Millan 2009), few have been actively investigated in wild birds or other wildlife. Approximately one third of the known pathogens of red-legged partridges originate in the wild, one third confined to captive bred birds and the remaining third in both captivity and the wild (Millan 2009). After pheasants are released, their pathogen loads increase, and new pathogens are also gained (Villanua *et al.* 2006a). Treatment is continued via supplementary feeders that also attract other birds (Millan 2009). While supplying food to wild birds is of potential benefit to biodiversity conservation (see section 4.2), if captive bred pheasants act as primary reservoirs for pathogens, feeding stations pose a threat for wild birds through the transfer of these pathogens.

The majority of detailed studies examine the interaction between pheasants, grey partridges and the endoparasite *Heterakis gallinarum*. Pheasants act as a reservoir of this parasite because they are largely resistant to it, passing it on directly or indirectly to grey partridges which have lower resistance and may suffer population declines as a result. Pheasants are also prolific transmitters of the bacteria responsible for Lyme disease, which infects passerines and humans via a tick vector, and carry strains of Avian Influenza which, though currently considered to be of low pathogenicity, have the potential to mutate to more highly pathogenic strains. Pheasants can also act as carriers of Newcastle disease with potential for transmission to wild birds and are commonly infected with the parasitic nematode

Syngamus trachea which is known to infect corvids and reduce reproductive success in house sparrows.

While game rearing farms do treat birds, there are few regulations, and therefore no consistent records of the levels at which pathogens are kept under control, and of course without statutory obligation there is likely to be wide-ranging variation between farms. Likewise, after release, free-ranging animals are no longer treated for parasites unless they feed at hoppers supplying medicated feed. At high release densities pheasants disperse further from release pens (Sage and Swan 2003), are less likely to regularly use hoppers containing treatment, and are therefore more likely to represent greater pathogen transfer threats to native wildlife than those released at low densities. The impacts of the medications in treated feed themselves on wildlife are also poorly understood. Antimicrobial resistance in pathogenic bacteria as a result of the widespread, largely prophylactic use of antibiotic medications in gamebird feed (and across the livestock rearing, veterinary and human health sectors) has also generated major concern in the UK and globally through the associated risks to human health. This practice is thought to have increased following a ban on the previously widely-used anti-protozoan product Emtryl (Dimetridazole) in 2003, for which antibiotics were the only effective alternative treatment. The gamebird sector has however begun to take considerable steps to reduce this practice and the associated risks in recent years.

4.6.2   Background

Conditions that promote parasite infestation

Pheasants and partridges are prone to high levels of parasitic infection (Draycott *et al.* 2006), which are promoted by environmental conditions associated with high stocking densities (Coles 1984, Jansson *et al.* 2001, Millan *et al.* 2004, Gortazar *et al.* 2006, Millan 2009). While most birds naturally act as hosts for many parasites, over-stocking and mismanagement in livestock industries can exacerbate parasite loads, allowing new pathogens to enter the system, and existing ones to become more ubiquitous (Tapper 1999). Captive bred gamebirds carry higher parasite loads than their wild counterparts because at higher stocking densities gut parasites are more likely to be picked up by other birds after they are excreted (Millan 2009). In the wild, infections tend to be self-limiting or 'sub-clinical' (e.g. with no outward symptoms) as the birds develop some degree of immunity or tolerance. However, when infections are overwhelming, or if the immune system is depressed, then 'clinical' disease occurs, where recognisable deleterious symptoms result in illness or mortality. Wild pheasants breed more successfully in Britain than captive-bred pheasants, and this has been attributed, in part, to higher parasitic burdens in the released captive-bred population (Draycott *et al.* 2006)

There is also a risk that diseases will be imported from other countries along with the gamebirds destined for release in the UK. However, there are measures in place to prevent the introduction of notifiable diseases including Avian Influenza (H5N1) and Newcastle disease through gamebirds imported from France (the primary source of gamebird imports destined for UK release; DEFRA 2010a).

Risks to native wildlife

Many introduced species carry with them associated parasites or diseases that native wildlife may be naïve to, particularly when the strain does not naturally occur within their range (Manchester and Bulloch 2000). Parasites and diseases are often specific to a narrow host-species range however, so threats of transfer from pheasants and red-legged partridges are in many cases likely to affect only closely related species such as grey partridge (Hudson 1997), black grouse (Cole *et al.* 2012) or red grouse (GCT 2003). Additionally, many of the diseases associated with gamebirds are restricted to conditions in rearing pens and are therefore not likely to represent a major risk to wildlife. Despite this, several

parasites thought to be spread by gamebirds may pose a threat to native animals. This is of concern since the introduction of diseases may reduce reproductive output and increase mortality rates and the risk of predation (Woodburn 1995). In addition to posing a threat to native wildlife, infected populations also form a reservoir of infection for domestic livestock, pets and humans (Kurtenbach *et al.* 1998a, Pennycott *et al.* 1998); see section 5.1.6.

Medication and antimicrobial resistance

Several veterinary medications are used to treat gamebirds during rearing, in release pens, and at supplementary feeders around release estates. Most farm-reared gamebirds are fed with enhanced feed mixes that include proteins, vitamins, minerals and other necessary nutrients. While many rearing farms treat signs of parasitic infection with anti-microbial coccidiostats, worming medications and antibiotics, others use them in everyday feed mix in a prophylactic manner[32]. The potential effects of medications on non-target species including other birds and invertebrates has not been studied, and as such is a key knowledge gap that would benefit from further research (Natural England 2009, Mustin *et al.* 2012).

In 2003 a widely used product called Emtryl,(Dimetridazole) used to treat and prevent the protozoans *Trichomonas* and *Hexamitiasis*, was banned following concerns regarding the potential for carcinogenic effects on humans following consumption of game treated with it (Davis 2004, Canning 2005). The cessation of its use prompted significant management and husbandry changes in gamebird rearing farms, primarily focusing on reducing stress and densities. It has also been suggested that the banning of this product, which was heavily relied upon during gamebird rearing and for which there was no similar effective replacement, led to the increase in prophylactic use of antibiotics, which was thought to improve gamebird resistance to protozoan pathogens despite having no direct effect on them (e.g. Elliott 2018).

Antibiotics are widely used both prophylactically and to treat clinical illnesses during gamebird rearing and post-release supplemental feeding (Seguino and Chintoan-Uta 2017, UK-VARSS 2019). There is considerable concern nationally and globally that this and similar widespread indiscriminate use of antibiotics throughout the agricultural, veterinary and human health sectors might lead to increases in anti-microbial resistance in bacteria, with knock-on effects for human health (Natural England 2009, O'Neill 2015, O'Neill 2016).

Wildlife may also be at risk from antimicrobial-resistant disease strains originating in and carried by released gamebirds (although resistant bacterial strains may also be spread from other farmed livestock; Madden and Sage 2020). Resistance of multiple bacterial strains to multiple antibiotic types long after their original antibiotic medication has been detected in samples from pheasants and red-legged partridges in Belgium and Spain (Devriese *et al.* 1996, Guerrero-Ramos *et al.* 2016), and LA-MRSA (livestock-associated methicillin-resistant *Staphylococcus aureus*) was reported in a pheasant in Scotland in 2017 (UK-VARSS 2019). Díaz-Sánchez *et al.* (2012b) found that antibiotic resistance in *E. coli* was much more frequent in farmed red-legged partridges (75%) compared to wild birds in Spain, meaning that with widespread indiscriminate use of antibiotics, farms rearing red-legged partridges for release could become a potential source of resistant *E. coli* in the environment.

There are no confirmed instances of antimicrobial resistance transfer from gamebirds to wildlife, although high levels of resistance potentially associated with medicated gamebird feeders were found in bank vole and wood mice populations in northwest England (Gilliver *et al.* 1999), and in foxes in Scotland (Carson *et al.* 2012) and buzzards in Portugal (Radhouani *et al.* 2010) which may have consumed gamebird carcasses containing veterinary antibiotics or antimicrobial resistant organisms (Madden and Sage 2020).

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[32] e.g. https://keeperschoice.co.uk/2020/07/01/flubenvet-protecting-gamebirds-from-worms/

In response to the threat of antimicrobial resistant disease infections, the UK government has committed to enforce a reduction in the use of antibiotics in livestock rearing (including gamebird rearing), veterinary and human medicine through two five-year action plans between 2013 and 2024 (Department of Health 2013, Global and Public Health Group 2019). The game shooting community has made good progress in starting to achieve these reductions, with an overall 52% reduction in antibiotic use between 2016 and 2018, and the use of antibiotics in game feeds falling by at least 70% (UK-VARSS 2019). Multiple organisations across the UK gamebird sector agree that there is still work to be done to continue to drive down all unnecessary antibiotic use however (Anon 2020).

4.6.3   Diseases and parasites carried by gamebirds

Heterakis gallinarum and other parasitic worms

In Britain, the most common endo-parasites that infect pheasants are the gastrointestinal worms *Heterakis gallinarum*, *Capillaria spp.,* and the tracheal worm *Syngamus trachea*. Intake of these parasites is via direct ingestion of eggs in soil or faecal particles, or indirectly via soil-feeding organisms including earthworms (Beer 1988, Draycott *et al.* 2006); there is therefore a potential risk of infection in wild birds and mammals ingesting soil invertebrates infected by close proximity to pheasant rear-release areas. Endoparasites can cause pathogenic effects themselves, and/or carry other pathogenic organisms that result in secondary infection. For example, the nematode worm *Heterakis gallinarum* is a transport host for the protozoan *Histomonas meleagridis*, which causes histomonosis, a disease associated with high mortality in untreated grey partridges and red-legged partridges (Potts 2009) and which has also been isolated in black grouse (Cole *et al.* 2012). Successful treatment of this disease in reared birds has however resulted in a large decline in the occurrence of histomonosis since the 1960s (Potts 2009).

Heterakis gallinarum: Parasite-mediated competition between pheasant and grey partridge

Pheasant and grey partridge share numerous parasites globally (Table 22). Tompkins *et al.* (1999), Tompkins *et al.* (2000a), and Tompkins *et al.* (2000b) studied the role of the nematode worm *Heterakis gallinarum* within spatially intersecting pheasant and grey partridge populations and found that levels of *H. gallinarum* in grey partridges on a pheasant release estate were significantly correlated with *H. gallinarum* levels in pheasants (Tompkins *et al.* 2000a). Grey partridges were negatively affected by the parasite, which resulted in reduced mass gain, decreased food consumption and impaired caecal activity (Tompkins *et al.* 1999, Tompkins *et al.* 2001). Tompkins *et al.* (1999), and Tompkins *et al.* (2001) also found that pheasants were largely unaffected by *H. gallinarum* apart from some impairment of caecal activity, though a later study by Sage *et al.* (2002a) found that the body mass, breast muscle mass and cloacal fat of pheasants in release pens was negatively affected by the parasite abundance carried by the birds.

Table 22. Endoparasites found in pheasant and grey partridge globally (after Vrezec 2006).

Parasite	Pheasant	Grey partridge
Trematodes		
Echinoparyphium cinctum	+	
Brachylaemus fuscatus	+	
Postharmostomum gallinum*	+	+
Cestodes		
Davainea proglottina		+
Raillietina echinobothrida	+	
Choanotaenia infundibulum*	+	+
Rhabdometra nigropunctata		+
Drepanidolepis anatina		+
Passeripelis crenata	+	
Nematodes		
Capillaria columbae*	+	+
Capillaria picorum	+	
Thominx contorta*	+	+
Trichostrongylus tenuis*	+	+
Syngamus trachea*	+	+
Ascaridia galli	+	
Heterakis gallinarum*	+	+
Ganguleterakis isolonche*	+	+
Cyrnea spinosa	+	
Acuaria hamulosa	+	
Dispharynx nasuta*	+	+

* Parasite species found in both hosts constitute potential or actual mediators in apparent competition between pheasant and grey partridge.

After experimental trials, Tompkins *et al.* (2000b) predicted that spatial overlap between grey partridge and pheasant populations greater than 57% would cause the exclusion of grey partridge, due to their lower parasite resistance. They further predicted that in the absence of a pheasant population, *H. gallinarum* would not persist in grey partridge populations because its reproductive rates would be greatly lowered and grey partridges are able to expel *H. gallinarum* through their gut at much faster rates than pheasants. Since *H. gallinarum* has been detected in wild grey partridge (Clapham 1935, Keymer *et al.* 1962), and Tompkins *et al.* (2000b) suggest that it cannot exist in grey partridge in the absence of a shared host, it is likely that parasite transmission to this species does occur from other reservoirs in the wild. As *H. gallinarum* is also unable to survive in red-legged partridges (Tompkins *et al.* 2002), pheasants are therefore implicated as being solely responsible for maintaining *H. gallinarum* infections in grey partridge the UK, being the only other gamebird with an ecological overlap with grey partridge.

The extent of the impact of this parasite on grey partridge is, however, disputed. Sage *et al.* (2002b) conducted further studies of *H. gallinarum* in grey partridge and found no negative effects on body condition or fecundity, suggesting that grey partridge are in fact tolerant to the parasite. The major

difference between these studies was that Sage *et al.* (2002b) injected the parasite directly into the birds within a controlled laboratory environment, while in the Tompkins *et al.* (1999, 2000a, 2000b) studies the parasite was transmitted naturally from pheasants to grey partridges within (1) a release pen, and (2) free-ranging on a pheasant estate. It is possible that *H. gallinarum* that have multiplied within pheasants are better adapted than laboratory strains to parasitize grey partridge to greater effect. Likewise, the Tompkins *et al.* (1999, 2000a, 2000b) studies are more representative of conditions in nature. Despite conflicting results, it therefore seems plausible that parasite transmission from pheasants at high densities may be impeding the current efforts to restore grey partridge numbers in the UK.

Potts (2010) and Potts (2009) have since found that infection rates by *H. gallinarum* have significantly fallen in adult red-legged partridges and grey partridges by more than 90%, and by 20% (not significantly) in adult pheasants, since 1951. They hypothesise that free-ranging domestic fowl (chickens, turkeys) were the most likely source of the parasite prior to the mid-1950s, and infection rates have reduced because these are now much less abundant. Draycott and Armenteros Santos (2012) found that prevalence of *H. gallinarum* in juvenile grey partridges was still relatively high in the autumn of 2010 on at least one site however, with 45% of 139 juveniles less than 5 months old containing the parasite, which had most likely been transmitted from infected pheasants. The grey partridge population sampled by Draycott and Armenteros Santos (2012) was undergoing a rapid population increase and no birds exhibited signs of illness or loss of body condition, so the authors thought it unlikely that the levels of parasitic infection were having a significant negative effect on the population. Ewald and Touyeras (2002) also found no spatial association between pheasant release and grey partridge population declines and concluded that parasites transferred between these species are therefore probably not an influential factor of the decline. This evidence further refutes the hypothesis that pheasants spread significant *H. gallinarum* to partridges.

The contrasting results above suggest that any effects of parasite-mediated competition between pheasant and grey partridge may be site- or context-specific (Chapman 2019). There are few studies examining these relationships in other species and other parasites, so further research in this area is needed.

Heterakis gallinarum: Gamebirds and moorland fringe habitat

Red grouse have undergone declines in recent years resulting in their classification on the UK amber list (Eaton *et al.* 2015). As red grouse is an important gamebird in moorland habitats, game managers in these areas have begun releasing pheasants and red-legged partridges at the edges of moorlands to compensate for declining grouse populations, to maintain shooting interests and to support associated jobs. In 2002, six red grouse were found infected with *H. gallinarum* on a Yorkshire moor (GCT 2003). Pheasants that had been released nearby were thought to be the most likely source of the infection (GCT 2003). It is currently unknown whether *H. gallinarum* could induce a population response in red grouse; where pheasant and red-legged partridge come into contact with similar species such as red grouse and black grouse on moorland fringe habitat, there are however grounds for concern (Cole *et al.* 2012). If *H. gallinarum* negatively impacts red grouse populations, and lower densities of red grouse contribute to estates initiating increased releases of other gamebirds at the moorland fringe, this may instigate a feed-back loop of considerable economic and environmental consequence.

Syngamus trachea

Syngamus trachea (gapeworm) is a parasitic nematode worm that causes syngamiasis in a range of avian hosts (Gethings *et al.* 2015b, Gethings *et al.* 2015a). *S. trachea* infections are relatively common in pheasant rearing and release pens, especially where high densities of birds are reared together (Gethings *et al.* 2015a) and even sub-clinical infections can result in reductions in pheasant body

condition (Gethings *et al.* 2015a). Gethings *et al.* (2015a) hypothesise that this could be the cause of poor reproductive success and survival of pheasants post-release (see section 2.1.1). The parasite can build up in the environment, with pheasant release pens which have been in use for longer (several years) having a higher abundance of *S. trachea* eggs (Gethings *et al.* 2015a, Gethings 2018). Stocking densities also influence the parasite load, with higher densities of pheasants resulting in more eggs (Gethings *et al.* 2015a). *S. trachea* can be transferred directly or indirectly; many invertebrates ingest *S. trachea* eggs and thus serve as hosts which may subsequently be ingested by wild birds. Syngamiasis has been recorded in house sparrows where it has both short- and long-term effects on reproductive success (Holand *et al.* 2015). Bandelj *et al.* (2015) also found *S. trachea* in 2.6% of 76 individual birds from 17 species of non-migratory European passerine in Slovenia, and it is present in corvids in the UK including crow, rook and jackdaw (Simon *et al.* 2011, Gethings *et al.* 2015a, Holand *et al.* 2015, Gethings *et al.* 2016). Corvids are often cited as the likely mode of transmission of *S. trachea* between pheasant release areas (Simon *et al.* 2011, Gethings *et al.* 2015a, Gethings 2018), but the clinical and population effects on these wild birds themselves has not been studied.

Trichomonas gallinae (trichomonosis) and other protozoans

Protozoans such as *Trichomonas spp.* are relatively common in captive bred pheasants (Tapper 1999), and are known to infect and lead to the death of wild bird species, particularly birds of prey, finches, pigeons and doves (Real *et al.* 2000, Duff *et al.* 2003, Hofle *et al.* 2004, Villanua *et al.* 2006b, Bunbury *et al.* 2008, Sansano-Maestre *et al.* 2009, Lawson *et al.* 2012). *Trichomonas* is passed from bird to bird, often at feeding and drinking stations (Gortazar *et al.* 2006, Villanua *et al.* 2006b, Lawson *et al.* 2012, Lennon *et al.* 2013), or after consumption of an infected animal (Real *et al.* 2000, Sansano-Maestre *et al.* 2009). Infection by *Trichomonas* usually causes deposits in the mouth lining which inhibit feeding and cause breathing difficulties. This eventually leads to death either through secondary infection, starvation, or a greater vulnerability to predation (Hofle *et al.* 2004, Villanua *et al.* 2006b).

T. gallinarum, the *Trichomonas* strain that most commonly infects pheasants and red-legged partridges (Pennycott 1998) is genetically distinct from the parasite *T. gallinae* which causes trichomonosis and resulting adult and nestling mortality in finches (Lawson *et al.* 2012), doves and pigeons (Columbidae), including woodpigeon, collared dove, stock dove and most notably turtle dove (Pennycott 1998, Lennon *et al.* 2013, Stockdale *et al.* 2015). This suggests that direct parasite transmission between gamebirds and Columbidae may be unlikely (Lennon *et al.* 2013). However, Columbidae are more likely to be infected on farms providing supplementary food for gamebirds (Lennon *et al.* 2013), and *T. gallinae* has now been identified in both moribund red-legged partridge and turtle doves, which were using supplementary gamebird feed on the same site, suggesting that such parasite transmission is possible (Stockdale *et al.* 2015). Parasites "jumping" hosts from one clade to another has been hypothesised as the mechanism by which trichomonosis emerged in finches: by *T. gallinae* jumping from Columbidae to passerines (Lawson *et al.* 2012). Screening of gamebirds may therefore be worthwhile to establish whether parasite transmission is a possible occurrence at shared food resources such as game bird feeders or grain spills in farmyards (Stockdale *et al.* 2015).

Lyme disease

Pheasants may increase the levels of Lyme disease in the British countryside as they are competent reservoirs for the bacterium which causes the disease (*Borrelia burgdorferi* sensu lato) which is transferred to humans and other vertebrates via a tick vector. Despite several studies implicating rodents as the most frequent carriers of the bacterium (Humair *et al.* 1993, Sinski and Karbowiak 1994, Sinski *et al.* 2006), pheasants are 55% more effective at transmitting *B. burgdorferi* to ticks compared to wood mice and bank voles (Kurtenbach *et al.* 1998b). This suggests that pheasants are more prolific transmitters of *B. burgdorferi* (Kurtenbach *et al.* 1998a), and are important in maintaining the transmission of Lyme disease spirochaetes in lowland woods in the UK (Hoodless *et al.* 1998,

Kurtenbach *et al.* 1998a). It is important to note that deer and foxes are also important hosts for *B. burgdorferi*, and the growing populations of these species is also likely to lead to higher levels of Lyme disease in the British countryside (Hudson 1997). This is highlighted by (Hoodless *et al.* 1998) who found that the exclusion of roe deer from woodland by fencing remained an effective means of reducing tick densities despite the fact that pheasants had unrestricted access to such areas.

The disease stage of the bacterium's life cycle is currently only known to occur in humans (see section 5.1.6) and in some cases cattle. Gryczynska *et al.* (2004) tested passerines for *B. burgdorferi* in Poland and detected the bacterium in 4.2% (53 of 1254) of the total birds caught, comprising 21% (9 of 42) of species tested. *B. burgdorferi* was found in 21% of tree pipits, 16% of dunnock, 13% of chaffinch, 9% of song thrush, 8% of nuthatch, 8% of hawfinch, 5% of robin, 4% of blackbird and 4% of wren. While there has been little investigation regarding the effect of *B. burgdorferi* infection in passerines, Gryczynska *et al.* (2004) recaptured just one infected bird (1.8%), compared to a recapture rate of 8.2% (99) in uninfected birds. They therefore suggest that infected birds may be subject to higher mortality rates than uninfected birds. The prevalence of Lyme disease has not been extensively tested in other wildlife populations, although Ginsberg (1994) suggest that it may pose a significant threat.

Newcastle disease

Newcastle disease virus, or avian paramyxovirus type 1, is a notifiable disease that can result in mortality of entire flocks and is commonly associated with open-air commercially reared poultry including gamebirds (Gortazar *et al.* 2007, Alexander 2009). There is considerable variation in the clinical signs of the disease in different species (Alexander 2009). Pheasants of any age can be infected by many different strains of Newcastle disease, but the clinical signs and levels of mortality in infected birds vary considerably (Aldous and Alexander 2008, Aldous *et al.* 2010). This means that pheasants can act as carriers, contracting the virus without any mortal effects, with potential for transmission to wild birds (Aldous and Alexander 2008, Aldous *et al.* 2010). It is highly probable that all wild bird species are susceptible to infection, as a review in the 1980s recorded at least 241 species of birds, representing 27 of the 50 orders of the class Aves, where Newcastle disease infections had been detected (Kaleta and Baldauf 1988), although the outcome of the infection may vary considerably between species (Alexander 2009).

In 2005, 9,000 pheasants were culled following an outbreak of Newcastle disease on a game estate in Surrey. The birds were sourced from a game farm in France which was later confirmed to be the origin of the outbreak. This prompted a temporary ban on live bird imports into the UK, and pheasants and poultry are now routinely vaccinated against Newcastle disease (Avery 2019).

Avian influenza

Avian influenza is another notifiable viral disease associated largely with poultry and waterfowl. Avian influenza has resulted in mortality of pheasants in UK rearing facilities, and is occasionally detected in wild pheasants across Europe (only one tested case in the UK), although testing of wild birds post-release is rare (Avery 2019, European Food Safety Authority *et al.* 2019). Pheasants, partridges and other captive-reared gamebirds have moderate likelihoods of carrying avian influenza viruses (H5 and H7 influenza A strains) sub-clinically with no symptoms (Hillman *et al.* 2019). There is therefore potential for mutation of these low-pathogenicity strains to high-pathogenicity strains, which could result in severe illness or mortality, and subsequent transmission to wild birds, mammals and humans post-release (Hillman *et al.* 2019).

Salmonella spp., Escherichia coli

Passerines (particularly finches) are commonly infected by various *Salmonella* strains, and often come into close proximity with gamebirds at feeders. Pennycott and Duncan (1999) suggest that gamebirds represent a source for the spread of *Salmonella pullorum* despite the disease being controlled in the poultry industry:

"…this infected semi-wild population would form a reservoir of infection for young pheasants produced by *S. pullorum*-free breeding flocks, and even if no clinical disease were apparent in the young birds, the release and subsequent catching up of these birds could allow infected birds to be reintroduced to the breeding pens… Additional problems include the widespread movement of pheasants and pheasant eggs on a local, national, and international basis, and the practice of custom hatching in which eggs from several different sites are incubated in a common hatchery and the chicks are then redistributed. The elimination of *S. pullorum* from gamebird flocks would therefore require substantial changes to be made to the current breeding, incubation, rearing and releasing practices."

Díaz-Sánchez *et al.* (2012a) also found *Escherichia coli* (*E. coli*), an avian and human pathogen, to be at much higher prevalence in farmed-reared and released red-legged partridge in Spain compared to wild red-legged partridges, where these pathogens were practically absent.

The impacts of these diseases on wild bird populations has however not been studied.

Respiratory diseases

Reared pheasants are susceptible to respiratory diseases (predominantly sinusitis and infectious bronchitis) both before and after release (Madden and Sage 2020). Two types of pathogen are primarily responsible for these diseases: *Mycoplasma gallisepticum* and multiple coronavirus strains, the latter of which can also lead to kidney disease in wild gamebirds (Lister *et al.* 1985, Cavanagh *et al.* 2002, Welchman *et al.* 2002, Draycott 2013). *M. gallisepticum* may be one of the most important pathogens of poultry and wild birds globally, and has been confirmed by studies conducted in Belgium, Spain, USA and Japan as infecting 8 bird species present in the UK (mallard, woodpigeon, magpie, grey heron, peregrine falcon, house sparrow, starling and tree sparrow; Sawicka *et al.* 2020). *M. gallisepticum* was also detected in adult rooks from gamebird rearing sites in the UK (Pennycott *et al.* 2005), and it is possible that transmission occurred when gamebirds and corvids came into close contact when feeding. The impact of infection on these species and whether it may lead to demographic or population-level impacts is unknown, although a study of 53 corvids on a British gamebird site were pheasants experienced persistent sinusitis problems found that 50% of rooks, 38% of crows and 13% of jackdaws were infected with *M. gallisepticum* with limited deleterious outward signs, indicating that in corvids at least infection by *M. gallisepticum* may be subclinical (Bradbury *et al.* 2000a, Pennycott *et al.* 2005).

4.6.4   Key knowledge gaps and recommendations

- The source of many pathogens are not well understood, and in many cases studies have not comprehensively tested the pathogen links between gamebirds and other wildlife. A community-level study comparing the parasite and pathogen burdens and any consequential impacts on breeding success and survival for a range of sedentary farmland birds at different distances from gamebird release sites, or on sites releasing different gamebird densities, would be useful.

- No known studies have investigated population impacts of disease carried by gamebirds on native birds in the UK.

- It is important to understand whether pheasant and red-legged partridges released on the moorland fringe habitat influence disease of red and black grouse.

- The potential effects of medications on non-target species such as other birds and invertebrates has not been investigated and has been identified as a key knowledge gap that would benefit from further research.

4.7   Impacts on predators and predation

4.7.1   Impact summary

There was a relatively small amount of evidence (19 scores evidenced by 17 sources) for impacts of gamebird release on predators and predation. The ecological impact scores associated with effects on predators and predation as a result of gamebird release were significantly negative on average ($z = -3.11$, $P = 0.011$; Fig. 12), with no positive impact scores associated with any of the secondary impacts themes (Table 2, Table 23). These scores reflect potential increases in predation pressure through positive impacts on predator abundance as a result of gamebird release, the effects of which are likely to be negative for prey species and hence scored negatively (see section 3). Much of the evidence under this theme is indirect however, with only 24% of scores based on directly studied impacts (Table 23). There is therefore an urgent need for further research on this topic.

Table 23. Summary ecological impact scores associated with impacts on predators and predation dynamics, including the distribution of scores at each score level, the estimated marginal mean score and 95% confidence limits (95% CL) returned by the Ordinal Logistic Model (OLM; only presented for the primary theme overall as sample sizes precluded OLM analysis for secondary themes), the median and interquartile range (IQR) for secondary themes, and the proportion of scores which originated from peer-reviewed research and for which there was direct evidence of an impact rather than a potential impact.

	Ecological impact score						
	-2	-1	0	1	2	Mean	95% CL
Impacts on predators and predation	1	15	3	0	0	-1.71	-3.15 – -0.26
Secondary theme	-2	-1	0	1	2	Median	IQR
Food source for predators	0	11	1	0	0	-1	-1 – -1
Predator abundance	1	3	0	0	0	-1	-1.25 – -1
Predation rates	0	1	2	0	0	0	-0.5 – 0

19 scores evidenced by 17 sources*: 71% peer-reviewed, 24% direct evidence
* (Macdonald 1980, Hill 1985, Reynolds and Tapper 1986, Reynolds and Tapper 1995, Swann and Etheridge 1995, Kenward *et al.* 2001, Baker *et al.* 2006a, Callegari 2006b, Post Office 2008, Turner 2008, Rice 2016, Madden and Perkins 2017, Roos *et al.* 2018, Sage *et al.* 2018, Aebischer 2019b, Pringle *et al.* 2019, Swan *et al.* 2020a)

Gamebirds represent a considerable prey base which supplements natural prey in the UK landscape. They are eaten by a wide range of UK predators (Table 24), and a substantial proportion of the gamebirds released do get predated (Table 25). Gamebirds also present a source of carrion when killed in other ways (vehicle collisions, disease). The abundance of key generalist predators – foxes and carrion crows – is particularly high in the UK relative to other European countries where gamebird releasing does not occur at the same scale or intensity, and the abundance of avian predators is increasing (Roos *et al.* 2018). Populations of ground-nesting birds are limited by predation, particularly by foxes and carrion crows, and it is likely that the high and increasing predator abundance is partly responsible (Roos *et al.* 2018, McMahon *et al.* 2020). A recent large-scale study has demonstrated positive spatial associations between the abundance of gamebirds and both the abundance and growth rates of avian predators and scavengers (corvids and buzzard; Pringle *et al.* 2019). Further research is needed to determine whether there are similar associations or impacts on mammalian predators, particularly foxes. No studies have yet investigated whether predation rates of native UK wildlife are

related to the intensity of local gamebird release (via a mechanism of enhanced predator abundance) in a rigorous, experimental way.

4.7.2   Background

There are three primary mechanisms by which gamebird release and management of their habitats may affect predator abundance and predation rates:

- Firstly, released gamebirds and those breeding in the wild may act as a supplementary food source for predators and scavengers. Readily available and abundant supplementary food may saturate the local food web and increase the carrying capacity of the environment, which may allow predator abundance to increase beyond natural limits (Robb *et al.* 2008, Newsome *et al.* 2015, Ainsworth *et al.* 2016). This mechanism is discussed below within the 'Food source for predators and scavengers' section (4.7.3).
- Secondly, beneficial management of semi-natural habitats on gamebird shooting estates may also enhance the environmental carrying capacity for predator populations by increasing the availability of suitable habitat and natural prey. Management of arable farmland and woodland for the benefit of released pheasants and partridges confers many benefits for native prey species (see section 4.2) meaning that they may be more abundance on such sites than in the surrounding countryside. This in turn may provide a source of prey for predator populations outside the period of gamebird release, when gamebirds are less available as a supplementary food source, potentially increasing predator fitness, breeding success and abundance on these sites.
- Finally, lethal predator control on game estates may act to reduce the abundance of some predators that can be legally culled (foxes, stoats, weasels, some corvids, and buzzards under specific licenses on individual sites). Such lethal control is local in scale and restricted to individual game estates and is also often seasonal with control activities concentrated during autumn and winter when gamebirds are released. This mechanism was previously discussed in section 4.2.7.

4.7.3   Food source for predators and scavengers

The naturalised breeding population of pheasants and red-legged partridges combined represents a biomass of at least 4,059 tonnes, according to abundance estimates from 2016: see section 2.1.1 (Blackburn and Gaston 2018, Woodward *et al.* 2020). The annual release of at least 57 million gamebirds equates to an additional 44,340 tonnes (Aebischer 2019b). At the point of release in autumn, the combined biomass of naturalised and released pheasants and red-legged partridges therefore probably exceeds an estimated 48,399 tonnes. This quantity will decrease as a result of shooting mortality (approximately 34% of the gamebirds released; Aebischer 2019b), but still represents a considerable unnatural addition to the biomass of prey potentially available to predators and scavengers in the UK countryside during July–August and beyond.

Table 24. Example sources from the peer-reviewed and grey-literature presenting evidence that gamebirds (pheasant and red-legged partridge) form at least some part of the diet of predator species in the UK. For each source we list (where available) the predator species studied, the importance of gamebirds in the diet, the season, study period and geographical location. Sources marked with * were identified by the systematic literature search; those marked with ** contributed an ecological impact score under the 'Food source for predators' secondary impact theme (see Methods). To increase the ease of comparison between studies, the importance (proportion) of gamebirds in predator diet is arbitrarily classified as Low (< 10%), Moderate (10–50%) and High (> 50%).

Source	Predator	Importance and evidence of gamebirds in predator diet	Season	Study period and location
**Macdonald (1980)	Fox	**Low–Moderate**: Gamebirds comprised 5.5% of the diet of foxes in the breeding season, rising to 10% during August releasing	Breeding & Release (~Apr–Aug)	1970s; Releasing site in Oxfordshire, central England
**Reynolds and Tapper (1986)	Fox	**Moderate**: Gamebirds comprised 21.5% by volume of the diet of foxes	Annual	1980s; Releasing sites in Dorset-Hampshire border, southern England
*Reynolds and Tapper (1995), Reynolds and Aebischer (1991)	Fox	**Moderate**: Gamebirds comprised 16.5% of fox diet	Annual	1985-1987; Releasing sites in Dorset, southern England. Prey type ID from fox scat.
**Baker *et al.* (2006a)	Fox	**Low–Moderate**: Wild galliformes (mostly pheasant) comprised 5% of the diet of adult foxes by mass of prey ingested. Cubs ingested 17% wild galliformes.	Adults: Annual average Cubs: Breeding (Mar–May)	1995-1996; Non-release site with release occurring on adjacent land in southern England. Prey type ID from fox scat.
McDonald *et al.* (2000)	Stoat	**Low**: Galliformes comprised 3.9% of identifiable prey items.	Annual average	1995–1997; Releasing sites across UK
**Swann and Etheridge (1995)	Buzzard	**High**: Remains of juvenile pheasants found at ~56% of 212 nests in areas where pheasant release occurred	Breeding (Apr–Jul)	1977–1989; multiple sites across two regions in north Scotland
**Kenward *et al.* (2001)	Buzzard	**Low**: Pheasant remains found in 15% of 40 buzzard nests (found on 7% of 91 nest monitoring visits). Pheasants comprised 2.6% of 233 prey remains found at nests.	Breeding (May–Jul)	1990–1995; Releasing site in Dorset, southern England
*Swan (2017), **Swan *et al.* (2020a)	Buzzard	**Low**: Pheasants comprised 4.9% (15.3% of the biomass) of 82 identifiable prey provisioned by adult buzzards to chicks in 20 nests. Pheasants were one of the 6 most important prey categories.	Breeding (Jun–Jul)	2015; multiple sites, Cornwall, southern England
Davis and Davis (1981)	Red kite	**Low**: Gamebirds "scarce" in prey remains at nest sites (present at 2 of 64 nests).	Breeding (Apr–Jul)	1975–1979; multiple sites, Wales
Carter and Grice (2002)	Red kite	**Moderate**: Pheasants identified as the target prey item in 12% of 117 field hunting observations in autumn/winter, and w identified in 10% of 346 pellets collected from winter roosts. Gamebirds accounted for 25% of 191 food items remaining at 9 nest sites, and in 11% of 183 pellets from monitored nests.	Autumn/Winter: Release & Shooting (~Aug–Feb) Nests: Breeding (~Apr–Jul)	Late 1990s; multiple sites, central England
Wildman *et al.* (1998)	Red kite	**Moderate**: Galliformes (pheasants) present in 15% of 40 pellets collected from roost sites in winter. Pheasants were prey items at 21% of nests in 1995.	Winter: Shooting (Jan–Apr) Nests: Breeding (~Apr–Jul)	1994–1995; multiple sites, northern Scotland
RSPB unpublished data[1]	Marsh harrier	**Moderate**: Gamebirds comprised 12% of 1924 identifiable prey items provisioned to 8 nests monitored with cameras.	Breeding (Jun–Jul)	2013-15; Non-release sites with release occurring on adjacent land eastern England

[1] Upcott, E., Bagguley, E., Cooke, S., Drewitt, A. L., Gooch, H., Meadows, M., O'Dowd, R., Smart, J. & Mason, L. R. (*in prep*) The impact of marsh harriers on wetland breeding waders.

Gamebirds, particularly pheasants which are released in the highest numbers, are exploited as a food source by mammalian and avian predators and scavengers throughout the year in the UK (Table 24). The relative importance of gamebirds in predator diets varies between species, individuals and sites however, and is influenced by the method of study, the season and availability of other prey in the environment (Table 24). In all cases, gamebirds appear to supplement predator diets even when not forming a high proportion of dietary intake (Table 24), which in many cases is dominated by more-ubiquitous native prey types such as rabbits and rodents (Reynolds and Tapper 1995, McDonald *et al.* 2000, Baker *et al.* 2006a, FERA 2012, Swan 2017). The dependence on gamebirds as a prey source may fluctuate if the availability of natural and gamebird prey types also fluctuates, but the majority of sources we found examined predator diet in the breeding season, by which time the abundance of released gamebirds present in the environment will be at an annual low (Table 24). Reynolds and Tapper (1995) found no seasonal variation in the proportion of gamebirds in fox diet however, indicating that similar numbers are predated across the seasons despite high density gamebird release in autumn; while McDonald *et al.* (2000) found that the proportion of gamebirds in the diet of stoats was highest in Autumn.

Table 25. Example sources from the peer-reviewed and grey-literature presenting evidence of predation as a major cause of mortality in released and wild-breeding populations of released gamebirds (primarily pheasants). For each source we list where available, the percentage of the monitored gamebirds that were predated and the type of predator species identified (or assumed), along with the season within which the predation occurred, and the gamebird release status of the study sites (NR = no gamebird release, R = gamebird release, A = no gamebird release on site but release occurring on adjacent sites). Evidence from two additional sources which investigated the impact of annual predation by foxes on spring pheasant biomass is also summarised. Sources marked with ** were identified by the systematic literature search and contributed an ecological impact score under the 'Food source for predators' secondary impact theme (see Methods). All studies were based in Southern or Eastern England.

Source	% predated	Predator	Season	Site	Study period
**Hill (1985)	61% of 23 Pheasants	Unknown	Breeding (Apr–July)	NR	1970-1983
**Kenward *et al.* (2001)	9.5% of 20,725 Pheasants	3.2% Fox 4.3% Buzzard 2% Other	Release (July–Oct)	R	1994–1995
**Turner (2008), Sage *et al.* (2018)	35.1% of 486 Pheasants	Fox	Release & Shooting (July–Feb)	R	2001-2003
**Sage *et al.* (2018)	49% of 707 Pheasants	Fox	Breeding (Mar–July)	R+A	1992-2013
	48% of 451 Pheasant nests	Fox			
Draycott (2013)	16–26% of 50 Pheasants	Fox and Stoat	Breeding (Mar–July)	R+A	2011–2012

Effects of fox predation on spring pheasant biomass:

Source	Evidence	Site	Study period
**Baker *et al.* (2006a)	Annual predation on pheasants by foxes equivalent to 34-81% of the estimated spring biomass	A	1995–1996
**Reynolds and Tapper (1986)	Foxes predated the equivalent of 75% of the total spring population of gamebirds on the study area	R	1980s

119

The importance of gamebirds in predator diets is generally higher for foxes than it is for raptors (buzzard, red kite, marsh harrier; Table 24). This is further supported by the considerable losses of gamebirds to fox predation post-release and during the breeding season (Table 25), while raptors such as buzzards have a relatively small impact on the numbers predated on the majority of sites (Kenward *et al.* 2001, FERA 2012). With the exception of shooting, predation is considered to be the major cause of mortality of released pheasants (Sage 2018a), although the exact proportions predated vary between sites and seasons: Table 25 (Sage *et al.* 2018). The effectiveness of legal predator control of foxes, mustelids and corvids does however influence the proportion of released birds predated, with higher-level control (regularly checked trap networks for smaller mammalian predators as well as fox and corvid control) resulting in a reduction in absolute predation of released pheasants by ~29% on average (average 59% predation under low-level control compared to 30% predation under high-level control; Sage *et al.* 2018).

The maximum proportion of released pheasants either predated or scavenged by foxes was 35%: Table 25 (49% predation by foxes was recorded for breeding pheasants; Turner 2008, Sage *et al.* 2018). If this figure is representative of the numbers predated across all UK pheasant releasing sites, this suggests that 16.5 million of the 47 million pheasants released annually may fall prey to foxes. 16.5 million pheasants represent around 14,000 tonnes, which is the entire annual food requirement for almost 100,000 adult foxes if pheasants were their sole food source (assuming 850g per individual pheasant, and an average annual adult fox food consumption of 140.5 kg: Baker *et al.* 2006a, Blackburn and Gaston 2018). If 5–21.5% of fox diet consists of pheasants (the range in proportions of gamebirds recorded in fox diet from the studies identified by this review: Table 24), then 16.5 million pheasants might therefore contribute to supporting between 460,000 and 2 million foxes. The pheasant surplus could therefore support the entire fox population of the UK, which is currently estimated at between 357,000 and 430,515 individuals (The Mammal Society 2013, Mathews *et al.* 2018).

Foxes are attracted to pheasant release pens by the high densities of available prey contained within (Robertson 1988). This is not necessarily the case with raptors however, which may predate gamebirds more opportunistically, although the evidence base is mixed. Callegari (2006b) found that buzzard abundance increased in areas around gamebird release pens in autumn and winter (October–March), while Mrlik and Koubek (1992) found that raptors did not concentrate in areas where pheasants were released, and Kenward *et al.* (2001) found that only 8% of buzzards that had a release pen within their home-range visited them frequently. Occasionally significant predation by raptors can occur, and pen characteristics may strongly influence predation by birds of prey (Kenward *et al.* 2001). Pheasant predation by buzzards is more likely in large pens with lower shrub cover, with deciduous canopies providing perching sites (Kenward *et al.* 2001), but in general buzzards may be more likely to scavenge dead gamebirds than kill them.

The high proportions of released gamebirds that are predated (Table 25) may in part be due to the rearing process, with artificially reared birds lacking predator evasion skills which would otherwise be gained from parental influence or experience of life in the wild (Musil and Connelly 2009, Robertson *et al.* 2017, Madden *et al.* 2018). Birds reared in farms do not survive as well in the wild as naturalised gamebird populations (Madden *et al.* 2018, Sage *et al.* 2018). Improvements in early-life, pre-release management could help develop the natural predator avoidance behaviours of reared gamebirds and reduce the numbers that are predated and therefore contribute to the supplementary food available for predators (Whiteside 2015, Whiteside *et al.* 2015, Whiteside *et al.* 2016, Madden *et al.* 2018, Hall *et al.* 2019).

Other sources of gamebird mortality include 5–13% of released pheasants killed through collisions with road vehicles, particularly between September and March (Turner and Sage 2004, Post Office 2008, Turner 2008, Madden and Perkins 2017, Roos *et al.* 2018). Vehicle collision mortality has increased in recent decades with 5.6 times as many pheasants killed on roads in the 2010s compared to the 1960s

reflecting the recent increase in large-scale releasing (Madden and Perkins 2017). Patterns of road mortality are seasonal and linked with periods of release: Madden and Perkins (2017) report peaks in pheasant road mortality corresponding with periods of release (autumn) as well as late winter/early spring possibly corresponding to the cessation of supplementary feeding after the shooting season, and male pheasants prospecting for breeding territories. Draycott (2013) and Turner (2008) found that 4–35% of released pheasants may also die as a result of disease post-release. Pain *et al.* (2019a) also estimate that 0.558% of pheasants die from lead poisoning annually, based on a study by Butler *et al.* (2005), equating to an estimated total 294,560 of those released (using updated estimates on the numbers released from Aebischer (2019b); see sections 4.3.4 and 5.1.2). If these proportions reported dying from vehicle collisions, disease and lead poisoning are representative, then it is possible that 4.5–22.8 million of the 47 million pheasants released each year may die in these ways and equate to 3,818–19,399 tonnes of available carrion.

The potential importance of gamebirds as a food source for predators and scavengers should therefore not be overlooked, particularly as availability peaks during the autumn and winter when other natural prey may be more scarce or harder to find. Thus, gamebird releasing has the potential to support predator populations through periods when food-related mortality may otherwise have restricted populations, and when body condition may be key for survival and breeding success the following spring (Robb *et al.* 2008, Newsome *et al.* 2015). For example, gamebird releasing in July/August coincides with the period when fox cubs are becoming independent and learning to hunt for themselves prior to dispersal (Lloyd 1980). Likewise, foxes mate in later winter and give birth in early spring (Lloyd 1980), so food availability in autumn and early winter may be crucial for vixen body condition during gestation and lactation, and subsequently influence the successful raising of large litters.

4.7.4   Impacts of gamebird release on predator abundance

Gamebirds at high densities may increase the numbers of predators because a food rich environment raises the equilibrium population that the landscape can support (Gibbons *et al.* 2007). Artificially abundant prey sources are therefore expected to enable predator populations to expand, particularly (as is the case in the UK) where predators have few direct competitors or predators of their own (e.g. Newsome *et al.* 2015). Other factors may also drive high and increasing predator abundance, including anthropogenic land-use change, a reduction in human persecution, recent re-introduction programmes or meso-predator release through the lack of apex predators (e.g. Newsome *et al.* 2017). The provision of gamebirds as supplementary food is likely to play a key role in exacerbating these other factors. However, the impact that released gamebirds as a source of supplementary food source may be having on the abundance of predators has rarely been directly measured so most of the evidence is either correlational or indirect.

Local behavioural impacts

Predator abundance increases locally at the point of gamebird release, with some predators anticipating the release of gamebirds and changing their behaviour to take advantage of seasonal pulses in this supplementary food source. The effects of attracting predators to release areas tend to be transitory and short-lived however (Kenward *et al.* 2001, Callegari 2006b). Foxes in particular are attracted to release pens: Robertson (1988) found that pheasant mortality was far higher immediately after release in Ireland, and suggested this may be a consequence of the high densities of pheasants within release pens attracting foxes prior to release. Once pheasants started to disperse away from the pen and densities decreased, predation pressure from foxes also decreased. Callegari (2006b) also found that buzzards were attracted to high density pheasant release areas at the time of releasing on one chalk grassland site, although in contrast Kenward *et al.* (2001) found that only a minority of buzzards associated with pheasant release pens resulting in heavy predation being restricted to a handful of sites

where pen characteristics and release factors may have made it easier for individual buzzards to kill pheasants.

Population-level impacts: mammalian predators

There is little direct evidence that the local behavioural responses of foxes to gamebird releases are translated into population level effects. However, foxes are thought to be more numerous in the UK than natural ecological systems would usually permit, with approximately 357,000–430,515 foxes in Britain at a density higher than in most other European countries (The Mammal Society 2013, Mathews *et al.* 2018, Roos *et al.* 2018, Sainsbury *et al.* 2019). It is plausible that this is partly linked to extensive releases of gamebirds, given that the scale of UK releasing greatly exceeds that anywhere else in Europe or North America (Arroyo and Beja 2002, Mustin *et al.* 2012). The density of foxes is also highest in England relative to the other UK countries, which is where the density of gamebird releasing is highest [refer to figures in Background].

The number of foxes killed on shooting estates from the National Gamebag Census (NGC), a proxy for fox abundance, increased by 203% between the 1960s when large-scale gamebird releasing first became common, and 2010: Fig. 23 (Aebischer *et al.* 2011)[33]. Since the 1990s, fox abundance has continued to rise in England (where gamebird releasing is most common) but has declined in Scotland, resulting in stability of numbers at the UK scale (Aebischer *et al.* 2011). The BTO's Breeding Bird Survey (BBS), the second primary source of data on UK fox population trends, suggests that abundance declined by 44% between 1996 and 2018 (Sainsbury *et al.* 2019, Harris *et al.* 2020). However, the BBS trend confidence intervals incorporate almost completely the NGC confidence intervals, suggesting that the two surveys may have measured the same underlying trend (Aebischer *et al.* 2011). The NGC trend may also be a more reliable measure of fox abundance, being based primarily on numbers of foxes killed at night when foxes are most active, compared to BBS data which is dependent on ad-hoc sightings during the day.

The NGC is the only source of data on national trends in stoat populations due to a lack of national (or even regional) surveys. The UK NGC index for stoat doubled between 1961 and 2009 (Aebischer *et al.* 2011), and it is possible that this is partly due to increases in the availability of released-gamebirds as prey, although most sources suggest that rabbit population changes have been most influential (Aebischer *et al.* 2011, Sainsbury *et al.* 2019, Harris *et al.* 2020).

There are no studies which directly link the abundance of mammalian predators to gamebird release activities on a national scale, although Porteus *et al.* (2019) have shown that fox populations can be locally supressed as a result of legal lethal control on game releasing estates (see section 4.2.7). Fox abundance on some game shooting estates may therefore be expected to be low. However, such lethal control is only effective when conducted persistently at a high level of effort, as culled foxes are rapidly replaced by immigrant individuals (e.g. Baker *et al.* 2006b, Porteus and McAllister 2018, Porteus *et al.* 2019). This implies that fox abundance in the wider countryside is likely to be high relative to the sites where culling occurs (Baker *et al.* 2006a). Gamebirds disperse away from the point of release, although typically not more than a few kilometres (Madden *et al.* 2018), so are likely to represent a supplementary food source for foxes in the wider countryside, outside the immediate release vicinity. Foxes themselves are also wide ranging, and will typically hold territories of 2.7–5.2km^2 over multiple landownership units (Reynolds and Tapper 1995, O'Mahony *et al.* 1999); gamebird release on one site may therefore be expected to elevate fox density over a wider area., and may be driving the high and potentially increasing abundance of foxes in the wider countryside, even if local fox abundance on game releasing estates themselves remain low.

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[33] http://www.gwct.org.uk/ngcmammals
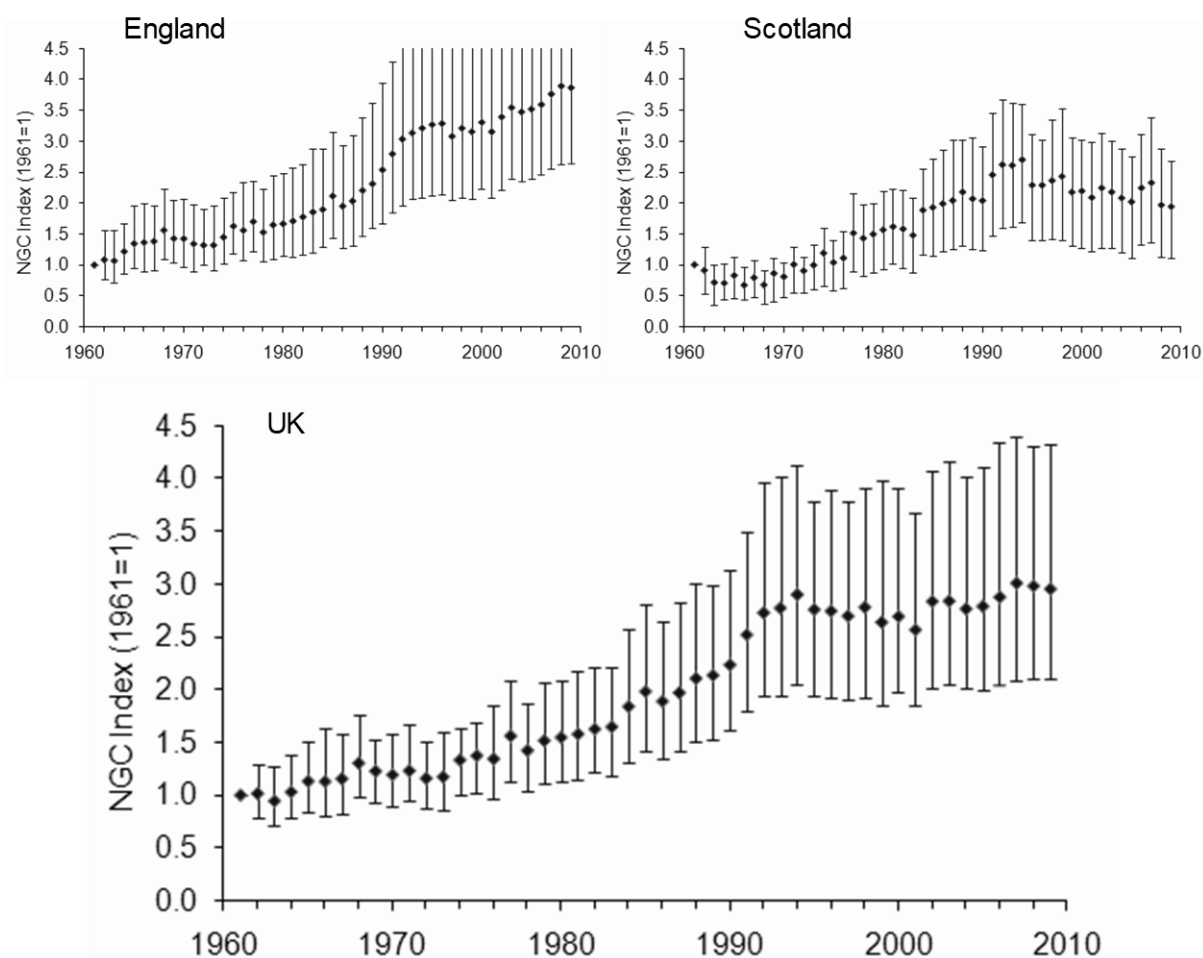
Fig. 23. National Gamebag Census (NGC) index for the red fox in England, Scotland and the whole of the UK from 1961 to 2009. Values are indices of the numbers killed per unit area on game estates, with index values in 1961 given an arbitrary value of 1. Error bars represent 95% confidence intervals. Reproduced from the Game and Wildlife Conservation Trust (Aebischer *et al.* 2011)[33].

Population-level impacts: Avian predators

As with foxes, the abundance of carrion crow (considered to be a common predator and scavenger of released gamebirds) is also higher in the UK than anywhere else in Europe (Roos *et al.* 2018). UK crow abundance has increased by 99% between 1970–2014 (Hayhow *et al.* 2017, Robertson *et al.* 2017, Roos *et al.* 2018). Similarly, the abundance of raptors whose distributions overlap the geographic spread of gamebird release activities and for which gamebirds or gamebird carrion are likely to form at least some part of their diet (buzzard, red kite, marsh harrier, goshawk: Table 24, Kenward 1977) have also increased (buzzard by 454% 1970–2014, red kite by 1,738% 1995–2018, marsh harrier by 479% 1985/89–2011/15, goshawk by 274% 1985/89–2011/15; Arraut *et al.* 2015, Hayhow *et al.* 2017, Roos *et al.* 2018, Harris *et al.* 2020).

A recent study tested for spatio-temporal associations between gamebird abundance/releases and the abundance of avian predators and scavengers across lowland Britain (Pringle *et al.* 2019). This study found an overall pattern of positive correlations between the abundance of released gamebirds and the abundance and/or abundance growth rates of buzzard, carrion crow, magpie, raven and jay (Table 26), indicating that these avian predators are generally more abundant, or have become more abundant, in areas where more gamebirds have been released. Gamebird releasing may therefore be an important driver of abundance, and abundance change, of avian predators and scavengers in lowland Britain.

Table 26. Summary of the spatial and temporal associations found by Pringle *et al.* (2019) between avian predator abundance, and abundance change, and gamebird abundance (both numbers of reared gamebirds providing a proxy for the numbers released, and the naturalised abundance of gamebirds the following breeding season which is closely determined by number released the previous autumn). Associations are from three different analyses numbered as in Pringle *et al.* (2019). Positive correlations are indicated by '+' and light grey shading, negative by '-' and dark grey shading, quadratic relationships where predator abundance was generally higher at higher gamebird abundance or biomass are indicated by 'U'. The relationship between red-legged partridge and crow in (3) was quadratic but essentially positive, hence is indicated by $+^U$. Only significant relationships ($p < 0.05$) are shown; blank cells indicate no significant associations. Similar patterns of associations were found for winter data (Pringle *et al.* 2019).

		Buzzard	Crow	Magpie	Raven	Jay
Breeding season predator abundance vs. number of reared (released) gamebirds (2)	Pheasant	+		+	U	+
	Red-legged partridge				U	+
Breeding season predator abundance vs. abundance or biomass of free-roaming gamebirds in the breeding season (3)	Pheasant	+	+	-		+
	Red-legged partridge	+	$+^U$	-		
	Total gamebird biomass	+	+	-		+
Predator abundance change vs. abundance or biomass of free-roaming gamebirds (4)	Pheasant		+		+	
	Red-legged partridge	+	+		+	+
	Total gamebird biomass		+		+	

The inclusion of jay by Pringle *et al.* (2019) has been questioned, with a concern that jay may not be truly influenced by gamebirds as a prey source. Jays are unlikely to kill gamebirds themselves, but may scavenge carrion from gamebird carcasses. Holyoak (1968) found little evidence that carrion featured in the diet of jays relative to other UK corvid species, but a more-recent study by Selva *et al.* (2005) found that jays scavenged on 44% of carcasses in woodland in Poland. The associations found for magpie are also inconsistent with those for the other species studied (Table 26). Pringle *et al.* (2019) suggest that this could be an artefact of lethal predator control depressing magpie numbers as part of the management for intensive gamebird releasing on shooting estates, and therefore a secondary impact of management for gamebirds on predators rather than the predators themselves.

The associations found by Pringle *et al.* (2019) are correlative and not necessarily causative, so other potential drivers of predator abundance need to be considered. Gamebird release areas tend to have a high proportion of semi-natural habitats managed to be beneficial for gamebirds (e.g. Firbank 1999, Duckworth *et al.* 2003, Oldfield *et al.* 2003), and this may provide improved nesting and hunting opportunities for avian predators (see section 4.2). Gamebird estates also often provide supplementary seed and grain which may be taken by some predators. However, Pringle *et al.* (2019) allowed for these effects in their analyses by controlling for habitat type (including four land cover classes including woodland, pastoral, arable and urban) and for wider habitat quality (average abundance and diversity of 59 other bird species), after which the generally positive correlations between gamebird and predator abundance were still apparent (Table 26). While it is possible that these relationships are not driven by the nutritional benefits of abundant gamebird prey to the predator species, the findings of Pringle *et al.* (2019) are suggestive of such a mechanism and highlight the need for further detailed investigation.

4.7.5   Predation rates: impacts on native prey species

Increases in the abundance of avian and mammalian predators could have implications for native UK prey species on a local scale, but few studies have directly investigated the potential links between gamebird release, predator abundance and the impacts on predation rates of native species.

Predators tend to concentrate on the most available prey source (Ferrari and Weber 1995, Lanszki *et al.* 2007, Delibes-Mateos *et al.* 2008). At the end of the shooting season (ends 1st February) when gamebird abundance has declined, it is possible that predators temporarily switch to other sources of prey and this coincides with the spring bird breeding season. Elevated predator abundance, caused by high densities of gamebirds, may therefore cause an imbalance in typical predator-prey relationships, resulting in increased predation of native birds or their nests, and this may negate any positive effects of predator control on game estates.

Predation is a key factor limiting populations of ground-nesting waders, seabirds and gamebirds themselves in the UK and more widely (MacDonald and Bolton 2008, Roos *et al.* 2018, Kämmerle and Storch 2019, McMahon *et al.* 2020), and many studies of lethal control or experimental removal of predators in multiple habitats have had positive impacts on birds (Smith *et al.* 2010); see section 4.2.7. In lowland farmland areas the experimental removal of predators in addition to beneficial habitat management has had positive impacts on breeding farmland passerines at small spatial scales (Stoate and Szczur 2001b, Donald *et al.* 2002, Stoate 2002, Stoate 2004, Stoate 2005, Stoate 2006, Stoate and Szczur 2006, Stoate 2007, Stoate *et al.* 2008, White *et al.* 2008, Stoate *et al.* 2009, White *et al.* 2014, Aebischer *et al.* 2016). In the absence of beneficial habitat management, predator control alone is predicted to produce a 166% increase in grey partridge equilibrium density (Aebischer and Ewald 2004), and increase grey partridge breeding densities almost threefold (2.8 times) when conducted in addition to supplementary food provision (Aebischer and Ewald 2010). Grey partridge productivity also increased by up to 3.5 times over three years as a result of lethal predator control (Reynolds *et al.* 1993, Tapper *et al.* 1996).

In other habitats Fletcher *et al.* (2010) found that the breeding abundance of lapwing, golden plover, curlew and red grouse declined by more than 17% annually on average in the absence of predator control on moorland, while the control of foxes, crows, stoats and weasels led to an increase in breeding abundance of more than a 14% on average. Reductions in foxes and crows also increased the breeding success of lapwing, golden plover, curlew, red grouse and meadow pipit by an average of three times (Fletcher *et al.* 2010). Curlew abundance, density and population change is also strongly positively associated with pheasant abundance and gamekeeper density on moorland, a relationship which is likely to be driven by predator control reducing fox and crow abundance on pheasant releasing or gamekeeper-managed sites (Douglas *et al.* 2014, Franks *et al.* 2017). Similarly, where predator densities are high on lowland wet grassland, lethal control of foxes and crows results in an increase in lapwing nest survival (Bolton *et al.* 2007), and mammalian predator exclusion results in substantial increases in lapwing nest survival and productivity (Malpas *et al.* 2013).

These studies illustrate that where predator abundance is high, such as when predator control or exclusion is not conducted (as is the case in the majority of the UK countryside; Baker *et al.* 2006b), negative impacts on bird populations occur as a result of higher predation rates. Elsewhere in Europe, high predator abundance as a result of supplementary increases in the amount of food available to predators have been linked to increased nest predation (Selva *et al.* 2014, Newsome *et al.* 2015), and researchers in Poland have found that carrion in woodland attracts predators and results in local reductions in prey species populations (Cortés-Avizanda *et al.* 2009). The provision of gamebirds as supplementary food may therefore have similar effects on predator-prey relationships.

Many of the studies above involved the control of both mammalian (usually foxes, stoats and weasels) and avian predators (usually corvids). However, a number of recent studies and reviews suggest that there is limited evidence that corvid predation limits bird population size (although it often does limit breeding success; MacDonald and Bolton 2008, Newson *et al.* 2010b, Madden *et al.* 2015, Roos *et al.* 2018), while there is growing evidence suggesting that mammalian predation (particularly by foxes) may limit both breeding success and population size of ground nesting birds (Roos *et al.* 2018, McMahon *et al.* 2020). The impacts of gamebird release on the abundance and predation associated with mammalian predators may therefore be of most concern.

However, Callegari (2006b) found no differences in total abundance, total species number or avian diversity between gamebird release and control non-release areas, which was hypothesised to indicate no effect of differences in predation pressure between these areas. Similarly, Reynolds and Tapper (1995) found no evidence on their study sites that increasing the autumn density of pheasants by releasing reared birds increased predation pressure by foxes on a wild-breeding population of grey partridge. These contrasting results suggest that further targeted research is urgently needed to disentangle these effects.

4.7.6   Key knowledge gaps and recommendations

- No studies have yet quantified the impacts of released gamebirds on the abundance of generalist mammalian or avian predators in an experimental or comparative way (although at least one correlative study has shown a link between avian predators and gamebird release), particularly whether the provision of released gamebirds as a supplementary food source may be increasing the environmental carrying capacity for these predators. The impacts on mammalian predators (especially foxes) are of particular concern.

- There is an urgent need for new field studies to test whether releases of gamebirds (1) enhance the local abundance of generalist predators (birds and mammals), and (2) lead to increases in predation pressure on ground-nesting birds by increasing the abundance of generalist predators.

5   SOCIO-ECONOMIC IMPACTS

5.1   Socio-economic results overview

Source publication year and authorship

We identified 28 sources which provided evidence for socio-economic impacts of gamebird releasing or shooting on humans in the UK (or where data from the UK were included in European or global studies) published between 1988 and 2020, 96% of which were published in the last three decades (1990–2020), and 64% published since 2010 (Fig. 24). We only include sources which were identified during our systematic literature review (see section 3).
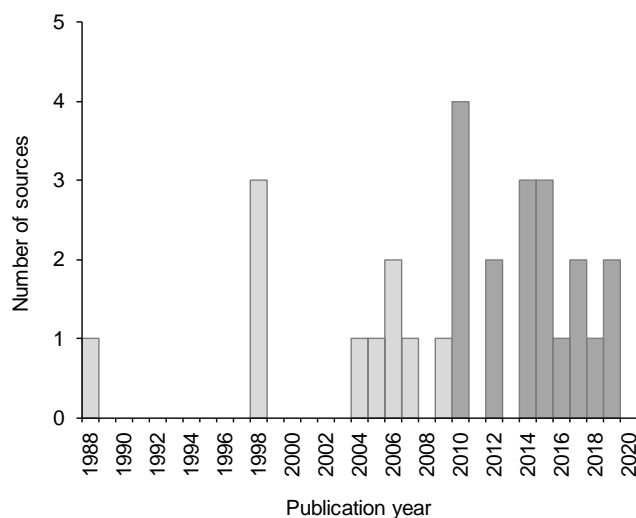
Fig. 24. The temporal distribution of sources which evidenced socio-economic impacts of gamebird release in the UK. Light grey bars indicated sources published up to 2010; dark grey bars are sources published 2010–2020.
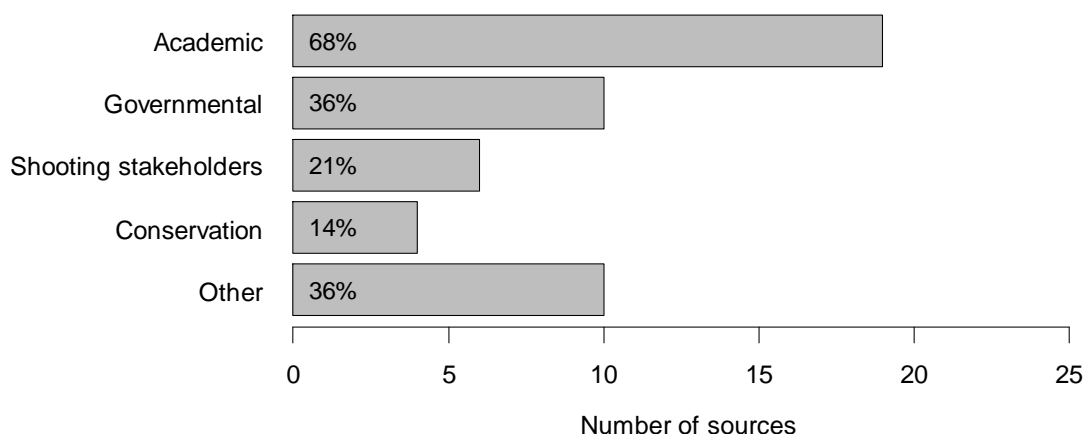
Fig. 25. Author affiliations for the 28 sources which provided evidence for socio-economic impacts of gamebird releasing or shooting on humans in the UK. Bars indicate the number of sources authored by at least one author from an organisation with a stake in the shooting industry ('Shooting stakeholders', e.g. GWCT, BASC), an academic institution ('Academic', e.g. university or other research institute), conservation organisation with no stake in the shooting industry ('Conservation', e.g. BTO, Wildfowl & Wetlands Trust), governmental department or advisory body ('Governmental', e.g. DEFRA, Natural England, Scottish Natural Heritage), or other affiliation ('Other', e.g. consultancies, unaffiliated individuals). Percentages indicate the proportion of the 28 sources with at least one author from each affiliation type; these do not sum to 100% as many sources had multiple authors with different affiliations.

Over two thirds (68%) of the scored sources were authored by at least one author affiliated with an academic institution. Authors from government departments and advisory bodies, and those affiliated with consultancies ('Other') contributed to 36% of sources, while authors affiliated with an organisation with a stake in the shooting industry contributed to 21%. Authors affiliated with conservation organisations with no stake in the shooting industry contributed to 14% of sources (Fig. 25).

5.1.1   Impact summary

The 'vote counting' exercise for socio-economic impacts associated with gamebird release indicates that there is more evidence for negative socio-economic impacts than for benefits: with 27 sources indicating negative effects, 1 source indicating a benign effect, and 10 sources indicating positive effects (Table 3, Table 27). There is no doubt that gamebird release is valuable economically, socially and provides employment opportunities to those supporting or directly involved in the shooting industry; the available published peer-reviewed and grey literature sources highlighting these benefits are relatively few in number, however. In contrast, the bulk of the published literature surrounding socio-economic impacts focuses on the impacts relating to lead consumption by humans, the potential for disease transmission and vehicle collisions, which are all inherently negative. Vote counting does not attempt to account for the relative magnitude of the impact evidence, nor the 'importance' of impacts to human society which is difficult to compare objectively across different impacts.

Table 27. The number of cases from literature sources providing evidence for positive, negative or benign socio-economic impacts of gamebird release, and the proportion of sources which were peer-reviewed research and for which there was direct evidence of an impact rather than a potential impact.

	Impact direction		
	Negative	Benign	Positive
Socio-economic impacts	27	1	10
Secondary theme			
Economic value	1	0	2
Employment	0	0	2
Social-cohesion and wellbeing for the shooting community	0	0	2
Lead consumption in humans	11	1	0
Disease transmission to humans	8	0	0
Vehicle and aviation accidents caused by gamebirds	6	0	0

Evidence from 28 sources*: 45% of votes peer-reviewed, 74% direct evidence

* (Hill and Robertson 1988, Hoodless *et al.* 1998, Kurtenbach *et al.* 1998a, Kurtenbach *et al.* 1998b, Turner and Sage 2004, Coburn *et al.* 2005, PACEC 2006, Langbein 2007, Post Office 2008, Chandramouli *et al.* 2009, EFSA 2010, Pain *et al.* 2010, Quy 2010, Williams *et al.* 2010, FSA 2012, Green and Pain 2012, Horigan *et al.* 2014, PACEC 2014, Taylor *et al.* 2014, Cromie *et al.* 2015, Green and Pain 2015, Lead Ammunition Group 2015, Rice 2016, Madden and Perkins 2017, Seguino and Chintoan-Uta 2017, Seguino *et al.* 2018, Green and Pain 2019, Hillman *et al.* 2019)

Very few sources attempted to separate the socio-economic impacts relating to gamebird releasing from those associated with other forms of shooting, so it is impossible to assess the economic value, contribution to employment or other socio-economic impact of gamebird releasing specifically. All shooting activities combined are likely to contribute substantially to the rural economy, and particularly to the community involved in shooting however. The exact value of the input from the shooting industry to the national UK economy is contested but may also be considerable. The UK shooting industry also supports a significant number of jobs, although again the exact figures are contested. Surveys of shooting participants suggest that if all UK shooting activities were to cease, many shooting participants would travel abroad to shoot and spend considerably less on alternative UK leisure activities, suggesting there would be an overall loss of leisure income to the UK economy. Some arable farmers do however report losing profit due to crop damage from pheasants, and the costs associated with the loss of pheasants and red-legged partridges to lead poisoning, and with road vehicle and aviation collisions, may reduce some of the economic benefits.

An estimated 430,000 people participated in some form of driven or walked-up game shooting in 2012/13, primarily males over the age of 40. Shooting is undeniably important for the social wellbeing of this demographic group, and their perception is that it improves the social cohesion of their local area. The risk of human exposure to lead from undetectable ammunition fragments in gamebird meat is however particularly high for this shooting community, with high-level consumption (>1-2 gamebird meals per week) linked to reductions in IQ and writing ability in children, and high systolic blood pressure, chronic kidney disease and high rates of spontaneous abortion in adults. This may also be a health risk for wider society, particularly in people who regularly consume pheasant and red-legged partridge meat.

There is also a risk to humans (although low) from zoonotic disease carried and transmitted by gamebirds, particularly Lyme disease and potentially also Avian Influenza, but also gastrointestinal and other diseases. Again, the risk is likely to be greater for those involved in the rearing, shooting and regular consumption of pheasants and red-legged partridges, but also has the potential to affect wider countryside users. Approximately 2.4–6.1 million pheasants may be killed on the roads each year, with numerous road traffic accidents reportedly caused by pheasants annually, some of which result in serious human injury or fatalities. Pheasants are also estimated to cost the British aviation industry at least £300,000 in damage annually; at least 1/3 of the total cost associated with bird strikes.

5.1.2   Economic value

Contributions by the shooting industry to the rural and national economies: Gross Value Added (GVA), Gross Output (GO) and contribution to Gross Domestic Product (GDP)

During focus groups assessing the opinions amongst shooting industry stakeholders (gamekeepers, beaters and pickers-up, shoot owners, and shooters), the financial input of pheasant shooting to rural communities was considered to be one of its primary benefits (Greenall 2007). If commercial gamebird shoots were to cease, they believe that there could be serious repercussions for the rural economy, affecting many social groups (Greenall 2007). One third of shooting participants reported that they would travel abroad to shoot and spend considerably less on alternative UK leisure activities, suggesting there would be an overall loss of leisure income to the UK economy (PACEC 2014). Natural Resources Wales concluded that shooting represented a positive contribution to local economies in Wales after an evidence review and public consultation in 2017 (NRW 2018). The effect of gamebird releasing and shooting on the rural economy is therefore likely to be positive, although no published literature details the contribution of shooting to the rural economy in a quantifiable way.

There are only two reports which summarise the national economic contribution of the UK shooting industry: PACEC (2006) based on representative survey data from 2004/5, and the more recent PACEC (2014) based on survey data from 2012/13. Both 'PACEC reports' summarise the results of a survey of the shooting industry repeated in different time periods and conducted on behalf of UK shooting and countryside organisations. Many of the values presented in these reports represent all UK shooting types including clay pigeon and target shooting, as well as all types of live quarry and pest-control shooting (PACEC 2006, PACEC 2014). Pheasant and red-legged partridge shooting is included, but is not examined separately. Some figures are associated with 'driven game' (PACEC 2006, PACEC 2014), which is likely to include the shooting of pheasants and red-legged partridges, as well as red grouse and some forms of waterfowl shooting. Occasionally, 'driven grouse' and 'other driven game (including duck)', the latter of which is assumed to primarily refer to pheasant and red-legged partridge shooting, are split (PACEC 2014).

PACEC (2014) state that approximately £2 billion Gross Value Added (GVA) in 2012/13 was attributable to the UK shooting industry as a whole. In perspective, this equates to approximately 17% of the GVA for the agriculture, forestry and fishing sector (£11.979 billion according to the Office for National Statistics[34]), 0.13% of the total UK GVA for 2013 (£1,590.569 billion), and 2% relative to the economic output associated with nature conservation and the natural environment (over £27.5 billion; RSPB 2011). This represents an increase of £0.4 billion since 2004/5, when GVA attributable to the shooting industry was calculated as £1.6 billion (PACEC 2006).

The PACEC reports were critically reviewed by academic economic specialists Paul Cormack and Professor Ian Rotherham of Sheffield Hallam University in 2014, commissioned by the League Against Cruel Sports (Martin 2012, Cormack and Rotherham 2014). Cormack and Rotherham (2014) sought evaluations from a range of experts in addition to their own, concluding that the PACEC reports represent a "substantial piece of research" which are "impressive as evidence-based advocacy statements" (Cormack and Rotherham 2014). Cormack and Rotherham (2014) however highlight some significant faults in the methods used to generate the values presented in the PACEC reports and attempt to provide some alternative values based on the data that PACEC present. Their main criticisms of PACEC include: a) guidelines from the Office for National Statistics on the inclusion of gross profits and losses were not followed in the estimation of GVA; b) there is a lack of transparency in the way in which calculations were undertaken; c) PACEC economic estimates ignore the existence and input of land-based subsidies available to the shooting industry (e.g. agri-environment payments); d) PACEC

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[34]https://www.ons.gov.uk/economy/grossvalueaddedgva/datasets/nominalandrealregionalgrossvalueaddedbalancedbyindustry

include an estimate of jobs supported or attributable to the industry in their economic calculation, which is not considered to be a valid economic category, e) much of the economic data included by PACEC is associated with alternative industries (e.g. transport, accommodation) and 'second round' expenditure (procurement of goods and services from the supply chain on the behalf of the shooting industry) which should not have been included in industry-specific calculations (Cormack and Rotherham 2014). Cormack and Rotherham (2014) therefore concluded that although the basic assertions made by PACEC – that sport shooting has a significant impact on the economy and communities involved – cannot be disputed, the exact findings of PACEC should be not taken at face value.

Using data presented by PACEC (2014), Cormack and Rotherham (2014) estimate that Gross Output (GO) from the shooting industry based on participant expenditure or provider income is between £887 million and £1.1 billion respectively (although both include values that are likely to have been double counted). The £887 million GO figure is based on £860 million of direct expenditure by participants on shoots and £27 million for the purchase of game. There are also indirect expenditures totalling a further £1,610 million in 2014, but most of this expenditure is not technically part of the sporting shooting industry itself (Cormack and Rotherham 2014). Cormack and Rotherham (2014) estimate the GVA attributable to the shooting industry as between £267 million and £480 million (depending on whether participant spending or provider income are used respectively), calculated at market prices (what the consumer pays) and not accounting for indirect taxes levied on the industry (which are not discussed in the report and are not available through HMRC) which would reduce the figures further. Gross Domestic Product (GDP) would not be more than the £267 million (Cormack and Rotherham 2014).

Neither the conclusions of the PACEC reports nor the critique by Cormack and Rotherham (2014) may be entirely unbiased; the former highly influenced by the shooting industry, the latter commissioned by an animal welfare organisation.

Profits and turnover associated with gamebird releasing and shooting

The financial costs of producing a commercial shoot advertising large bags of pheasant and red-legged partridge are considerable if sufficient gamebirds are to be produced (Greenall 2007). The cost of eggs and poults are very similar for pheasant and red-legged partridge (Steel and Draycott 2014); in 2005 eggs were around £0.40 each, day-old chicks costed between £0.60 and £1.00, and six- to seven-week old poults were £3.50–£4.00 (Bicknell *et al.* 2010). In 2018/2019 the average pheasant poult cost £3.75 (Teanby *et al.* 2019), and the average total cost per bird "put down" (i.e. released) was £14.13, which includes all the costs incurred during rearing, feeding, gamekeeper salaries and shooting (Teanby *et al.* 2019). As only a proportion of the birds 'put down' are shot, the cost per bird shot was higher, averaging £39.24 (Teanby *et al.* 2019). In comparison, the average income per bird shot on shoot days was £36.92 in 2018/19, so many shoots may make a loss (PACEC 2006, Greenall 2007, Teanby *et al.* 2019).

Many of the pheasants and red-legged partridges released in the UK originate from eggs and chicks bred in France and imported (Canning 2005, Rutley 2019, Madden and Sage 2020), and 80% of shot birds may be exported back to France for human consumption (Canning 2005), although game is now more widely consumed in the UK. To the food market, a pheasant or red-legged partridge is worth relatively little, and prices have halved in recent years: prices for shot game fell from £0.50–£0.60 to £0.25 per bird between 2011/12 and 2017/18 for both pheasant and red-legged partridge (Steel *et al.* 2018). The increase in numbers of birds released has created an imbalance between game meat supply and demand, with game dealers taking only 48% of shot game on average (Steel *et al.* 2018). As a result, some shoots are unable to sell the shot birds, and in 2017/18, 46% of shoots were supplying their game dealer free of charge and 12% were paying the game dealer to collect them (Steel *et al.* 2018).

Costs to shooters (called 'guns') vary, but the most frequent price for a day's shooting was £401–£600 in 2004–2007 (27.5% of guns), ranging from less than £100 (2.5% of guns) to more than £1,000 (7.5% of guns). The average pheasant bag size (number of pheasant shot) per day's shooting reported by shooting industry stakeholders 2004–2007 was approximately 200 birds (Greenall 2007), equating to an income per day's shooting of £7,384 at current market prices (Teanby *et al.* 2019). Of the UK shooting providers (not restricted to gamebird shooting) surveyed by PACEC (2014), only 16% reported making any profit from their shooting operation, while 62% reported their shooting operation broke even, and 22% reported making a loss.

Other economic impacts of gamebird releasing and shooting

The evidence above suggests largely positive economic impacts from the shooting industry, including gamebird releasing and shooting, on the rural and national UK economy, although the magnitude of this impact is disputed. There are however a number of negative economic impacts associated with pheasant and red-legged partridge releasing. For example, gamebirds may damage crops with a resulting loss of profits for arable farmers (Rice 2016). On Jersey (where pheasants are naturalised but no longer released), 33% of farmers believe pheasants cause severe damage to their crops and 27% believe pheasants cause a substantial loss of profit on their yield of brassicas, cereals, fruit and potatoes (Rice 2016). Monocot plant leaves form a very high proportion of pheasant diets (45–87% depending on the availability of supplementary cereal grain feed; Hoodless *et al.* 2001), much of which may be winter-sown wheat and other cereal shoots in areas where pheasant releasing and arable farming overlap (Hill and Robertson 1988). There are also considerable costs associated with road and aviation collisions with pheasants (Williams *et al.* 2010); see section 5.1.7.

Lead poisoning from gunshot ingestion (see section 5.1.5 and 4.3.4) may result in the loss of 0.558% of pheasants and 0.323% of red-legged partridges annually (Butler 2005, Butler *et al.* 2005, Pain *et al.* 2019a). Pain *et al.* (2019a) estimated the replacement cost of these affected pheasants and red-legged partridges to be approximately £3.164 million (€3.451 million) in the UK, based on numbers of gamebirds released in 2006, breeding gamebird population estimates from 2009 and production costs of gamebirds from 2016/17. Using the same method, updated using the numbers of gamebirds released in 2016 (Aebischer 2019b), breeding population estimates from 2016 (Woodward *et al.* 2020) and total overall costs per bird put down in 2018/19 (Teanby *et al.* 2019), we estimate the cost of replacement to the UK shooting industry from lead-poisoning related mortality of pheasants and red-legged partridges to be between £4.390 million and £4.531 million annually (Table 28).

Table 28. Estimated replacement costs of pheasants and red-legged partridges affected by lead-poisoning related mortality from ammunition sources in the UK, using the method presented by Pain *et al.* (2019a) updated using more recent published figures.

	% estimated as dying	Population (individuals)			Number of birds estimated as dying	Cost per bird	Cost of replacement
		Released [c]	Breeding	Total			
Pheasant	0.558 [a]	47,000,000	2,800,000–4,600,000 [d]	49,800,000–51,600,000	277,884–287,928	£14.13 [f]	£3.927 million–£4.068 million
Red-legged partridge	0.323 [b]	10,000,000	145,000 [e]	10,145,000	32,768	£14.13 [f]	£0.463 million
Total							£4.390 million–£4.531 million

[a] Butler *et al.* (2005); [b] Butler (2005); [c] Aebischer (2019b); [d] Pheasant breeding population lower estimate based on 2,300,000 females in 2016 from Woodward *et al.* (2020) and assuming a sex ratio of 1:4.6 males to females as used by Pain *et al.* (2019a); upper estimate based on a potentially more accurate pheasant sex ratio of 1:1, as many males remain unmated in the breeding season, despite others holding large harems of females according to Hill and Robertson (1988). [e] Red-legged partridge breeding population estimate based on 72,500 territories in 2016 from Woodward *et al.* (2020), assuming one pair on each territory. [f] Total overall cost per pheasant released in 2018/19 (£14.13) reported by Teanby *et al.* (2019) assumed to be similar for red-legged partridge.

5.1.3   Employment

As with economic impacts (section 5.1.2), the PACEC reports (PACEC 2006, PACEC 2014) are also the main available source of information on employment associated with shooting in the UK. PACEC (2014) suggests that the shooting industry supported a total of 74,000 full-time equivalent (FTE) jobs in the UK in 2012/13, 35,000 of which were directly linked to shooting (e.g. gamekeepers, beaters, shoot managers, accommodation staff, etc.) and 39,000 were indirectly linked (e.g. suppliers, "downstream" jobs such as game dealers and craftsmen, and "rest of supply chain" jobs including those associated with hospitality, gun dog welfare and travel; PACEC 2014). This is an increase in employment of approximately 4,000 FTE jobs since 2004/5 (PACEC 2006). For perspective, 74,000 FTE jobs equates to approximately 11% (those employed directly) and 13% (those employed indirectly) of people employed in the UK Agriculture, Forestry and Fishing sector (estimated at 308,000 people according to the UK Office for National Statistics[35]), 0.25% (direct and indirect employment combined) of all employed people in the UK in April–June 2013 (29,861,000 people), and 21% relative to the number of FTE jobs supported by nature conservation and the natural environment (750,000 FTE; RSPB 2011). In reality the 74,000 FTE total employment figure reported by PACEC (2014) cross-cuts a number of inter-related employment sectors including Agriculture, Forestry and Fishing, Accommodation and Food Services, and Leisure and Tourism for example[2,36].

Many of the paid jobs associated with shooting are seasonal or part-time (e.g. 51% and 48% of gamekeepers responding to a national survey in 2011 and 2019 were 'amateur' or part-time; National Gamekeepers' Organisation 2011, Ewald and Gibbs 2020), so up to 350,000 people may actually benefit from some form of employment in the shooting industry annually (PACEC 2014). Much of the direct employment associated with shooting activities is of benefit to the local community, with 84% of shooting providers' employees living locally (within 10 miles; PACEC 2014), and 64% of shooting providers stating that all (100%) of their staff live locally (PACEC 2014). 81% of the shooting community

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[35]https://www.ons.gov.uk/employmentandlabourmarket/peopleinwork/employmentandemployeetypes/datasets/employmentbyindustryemp13

[36]https://www.ons.gov.uk/peoplepopulationandcommunity/leisureandtourism/articles/tourismemploymentsummaries/characteristicsoftourismindustries2014

believes that shooting contributes to local employment and skills (PACEC 2014). 84% of shooting industry jobs are based in England, 12% in Scotland, 3% in Wales and 1% in Northern Ireland (PACEC 2014).

However, as with the economic figures presented in the PACEC reports, the employment figures above have been questioned by Cormack and Rotherham (2014). Cormack and Rotherham (2014) highlight that the PACEC employment figures may have been inflated by the inclusion of some items which are inappropriate, particularly the inclusion of some more ambiguous first round suppliers under the indirect employment category (Cormack and Rotherham 2014). They suggest that in addition to the direct jobs provided by the shooting industry (35,000 FTE), only indirect 'downstream' first round supplier jobs which are integral to the industry or could have been provided by the industry but have been outsourced should be included (equating to 2,152 FTE jobs including land management services, sales and marketing, membership, game farming, magazines, feed and fencing, track maintenance, crafts; PACEC 2014). According to Cormack and Rotherham (2014), employment where an unidentifiable quantity is linked to the shooting industry, but which could feasibly have taken place anyway, could also potentially be added (4,763 FTE jobs including firearms and ammunition, dog training, vet fees, vehicle maintenance; PACEC 2014). Any items which have their own separate existence (such as accommodation, food, travel, general goods and services, utilities and communication) should not have been included in industry-specific employment calculations however (8,875 FTE jobs according to PACEC 2014). Cormack and Rotherham (2014) recalculate the employment figures from PACEC based on these assertions, and arrive at an estimate of between 37,000 and 42,000 FTE jobs supported by the shooting industry as a whole; considerably lower than the 74,000 FTE jobs reported by PACEC (2014). Additionally, many of the direct jobs included by PACEC (2014) may not actually be treated by the industry as employment, often being poorly paid (below minimum wage) and organised on a casual basis (short-term employment of two weeks or less such as for beaters or pickers-up) without contracts or formal employment agreements (Cormack and Rotherham 2014, Strutt & Parker *et al.* 2017).

However, all of the employment estimates described above relate to the entire UK shooting industry, of which non-native gamebird shooting forms only a constituent part (55% of UK shooting providers offer driven lowland game shooting, which includes duck, pheasant and red-legged partridge as quarry species, but may also provide other types of shooting; PACEC 2014). We found no published literature stating the level of employment specifically associated with the release, shooting and management of non-native gamebirds in the UK.

5.1.4   Social cohesion and wellbeing for the shooting community

Approximately 430,000 people participated in some form of driven or walked-up game shooting in the UK in 2012/2013 (PACEC 2014), equivalent to 0.7% of the estimated UK population of 64 million in 2012/2013[37]. The majority of this shooting community are male (96%) and over the age of 40 (85%; PACEC 2014).

For this community, shooting is important for their social wellbeing and enjoyed as a traditional pastime with many social benefits; it is also perceived by them as important for the cohesion of the wider local community. 97% of shooting participants believe that shooting positively contributes to their wellbeing for example, and 87% of participants agree that shooting positively contributes to the social fabric of the local area (PACEC 2014).

However, we found no published studies that assessed the impacts of shooting on human wellbeing or other social factors from a wider representative sample of the population (i.e. including both shooters

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[37] www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationestimates/ datasets/populationestimatesforukenglandandwalesscotlandandnorthernireland

and non-shooters). The conclusion that shooting has a positive impact under this secondary theme is therefore restricted to the impacts on the shooting community and possibly may not apply to local host communities or wider society (Cormack and Rotherham 2014).

5.1.5   Lead consumption in humans

Lead is an environmentally polluting, toxic, non-essential metal that has no beneficial effects in living organisms, instead acting as metabolic poison which accumulates in living tissues (Green and Pain 2019, Pain *et al.* 2019b). Lead is toxic to humans, with lead exposure posing a substantial threat to human health, with no clinical threshold of lead that is considered 'safe' in the human body (Iqbal *et al.* 2009, Quy 2010). Lead was removed from water pipes, paint and petrol in the UK during the twentieth century, chiefly due to evidence of its negative effects on human health and the environment[38] (Green and Pain 2019, Jones *et al.* 2019). Shotgun cartridges containing large quantities of lead gunshot pellets are still the most common type of ammunition used for shooting gamebirds and other live quarry species in the UK (Pain *et al.* 2015). Ammunition-derived lead in game meat destined for human consumption is therefore one of the very few remaining routes of human exposure to high concentrations of lead and subsequent lead poisoning in the UK and across Europe (EFSA 2010, Green and Pain 2019).

Humans risk exposure to high lead concentrations when consuming meat from game animals (including pheasants and red-legged partridges) which were killed with lead gunshot, or which ingested lead during their lives and thus contain accumulated lead in their tissues. Concentrations of lead in gamebird meat is higher than in large game (e.g. deer), where the wounded parts of the carcass, and thus many of the embedded lead fragments, are removed during processing; this is more difficult in pheasant and red-legged partridge carcasses because of their smaller size and the smaller size of lead pellets which are more difficult to remove (FSA 2012). The concentrations of lead in game meat (including pheasants and partridges, but also including other game) is also very high relative to other food types (EFSA 2010).

Pain *et al.* (2010) sampled oven-ready pheasant and red-legged partridge meat from birds shot in the UK and purchased from UK supermarkets, game dealers and shoots, and found an average 3.32 lead gunshot per pheasant and 2.12 lead gunshot per red-legged partridge sold for human consumption. Lead gunshot pellets themselves are relatively easy to detect and remove prior to game-meat consumption, but Pain *et al.* (2010) used X-rays to show that a considerable quantity of lead fragments, possibly created as gunshot shatters on impact or passes through the bird, remain within game meat and are too small to be detected by the consumer: 82% of 22 pheasants and 65% of 26 red-legged partridges tested contained more than one of these small fragments of lead and 14% of pheasants contained more than 15. These fragments remain largely unchanged through cooking or result in lead leaching into cooking liquids and sauces (Pain *et al.* 2010). Human consumption of gamebird meat which was killed with lead gunshot is therefore almost certain to result in some level of lead exposure (EFSA 2010, Pain *et al.* 2010, Green and Pain 2015, Green and Pain 2019).

As described in section 4.3.4, pheasants and red-legged partridges also ingest lead gunshot which they mistake for dietary grit, and at least 3% of pheasants and 1.4% of red-legged partridges may have lead pellets in their digestive tracts and correspondingly elevated bone lead levels (Butler 2005, Butler *et al.* 2005). It is possible that the total proportion of gamebirds with an accumulation of lead in their tissues is higher than this, but undetected due to the short lifespan of lead pellets in the gizzard (Potts 2005, Mateo 2009); see section 4.3.4. Gamebirds that have ingested lead gunshot may also have an increased likelihood of being shot due to behavioural impacts of the lead on the birds themselves (Pain *et al.* 2019a). Although the gizzard and digestive tract are usually removed along with most of the other internal organs during butchery, it is possible that lead may be found accumulated within tissues

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[38] https://en.wikipedia.org/wiki/Lead-based_paint_in_the_United_Kingdom

destined for human consumption or retained during cooking, including the liver, kidneys and bones (e.g. when roasting or stewing a whole carcass), even in birds not shot with lead.

The consumption of lead-contaminated game meat has been linked to elevated lead levels in human blood and bone high enough to be associated with cardiovascular effects such as increases in systolic blood pressure, as well as chronic kidney disease, increases in spontaneous abortion in pregnant women and developmental neurotoxicity in children in the UK (Kosnett 2009, EFSA 2010, Green and Pain 2015, Green and Pain 2019). Exposure of children to relatively low lead levels from a young age is associated with a one point reduction in IQ and reduced educational attainment (e.g. lower Standard Assessment Tests writing scores), and an increase in antisocial behaviour and hyperactivity (Chandramouli *et al.* 2009, EFSA 2010, ACCLPP 2012, Green and Pain 2012, Green and Pain 2015, Green and Pain 2019).

However, the extent of these deleterious effects on human health are highly dependent on the rates of exposure, in turn determined by the frequency of consumption. There is a relatively low risk when consuming game meat irregularly (Coburn *et al.* 2003, Coburn *et al.* 2005, EFSA 2010, Quy 2010). However, 'high' consumption rates equating to <1 gamebird meal a week in children and 1.2–6.5 gamebird meals a week in adults are linked to the negative health and neurological impacts described above (EFSA 2010, Green and Pain 2012). An adult regularly eating pheasant shot with lead ammunition (two 100g portions a week) could increase their dietary exposure to lead by up to 8 times the background exposure from the rest of their diet. In children less than five years old, the dietary exposure could be up to 5 times the background exposure from two 30g portions a week (EFSA 2010, FSA 2012). Adults consuming 1.3–9.3 partridge meals (150g) or 6.5–10.7 pheasant meals (150g) per week would likely exceed the international Provisional Tolerable Weekly Intake (PTWI) of lead recommended by FAO/WHO (Pain *et al.* 2010). The proportion of the PTWI that would be accounted for by consumption of 88g of gamebird meat per day would be 50–294% for red-legged partridge or 45–67% for pheasant (Pain *et al.* 2010). Regular consumption is most common amongst the shooting community (shooters themselves, gamekeepers, estate managers, and their families), so this is the group most at risk from negative health impacts (FSA 2012, Green and Pain 2015, Green and Pain 2019). There is also a risk of consumption for a wider cross-section of society consuming shop-bought pheasant and red-legged partridge meat killed with lead gunshot (Green and Pain 2019), although the frequency of consumption, and therefore associated health risks, may be lower.

The prevalence of gamebird meat consumption may be decreasing in particularly at-risk groups, but 2.7% of women of childbearing age surveyed in the UK in 2008–2010 reported eating game birds at a mean intake of 19.5g per day (Taylor *et al.* 2014), and 77% of children under five consumed game meat in Scottish families where game meat consumption was higher than average in 2011 (FSA 2012). 27% of shooting providers (including those providing target or clay shooting as well as live quarry shooting) reported eating gamebirds weekly during the shooting season in 2012/13, and 97% of edible quarry shot in the UK in 2012/13 is estimated to have entered the food chain (PACEC 2014). Thousands of children and adults in the UK, particularly high level consumers from the shooting community, are therefore calculated to be at risk of consuming high enough levels of ammunition-derived lead from gamebird meat to potentially cause them physiological and neurodevelopmental harm (Green and Pain 2015, Lead Ammunition Group 2015). More are estimated to consume enough lead to potentially cause them a range of low level but nonetheless harmful health effects (Lead Ammunition Group 2015). Across Europe (including the UK), 5 million high-level consumers may be at risk of deleterious health effects from consuming lead in game meat (Green and Pain 2019).

UK sales of game meat have risen over time, with 31% growth in sales of feathered game between 2006 and 2009 (FSA 2012) and an increasing social trend towards the cooking and consumption of pheasant and red-legged partridge meat, as well as other game. The regular consumption of game

meat by an estimated 14% of the UK population according to the British Game Alliance[39], is therefore concerning if the animals were shot with lead. There has however been considerable progress in recent years in the development of lead-free ammunition, and Waitrose, possibly Britain's largest game retailer, announced it would not sell gamebirds shot with lead ammunition by the 2020/21 shooting season, with some catering outlets also no longer serving game shot with lead (Lead Ammunition Group 2020, Pain *et al.* 2020). There is still resistance to change from part of the shooting community however (Newth *et al.* 2019), with several sociological and political barriers combining to inhibit both compliance with existing regulations and a transition to wider use of non-toxic ammunition (Cromie *et al.* 2015, Newth *et al.* 2019). The threat of lead poisoning to humans from game meat consumption in the UK is unlikely to be fully resolved until Government action is taken to regulate or ban the sale and use of lead ammunition across all habitats and all forms of shooting (Pain *et al.* 2020).

5.1.6   Disease transmission to humans

Non-native gamebirds, similar to many native UK species, carry zoonotic diseases that can infect humans. Their incidence and associated human health risks are generally low but are notable particularly for more vulnerable members of the population such as infants and the immunosuppressed. Many have the potential to infect people involved in the rearing and hunting of released birds, and the storage, preparation and consumption of their meat, which is most likely to be those associated with the shooting community. People outside the shooting community could be affected when preparing and consuming gamebird meat, although the risk is generally low. Some diseases, such as Lyme disease, have the potential to affect any users of the countryside, so could impact on society more widely. These diseases may also infect other UK wildlife, so are discussed in more detail in section 4.6.

Lyme disease

Pheasants, in addition to wood mice, grey squirrels and deer species, are reservoirs for tick-borne Lyme disease in lowland woodlands (Hoodless *et al.* 1998, Kurtenbach *et al.* 1998a, Kurtenbach *et al.* 1998b) and may increase the risk of Lyme disease exposure in humans (Hoodless *et al.* 1998); see section 4.6. More than 50% of engorged ticks collected from pheasants in a woodland in southern England were infected with Lyme disease spirochetes (Kurtenbach *et al.* 1998b). Incidence is generally low, but laboratory confirmed cases of Lyme disease in humans have been increasing in frequency in England and Wales, with 1,579 laboratory confirmed cases in 2017 (a mean annual infection rate of 2.7 per 100,000 people) compared to 268 in 2001 (infection rate of 0.5 per 100,000 people; PHE 2018). An additional 1,000–2,000 cases in England and Wales are thought to be treated by the National Health Service without laboratory diagnosis annually (PHE 2018). Cases of Lyme disease have remained relatively stable in Scotland and Northern Ireland over the same time period (HPS 2016, PHA 2017, HPS 2018). Due to modern medicine, Lyme disease is rarely fatal, but can cause severe neurological, rheumatological, cardiac and other complications if left undetected or untreated (Dillon *et al.* 2010, Dubrey *et al.* 2014). Human exposure risk to Lyme disease is negatively correlated with vertebrate species diversity in north America (Ostfeld and Keesing 2000, Schmidt and Ostfeld 2001, Keesing *et al.* 2006), meaning that where few vertebrate species dominate, human Lyme disease incidence is higher. Therefore, where pheasants are released at high densities it seems plausible that Lyme disease may be more prevalent, although this hypothesis has not yet been tested in the UK.

Avian influenza

Gamebirds are also carriers of low-pathogenicity avian influenza, which could potentially mutate to a high-pathogenicity strain and pose a risk to humans. Avian influenza has resulted in mortality of

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[39] e.g. https://www.britishgamealliance.co.uk/bga-secures-new-markets-for-game-with-one-of-the-uks-largest-food-manufacturers/

pheasants in UK rearing facilities and has been detected in wild pheasants in the UK in at least one case, although testing of wild birds post-release is rare (Avery 2019, European Food Safety Authority *et al.* 2019). Pheasants, red-legged partridges and other captive-reared gamebirds have moderate likelihoods of carrying low-pathogenicity avian influenza viruses (H5 and H7 influenza A strains) sub-clinically with no symptoms, with gamebird holdings testing positive for these strains across Europe more than average when compared with other poultry types (positive detection on 1.02% of gamebird holdings tested compared with a median incidence of 0.34% across other types of poultry holding; Hillman *et al.* 2019). Similar low pathogenicity H7 strains have resulted in a relatively small number (>100) of cases of human infection since 2002 across the UK, Netherlands, Italy, Canada and the United States, usually originating from domestic poultry flocks, with clinical symptoms ranging from conjunctivitis and mild upper respiratory illness to pneumonia (Nguyen-Van-Tam *et al.* 2006, Belser *et al.* 2009). Although these H7 infections have resulted in a smaller proportion of hospitalisations and deaths in humans than those caused by the more well-known H5N1 (high pathogenicity) strain, some subtype H7 strains appear more adapted to human infection and transmission (Belser *et al.* 2009). Anyone regularly in contact with pheasants during rearing and release may therefore be at a low risk of contracting similar avian influenza strains. There is also potential for these low-pathogenicity strains to mutate to high-pathogenicity strains, with higher risks of humans infection resulting in severe illness or mortality (Hillman *et al.* 2019); if such a mutation occurred in pheasants post-release then the high density of these gamebirds in the UK environment as carriers of the disease would be of concern.

Other diseases

There is a low risk of human infection by *Campylobacter*, *Salmonella*, *E. coli*, *Toxoplasma gondii and Mycobacterium avium* from hunting, storage, preparation and consumption of gamebirds. *Campylobacter jejuni,* and *Campylobacter coli*, *Salmonella* spp. and *Escherichia coli* are among the most common bacterial food-borne human gastrointestinal diseases worldwide (Horigan *et al.* 2014, Seguino and Chintoan-Uta 2017, Seguino *et al.* 2018), to which there is a low risk of exposure from infected gamebirds (Coburn *et al.* 2005, Horigan *et al.* 2014, Seguino and Chintoan-Uta 2017, Seguino *et al.* 2018). Hunting, storage, preparation and consumption of red-legged partridge meat confers a low risk of infection from *Campylobacter, Salmonella* and *E. coli* in the UK for example (Horigan *et al.* 2014). *Campylobacter* was also found to be present in 37% of 287 pheasants sampled by Seguino and Chintoan-Uta (2017) and Seguino *et al.* (2018) in Scotland, indicating a similar potential risk to humans through the consumption of pheasant meat, although the overall risk to public health from pheasant meat is considered to be low (Seguino and Chintoan-Uta 2017, Seguino *et al.* 2018).

There is also a low risk of exposure to *Mycobacterium avium* from infected gamebirds, which causes infection in those who are immunocompromised or those with severe lung disease (Coburn *et al.* 2005). Similarly, there is a potential risk to humans from *Toxoplasma gondii* infection during the hunting, storage, preparation and consumption of red-legged partridge (Horigan *et al.* 2014). *Toxoplasma gondii* is an intracellular parasite that causes the infectious disease toxoplasmosis which may result in serious and occasionally fatal illness in infants and those with weakened immunity. It has few observable symptoms in healthy adults but has been associated with numerous subtle adverse or pathological human behavioural alterations (Flegr *et al.* 2014, Cook *et al.* 2015).

Antibiotic resistance

Antibiotics are widely used both prophylactically and to treat clinical illnesses during gamebird rearing and post-release supplemental feeding (Seguino and Chintoan-Uta 2017, UK-VARSS 2019). There is considerable concern nationally and globally that this and similar widespread indiscriminate use of antibiotics throughout the agricultural, veterinary and human health sectors might lead to increases in anti-microbial resistance in bacteria, with knock-on effects for human health (Natural England 2009, O'Neill 2015, O'Neill 2016).

Resistance of multiple bacterial strains to multiple antibiotic types long after their original antibiotic medication has been detected in samples from pheasants and red-legged partridges in Belgium and Spain (Devriese *et al.* 1996, Guerrero-Ramos *et al.* 2016), and LA-MRSA (livestock-associated methicillin-resistant *Staphylococcus aureus*) which is known to infect humans was reported in a pheasant in Scotland in 2017 (UK-VARSS 2019). Díaz-Sánchez *et al.* (2012b) found that antibiotic resistance in *E. coli* was much more frequent in farmed red-legged partridges (75%) compared to wild birds in Spain, meaning that with widespread indiscriminate use of antibiotics, farms rearing red-legged partridges for release could become a potential source of resistant *E. coli* in the environment. Seguino and Chintoan-Uta (2017) identified resistance to at least one type of antibiotic in *Campylobacter spp.* commonly isolated in clinical human infections in 42% of 287 pheasants sampled across five pheasant rearing facilities in five Scottish regions, with the prevalence of antimicrobial resistance varying from 22% to 89% between shooting estates. On the estate that had the highest level of antimicrobial resistance, 67% and 11% of samples also had double and triple antimicrobial resistance respectively (resistant to two or three different antibiotic groups; Seguino and Chintoan-Uta 2017). The risk of antimicrobial resistant human *Campylobacter* infection posed by pheasants may therefore be high within certain geographical areas (Seguino and Chintoan-Uta 2017). Further studies are required to better define the risk posed by antimicrobial resistant *Campylobacter* strains to humans, which have the potential for a major impact on a local scale (Seguino and Chintoan-Uta 2017, Seguino *et al.* 2018).

The risks of antibiotic resistance developing through the indiscriminate use of antibiotics in gamebird rearing practices are starting to diminish due to UK government, veterinary, shooting industry and stakeholder initiatives to reduce their usage (Department of Health 2013, Global and Public Health Group 2019); see section 4.6

5.1.7   Vehicle and aviation accidents caused by non-native gamebirds

Road vehicle collisions and aviation accidents caused by released gamebirds, particularly pheasants, are relatively common in the UK and can result in serious human injury and fatalities, and economic costs from insurance, damage and mitigation. Pheasants may be the most likely of all UK bird species to die on UK roads, accounting for 38% of all ad hoc sightings of roadkill wildlife reported by the UK public during 2013–2016 (Madden and Perkins 2017). 5.6 times as many pheasants were killed on UK roads in the 2010s compared to the 1960s (Madden and Perkins 2017), reflecting the recent increase in large-scale releasing (Fig. 2 in section 2.1.1). Annual peaks in pheasant road mortality correspond with the releasing period in autumn, with a further peak in late winter/early spring corresponding with the period when male pheasants are prospecting for territories (this is also the time when supplementary feeding of pheasants is stopped on many game estates after the shooting season and birds need to disperse to find food; Madden and Perkins 2017).

In studies of pheasant survival, the proportion killed on UK roads through collision with vehicles is estimated to be in the range of 5–13% (Hill and Robertson 1988, Turner and Sage 2004, Turner 2008). This equates to between 2.4 and 6.1 million pheasants killed on the road each year (assuming 47 million are released`: Aebischer 2019b), similar to the Post Office (2008) estimate of 3 million pheasants killed on the road annually. Pheasants are reported as causing 12% of road traffic accidents (87) across 14 English counties 1999–2003, 6% of which (5) resulted in human mortality or serious injury (Langbein 2007). The insurance and damage costs of these collisions are unknown but likely to be substantial.

Pheasants are estimated to cost the British aviation industry at least £300,000 in damage annually, based on evidence for at least 10 aviation bird strikes caused by pheasants in 2009 at an estimated cost of £30,000 per incidence (although the species causing aviation strikes could not be determined in 60% of cases, so the number of pheasant-related cases may be higher; Williams *et al.* 2010). This

equates to at least 1/3 of the total cost associated with non-native aviation bird strikes in the UK annually (estimated at £500,000; Williams *et al.* 2010). Gamebirds (pheasants, partridges and quails combined) are listed as one of the top 20 bird groups causing aviation strikes in the UK across 2012–2016, with 40–50 aviation strikes attributed to gamebirds during this period, although overall incidences of gamebird strikes were very low relative to other native species (e.g. gulls, swallows and martins, pigeons and doves: ~3000 strikes combined; CAA 2016).

5.1.8   Key knowledge gaps and recommendations

- No truly independent assessment of the economic impact of pheasant and red-legged partridge releasing and shooting in the UK has been conducted, so the potential benefits to the rural and national economy from this specific practice are difficult to quantify with confidence.

- We found no published studies that assess the impacts of shooting on human wellbeing or other social factors from a representative sample of the population, or specifically in relation to non-native gamebird shooting. A wider survey of people's perceptions of the social value of this type of shooting, conducted across a sample of different demographic and interest groups would be of benefit to determine the views of the wider UK society.

- An assessment of the human and economic costs of road vehicle collisions involving pheasants and red-legged partridges, perhaps through a compilation of UK insurance claims relating to gamebird collisions, would help to better indicate the scale of the impact of gamebirds on UK road users.

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