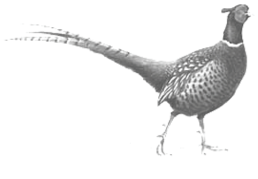
Impacts of non-native gamebird release in the UK: a review

RSPB Research Report Number 40



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Objectives of this review

This review aims to investigate the extent and impacts of non-native gamebird release in the UK, and subsequently draw attention to areas for conservation concern, and identify areas of research priority. This review also highlights the lack of substantial knowledge regarding the larger scale impacts of gamebird release in the UK. While evidence points to several potential mechanisms that may cause population level responses of wildlife, these have not been thoroughly investigated. Here, potential mechanisms are therefore explored, and available evidence are presented.

Research methods for this review

Data compilation for this review involved literature searches, particularly scientific publications, and contacting the various organisations of interest to this review. At the Game and Wildlife Conservation Trust (GWCT), Dr. Rufus Sage provided feedback, and contact was made within the RSPB, the BTO (British Trust for Ornithology), Avon Wildlife Trust, Buglife and Butterfly Conservation. In most cases, impact studies are conducted on pheasant and not red-legged partridge, and therefore many of the discussions in this review are derived from information only available for pheasants. In many cases impacts associated with pheasants are likely to be similar for red-legged partridge because their ecology is largely comparable. However, red-legged partridge are smaller, and therefore require lower levels of certain resources, and the scale of red-legged partridge release is substantially smaller than pheasant release, and therefore pheasants are of primary interest to this review.

Executive summary

The impacts of gamebirds and gamebird shooting practices span multiple disciplines, but few have been extensively investigated. The data available show that at high densities of gamebird release, negative environmental impacts are likely to occur, and may in some cases be severe. In the majority of cases however, where densities are moderate, it is likely that impacts are minor or may be offset by beneficial habitat management. In areas where good habitat management is combined with low release densities, or in areas that work to promote breeding populations of gamebirds, impacts may be largely positive.

Game estate management

- 35 million pheasants and 6.5 million red-legged partridge are released each year in the UK.
- Game estate habitat management includes woodland sky-lighting, planting cover-crops, conservation headlands, and more.
- It is likely that game estate management, including woodland and farmland habitat management, provision of supplemental food, and predator control increases the numbers of some bird groups, particularly warblers, finches and ground feeders.
- Estate management also benefits some small mammals, particularly wood mice and bank voles.

Direct impacts of gamebirds

- Gamebirds, mostly pheasants, modify woodland ground flora within release pens, through browsing and soil enrichment.
- Pheasants at high densities can modify hedgerow and hedgebank floral structure, and this may have knock-on effects for hedge nesting birds.
- Pheasants reduce the biomass of overwintering ground-active invertebrates and caterpillars that are important food resources for breeding birds.
- Breeding gamebirds may compete with native birds for invertebrate resources.

- Gamebirds on moorland fringe habitat threaten rare and endangered bryophyte communities, and may impact on red grouse and other fragile moorland bird species.

Gamebirds and disease

- Pheasants may spread numerous parasites to wild birds, particularly at feeders.

Shooting practices

- Birds of prey and other animals suffer lead poisoning following the consumption of gamebirds that are shot but not collected.
- Spent lead shot on game estates is ingested by some birds, leading to poisoning at sufficient concentrations.
- Lead shot in the environment may escalate the food chain from soil invertebrates to small mammals to predators.
- Unintentional by-catch of grey partridge results in population impacts on this declining species.

Impacts on predators and predation dynamics

- Predator abundance may be increased by excess prey abundance in the form of gamebirds. Predators such as foxes and corvids may become more ubiquitous, and protected predators, such as raptors may also benefit.
- At the end of the shooting season gamebirds may be reduced to such an extent that predators sustained at elevated numbers due to abundant overwinter prey, may switch to other prey types. This period of low gamebird abundance coincides with the nesting season for most bird species, and over-abundant predators may have detrimental effects on nesting birds.
- Some gamekeepers persecute protected predators such as birds of prey, particularly buzzards and goshawks, which are perceived as threats to gamebirds.

Socio-economic impacts

- Estimates suggest that the shooting industry is worth £2 billion each year, and supports 70,000 full time jobs.
- Nearly half a million (480,000) people are estimated to shoot game.
- 0.9-1.8 million pheasants collide with vehicles each year, with unknown insurance costs.
- Pheasants may increase the risk of exposure to Lyme disease in humans.
- Lead shot in game for human consumption regularly goes undetected, and people can suffer lead related illness.

Table I. Details of the studies (numbered and referenced below table), and summary of impacts and potential impacts of non-native gamebirds at high densities (pheasant & red-legged partridge). Approx. 40 million released each year in the UK.

Gamebird release associated impact or activity (numbers refer to sections in the text)	*Impact evidence - positive or benign(b)	*Impact evidence - negative	Expected impact[a]	Geographic scale	Notes and species affected
2. Game estate management					
2.3 Woodland creation & retention	3,62[b],63		+ Mod	Unknown	Woodland wildlife likely to be benefited
2.4 Woodland management	1,2,4,8,9,11, 14,20,62[b],64, 66[b],67[b],70[b]	62	+ Mod - Low	Widespread	+ Butterflies may benefit from opening of woodland rides. Some woodland birds and small mammals can benefit from sky-lighting, and increased densities in field layer flora. - Tits and other species of dense broadleaf canopy may be detrimentally affected.
2.4 Farmland management (includes hedgerow management)	1,3,7,16,18, 19,20,37,38, 39,40,41,42, 53[b],62[b],63,65		+ Mod	Widespread	Conservation headland and cover crops etc benefit farmland birds, butterflies, ground nesting birds, and invertebrates. Greatest benefits for grey partridge.
2.5 Legal predator control	1,5,6,7,10, 11,13,15,71		+ Mod	Widespread	Nesting birds may benefit, particularly ground nesting.
2.6 Supplemental feed	1,8,12,17,66	43,44,66	+ Mod - Low	Widespread	+ Seed eating birds, thrushes, and some small mammals benefited. - Some birds avoid feeders. Feeders may represent point of disease transfer.
3. Direct impacts of gamebirds					
3.3 Browsing by gamebirds	53[b]	32,33,34	- Mod	Widespread	Highly omnivorous, can reduce plant species richness, ground cover, and hedge leaf density, and cause the exclusion of some perennials. Changes to hedge structure may modify nesting habitat for some birds, and cause a reduction in host plants for some lepidopterans.
3.4 Predation of overwintering invertebrates by gamebirds	35[b]	67	- Mod	Widespread	- Ground-active invertebrate biomass lower in pheasant release areas in late spring. Caterpillar biomass lower in areas where pheasants released. This may reduce chick provisioning resource abundance for breeding birds.

Table I. cont.

Gamebird release associated impact or activity (numbers refer to sections in the text)	*Impact evidence - positive or benign(b)	*Impact evidence - negative	Expected impact[a]	Geographic scale	Notes and species affected
3.4 Breeding gamebird competition with native wildlife			Unknown	Unknown	Breeding gamebirds may compete with farmland birds for chick food resources, particularly as pheasants reduce invertebrate biomass. Not studied, but likely.
3.5 Soil enrichment		32,36	- Mod	Widespread	Game birds at high densities increase soil phosphate and potassium levels, changing plant species composition in woodlands, and also threaten rare bryophyte species in moorland fringe habitat.
4. Gamebirds and disease					
4.5 Released gamebirds spread disease	22[b],26[b]	21,23,24,25, 27,28,29,30, 31,43,44,55,	- Mod	Widespread	Released game birds pass on parasites to wild birds especially at feeders. Grey partridge particularly at risk, as well as seed-eaters, pigeons and in some cases birds of prey.
5. Shooting practises					
5.3 Consumption of lead poisoned prey		45,46	- Mod	Widespread	Birds of prey and scavengers feeding on lead poisoned prey suffer secondary poisoning.
5.4 Lead directly ingested by wildlife		47	- Mod	Widespread	Ingested by galliforms including grey partridge, pigeons and doves. Causes poisoning leading to death at sufficient concentrations.
5.5 Lead concentrations in the environment		48	Unknown	Unknown	Enables lead transfer through food chain from primary consumers up Lead absorbed by earthworms, in turn absorbed by small mammals, which are in turn consumed by predators.
5.7 Grey partridge by-catch		74,75	Moderate	Unknown	Grey partridge may suffer unintentional population impacts.
6. Impacts on predators and predation					
6.3 Predator abundance increased	68[b],72[b]		+ Mod	Widespread	+ Abundant game birds may provide important food source for protected predators (e.g. birds of prey). - May increase the numbers of various pest predators (e.g. fox). At the end of the shooting season, predators have to switch to other prey sources, coinciding with the nesting season of native birds. Not studied.

Table I. cont.

Gamebird release associated impact or activity (numbers refer to sections in the text)	*Impact evidence - positive or benign(b)	*Impact evidence - negative	Expected impact[a]	Geographic scale	Notes and species affected
6.4 Illegal persecution of predators by gamekeepers		73	Unknown	Widespread	Some gamekeepers kill protected predators such as birds of prey. Largely unquantified, but known to be widespread.
7. Socio-economic impacts					
7.3 Economic input	58		+ Mod	Widespread	Shooters spend £2 billion each year on goods and services.
7.3 Employment	58		+ Mod	Widespread	Game shooting (all types), supports about 70,000 full time jobs.
7.4 Traffic accidents caused by gamebirds		59,60,61	Unknown	Widespread	Between 2.7-5% of released pheasants are killed on the road (0.9-1.8 million each year).
7.5 Disease passed indirectly from gamebirds to humans		54,55,56,69	- Low	Localised	Abundant pheasants that harbour the lyme disease parasite may increase the risk of lyme disease exposure in humans.
7.6 Lead in game for human consumption	52[b]	49,50,51	- Low	Widespread	Lead shot and fragments can go undetected in food. People who regularly eat game can suffer from lead related illness.

*Only studies within the UK, involving species found in the UK, or involving pheasant/red-legged partridge are cited.

[a] Where data are insufficient, expected impact is based on biologically likely mechanisms, and therefore may not relate directly to the evidence column.

[b] Study shows no affect (benign).

Evidence

1. Stoate & Szczur 2001a. 2. Draycott *et al.* 2008. 3. Arroyo & Beja 2002. 4. Robertson *et al.* 1988. 5. Stoate *et al.* 2008. 6. Stoate *et al.* 2009. 7. Stoate 2007. 8. Hoodless *et al.* 2006. 9. Hoodless & Draycott 2006. 10. Stoate 2006. 11. Stoate & Szczur 2006. 12. Draycott 2005. 13. Stoate 2005. 14. Draycott & Hoodless 2005. 15. Stoate 2004. 16. Parish 2004. 17. Brickle 1997. 18. Stoate & Parish 2002. 19. GCT 2001. 20. Hinsley *et al.* 1999. 21. Gryczynska *et al.* 2004. 22. Ewald & Touyeras 2002. 23. Tompkins *et al.* 1999. 24. Tompkins *et al.* 2000a. 25. Tompkins *et al.* 2000b. 26. Sage *et al.* 2002b. 27. Villanúa *et al.* 2007. 28. Hofle *et al.* 2004. 29. Real *et al.* 2000. 30. Sansano-Maestre *et al.* 2009. 31. GCT 2003. 32. Sage *et al.* 2005. 33. Sage *et al.* 2009. 34. Woodburn & Sage 2005. 35. Callegari 2006. 36. Rothero 2006. 37. Sotherton 1991. 38. Thomas *et al.* 1992. 39. Rands 1985. 40. Rands 1986. 41. Potts 1997. 42. Brickle *et al.* 2000. 43. Pennycott 1998. 44. Pennycott & Duncan 1999. 45. Pain *et al.* 1995. 46. Pain *et al.* 2007. 47. Potts 2005. 48. Ma *et al.* 1991. 49. Iqbal *et al.* 2009. 50. Kosnett 2009. 51. Tranel 2009. 52. Haldimann *et al.* 2002. 53. Hoodless & Draycott 2007. 54. HPA 2009. 55. Ostfeld & Keesing 2000. 56. Schmidt & Ostfeld 2001. 57. Keesing *et al.* 2006. 58. PACEC 2006. 59. Turner & Sage 2004. 60. Hill & Robertson 1988. 61. PostOffice 2008. 62. Firbank 1999. 63. Oldfield *et al.* 2003. 64. Fuller & Warren 1993. 65. Rands 1985. 66. Davey 2008. 67. Pressland 2009. 68. Mrlik & Koubek 1992. 69. Kurtenbach *et al.* 1998a,b. 70. Warren 1989. 71. Gibbons *et al.* 2007. 72. Kenward *et al.* 2001. 73. RSPB 2009. 74. Watson *et al.* 2007. 75. Aebischer & Sotherton 2002.

1. Background

1.1 Annual release of non-native gamebirds into the UK countryside

1.1.1 Facts and figures

More than 40 million non-native gamebirds are released into the British countryside each year (PACEC 2004). In 2009, over 37 million common pheasants (hereafter pheasant), and 13.8 million partridges were registered on DEFRA's Great Britain poultry register. As these species are released into lowland habitats, the majority of holdings are in England. Nonetheless a significant proportion are held in Scotland (4 million pheasants, 1 million red-legged partridges), and Wales (2 million pheasants, 0.4 million red-legged partridges). In 2004, it was estimated that a total of 35 million pheasants and 6.5 million red-legged partridge were released for game shooting, of which just 15 and 2.6 million were shot respectively (Fig. 1.1.1). The density of released pheasants on game estates has continued to increase since intensive rear and release operations began in 1961. The biomass of pheasants released annually is approximately 41,000 tonnes, which is 16 times greater than that of any other bird species in the UK with 2,600 tonnes of woodpigeon, and red-legged partridge release biomass (3,200t) is now also greater than woodpigeon (Baker *et al.* 2006a). Assuming that the figures reported in Turner & Sage (2004) are representative of all released pheasant mortality, then approximately 12.6 million (36%) pheasants are predated each year, 1.8 million (5%) die of other causes such as road accidents and disease, and approximately 5.6 million (16%) survive into the spring and onto the following shooting season (Turner & Sage 2004). This equates to a surplus biomass in late summer of approximately 6,500t of pheasants (Table 1.1.1).

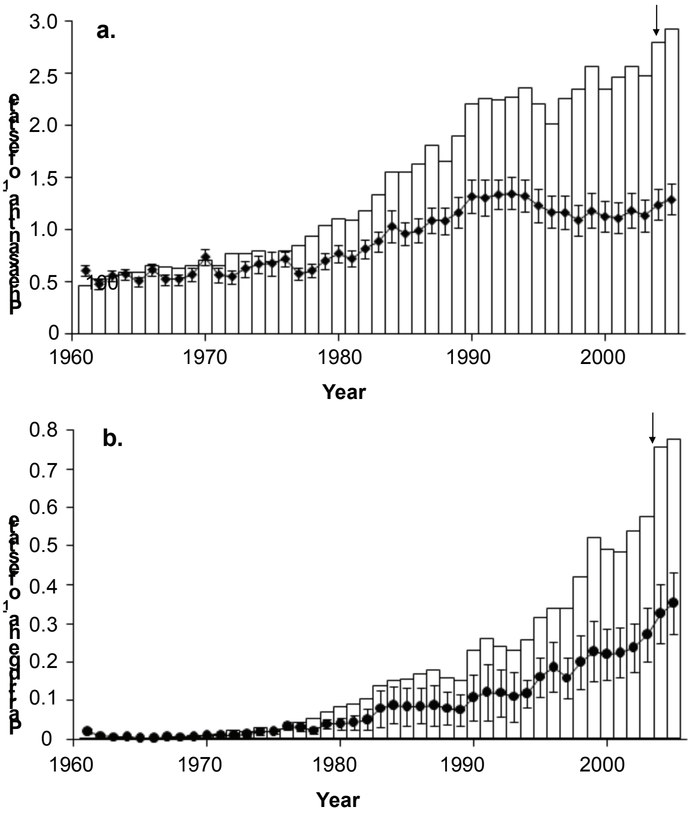

Fig. 1.1.1. Annual density of a. pheasants and b. red-legged partridges released per hectare of game estate (bars), and numbers shot (dots ± 95% CI) from 1961 to 2005 in the UK. In 2004 (depicted by arrow), it was estimated that 35 million pheasants were released and 15 million shot, and 6.5 million red-legged partridges were released, and 2.6 million shot (PACEC 2006) (graphs from www.gct.org.uk).

Despite continued increases in release density, the proportion of pheasants shot has declined since about 1990 (Fig. 1.1.1a). This means that although game managers are spending more time, effort and money releasing greater numbers of pheasants, their financial returns are dropping. The mechanisms behind this are unclear, but may be caused by lower survival rates in pheasants. Changes to pheasant survival may be due to greater levels of predation or disease, however the data on fox cull levels for the same period do not support this (Davey & Aebischer 2009). Data are not available on disease incidence in pheasants but in 2003 a widely used and important protozoan parasite preventative was banned. Additionally, this period coincides with a reduction in the level of compulsory set-aside, and a subsequent increase in the planting of game cover crops which are used to lead 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 (N. Aebischer pers. comm.).

Table 1.1.1. Approximate mean annual fate of released pheasants in the UK. Data after a. PACEC 2006 and b. Turner & Sage 2004. Excludes birds that die prior to release.

Cause of death	Number (millions)	Percent	Biomass (tonnes)
Shot[a]	15	43	17,430
Predated/scavenged[b]	12.6	36	14,600
Other (road and disease)[b]	1.8	5	2100
Survive to following season[b]	5.6	16	6,500
Total	35	100	41,000

1.1.2 *Non-native gamebird species*

The native range of the pheasant, and its ~30 subspecies extends throughout the Asian continent from eastern China to south-western Russia. In Britain, pheasants have been closely associated with woodlands and farmland since the 15th century when they became naturalized. By the early 1800s they had become an important quarry species when they were also introduced across much of western Europe including further releases in the British Isles (Hill & Robertson 1988). In the 1960s, the development of artificial rear and release methods led to a substantial increase in the number of pheasants in Britain (Tapper 1992). Pheasant releases now appear to be widespread and quite evenly distributed throughout lowland Britain, and largely absent from uplands (Figs. 1.1.2.1 & 1.1.2.2). The breeding population of pheasants in the UK is estimated at 1.8-1.9 million females (Baker *et al.* 2006a), and has continued to increase. At least six subspecies are now found in the UK, and it seems that the breeding population consists of a conglomerate of numerous phenotypes (Hill & Robertson 1988).

The red-legged partridge is native to Spain, Portugal, Andorra, and southern France, and was introduced into the UK around 1770. For rear and release operations, this species is preferred over the native grey partridge for economical reasons. The breeding population of red-legged partridges in the UK is now estimated at 72,000-200,000 territories (Baker *et al.* 2006a), and has fluctuated over the last 50 years. In Britain, releases seem to be slightly less wide-ranging than pheasant releases, and in much lower numbers. Figs. 1.1.2.1 & 1.1.2.2 gauge gamebird release distribution in different ways, and are from data seven years apart. While Fig. 1.1.2.1 does not perfectly represent the spatial distribution of gamebird releases (because it includes rearing farms as well as release estates), it does provide a reasonably accurate measure of release spatial distribution. It is unknown whether data from the Breeding Bird Survey (Fig. 1.1.2.2) reflect release distribution, but given that there has been a sharp increase in red-legged partridge release numbers since 2003 (Fig. 1.1.1), the disparity between these maps could be pertaining to a more widespread release distribution since 2003. A relatively recent pattern has been the release of red-legged partridge on moorland fringe. These releases may reflect an attempt to compensate financially for declining grouse bags that some estates have experienced in recent years. Despite this trend becoming increasingly widespread, there are little information on the numbers of birds or the areas involved.

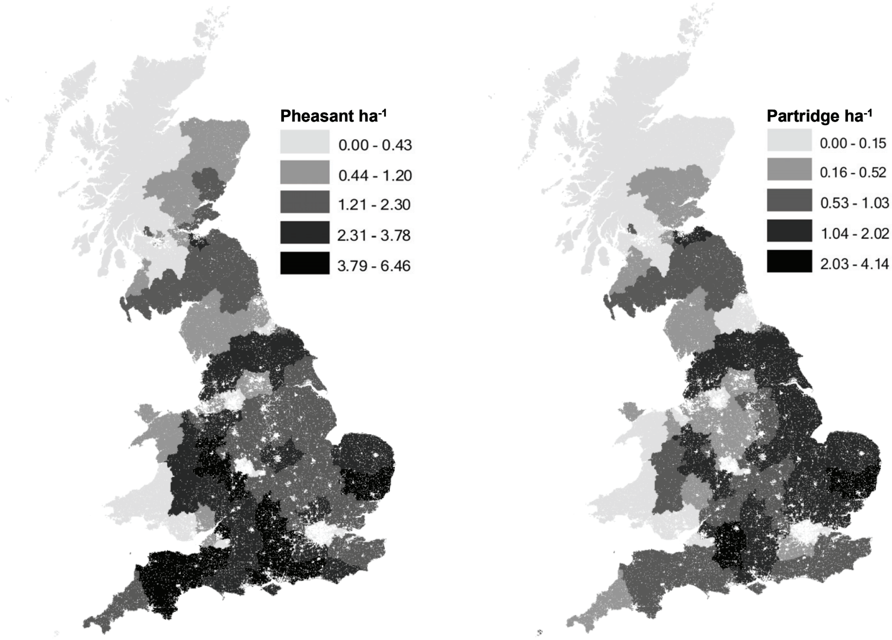

Fig. 1.1.2.1. Density of pheasant and partridges (includes red-legged and grey) registered on the DEFRA Poultry Register in 2009. All holdings with >50 birds are required to register, including release estates and rearing farms. Density per ha is by county and excludes urban areas.

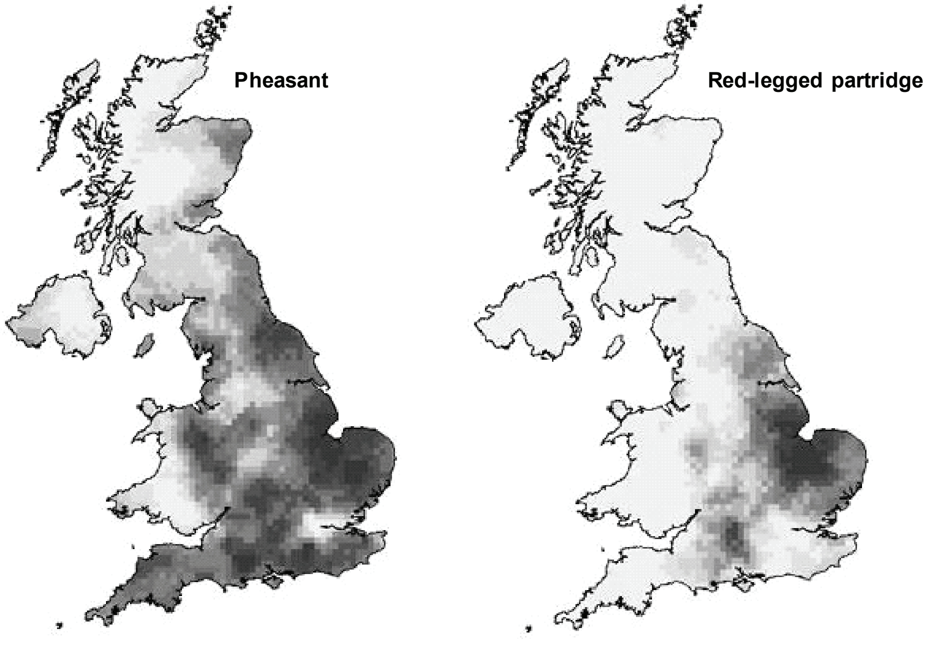

Fig. 1.1.2.2. Relative abundance of pheasant and red-legged partridge in the UK. Darker and lighter squares represent higher and lower abundance respectively. Data are from the Breeding Bird Survey spring 2003, and do not represent release densities. However, these data do serve as a proxy for release density, as more birds released in summer are likely to correlate with greater numbers of birds in spring in the same area, as these species are relatively sedentary (after Newson & Noble 2003).

1.1.3 *Regulation*

The practice of releasing non-native gamebirds has been occurring for over 100 years and is largely exempt from regulation. Pheasant and red-legged partridge are not listed on schedule 9 of the Wildlife and Countryside Act of 1981 (as amended), and are as such not considered in a policy sense as Invasive Non-Native Species (INNS). The justification for this however, is not ecological, but economical, and is therefore counter-intuitive to the purpose of the Act. Therefore, the validity of such an exemption is questionable. An analogous example of justified exemption is the agricultural livestock industry. However, livestock are carefully controlled, individually 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 hobby, and does not supply a significant proportion of the UK food industry (PACEC 2006). The gamebird industry therefore falls outside of regulations covering the release of non-native species, despite the negative impacts associated with high density release operations. In contrast, the release of non-native gamebirds in the Netherlands was banned on animal welfare grounds, following the Netherlands Flora and Fauna Act of 2002.

1.2 Impacts of large scale non-native gamebird release

1.2.1 *Concerns in the literature*

The effect of released gamebirds on other species was considered one of the 100 most important ecological questions with policy relevance in the UK by Sutherland *et al.* (2006). Additionally, Fuller *et al.* (2005) suggest that high pheasant densities may have negative impacts on woodland birds in the UK, and effects may include impacts on the structure and composition of woodland ground vegetation, the spread of disease and parasites, and competition for food. To date these potential effects have not been widely studied, and what little evidence does exist has not been synthesised. As non-native species that are being released at extreme quantities, research into the impacts of this practice should be considered a priority.

The release of thousands of gamebirds in an area could alter the predator community, the parasite community, habitat structure and composition, and the abundance of invertebrates and food for native wildlife. Shooting itself also has potential impacts. Most shooting on lowland estates uses lead ammunition that can poison wildlife directly, or indirectly, and occasionally native species are mistakenly shot, particularly grey partridge (Watson *et al.* 2007).

1.2.2 *Density*

Sage (2003) suggests that many of the concerns associated with the release of gamebirds can be attributed to estates operating high release densities, and this is likely to be the case. In response to this, the Game and Wildlife Conservation Trust (GWCT) have been working to test, and recommend the highest standards of gamebird release practices, including lower release densities (see box 1.3.2). However, despite these recommendations, evidence suggests that release densities have continued to rise since the 1960s (Fig. 1.1.1). It is therefore likely that the negative impacts of gamebird release are significant and widespread. Best practices inevitably require greater financial input from game managers whose profit margin is usually small (PACEC 2006).

The GWCT recommend 700-1000 pheasants per hectare of release pen, or just over 14m^2 per bird (Sage & Swan 2003). However, most release pens are stocked at levels greater than 2000 birds per hectare, with some reaching closer to 5000 birds per hectare (Sage *et al.* 2005). This is therefore of concern to conservationists and land managers, as many of the available studies suggest that negative environmental impacts are experienced at pen stocking densities >1000 birds per hectare. Outside of release pens, gamebird densities are not widely documented. Fig. 1.1.1 estimates densities at about 2.8 pheasant and 0.8 red-legged partridge per hectare of game estate. However, these averaged figures do not represent the clustered distribution of gamebirds throughout estates, as they congregate at feeders, roost in release pens and other areas, and often follow linear boundaries. Gamebirds can therefore be densely aggregated outside of release pens, and it is in these high density areas that negative impacts are most likely to occur.

1.3 Game shooting organisations

1.3.1 GWCT and BASC

The Game & Wildlife Conservation Trust (GWCT), (formerly The Game Conservancy Trust (GCT)) conduct scientific research and develop and test management techniques that benefit game and other wildlife. The GWCT provide training and advice to farmers, gamekeepers and land managers on methods to improve game estate biodiversity, and primarily how to advance game management practices. They operate several experimental sites to test and research these novel techniques in the interest of benefiting gamebirds, biodiversity, and mitigating the impacts of rear and release activities. The Loddington Estate in Leicestershire is a 333 ha farm managed by the GWCT. The estate provides long-term data on wildlife in relation to an experimental set of game management practices, such as conservation headlands, cover crops and many more. Another estate in the Sussex Downs forms the main location for Grey Partridge research. The GWCT additionally collaborate with numerous estates across the UK used as study sites for specific projects.

The British Association for Shooting and Conservation (BASC) advocate ‘green shoots’ where estates work to maximise biodiversity, and the breeding potential of gamebirds, negating the requirement for large rear and release operations. Additionally, BASC provide shooting codes of practice and advocate the conservation of grey partridge. The Game Farmers Association further provide guidelines for animal husbandry and the control of disease and other rearing related issues. There are many other game associated organisations, but none are actively involved in the biodiversity issues of game shooting in the way that the GWCT (and BASC to a certain degree) are, and therefore hold little relevance here.

1.3.2 Guidelines from the GWCT

Of primary interest to this review are the GWCT’s guidelines for minimising the direct impacts of released gamebirds, and maximising the benefits from game estate management. They advocate numerous management practices, and have produced multiple guideline documents for farmland and woodland management, and procedures to benefit specific species. These are available at their website (www.gct.org.uk), and are summarised in Box 1.3.2.

Box 1.3.2. GWCT – Guidelines for sustainable gamebird releasing (Sage 2007)

Woodland release pens

- We have now demonstrated that woodland management for shooting, such as encouraging native shrubs and sky-lighting among mature trees, usually improves habitat for a broad range of woodland wildlife and should be encouraged unless there are other particular conservation objectives.
- We recommend that no more than 1,000 pheasants should be released into each hectare of release pen. This is the point at which undesirable plants become prevalent in pens and where effects on woodland plants that are exposed in late summer and early winter become more marked. In sensitive woods, for example, ancient semi-natural woods, the maximum release should be 700 birds per hectare of pen. This will minimise the effect of releasing to sensitive native woodland plants.
- To provide sufficient scope for benefits, pheasant release pens should not take up more than about one-third of the total woodland area on an estate or shoot holding. This should include any woodland whether or not it has existing nature conservation interest.
- The 'total woodland area' used in this calculation could include scrub patches, substantial hedgerows with trees, shelter belts and new woodland plantings, but it should not include the central part of large woodland blocks where there is no game interest.
- We are currently unsure of the extent to which the woodland ground flora in a disused release pen will revert to its original state. We recommend not moving release pens unless there are obvious conservation benefits to be gained (or if there are overriding husbandry reasons). We are studying this and may update this guideline.
- Birds released into woodland pens should be encouraged to leave the pen for at least part of the day once they are used to roosting in trees.

Hedgerows

- Don't allow releases of more than 1,500 birds to funnel through one main hedgerow linking releasing and holding areas, particularly if that hedgerow contains a good ground flora and is home to valuable wildlife. For larger releases, use several linking hedgerows or plant cover crop 'lead-in' strips to widen linking habitats.
- Remember to protect valuable hedgerows from crop sprays, livestock browsing and cultivation. There is no point in worrying about pheasant damage if conservation measures are undone by, for example, access by livestock.

Red-legged partridges

- Red-legged partridge release pens should, wherever possible, be placed in game cover planted on arable or improved grassland, rather than on semi-natural or unimproved grassland sites. We found no evidence that red-legged partridges released onto improved land, had any effect on nearby chalk grassland onto which the partridges later spread.
- Avoid placing release pens or partridge feeders next to high conservation value hedgerows. Allow a buffer zone of game crop or other cover to keep concentrations of birds away from the hedge.

Game crops

- We have shown that game crops provide valuable cover and food for many farmland and woodland birds in winter. In general, kale, quinoa and cereal mixtures offer the greatest value to these birds.
- The value of game crops to wild birds in winter is greatly enhanced if game feeders in these crops are kept topped up through the late winter and spring.
- Long-term game cover plots can sometimes spoil adjacent woodland edges and hedgerows, so it is best to move them if this appears to be happening. It is best to avoid maize as it can encourage rats. Keep rats (and rabbits) under control.

The precautionary approach

- Always avoid placing release pens directly onto or close to particularly sensitive locations. For example, reptile breeding or hibernation sites, and patches of woodland that have a notable ground flora or insect fauna.
- Delaying the time of release can avoid some potential conflicts. For example, some reptile and butterfly colonies may be vulnerable in mid-to-late summer but less so if the gamebirds are released later. If in doubt, take expert advice.

Case study - pheasant releasing

- Our most important recommendations relate to the total release pen area in relation to maximum stocking densities in these pens. Effectively this limits the numbers of pheasants that should be released on any one shoot. It is important to remember, however, that the decision to release pheasants on any area is dependent on a range of factors. Here we provide general guidance based on broad conservation issues alone.
- Using this approach, on a 450 hectare estate with 30 hectares of woodland (typical for lowland England) we recommend that the total area of release pens should not exceed one third of this, or 10 hectares. This means that a small woodland can be mostly release pen as long as it is balanced by un-penned woodland elsewhere on the holding. Depending on the conservation value of the woods, it would then be appropriate to release no more than between 7,000 and 10,000 pheasants into this 10 hectare area.

1.4 Lowland gamebird shooting industry

1.4.1 Rear and release activities

Approximately 40% of pheasants, and 90% of red-legged partridges come from France, normally as eggs or as day-old chicks. The EU transport limit on chicks is 24 hours providing it is completed within 72 hours of hatching. There is also a small trade in 6-8-week old poults from France, but this is likely to represent only a fraction of the trade. The maximum journey time of 12 hours for these older birds makes much of the UK out of range for French game farms, thus limiting this particular trade. The remaining poults are reared within the UK. However, in 2010, the British government plan to make considerable changes to way gamebirds are reared, and the new Code is claimed to be 'unworkable' by the Game Farmers Association (GFA). If this Code is passed, the GFA predict that much of the trade will rely more heavily on birds from France.

At 6-8 weeks old, during July-August, poults are placed in open-topped woodland release pens which can be up to 5 ha in size. Woodland release pens are fenced-off areas where birds are slowly accustomed to natural surroundings, and are protected from mammalian predators. At about 12 weeks old, birds are encouraged to venture out of pens, but usually return to roost at night. As the birds get older they spend more and more time foraging in woodlands and fields adjacent to release pens. Adult birds eat a greater 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. In release pens, birds can be stocked at densities from 500-5000 ha^{-1}, but on average about 2000 ha^{-1}, double the maximum level recommended by the GWCT. It is estimated that about 10% of birds die before being placed in release pens, and a further 5-25% die prior to shooting (predation, disease etc).

1.4.2 Shooting providers

About 480,000 people shoot live quarry for sport, the majority of which shoot gamebirds. The entire game shooting industry is estimated to be worth £2 billion each year, a significant proportion of which is derived from lowland gamebird shoots (PACEC 2006). There are approximately 7,000 shooting estates in the UK, and 30,000 lowland game shooting providers. These are supplied by about 400 rearing farms in the UK, and about 2,500 smaller premises rear birds for release on their own estate (plus French farms). 83% of shooting providers rely on released birds, and larger rearing farms may produce up to 3 million eggs a year.

The shooting season for red-legged partridges and pheasants begins on the 1st September and 1st October respectively, and ends on the 1st February. The numbers of birds released varies between shoots, depending on the interest and business plan of the owner, and the size of the estate. Medium to large shoots are generally prepared to invest more in game management. The number of birds that clients wish to shoot in a day varies, and currently 'large bag' days are under some scrutiny within the industry. Few shoots advertise, and the demand for shooting apparently remains unchanged (Wong & Dickinson 2003).

The cost of eggs, and poults are very similar for pheasant and red-legged partridge. Eggs are around £0.40 each, day-old chicks cost between £0.60 and £1.00, and six to seven week-old poults are £3.50-£4.00. In 1999 the cost of rearing a bird from egg to adult was approximately £2.50 (Canning 2005, Wong & Dickinson 2003). To the food market, a pheasant is worth only around £0.50, because the market is driven by the sale of shooting not meat. Costs to shooters (called guns) vary, but typically £25 - £35 is paid per bird shot, and an individual may shoot up to 40 birds per day (i.e. £1000-£1400 per day). 80% of shot birds are exported to France for human consumption (Canning 2005), although game is now becoming more widely consumed in the UK.

1.4.3 Area under management for gamebird shooting

Shooting estates cover a total of 15 million hectares, of which approximately 2 million ha are actively managed (Fig. 1.4.3). Approximately 30,000 ha (3% of total woodland area) of woodland in England are enclosed in the approximately 13,000 release pens (PACEC 2006). Pens are found in 5% of woodlands >0.2 ha in the UK (Sage *et al.* 2005). Fig. 1.4.3 shows the extent to which management practices are implemented on game estates. However, figures are not available to compare this with non-game estates at the national scale. It is therefore difficult to establish whether management of these habitats is game motivated or simply typical of most estates (but for woodland see section 2.3.1).

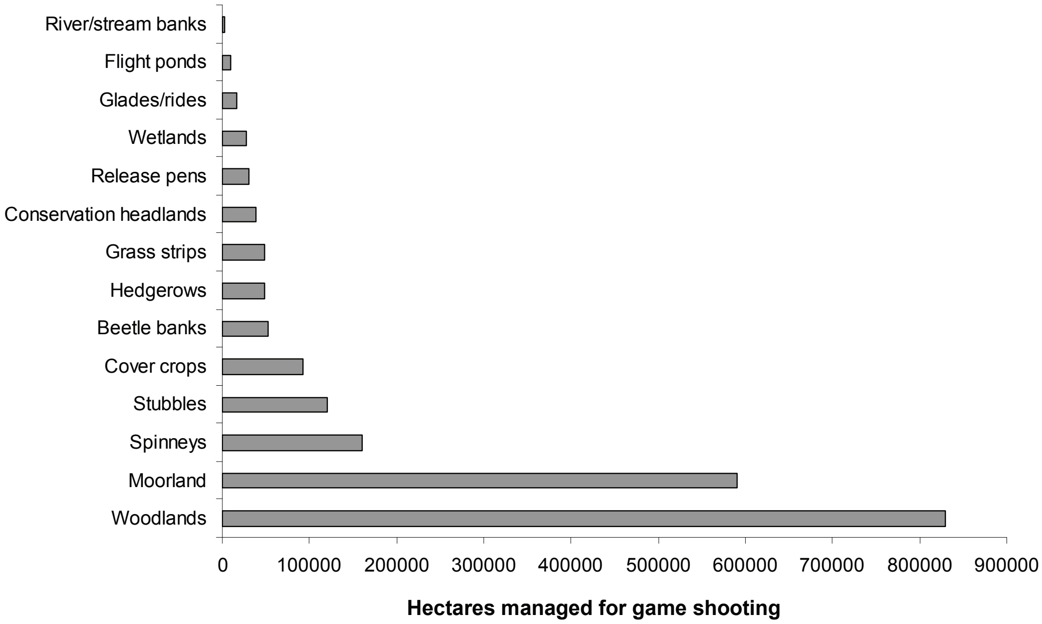

Fig.1.4.3. Estimated area (ha) of various land management types actively managed for sport shooting in the UK (after PACEC 2006).

2. Game estate management

2.1 Summary

Many estate managers actively improve woodland and farmland to assist the survival of their game stock. On farmland, some managers encourage invertebrate habitat such as conservation headlands to provide chick food resources for breeding gamebirds. In woodlands, managers may open-up rides, and encourage ground flora as well as plant new woodland. Additionally, predator control and supplemental feeding are generally common game management activities, and it is thought that these habitat management practices largely benefit other wildlife.

2.2 Background

Most game estate managers actively manage their land in some way to benefit their released stock. Gamebirds surviving beyond one shooting season into the next lower the number of birds that need to be released, and therefore reduce output costs, and likewise stronger, fitter birds make better sport for shooting. Game managers undertake woodland habitat improvement such as widening woodland rides, sky-lighting to increase understorey light levels, encouraging ground flora, and planting new areas of woodland. On farmland, game managers maintain hedgerows, plant cover crops, operate conservation headlands and use several fallow types to create nesting and foraging habitat for breeding gamebirds. Game managers also undertake predator control and provide supplementary food to improve the survival of gamebirds. Many of these activities are likely to be beneficial to other wildlife. Game estates usually contain more woodland, more ponds and lakes, but less grassland than non-game estates (Stark *et al.* 1999). It is estimated that shoot providers spend £250 million a year on habitat management (PACEC 2006).

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 (Amar *et al.* 2006, Fuller *et al.* 2005). Several bird species may therefore benefit from game estate management in woodlands, where shrub layer and field layer vegetation is often encouraged. Wild birds are also likely to benefit from game management in farmland, where important invertebrate habitat is often planted, and this is likely to provide an important resource for nesting birds.

2.3 Habitat management on game estates

2.3.1 Woodland management on game estates

Game estate managers retain more woodland on their land than non-game estates (Firbank 1999, Oldfield *et al.* 2003). Oldfield *et al.* (2003) found on average 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. Game estates are also more likely to plant new woodlands, particularly broadleaf woodland (Firbank 1999, Oldfield *et al.* 2003). A questionnaire by the GWCT of 150 game managers (presumably GWCT members) found that 81% had conducted some kind of woodland management, and 30% had planted or coppiced woodland for game management purposes.

Many traditional woodland management techniques that have been largely phased-out elsewhere are retained within woodlands managed for game. Coppice is more commonplace in game managed woodlands, and this technique has been shown to benefit various wildlife, including a variety of birds (particularly migrants) and small mammals such as dormice (Clarke & Robertson 1993, Fuller & Green 1998, Fuller *et al.* 1989, Fuller & Warren 1993, Tapper 1999). Adult pheasants spend much of their time near woodland edges (Robertson 1997), and therefore estate managers often create large open rides, that increase light levels, and provide cover from predators that is thought to be more beneficial than farmland edge habitat. In a study of 159 woodlands, Draycott *et al.* (2008) found that woodlands managed for pheasants had a more open structure with 2-7% less canopy cover, a denser herb layer with 5-58% more ground vegetation, more grass, and less moss. Although it is widely considered that game estates encourage the woodland shrub layer, Draycott *et al.* (2008) found no differences between game and non-game estates. Additionally, in a study of 26 sites, Firbank (1999) found no differences in canopy density, or shrub and field layers. The study by Draycott *et al.* (2008) is more recent and extensive, and is therefore more likely to closely represent the current nationwide average.

2.3.2 Farmland management on game estates

Many game estate managers operate various farmland habitat management practices to benefit free-ranging and sometimes breeding gamebird stock. These include cereal field margins such as conservation headlands (Sotherton 1991) and cover crops, and the maintenance of hedgerows, grass margins and 'beetle banks' (Tapper 1999, Thomas *et al.* 1992). Conservation headlands are a strip of crop at field margins that is left untreated from herbicides and pesticides to encourage weeds 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. Game estates are more likely to plant hedgerows (Oldfield *et al.* 2003), and Firbank (1999) found that hedgerows on game managed estates are more often connected to woodlands.

On farmland, conservation headlands contain broadleaf weeds that support a rich invertebrate fauna, providing a fundamental food source for the young of several wild bird species during the breeding season in spring and early summer. Conservation headlands may increase pheasant and red-legged partridge breeding productivity, and have been shown to increase productivity in native birds such as the red-listed grey partridge (Potts 1997), and provide foraging habitat for species such as the red-listed corn bunting (Brickle *et al.* 2000). Beetle banks, grass margins and cover crops also provide foraging and breeding habitat for gamebirds, and these areas are important for other wildlife, particularly finches, and wood mice (Tapper 1999, Thompson & Sage 2000).

It is unknown how widespread these management practices may be (but see section 1.4.3), and the majority of shooting estates (83%) rely solely on released pheasants (PACEC 2006), and therefore have little incentive to encourage breeding habitat and promote invertebrate fauna for chicks. Hinsley *et al.* (1999) found that in areas where shooting was mainly based on released pheasants rather than wild breeding pheasants, management of crop and crop boundaries, showed few consistent differences between game and non-game sites, reflecting the lack of widespread adoption of these farmland management practices motivated by game shooting. Additionally, with the recent incentive-driven agri-environment schemes, discrepancies between game and non-game estates are now likely to be less distinct. Agri-environment schemes pay landowners to undertake land management practices that benefit biodiversity. Game estates that receive these rewards are effectively double paid, by agri-environment incentives and by shooting revenue. Therefore, 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.

2.4 Effects of game estate habitat management on wildlife

2.4.1 Effects of game estate habitat management on invertebrates

A reduction in invertebrates caused by agricultural intensification has been identified as a major component in the decline of many farmland birds (Chamberlain *et al.* 2001, Fuller *et al.* 1995, Fuller *et al.* 2005, Vickery *et al.* 2009), and field margins managed for biodiversity conservation have been considered as a method to help reverse these declines (Vickery *et al.* 2009). Decreased use of pesticides at the edge of fields in particular augment the abundance of invertebrate species essential to the rearing of gamebird and other bird nestlings. Fallow strips such as beetle banks 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). It is therefore likely that in areas where these farmland habitat improvements are made, many species benefit.

Pheasant managed woodlands contain more butterflies than non-game woodlands (Robertson *et al.* 1988), and conservation headlands contain more butterflies than normal headlands (Tapper 1999). In woodlands on game estates, butterflies benefit from the opening of rides, but pheasants also feed on caterpillars and may reduce their numbers (Pressland 2009) (section 3.4).

2.4.2 Effects of game estate habitat management on birds

Most studies have reported an overall increase in the abundance of many woodland and farmland bird species on game estates compared with non-game estates (Davey 2008, Draycott *et al.* 2008, Hinsley *et al.* 1999, Stoate 2007) (Table 2.4.2, Fig. 2.4.2). The exact mechanisms for these increases have not been thoroughly tested, though it is likely they are due to a number of factors including habitat management, habitat creation, supplemental food, and predator control. In a further comparison between game-managed and control sites, nationally declining passerines increased over a seven year period on the game-managed site, but did not change at the adjacent non-game sites (Stoate & Szczur 2001a). However, in a more wide ranging woodland study, there were no significant differences found in bird abundance estimates between sites where pheasants were released, and non-game sites across the UK (Davey 2008). This study used data from the Repeat Woodland Bird Survey (RWBS), and compared pairs of woods that contained release pens with woods that did not. Habitat variables between these game and non-game woods differed only slightly and the results suggest that habitat management was not widely undertaken in these game woods. However, the RWBS focused on larger woods that are unlikely to be managed for pheasants towards their centre (where most point counts and habitat surveys were conducted), as pheasants require woodland edge habitat.

The affects of game management on woodland birds is therefore not well understood, and survey design may be a contributing factor towards this. The study by Draycott *et al.* (2008) is probably the most extensive and appropriately designed, but more subtle effects on individual species cannot be investigated from this study, as their analysis is restricted to family level, presumably due to low detection rates. However, Fig. 2.4.2 and Table 2.4.2 bring together all existing studies to investigate consistencies in bird species responses to game management. It appears that in general finch species and ground feeders are more abundant in game-managed estates. Warblers seem to be variable in their responses, but are generally more abundant in game estates. Tits however, appear to be less abundant or unaffected by game management.

It is likely that ground feeders and granivorous species such as many of the finches benefit from supplemental feed on game estates, and some warblers respond positively to the changes in woodland structure, whereas others do not (Table 2.4.2). Species characteristic of broadleaf woodland canopies such as tits, may be less abundant or show no differences in woodland managed for game, and this is explored further in section 2.4.3.

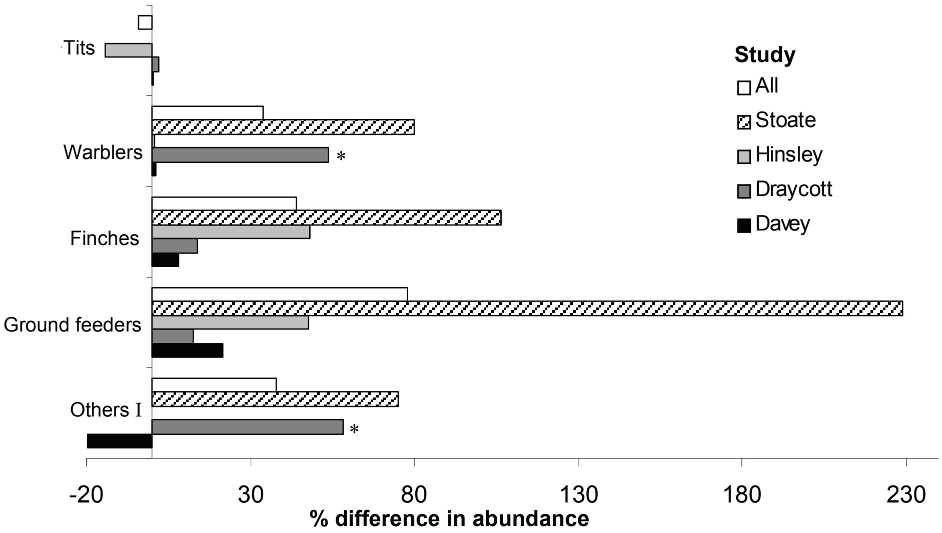

Fig. 2.4.2. Pooled percent difference in abundance of bird groups between game managed and non-game managed estates from four studies. Positive (+ve) and negative (-ve) figures represent greater and lower abundance in game sites respectively. Bird groupings as follows: Tits = Blue tit, great tit, marsh tit, long-tailed tit, coal tit. Warblers = Blackcap, willow warbler, chiffchaff, garden warbler, wood warbler, whitethroat, goldcrest. Finches = Chaffinch, greenfinch, bullfinch, linnet. Ground feeders = blackbird, song thrush, mistle thrush, robin, dunnock, wren. Others I = Great spotted woodpecker, green woodpecker, nuthatch, treecreeper, spotted flycatcher. Data after Draycott *et al.* 2008, Stoate 2007, Davey 2008, Hinsley *et al.* 1999. See footnote of Table 2.4.2 for study descriptions. * significant at the p <0.05 level.

Table 2.4.2. Percent difference in mean bird abundance between game-managed, and non-game sites. Figures close to zero show no difference from non-game sites. Positive (+ve) and negative (-ve) figures represent greater and lower abundance in game sites respectively. Species groupings are from Draycott *et al.* 2008, except others II, which are miscellaneous, unrelated species, and are therefore not pooled in 'all'.

Species	Study				All
	Davey[1]	Hinsley[2]	Draycott[3]	Stoate[4]	
Finches (all)	8	48	14	107	44
Bullfinch	-	-		100	100
Chaffinch	8	48		70	42
Greenfinch	-	-		-	?
Linnet	-	-		150	150
Ground feeders (all)	22	48	13	229	78
Blackbird	8	31		117	52
Dunnock	115	-27		213	100
Mistle thrush	-10	-		-	-10
Robin	1	6		-	4
Song thrush	10	150*		357	172
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)	1	1	54*	80	34
Blackcap	-4	-14		-	-9
Chiffchaff	13	-50*		-	-19
Garden warbler	11	-		-	11
Goldcrest	-5	67		-	31
Whitethroat	-	-		80	80
Willow warbler	-	-		-	?
Wood warbler	3	-		-	3
Others I (all)	-20	-	58*	75	38
Grt spotted w.pecker	-	-		-	?
Green w.pecker	-	-		-	?
Nuthatch	0	-		-	0
Spotted flycatcher	-	-		75	75
Treecreeper	-39	-		-	-39
Others II					
Jay	-26	-	-	-	-26
Redstart	-41	-	-	-	-41
Reed bunting	-	-	-	0	0
Skylark	-	-	-	3	3
Tree sparrow	-	-	-	133	133
Woodpigeon	-	-	223*	-	223
Yellowhammer	-	-	-	-5	-5

- Species not studied. Cells are left blank where the species is included in the study, but pooled in analysis.

* significant at the $p < 0.05$ level.

[1] Davey 2009 - Woodland study that 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. Did not test for significant difference at the species or family level.

[2] Hinsley *et al.* 1999 - Woodland study that tested bird abundance between game and non-game pairs.

[3] Draycott *et al.* 2008 - Woodland study that tested bird abundance between game and non-game pairs. Did not report figures for individual species except woodpigeon.

[4] Stoate 2007 - Predominantly farmland study. Tested bird abundance before game management with bird abundance after nine years of game management. One site only. Did not test for significance.

2.4.3 *Expected responses of birds to game estate management*

Tables 2.4.3a & b use the known habitat associations of various woodland and farmland birds to predict responses to habitat management for gamebirds. From these findings it is difficult to extrapolate abundance responses as positive and negative associations do not relate directly to bird abundance in quantitative terms. Despite this, some consistent patterns can be seen within bird groups, and these can be compared with the findings in Fig. 2.4.2 & Table 2.4.2.

Finches are most likely to be represented by chaffinch in most studies, and is the only species sampled in two or more of the studies in Fig. 2.4.2 & Table 2.4.2. Chaffinch are widespread generalists, and are unlikely to be effected by woodland management for game. Being granivorous, finches probably benefit from supplemental food on estates (although this has not been directly quantified), but all studies show consistent increases in finches across species.

Ground feeders may benefit from less canopy on game estates, because game woodlands are often dominated by mature trees that have been thinned. Therefore, the non-linear association with canopy in these species indicates a positive response to this management practice. Ground feeders are also likely to benefit from supplemental food, and these hypotheses are supported by the available studies.

Tits on the other hand, seem to have few strong associations with game management practices. The specialist marsh tit may be most likely to respond to game management, and is likely to respond positively to increases in shrub and field layer abundance in game-managed woodlands (Carpenter *et al.* 2010). Great tit and blue tit are likely to be unaffected by management because they are habitat generalists (Smart *et al.* 2007).

Warblers that show more variable responses to game woodland management may be responding to changes in ground flora depending on individual species ecology. Willow warbler are associated with early successional woodland habitats, and may therefore profit from game management with greater light levels, and associated floral composition. Chiffchaff however, select older, more mature woods (Smart *et al.* 2007), and Hinsley (1999) suggest that they are negatively affected by game management. In general, warblers require ground or understorey cover for nesting, which is more abundant in game-managed woods, and the studies show that in general warblers benefit from game management (Draycott *et al.* 2008).

Species such as nuthatch, treecreeper spotted flycatcher and woodpeckers seem to respond negatively to lower grass abundance, and this may be due to foraging requirements. Green woodpeckers forage for ants on the woodland floor and are therefore positively associated with a less dense field layer (Carpenter *et al.* 2009). The data available show mixed responses however, and the expected response to game management is currently unclear, particularly as the species within this group have varying ecology and are therefore likely to exhibit differing responses.

On farmland (Table 2.4.3b) most species are likely to respond positively to conservation headlands, beetle banks, grass margins and cover crops for breeding season foraging, and corn bunting and grey partridge may also use these for nesting habitat (Vickery *et al.* 2004). Game cover crops, are also likely to be important for winter foraging in most farmland birds, and supplemental food is likely to benefit granivorous species.

Table 2.4.3a. Expected responses of breeding season woodland birds to woodland habitat differences in game managed woodland, and other practices typical of game management. 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.

	Finches				Ground feeders					Tits						Warblers						Others I[a]				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
Game managed woodland[1]																															
More grass[2]		+			-	-	-	-	-	-		-		-		-	-		-	+		-	+	-	-	-		+			+
More herbs[2]			+		+					-				-			+	-		-			+					-		+	
Less moss[2]		-			-		+	+						+		+	+	-	+	-		+						-		-	
Less canopy cover[2]	+	+		+	∩	∩	∩	∩	∩	+		∪	+	+					∩	+	-	+	-		∩						+
Other game management																															
Supplemental food[3]					+	+	+	-	+	+										-		+									
Predator control	Insufficient evidence																														

[1] Expected responses of birds to game-managed woodland characteristics, derived from the woodland habitat associations in Smart *et al.* 2007, and Carpenter *et al*. 2009.

[2] Woodland characteristics from Draycott *et al.* 2008 that are significantly different in game-managed woodland compared with non-game woodlands.

[3] Species responses to supplemental feed at pheasant hoppers from Davey 2008.

[a] species grouping from Draycott *et al.* 2008 to enable direct comparison with Table 2.4.2 and Fig. 2.4.2.

Table 2.4.3b. Expected responses of farmland birds in terms of nesting resource, breeding season foraging, and winter foraging to farmland habitat differences in game managed farmland, and other practices typical of game management. Black and grey cells indicate strong and weaker associations respectively (only positive responses are available). White cells indicate no expected response.

	Corn bunting	Grey partridge	Linnet	Reed bunting	Skylark	Starling	Tree sparrow	Turtle dove	Whitethroat	Yellowhammer	Yellow wagtail
Game managed farmland[1]											
Conservation headlands[2]											
Nesting resources											
Breeding season foraging	■ +	■ +	▨ +	▨ +	▨ +		▨ +	▨ +	▨ +	▨ +	▨ +
Winter foraging								na	na		na
Beetle banks[2]											
Nesting resources	■ +	■ +		▨ +	▨ +						
Breeding season foraging		■ +		▨ +	■ +		▨ +			▨ +	▨ +
Winter foraging								na	na		na
Grass margins[2]											
Nesting resources	■ +	■ +		▨ +					■ +	■ +	
Breeding season foraging	▨ +	■ +	▨ +	■ +		▨ +	▨ +		■ +	■ +	▨ +
Winter foraging								na	na		
Game cover crops[2]											
Nesting resources		▨ +									
Breeding season foraging	▨ +	▨ +	▨ +	▨ +		▨ +	▨ +	▨ +	■ +	■ +	▨ +
Winter foraging	■ +	■ +	■ +	■ +	▨ +	▨ +	■ +	na	na		na
More hedge[2]											
Nesting resources		▨ +	■ +				▨ +	▨ +	■ +	■ +	
Breeding season foraging							▨ +		▨ +	▨ +	
Winter foraging								na	na		na
Other game management											
Supplemental food[3]	▨ +		▨ +	▨ +						▨ +	
Predator control	Insufficient evidence										

[1] Typical game farmland management practices from PACEC 2006, that are expected to be more ubiquitous on game managed estates.

[2] Expected responses of birds to game-managed farmland characteristics, derived from the farmland habitat associations in Vickery *et al.* 2004.

[3] Species responses to supplemental feed from Brickle 1997.

na – Not applicable, migratory birds not present during winter in the UK.

2.4.4 *Effects of game estate habitat management on small mammals*

Small woodland mammals may be affected by game-management such as supplemental food and modifications to field layer floral characteristics. In a study of 19 game-woods, wood mouse and bank vole populations increased with proximity to feed hoppers year round, and small mammal distribution was correlated with release pens (Davey 2008). In autumn wood mice were more ubiquitous near pens, but common shrew abundance was lower near release pens (Davey 2008). Shrews may be more sensitive to disturbance, and wood mice may be responding to increased resource levels in 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 section 2.6.1).

2.5 Predator control

2.5.1 *Predator control on game estates*

Game estate managers regularly control predators especially foxes, and carrion crows, but including rook, jackdaw, jay, magpie, rat and stoat, and birds of prey are also illegally killed (see section 6.4). It is likely that predator control is one management activity that is undertaken by a considerable proportion of game estates, but is seasonal on most estates. Estates that do not rely on breeding gamebirds are less likely to operate consistent predator control after the shooting season. This coincides with the breeding period for most native birds, the season when predator control is likely to be most beneficial. The extent of benefits from predator control on game estates is therefore likely to be largely governed by the intensity and consistency of culling (also see section 6).

2.5.2 *Species that may benefit from predator control*

Ground-nesting birds are at particular risk from mammalian predators (Gibbons *et al.* 2007), and the nests of all passerines may be susceptible to predation by corvids. In general, there is little evidence to suggest that predation induces a population response in most species, because predator-prey dynamics are usually governed by a 'doomed surplus' mechanism (Gibbons *et al.* 2007, Newson *et al.* 2010). Several studies by the GWCT suggest that predator control may increase the abundance of some bird species on game estates including blackbird, chaffinch, skylark, spotted flycatcher, linnet, bullfinch, dunnock, songthrush, whitethroat, and yellowhammer (Stoate 2005, 2007, Stoate *et al.* 2008, 2009). In a study of breeding bird predation on game, and non-game estates, carrion crow breeding density was significantly negatively correlated with nest survival in blackbird, song thrush, dunnock and yellowhammer, and magpie density was also significantly correlated with nest success in blackbird and song thrush (Stoate & Szczur 2001a). Due to predator control, carrion crow and magpie did not breed on the game estate. However more extensive studies e.g. Newson *et al.* (2010), have found little evidence to suggest that avian predators have population level impacts on avian prey at a nationwide scale. This suggests that the findings of the GWCT referred to above may be either circumstantial, or the result of estate management.

There are differences in the philosophy of predator control between conservation practitioners and gamekeepers. For example, studies show that although reducing predator numbers during the breeding season may increase prey numbers at the end of the breeding season, this rarely results in increased breeding numbers in the subsequent season (Gibbons *et al.* 2007). Gamekeepers on the other hand are often more concerned about producing a surplus for shooting in the autumn/winter, and this is more achievable than increasing breeding numbers (Gibbons *et al.* 2007), yet does not benefit biodiversity conservation.

2.6 Supplemental feeding

Supplemental feed is provided on release estates to increase the survival of gamebirds and to ensure that breeding birds enter the nesting period in good condition. Food is normally provided in feed hoppers placed throughout pheasant woods, or less commonly by dumping piles of grain spoil or spreading grain along straw covered woodland rides. Although this food improves the condition of adult pheasants, and yields higher winter densities (Robertson *et al.* 1993), it does not increase breeding productivity, because chicks require a protein rich diet consisting principally of invertebrates (Sage *et al.* 2002a). The data available suggest that wild birds and mammals use supplemental food provided to gamebirds. However, pheasant hoppers may provide a point of parasite transfer to native wildlife, and pheasant feed in some cases contains levels of certain chemicals that may harm wildlife.

Following release at around 6-8 weeks of age, a protein rich grow mix is provided 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 hoppers. Pheasants that are fed for too long on feed mixtures do not adapt well to being released. Conversely, if 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).

2.6.1 Food for wildlife

Many previously common farmland birds have declined in recent years, and for many granivorous species this has often been attributed to a shortage of food during the winter months (Fuller *et al.* 1995). Species such as corn bunting, yellowhammer, and linnet have undergone declines in recent years, but may be benefiting from the provision of supplemental food at pheasant hoppers. Supplemental food is often considered beneficial to passerines (Hinsley *et al.* 1999, Stoate & Szczur 2001a), though evidence to support this at a meaningful scale is currently lacking (Arroyo & Beja 2002). In a study of 20 game-managed woodlands, abundance of ground feeding birds such as blackbird, dunnock, robin and wren increased with pheasant hopper density, as well as nuthatch and blue tit. However, song thrush and willow warbler abundance decreased with hopper density (Davey 2008). Another study of hopper use by wildlife showed that only ~22% of the time spent feeding at hoppers was by gamebirds. The remaining time spent at hoppers by wildlife was ~45% passerines, ~6% corvids, ~9% columbids and ~18% mammals (Draycott 2005). Furthermore, in an agriculture dominated estate, corn bunting, yellowhammer, reed bunting and linnet all used areas with pheasant hoppers more than most other areas, and habitat types (Brickle 1997). It therefore seems evident that many farmland and woodland passerines may feed on, and benefit from supplemental feed provided at pheasant hoppers, although it is unknown whether this is likely to benefit nationwide populations.

Small mammals may benefit from the provision of supplemental food (Flowerdew 1972). Wood mouse and bank vole populations increase with proximity to feed hoppers year round (Davey 2008). The level of excess grain (surplus to pheasant requirements) in hoppers increases the overwinter survival of wood mice, but excess grain at the end of the shooting season may have negative implications for small mammals. Davey (2008) suggests that this may be due to increased disease, predation and intraspecific competition around hoppers, and predation is likely to be an important factor after the shooting season when gamebirds are scarce (also see sections 2.4.4 and 6.3.1).

2.6.2 *Risks for wildlife feeding at pheasant hoppers*

Birds congregating at predictable feeding sites may increase the potential for predation and the spread of infectious diseases, or the contraction of parasites from pheasants (Pennycott 1998, Siriwardena *et al.* 2007). An unpublished veterinary account found 15 dead starlings under a roost in East Anglia, and tests revealed that they died of poisoning from an ionophore antibacterial agent called lasalocid. Lasalocid has not been detected in wild birds before, although the costs of testing have thus far prohibited extensive testing. Lasalocid is an additive in pre-mixes for young gamebirds and poultry, for the prevention of coccidiosis caused by the parasite *Eimeria spp* (see section 4). As poultry farms are subject to stricter regulation than pheasant release operations, it is more likely that these starlings fed on the pre-mix within or near a pheasant pen. Lasalocid is banned for use on egg-laying poultry in the EU because of health concerns relating to trace levels of lasalocid in eggs for human consumption.

2.7 Conclusions

Despite recommendations by the GWCT and others, best practice guidelines are likely to be adopted by only a fraction of game estate managers, particularly estates that work to encourage breeding gamebirds (17% of estates). Therefore some of the beneficial management practices mentioned here are not applicable at a large scale, and may have little impact at a national scale. However, it is likely that where it is implemented, habitat improvement by game estate managers is beneficial for certain wildlife.

Provision of supplemental feed to gamebirds can be considered primarily beneficial to wildlife if it is free from chemicals. Pure grains at pheasant hoppers are likely to aid wildlife despite the risks of disease transfer. However, enriched pre-mixes that are often provided in open-topped pheasant pens may represent substantial risks for wild birds. While it is necessary for disease in gamebirds to be controlled with the use of feed additives, game release estates and rearing farms should operate strict bio-security to prevent accidental poisoning of wildlife via supplemental food. Predator control on game estates may be beneficial to other wildlife, but there is still a lack of clear understanding as to whether reduced predator populations increase breeding numbers season-to-season, and therefore the significance of predator control remains somewhat unknown (Gibbons *et al.* 2007).

2.8 Key knowledge gaps and recommendations

- Data are not available to compare the actual quantified difference in area of various habitat management types between game and non-game estates.
- With the above data collected, estates could be scored in terms of their management, and then studies conducted on a sample of each of the scores.
- To fully understand the effects of habitat management on wildlife, the above data should be used to carry out experimentally designed studies.
- There is a lack of quantification of game conservation motivated uptake of agri-environment schemes.
- The impacts of novel dietary supplements, consumed by wildlife are unknown, and should be more extensively studied.

3. Direct impacts of gamebirds

3.1 Summary

At high densities, gamebirds may modify microhabitats such as the lower portions of hedges and woodland ground flora, as well as reduce hedgebank species richness and cause soil enrichment. This may affect the nest sites of birds such as yellowhammer that nest in the lower portion of hedgerows, and could subsequently reduce productivity. Released pheasants reduce the biomass of ground-active invertebrates and caterpillars, and as such, breeding gamebirds may act as a source of competition for other breeding birds during the nesting season, as both require high levels of invertebrate prey to feed to their young.

3.2 Background

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.* 2005). Outside of release pens, pheasants often make regular journeys to and from pens, following linear boundaries such as hedgerows and ditches, where they browse as they travel. Natural England cites game management and pheasant rearing as responsible for adverse conditions in 473 ha (0.4%) of SSSI's in England (Natural England 2010).

Pheasants are omnivorous, feeding on leaves, grain, seeds, berries and other fruits, green shoots, arthropods, molluscs and occasionally small vertebrates (Collinge 1927). Pheasants dig around in soil to expose new shoots as well as earthworms, and therefore create a certain amount of soil disturbance in addition to general browsing activities. Pheasant diet appears to be highly variable, and differs both temporally and spatially, with several studies showing varied results (Hill & Robertson 1988). Some trends 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, which is likely to represent availability rather than temporal selection (Pressland 2009). However, it seems that reared pheasants are usually more reliant on grain feeds provided at hoppers, and exhibit a less variable diet than wild pheasants. When birds are released at high numbers, they tend to disperse further from pens (Sage & Swan 2003), and often spill-over into adjacent farmland where they may damage crops, particularly oil seed rape, barley and potatoes. Additionally, breeding pheasants require large quantities of invertebrates to feed their young during the spring, therefore potentially competing with native breeding birds. In a study of 183 pheasant stomachs over a 12-month period, 37% of biomass consisted of arthropods, and 63% vegetable matter (Collinge 1927).

3.3 Browsing by gamebirds

3.3.1 Impacts of browsing by gamebirds on floral composition

Pheasants feed on multiple plant parts in Britain including fruits of apple, raspberry, blackberry, and hawthorn. They will also feed on acorns, hazelnuts, roots, bulbs, rhizomes, buds and tubers (Dalke 1937, Hill & Robertson 1988). In a comparison of ground flora between pheasant release pens and control plots, density of low cover (<0.5m) was reduced in pens, and there was a reduction in cover of shade tolerant perennials and winter-green perennials, and an increase in bare ground and species characteristic of disturbed, fertile soil (Sage *et al.* 2005). Subsequently, species diversity was lower in release pens compared with control sites. Pheasant stocking density had the strongest effect on vegetation characteristics with negative impacts usually seen at densities greater than 1000 ha^{-1}. Age of release pen and pen size were also important factors (Fig. 3.3.1.1). A similar study by Low *et al.* (2003) compared ground and field layer vegetation in a pheasant pen that had not been used for four years, with control plots (Fig. 3.3.1.2). This study found differences in floral composition within the pen, suggesting that even several years after pheasant use, differences in floral composition are still apparent. Eleven herb species including the protected bluebell were found in the adjacent control plots, but not in the old pen (Low *et al.* 2003).

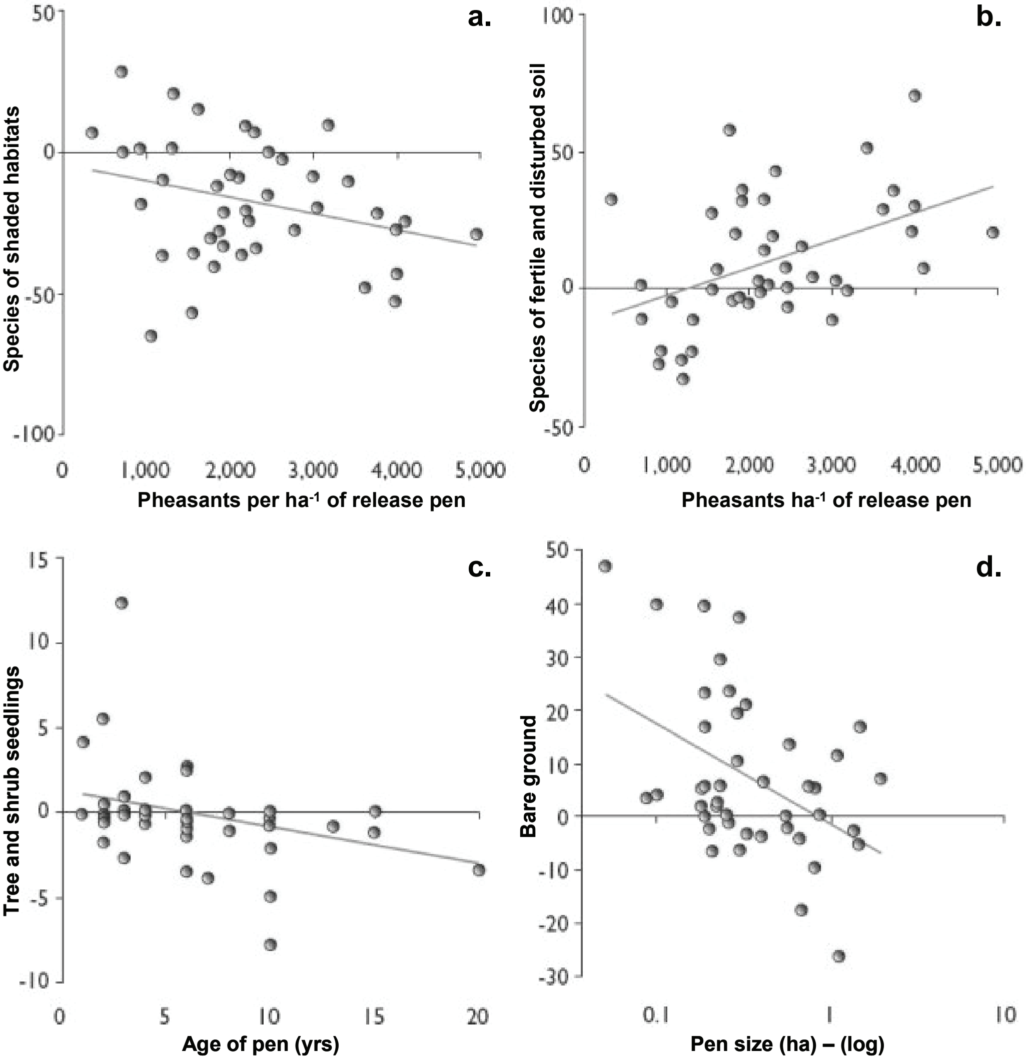

Fig. 3.3.1.1. Relationship between pheasant density per hectare of pen, and abundance of ground flora species characteristic of a. shaded soils (18 species) and b. disturbed, fertile soils (21 species). a and b show that negative effects are usually seen at densities > 1000 birds ha^{-1} of release pen. c. Relationship between age of pen (yrs) and abundance of tree and shrub seedlings (7 species). d. Relationship between log pen size (ha) and bare ground (% difference between pens and control). After Sage *et al.* 2005.

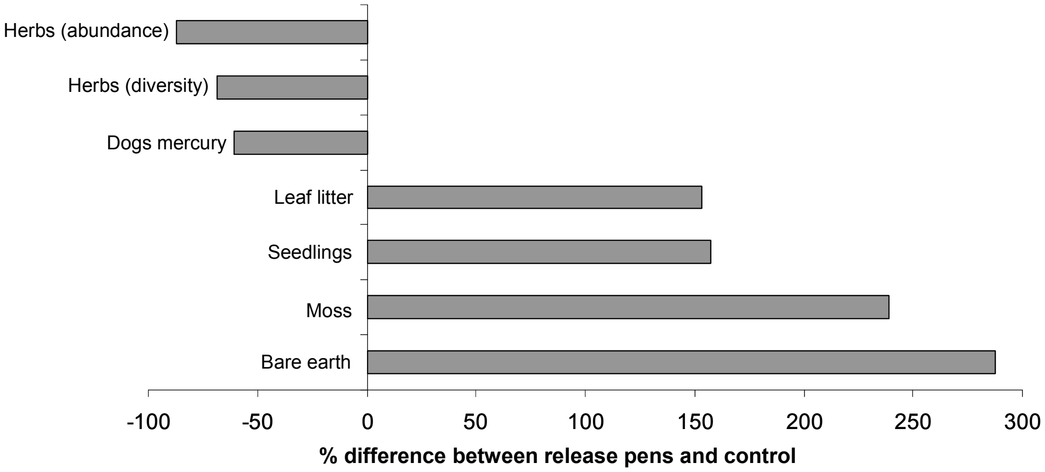

Fig. 3.3.1.2. Percent difference of ground flora characteristics between a pheasant release pen that had not been used for four years, and control plots. Positive (+ve) and negative (-ve) figures represent more and less in pheasant pens than control respectively. After Low *et al.* 2003.

Outside of release pens pheasants cause increased weed species richness and bare ground, and reduced numbers of stable perennials on hedgebanks close to release pens (Sage *et al.* 2009). In this study, release density was a highly influential factor regarding browsing impacts by pheasants. Hedgebanks in areas where >1000 pheasants were released, consistently contained fewer shrub and seedling species, and reduced shrub leafiness in lower portions (first 20cm) of hedges. Woodburn & Sage (2005) also found a strong negative correlation between pheasant release numbers, and the lower (11-40cm) herb cover of hedgerows adjacent to pens (Fig. 3.3.1.3.). Stratified by regional farming patterns (grassland or arable), these effects on weeds and stable perennials were consistent (Sage *et al.* 2009).

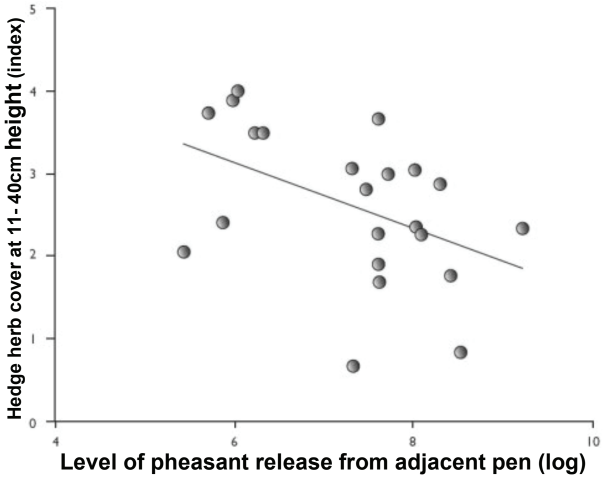

Fig. 3.3.1.3. Relationship between pheasant release numbers from pens (log), and adjacent hedgerow herb cover between 11-40cm in height, after Woodburn & Sage 2005.

3.3.2 *Potential impacts of browsing 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 micro-habitats, may affect nest sites for birds that nest on hedgebanks or in the lower portion of hedges. Species such as the red listed yellowhammer, and amber listed common whitethroat could be affected. Yellowhammer regularly nest in the lower portion of hedgerows, or in the vegetation below hedgerows, and are positively associated with wide grass margins and species rich verges adjacent to hedges that provide good foraging habitat (Bradbury *et al.* 2000, Green *et al.* 1994, Hinsley & Bellamy 2000).

Since 1966, yellowhammer numbers have declined by over 50% in the UK (Baillie *et al.* 2009). The primary understanding for this decline is a reduction in winter food availability caused by agricultural intensification (Gillings *et al.* 2005), as well as reductions in clutch size, brood size and nest success (Leech & Barimore 2008). On a localised scale, nest success could be related to a reduction in hedge herb cover and leaf density, and lower species diversity of perennials on hedgebanks may affect suitable nesting sites, and increase nest failure rates. Nests in sparsely vegetated hedges will succumb to higher predation rates as they are more easily detected and accessed by predators (Hinsley & Bellamy 2000).

On an estate where gamebirds were not released, Bradbury *et al.* (2000) found that yellowhammer nestling predation rate was significantly lower in nests situated in hedges than in all other nesting sites. In contrast, at the GWCT's Loddington game estate, Stoate & Szczur (2001b) found that predation rates were higher in nests located in hedges, and this could be due to changes in hedge structure caused by pheasant browsing. Although, compared with two adjacent non-game estates, Stoate *et al.* (1998) found that overall nest survival was higher on the game-managed estate, this is likely to be due to estate management. Despite this, during the period 1992-2006 the Loddington estate experienced a decline in yellowhammer abundance (Stoate 2007).

Increased predation in hedge located nests may not be the only mechanism by which predation rates could increase where hedge or hedgebank structure is unsuitable. Yellowhammer nest site selection changes over the course of the season in response to changes in vegetation (Bradbury *et al.* 2000). Earlier in the season, nests are positioned in grass or ditches below hedges, and later in the season, they are positioned in the lower portion of hedges (Bradbury *et al.* 2000, Stoate & Szczur 2001b). When hedges are not suitably vegetated, birds may choose to nest on the ground as they do earlier in the season. This may subsequently increase the average predation rate for the season, due to greater vulnerability to nest predation by mammalian predators such as foxes on the ground. Additionally, less experienced birds, nesting late in the season may select unsuitable hedges, and suffer increased nest predation as a result. While feasible, this mechanism is likely to be highly dependent on hedge vegetation type, as pheasants are unlikely to browse all hedge plant species. Additionally, yellowhammer often nest in grasses directly below hedgerows that are unlikely to be affected by pheasants.

Where impacts to hedgerows do occur, there are likely to be other consequences from a reduction in hedge herb cover and leaf density. Over the period 1965-2007, average yellowhammer first nesting date has shifted by about eight days, so they are now laying later in the season (Baillie *et al.* 2009). Later laying may be due to numerous factors, including reduced suitability of hedgebank and hedge vegetation in the early breeding season. Pheasant browsing could therefore have knock-on consequences for the number of breeding attempts possible each season. At a national scale, it is unlikely that these mechanisms have driven a population response, and are likely to be highly localised. However, the habitat modifications caused by pheasants may represent one of many components in the decline of yellowhammer, and potentially other similar species as well. In contrast, the provision of supplemental feed (see section 2.6) early in the breeding season, may encourage earlier breeding attempts in many granivorous farmland birds, and may balance any localised negative effects.

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 (Hinsley & Bellamy 2000, Parish *et al.* 1994, 1995). Therefore, where gamebirds released at high densities reduce hedgerow and hedgebank species richness, indirect effects may be seen in a range of farmland birds. Additionally, high densities of pheasants can result in the loss of larval food plants for Fritillary butterflies such as *Viola* spp. (Clarke & Robertson 1993, Ludulf *et al.* 1989), although game estate habitat management may increase the numbers of butterflies (see section 2.4.1 and 3.4.2).

3.4 Predation of invertebrates and vertebrates by gamebirds

3.4.1 Composition of fauna in gamebird diet

Adult pheasants and red-legged partridges consume a wide variety of fauna, mostly invertebrates but some vertebrates also. Invertebrate prey includes various beetles, spiders, ants, caterpillars, slugs and snails, earthworms, and flies among many others (Callegari 2006, Clarke & Robertson 1993, Collinge 1927). These protein-rich foods form up to 37% of the annual diet of adult pheasants (Collinge 1927). Additionally, gamebird chicks require a high protein diet consisting primarily of arthropod prey, particularly sawfly larvae that form a vital part of the pheasant chick diet (Robertson 1997). Like adult pheasants, chicks also require beetles, plant bugs, and caterpillars (Table 3.4.3).

3.4.2 Impacts of arthropod predation by gamebirds

It is probable that at high release densities, gamebirds will have short-term impacts on invertebrate abundance. The only comprehensive study tested invertebrate differences between 17-game-managed and 17-paired control estates (Pressland 2009). Game-managed sites released between 125 and 5000 pheasants per season, at densities of 6-255 ha^{-1} of estate. Ground-active invertebrates were sampled at woodland edge habitats, with pit-fall traps 2m either side of the woodland-field frontier. The overall biomass of ground-active invertebrates was lower in spring (May to June) where pheasants were released compared with control sites, and this probably represents predation of overwintering invertebrates during the shooting season when gamebirds are abundant (Pressland 2009). Specifically, carabidae (ground beetles), curculionidae (weevils) and tetragnathidae (long-jawed spiders) exhibited significantly lower biomass in spring at field edges where pheasants were released, and carabidae also showed a lower biomass within woodlands in autumn where pheasants were released. In contrast, chrysomelidae (leaf beetles) exhibited a greater overall biomass in areas where pheasants were released, possibly due to management, or to decreased competition from invertebrate groups that were less abundant where pheasants were released. In autumn, immediately after pheasant release (October), there were no overall differences in ground-active invertebrate biomass, and this was probably due to the lack of pheasants during the summer months.

Pressland (2009) also studied pheasant impacts on the biomass of woodland lepidoptera larvae (butterflies and moths), and found a significant negative relationship between pheasant release densities and caterpillar biomass (Fig. 3.4.2). There were some differences in the exact variables included in closely competing models, with plant diversity and temperature in one, and pheasant release density in another, but these models could not be separated statistically. This study therefore provides strong evidence for an impact of pheasant predation on the biomass of caterpillars that are an important food resource for many nesting birds in spring (Evans *et al.* 1997). Using data from the UK Butterfly Monitoring Scheme, Pressland (2009) also compared abundance and diversity of day-flying adult lepidoptera between game-managed and non-game woodlands. No differences were detected, and game management did not predict abundance or diversity of adult lepidoptera in multivariate analyses. The mechanisms behind this are not explicitly clear as caterpillars are less abundant in game-managed woodlands, but it is probable that immigration or reduced intraspecific competition in caterpillars may be important. Notably, game-woodlands did not contain more adult butterflies than non-game woodlands, contrary to less comprehensive studies that found otherwise (Clarke & Robertson 1993, Robertson *et al.* 1988).

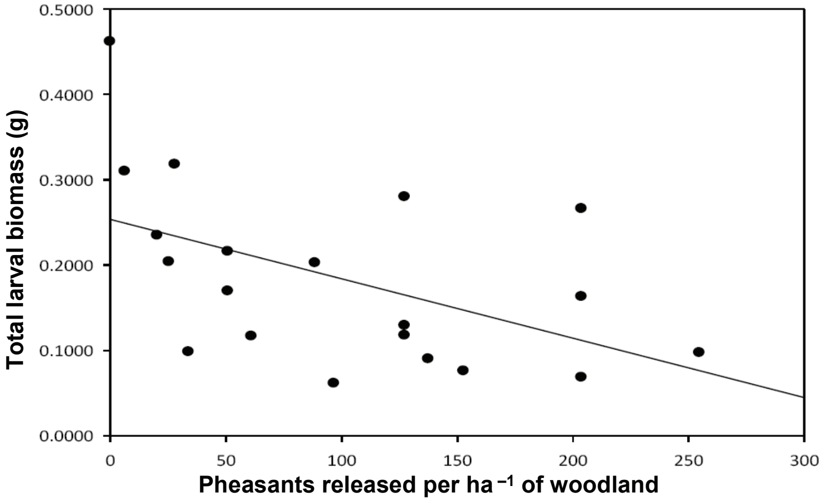

Fig. 3.4.2 Relationship between lepidopteran larval biomass, and pheasant release density ($R^2 = 0.258$, $F_{1,18} = 6.245$, $p = 0.022$) from Pressland 2009.

Callegari (2006) found no effect of pheasant and red-legged partridge on the numbers of beetles, spiders, and Adonis blue butterflies in open plots compared with plots where gamebirds were excluded on chalk grassland. It is unknown at what densities these gamebirds were released, making it difficult to infer these results in the wider context. However, the sample size for Adonis blue butterflies was insufficient to detect differences, and the subterranean nature of Adonis blue larvae make them largely inaccessible to gamebirds, and therefore this species is unlikely to be negatively impacted by gamebirds. Additionally Callegari (2006) sampled just one site, and therefore the results found in the study by Pressland (2009) are substantially more reliable. It therefore seems probable that pheasants impact upon ground-active invertebrate and caterpillar populations during the shooting season, reducing this resource for birds in the subsequent breeding season.

3.4.3 *Resource competition with breeding gamebirds*

Wild pheasants and red-legged partridges breed during the same months as most breeding birds in the UK. The chicks of gamebirds and many native birds require a protein rich, invertebrate diet to feed their young, and a reduction in invertebrate abundance caused by agricultural intensification is thought to be one of the leading causes of recent population declines and range contractions in many farmland birds in the UK (Fuller *et al.* 1995, Vickery *et al.* 2009). Resource competition for invertebrates may lower the breeding productivity of native birds where gamebirds breed in abundance. Table 3.4.3 demonstrates the overlap in the importance of various arthropod species in the chick diet of pheasant and three native, declining farmland bird species. This shows a large degree of overlap between pheasants and the three native species examined. In particular, yellowhammer, and corn buntings may compete with pheasants for sawflies and caterpillars, and grey partridges may compete with pheasants for aphids, bugs and flies. There are no data available on actual quantities of invertebrate prey items consumed, however as pheasant are approximately 20 times the mass of corn buntings and yellowhammer, and 3 times the weight of grey partridge, it is likely that pheasant chick arthropod consumption is correspondingly higher than 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 non-native gamebird chicks could therefore play an important role in reducing food availability for other farmland birds.

Table 3.4.3. Chick dietary composition (% in diet) of arthropods for pheasant (Hill & Robertson 1988), grey partridge (Green 1984), yellowhammer (Stoate *et al.* 1998), and corn bunting (Brickle & Harper 1999). Farmland species 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

3.5 Soil enrichment

3.5.1 *Soil enrichment by pheasants in woodland release pens*

Pheasants stocked at high densities (particularly in release pens) inevitably lead to soil enrichment caused by excretion and increased disturbance. This can significantly increase phosphate and potassium levels, but does not modify magnesium or soil pH (Sage *et al.* 2005). As a consequence, Sage *et al.* (2005) found that plants such as stinging nettle and rough-stalked meadow grass are unaffected by the presence of pheasant release pens, but plants characteristic of disturbed soils, such as annual meadow grass, broadleaf dock and common chickweed, are often more abundant in release pens. Consequently, stable perennials are less abundant in pheasant release pens, and Low *et al* (2003) demonstrate that release pen floral composition may not recover for several years following use by pheasants (see section 3.3.1).

3.5.2 *Soil enrichment by red-legged partridge on moorland fringe*

In recent years red-legged partridges and pheasants have been released more frequently on the edge of upland moorland, and this pattern may reflect an attempt to compensate for declining grouse bags. Craig Leek, a SSSI in Scotland is home to an extremely rich bryophyte community with eight Nationally Rare species, and one Red Data Book species. Recently, a game estate adjacent to Craig Leek began releasing red-legged partridges. These birds roosted on the crags at Craig Leek, causing soil eutrophication with detrimental effects to the fragile bryophyte community there (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.

Moorlands host internationally important habitats, and a number of important bird species, many of which are either of conservation concern (e.g. upland waders, black grouse) or are of economic importance (e.g. red grouse). Additionally, population trends for birds associated with moorland habitats show some of the greatest declines of all UK bird species. Many upland waders and gamebirds have undergone major declines in the last 20 years (Sim *et al.* 2005). Red grouse has recently been added to the new revised UK BAP priority list, together with Curlew and Lapwing (Eaton *et al.* 2009). Thus, upland bird populations are likely to be particularly vulnerable to any negative influences in their environment caused by gamebirds and associated activities. There is therefore a pressing need to understand the consequences of gamebird releases on this fragile and internationally important ecosystem (also see section 4.5.4).

3.6 Conclusions

At high release densities, it appears that gamebirds may significantly impact woodland ground flora in release pens. More importantly though, gamebirds may also modify microhabitats outside of release pens, and the conservation implications of this, particularly for farmland and moorland fringe species may be substantial. Gamebirds may even browse important protected plants including bluebells, and possibly others. Pheasants both reduce the biomass of ground-active invertebrates and caterpillars, that are important breeding resources for nesting birds, and additionally breeding gamebirds may directly compete with native species for invertebrates to feed their young.

Most game organisations are working to increase gamebird breeding numbers, and with a larger breeding stock, resource competition for chick nutrition could become more widespread. Efforts to increase breeding productivity in gamebirds are usually in conjunction with management that includes provision of extra foraging habitat, and may therefore balance the potential negative impacts. However, given that gamebirds decrease the abundance of overwintering invertebrates, efforts should be made to reconcile the negative effects of gamebird invertebrate predation.

3.7 Key knowledge gaps and recommendations

- There is a fundamental lack of follow-up on the consequences of direct gamebird impacts on native biodiversity. Changes in woodland lower-level structure may have knock-on effects for numerous groups of wildlife.
- The effects of different pheasant densities should be studied in relation to Yellowhammer nesting ecology and success.
- It is unknown whether reduced invertebrate biomass caused my gamebirds in one year, leads to a subsequent proportional reduction in the same area for the following year. The impacts of gamebirds on invertebrates should be sufficiently studied to detect population level impacts year-on-year.

4. Gamebirds and disease

4.1 Summary

Studies that demonstrate or even imply the transfer of parasites from non-native 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 wildlife of hosting various parasites have not been extensively quantified, except where significant outbreaks have occurred, though even in these cases the source of the outbreak is rarely well understood. The most detailed studies show that pheasants appear to act as a reservoir of *Heterakis gallinarum* which they pass on, directly or indirectly to grey partridges that are not resistant to the parasite. Pheasants are also prolific transmitters of the bacteria responsible for Lyme disease, which infects passerines and humans via a tick vector.

4.2 Background

4.2.1 Conditions that promote parasite infestation

Pheasants and partridges are prone to high levels of parasitic infection (Draycott *et al.* 2006), and at high stocking densities, environmental conditions promote parasite infestation and therefore increase the risk of disease outbreak (Coles 1984, Gortazar *et al.* 2006, Jansson *et al.* 2001, Millan 2009, Millan *et al.* 2004). Captive bred animals carry much higher parasite loads than their wild counterparts (Millan 2009), because at higher stocking densities, gut parasites are more likely to be picked up by another bird after they are excreted. Wild pheasants are much more successful breeders in Britain than captive bred pheasants, and this has been attributed in part to these discrepancies in parasitic burdens (Draycott *et al.* 2006). In the wild, infections tend to be self-limiting 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. While most birds naturally act as hosts for many parasites, over-stocking and mismanagement in livestock industries can cause problems to be exacerbated, allowing new pathogens to enter the system, and existing ones to become more ubiquitous (Tapper 1999). In reference to *Salmonella pullorum* in released pheasants, Pennycott & Duncan (1999) state:

> "… 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."

There are measures in place to prevent gamebirds imported from France introducing notifiable disease including avian influenza (H5N1) and Newcastle disease. These include surveillance in France and the requirement of official veterinary health certification on all imports. Additionally, in France enhanced biosecurity is being enforced on game farms in areas identified as being at risk from migrating birds. There is currently no evidence that the H5N1 virus is present in the gamebird exporting region of western France, however, should the virus be found, EU law would ban the trade in live gamebirds, at least temporally (DEFRA 2010). In 2005, 9,000 pheasants were culled following an outbreak of Newastle disease on a game estate in Surrey. The birds were sourced from a game farm in France, that was later confirmed to be the origin of the outbreak. This prompted a temporay ban on live bird imports into the UK. Newcastle disease also affects pigeons and grey partridges.

4.2.2 Risks to native wildlife

Non-native introductions into the wild include associated parasites that native wildlife may be naïve to, particularly when the strain does not naturally occur within their range (Manchester & Bullock 2000). However, parasites are often specific to narrow ranges of host species, so threats of transfer from pheasants and red-legged partridges are in many cases likely to only affect closely related species, such as grey partridge (Hudson 1997). Additionally, many of the diseases associated with gamebirds are restricted to conditions in rearing houses, and are therefore not likely to represent a major risk to wildlife. Despite this, a number of 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 even people (Kurtenbach *et al.* 1998a, Pennycott *et al.* 1998). Pennycott & Duncan (1999) suggest that gamebirds represent a source for the spread of *Salmonella pullorum* despite the disease being controlled in the poultry industry. Passerines (particularly finches) are commonly infected by various *Salmonella* strains, and often come into close proximity with gamebirds at feeders.

4.2.3 Medications

A number of medications are used to treat gamebirds during rearing, in release pens, and at feeders around release estates. Most farm-reared gamebirds are fed with enhanced mixes that include proteins, vitamins, minerals, energy and other necessary nutrients. While many rearing farms treat signs of parasitic infection with coccidiostats and antibiotics, others use them in everyday feed mix to act as a preventative. All rearing farms and release pens are required to operate high levels of bio-security to prevent transfer in either direction. In 2003 a widely used product called Emtryl was banned following concerns regarding the human consumption of game treated with this product. Emtryl was used to treat and prevent the protozoans *Trichomonas* and *Hexamitiasis*, however the cessation of its use prompted significant management and husbandry changes in gamebird rearing farms, primarily focusing on reducing stress and densities.

4.3 Endoparasites - helminthes and protozoans

In Britain, the most common helminthes 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). Few studies have related helminthes to other wildlife with the exception of that which is discussed in section 4.5.3.

Protozoans such as *Trichomonas* and *Histomonas*, 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 (Bunbury *et al.* 2008, Duff *et al.* 2003, Hofle *et al.* 2004, Villanúa *et al.* 2006b). *Trichomonas gallinae* is passed from bird to bird, often at feeding and drinking stations (Gortazar *et al.* 2006, Villanúa *et al.* 2006b). This parasite is most common in gregarious species such as finches and sparrows, but may also be passed to birds of prey after consumption of an infected animal (see section 4.5.3). Infection by *T. gallinae* 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, Villanúa *et al.* 2006b)

4.4 Lyme disease

4.4.1 *Lyme disease and pheasants*

Pheasants may increase potential levels of Lyme disease in the British countryside as they are competent reservoirs for the bacteria *Borrelia burgdorferi* sensu lato which is transferred to humans and other vertebrates via a tick vector. Compared with rodents, pheasants have been shown to be 55% more effective at transmitting *B. burgdorferi* to ticks (Kurtenbach *et al.* 1998b), despite several studies that show rodents are more frequently carriers of the bacterium compared with birds (Humair *et al.* 1993, Sinski & Karbowiak 1994, Sinski *et al.* 2006). This suggests that despite higher infection rates in rodents, pheasants are more prolific transmitters of *B. burgdorferi*. It is important to note that deer and fox are also important hosts for *B. burgdorferi* and their expansion is also likely to lead to higher levels of potential Lyme disease infection in the British countryside (Hudson 1997).

4.4.2 *Lyme disease and wildlife*

Many birds and rodents act as reservoirs for *B. burgdorferi*. The disease stage of the bacterium's life cycle is currently only known to occur in humans and in some cases cattle. However, the prevalence of Lyme disease has not been extensively tested in wildlife populations, though Ginsberg (1994) suggest that it may pose a significant threat to wildlife populations. Nevertheless, this has not yet been substantiated.

Gryczynska *et al.* (2004) tested passerines for *B. burgdorferi* in Poland. The bacterium was detected 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 affect 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 are subject to higher mortality rates than uninfected birds.

In humans, exposure risk to Lyme disease, and vertebrate species diversity in north America are negatively correlated (Ostfeld & Keesing 2000, Schmidt & Ostfeld 2001); i.e. where just few vertebrate species dominate, Lyme disease incidence is higher. Therefore, where gamebirds are released at high densities, it seems plausible that Lyme disease may be more prevalent. This hypothesis has not yet been tested further or in the UK though. Keesing *et al.* (2006) further demonstrate that this is not only relevant to Lyme disease, as any parasite which is involved in the regular evolutionary arms race is likely to benefit from a reduction in species diversity caused by the dominance of just a small number of species (also see section 7.4).

4.5 Cases of parasite transfer from non-native gamebirds

4.5.1 *Pheasant and grey partridge*

Pheasant and grey partridge share numerous parasites (Table 4.5.1.), but the origin of the parasites is usually unknown. A study by Ewald & Touyeras (2002) investigated the spatial relationship between pheasant release pens and red-listed grey partridge population parameters, with respect to the grey partridge decline in the UK. They found no evidence to suggest that pheasant release influenced grey partridge declines, and therefore conclude that parasites transferred between these species are not an influential factor of the decline. In contrast, Tompkins *et al.* (1999, 2000a, 2000b) studied the role of the macro-parasite *Heterakis gallinarum* within spatially intersecting pheasant and grey partridge populations. Field studies showed that levels of *H. gallinarum* in grey partridges on a pheasant release estate were significantly correlated with *H. gallinarum* levels in pheasants. This study showed that grey partridge were not tolerant of the parasite, which lowered fecundity and was significantly negatively correlated with body condition. After experimental trials, a model predicted that spatial analogy between grey partridge and pheasant populations greater than 57% would cause the exclusion of grey partridge, due to lower parasite resistance. The model further predicted that in absence of a pheasant population, the parasite is not present in grey partridge. This is because *H. gallinarum* is unable to persist in grey partridge alone as its reproductive rates are greatly lowered, and grey partridge are able to expel *H. gallinarum* through their gut at much higher rates than pheasants. Since *H. gallinarum* has been detected in wild grey partridge (Clapham 1935, Keymer *et al.* 1962), and this model suggests that it cannot exist in grey partridge in the absence of a shared host, it is likely that some parasite transmission to this species does occur from other reservoirs in the wild. Tomkins *et al.* (2002) also found that *H. gallinarum* is similarly unable to persist in red-legged partridges, and therefore they attribute pheasants as the primary reservoir of this parasite in UK gamebirds.

In response to Tompkins *et al.* (1999, 2000a, 2000b), 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 tolerant. The major difference between these studies was that Sage *et al.* (2002b) injected the parasite directly into the birds within a controlled environment. In the Tompkins *et al.* (1999, 2000a, 2000b) studies, the parasite was transmitted ad-hoc 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 the similar grey partridge to greater effect. Likewise, the Tompkins *et al.* (1999, 2000a, 2000b) studies are more representative of conditions in nature. Despite these conflicting results, it seems plausible that parasite transmission from pheasants at high densities may be impeding the current efforts to restore grey partridge numbers in the UK.

Table 4.5.1. Endoparasites of pheasant and grey partridge (after Vrezec 2006).

Parasite	Pheasant	Grey-partridge
Trematoda		
Echinoparyphium cinctum	+	
Brachylaemus fuscatus	+	
*Postharmostomum gallinum**	+	+
Cestoda		
Davainea proglottina		+
Raillietina echinobothrida	+	
*Choanotaenia infundibulum**	+	+
Rhabdometra nigropunctata		+
Drepanidolepis anatina		+
Passeripelis crenata	+	
Nematoda		
*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.

4.5.2 *Red-legged partridge and little bustard*

With the exception of the above example, there is little other evidence available for the direct transfer of parasites from non-native gamebirds to native wildlife in the UK. One study in Spain describes a nematode infected little bustard, living adjacent to a red-legged partridge estate (Villanúa *et al.* 2007). The game estate released about 3000 birds annually and was less than 5km from where 14 bustards were captured. Of the 14 bustards captured for radio-tagging, one was underweight and died during capture. An autopsy revealed three males and two gravid females of the nematode *Eucoleus contortus,* which is not known to inhabit members of the bustard family. In contrast, *E. contortus* was present in 7.7% of red-legged partridges from the neighbouring hunting estate (Villanúa *et al.* 2007). This case may be of concern for the great bustard UK reintroduction project that began in 2004 (see Dawes & Szekely 2008).

4.5.3 *Red-legged partridge, woodpigeons and Bonnelli's eagle*

Several studies in Spain have highlighted the dangers of protozoan pathogens being passed from congenial wildlife at supplementary feeders. An outbreak of *T. gallinae* in woodpigeons on a game estate is thought to have been transmitted via red-legged partridges at feeders (Hofle *et al.* 2004). *T. gallinae* has also been found to cause high mortality in Bonelli's eagle chicks, believed to have been contracted from the consumption of infected prey (Real *et al.* 2000, Sansano-Maestre *et al.* 2009).

4.5.4 Gamebirds and moorland fringe habitat

Red grouse have undergone declines in recent years resulting in their classification on the UK amber list. 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, and maintain shooting interests and associated jobs. In 2002, six red grouse were found infected with *Heterakis gallinarum* on a Yorkshire moor. Pheasants that had been released nearby were thought to be the most likely source of the infection, as *H. gallinarum* is common in pheasant yet very uncommon in the red grouse (GCT 2003). It is currently unknown whether *H. gallinarum* could induce a population response in red grouse, however where pheasant and red-legged partridge come into contact with red grouse and fragile populations of waders on moorland fringe habitat, there is grounds for concern. If *H. gallinarum* negatively impact red grouse populations, and lower densities of red grouse contribute to estates initiating releases of other gamebirds at moorland fringe, then there may be a feed-back loop of considerable economic and environmental consequence.

4.6 Conclusions

Pheasants and red-legged partridges are subject to parasites loads from numerous different groups, including protozoa, helminthes (nemamtodes, 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, and therefore understanding direct associations regarding parasite transfers are either anecdotal, or simply not possible with the available literature. 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 parasite loads increase, and new pathogens are also gained (Villanúa *et al.* 2006a). Treatment is continued via supplementary feeders that also attract other birds (Millan 2009). While supplying food to wild birds is of benefit to biodiversity conservation (see section 2.6), if captive bred pheasants act as primary reservoirs for parasites, feeding stations pose a threat for wild birds through the transfer of these parasites. Additionally, as reservoirs for Lyme disease, pheasants may pose a disease threat for humans (see section 7.4)

While game rearing farms do treat birds, there are no regulations, and therefore no consistent records of the levels at which parasites 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 treated feed. At high release densities, pheasants disperse further from release pens (Sage & Swan 2003), and are therefore less likely to use hoppers containing treatment regularly. Therefore, pheasants released at high densities are likely to represent greater parasite transfer threats to native wildlife than those released at low densities.

4.7 Key knowledge gaps and recommendations

- The source of many parasites are not well understood, and in many cases studies have not comprehensively tested the parasite link between gamebirds and other wildlife, making the evidence presented here largely anecdotal or isolated.
- No known studies have investigated population impacts of disease on birds in the UK.
- It vitally important to understand whether red-legged partridges released on the moorland fringe habitat may influence disease and/or parasites of red and black grouse, and the fragile wader species in this habitat.
- A very useful study would test a few sedentary wildlife species for multiple parasites on game estates compared with control estates.

5. Shooting practices on game estates

5.1 Summary

Effects of shooting practices on game estates include unintentional by-catch, and the use of toxic lead shot. Lead poisoning via the ingestion of lead shot can act directly or indirectly. Galliforms ingest shot that they mistake for grit, and raptors can suffer secondary poisoning following the consumption of lead ladened prey. Lead may also be absorbed into the soil where it is absorbed by soil invertebrates that can subsequently pass-on the lead through the food chain. Lead poisoning can cause quick or gradual deaths, and cause body abnormalities as well as lowering chick survival rates. On shooting estates, unintentional by-catch of wild grey partridge can severely impact breeding populations.

5.2 Background

Wildfowling practices are subject to stringent safety procedures, however environmental effects of shooting are often overlooked. Misidentified birds may be shot unintentionally, and toxic lead shot is frequently used. Spent lead shot may be found in abundance on the ground of typical shooting locations. Each cartridge contains ~280 shot, most of which does not penetrate the game animal, leaving several hundred thousand shot per hectare of soil (De Francisco *et al.* 2003). By 2000, the use of lead ammunition over wetlands in Britain was banned, following abundant evidence that the use of lead shot causes high levels of lead poisoning in waterfowl (Mateo 2009, Pain *et al.* 1992, Scheuhammer & Norris 1996). Lead poisoning in other habitats has been less widely reported, and hence lead shot is still prevalent in hunting activities of non-wetland game in the UK. Newly emerging evidence is now showing that similar effects (though possibly less marked) are occurring within a wide variety of non-wetland game and other wildlife (Fisher *et al.* 2006, Pain *et al.* 2007, Pain *et al.* 2009). Many bird species have been found with high concentrations of ingested lead shot in their digestive systems, or elevated bone or organ lead levels. These include 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 ingesting lead or suffering lead poisoning from ammunition, including 10 Globally Threatened or Near Threatened Species (Pain *et al.* 2009).

5.2.1 Species most at risk from lead

Two non-wetland bird groups are at particular risk from lead poisoning. Galliforms ingest lead fragments or pellets that they misidentify as the small stones or grit that they use to assist digestion in their gizzards (Fisher *et al.* 2006). In contrast, raptors are poisoned by ingesting lead shot or bullet fragments present in their prey (Kendall *et al.* 1996, Pain *et al.* 2009).

In the UK, four red-listed bird species have been documented as suffering from lead poisoning: grey partridge, herring gull, white-tailed eagle and hen harrier; and a further four amber-listed: red kite, western marsh-harrier, honey buzzard and golden eagle (Knott *et al.* 2009).

5.2.2 Mechanisms of lead poisoning

Lead toxicosis occurs following ingestion at sufficient concentrations. After lead is ingested, it spends approximately two weeks in the bloodstream, and then begins to be deposited into soft tissues such as the liver and kidneys, where it may remain for several more weeks. A small amount lead is expelled through excretion, but the remaining lead is eventually absorbed into bones following a similar path to calcium. Lead is subsequently retained in bones for up to tens of years, and can be mobilized back into the bloodstream as bones change throughout life. Younger individuals are therefore at greater risk from low concentrations of lead as bones are actively growing. When lead-poisoned animals are consumed by scavengers, predators, or humans, lead is passed-on through the food chain. Secondary lead toxicosis is therefore more likely when lead is still mobile in soft tissues, suggesting that during the hunting season this is more problematic.

Lead exposure may also cause less direct effects, such as thinning in eggshells (Grandjean 1976), deficiencies in body size, and in brain, liver and kidney weights, as well as reduced survival rates and deficiencies in major organ function (Hoffman *et al.* 1985a, Hoffman *et al.* 1985b).

5.3 Lead and birds of prey

Raptors are of particular concern regarding the threat of lead poisoning in the UK. Birds of prey may ingest lead from un-retrieved game, injured game, or from birds that have ingested shot themselves (Fisher *et al.* 2006, Pain *et al.* 2009). Scavenging raptors are therefore at high risk from secondary poisoning. Kites, vultures and other carrion feeders are more likely to feed on un-retrieved game, and in cases where carcass densities are high, removal by scavengers can be rapid (Pain 1991).

The digestive chemicals in birds of prey are highly acidic to break-down prey items, and are therefore efficient at dissolving lead that is subsequently absorbed into the bodily system, instead of being passed through the gut (De Francisco *et al.* 2003, Pain *et al.* 2007). In many cases, birds in apparently good physical condition suffer illness and quite sudden death following lead ingestion (Gill & Langelier 1994). Regurgitation of pellets 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). Repeated lead exposure in bald eagles still resulted in lead poisoning despite most shot being regurgitated via pellets (Pattee *et al.* 1981).

5.3.1 Lead poisoning in birds of prey 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 in one peregrine falcon (4% of species sample) and one buzzard (2% of species sample). An additional one peregrine and one buzzard had liver lead concentrations of 15-20 ppm, reflecting unusually high absorption of lead. No individuals of any other species had >15 ppm, although some had 6-15 ppm. Pain *et al.* (1995) suggest that these birds were likely to have ingested lead gunshot from their prey. In sparrowhawks, for which a large sample was available, lead concentrations in all but one individual were low (<2.6 ppm). Lead concentrations in juveniles were significantly lower than in adults, and were lowest in recently fledged individuals.

In a survey of 49 wild red kite carcasses in the UK, six (12%) had liver lead concentrations sufficient to cause mortality. Four (9%) of these were caused by the ingestion of spent lead ammunition, and two were intentionally poisoned (Pain *et al.* 2007).

5.3.2 Lead poisoning in birds of prey overseas

While studies overseas are more numerous, there are no reasons to suggest that effects on British species should fundamentally differ. In the United States and Canada, lead poisoning in bald eagles and golden eagles accounts for an estimated 10-15% of post-fledging mortality (Scheuhammer & Norris 1996), and Elliot *et al.* (1992) found that 37% of sick, injured or dead bald eagles exhibited significant lead exposure. In Spain one golden eagle died after ingesting 40 lead shot (Cerradelo *et al.* 1992), and ingested lead shot is reasonably common in Spanish imperial eagle and red-kite (Mateo *et al.* 2001).

5.4 Lead and Galliforms

In the UK and Europe lead poisoning has been detected in pheasant (Butler *et al.* 2005), and grey partridge (Potts 2005), and in the USA numerous other galliforms have been discovered with elevated concentrations of ingested lead, including Chukar partridge, scaled quail, northern bobwhite quail, ruffed grouse and wild turkey (Fisher *et al.* 2006). Of 1318 grey partridge carcasses examined between 1947-1992 in southern England, 18 (1.4%) were killed by elevated lead exposure (Potts 2005). In this study, birds that ingested one to three shot suffered mortality from lead toxicosis 38% of the time, however birds that ingested four or more shot died from lead toxicosis in 100% of cases.

5.5 Lead in the environment

5.5.1 *Ingested lead by other birds and wildlife*

Lead concentrations in passerines and mammals were tested on a military shooting range in Georgia, USA. 33% (24 of 72) of birds and mammals indicated lead exposure of clinical or sub-clinical levels (Lewis *et al.* 2001). Lead toxicosis related deaths were found in a group of yellow-rumped warblers, a grey-squirrel, and a solitary vireo, and elevated (>1ppm wet weight) lead levels were detected in 11 of the 22 species tested (5 mammals, 6 birds). Lewis *et al.* (2001) suggest species that feed on seeds that look similar to lead shot are more often affected. Although this study suggests that many wildlife groups are at high risk from lead poisoning, the amount of shot expelled at a firing range (5 million rounds per year), is likely to be higher than that on most game shooting estates, and therefore wildlife encounter rates with lead shot will be considerably lower on game estates. However, estates that operate particularly high density shooting may experience similar levels of lead poisoning in wildlife to that seen at this shooting range. In wetlands it is estimated that lead shot debris density varies from 40,000-180,000 pellets per hectare in the first 20cm of soil, with some sites reaching 300,000 pellets per hectare (De Francisco *et al.* 2003). Mateo (2009) reports lead shot levels as high as 399/m^2, (equivalent to 3,990,000 per hectare) in southern Europe.

5.5.2 *Lead ascending the food chain*

Lead may ascend multiple food chain levels. In two nature reserves adjacent to industrial areas of metal pollution in Holland, shrews and voles suffered high levels of lead exposure (Ma *et al.* 1991). Shrews exhibited particularly high lead levels, which were ingested following consumption of contaminated lumbricid earthworms, and other invertebrates. Voles on the other hand absorbed the lead from contaminated grasses and bryophytes. Lead levels were considerably higher in shrews than in voles, which is consistent with other insectivores such as moles (Ma 1987). These data indicate that on high intensity shooting estates, soils could become lead contaminated and species that feed on invertebrates, particularly earthworms, in areas exposed to lead debris, may be at risk from secondary lead contamination in the UK. Soil invertebrates form the initial stage at which lead can then enter the wildlife foodchain from contaminated soils. Therefore, blackbirds, robins, and thrushes, feeding on game estates may be at risk from secondary poisoning. This is likely to be most marked in areas with low soil pH, such as heathlands and conifer woodlands where high acidity may absorb lead pellets into the soil most effectively (De Francisco *et al.* 2003). Subsequently, birds of prey that feed on contaminated small mammals or passerines may ingest lead in the process.

5.6 Population level responses to lead

No known studies have directly quantified population level responses to lead poisoning on lowland game shooting estates. It is likely that the populations of some rare species may be heavily impacted however. Lead toxicosis is thought to be a major cause of the decline in the critically endangered Californian condor, and has severely hindered re-introduction efforts

(Cade 2007, Fisher *et al.* 2006). On wetlands, Mateo (2009) suggests a population level response to lead poisoning in several waterfowl.

5.7 Unintentional by-catch of grey partridge

Unlike red-legged partridges, grey partridges are native to the UK, but have suffered declines of more than 80% over the last 40 years. Nonetheless grey partridge are still a quarry species and are shot both intentionally and unintentionally (Watson *et al.* 2007). The GWCT recommend that grey partridge should only be shot where wild densities are >20 per 100 ha (0.2 ha^{-1}) (Aebischer & Sotherton 2002). However, commercial shooting of red-legged partridges often results in density-independent mortality of grey partridges, because the number of shoot days depends on the number of birds released not wild bird density (Watson *et al.* 2007), and these two species can be difficult to differentiate in flight. Therefore, shooting activities may result in population impacts on wild grey partridges, even where they are released. In a comparison across three shooting estates, and one non-shooting estate by the GWCT, unintentional grey partridge by-catch ranged from 37-91%, and reduced overall population numbers by 16-71% on the shooting estates. In contrast, on the non-shooting estate, grey partridge numbers remained stable over the study period (Aebischer & Sotherton 2002). Watson *et al.* (2007) suggest that warning signals such as whistles should be used by beaters (in driven game shoots), when non-target species are approaching the gun line, but this is not likely to be 100% effective even where implemented. Therefore, following guidelines set out by the GWCT, grey partridge should only be shot where they are found in sufficient densities, and to mitigate overshooting, only a proportion of available red-legged partridges should be shot, e.g. in 1977 Potts suggested just 40% (Aebischer & Sotherton 2002). From a financial perspective, game managers are unlikely to release 60% more red-legged partridges, and therefore by-catch mitigation remains a problem which may be difficult to resolve.

5.8 Conclusions

Birds of prey are particularly at risk from spent lead ammunition on game shooting estates. In the UK, many raptor species have suffered various anthropogenic impacts and all raptor species that feed on game could potentially be exposed to lead poisoning. Some individuals may target game animals as food resources and in these cases may show the highest lead concentration and risk of toxicosis (Komosa & Kitowski 2008). The likelihood of lead exposure may vary according to the proportion of game in the diet, the size of game taken, the season, and the local hunting intensity (Pain *et al.* 1993). Birds that have been exposed to lead in a particular area may also be less likely to return to that area to breed (Mateo *et al.* 1999). Many good alternatives to lead ammunition, mainly made from steel or copper, have been developed, and succeed at replacing most game hunting ammunition types (Knott *et al.* 2009, Oltrogge 2009). However, these will not be adopted without regulation, as they currently tend to be more expensive. In addition to lead associated issues, measures should be taken to reconcile the apparent impact of shooting activities on grey-partridge populations.

5.9 Key knowledge gaps and recommendations

- There is no known quantification of population level responses to lead poisoning in the UK.
- Levels of spent lead shot in the British countryside are not known.
- Thorough testing of a range of wildlife and the environment (soils) within game and non-game estates is an urgently required study.
- Methods of by-catch mitigation should be implemented on shooting estates where grey partridge are present.

6. Impacts on predators and predation dynamics

6.1 Summary

Subsidized populations of easy prey and carrion such as gamebirds, increase food resources for generalist predators, and predators of conservation interest, and are likely to subsidize the carrying capacity of predator populations. At the end of the shooting season when gamebirds are less numerous, it is likely that predators have to switch to other prey types, with potentially considerable impacts on wild populations of native wildlife.

6.2 Background

There are two primary mechanisms by which gamebird release and management may affect predators. Firstly, ~44,000 tonnes of prey and carrion released into the countryside every year represents a significant and unnatural food availability. Birds reared in farms do not persist well in the wild, and this may be because predator evasion skills are not well developed in captivity (Musil & Connelly 2009). For example, pheasants are regularly seen browsing in the centre of open fields, far from cover, and at all times of the day. This saturation of easy prey is likely to increase survival and productivity of predators that feed on gamebirds, and subsequently increase their numbers, particularly given that gamebirds are abundant in winter when predator populations are most likely to be limited by food. Also, as many gamebirds are killed in vehicle collisions and are shot but not retrieved, gamebird carrion represents a substantial food source. However, little is known about the impact of this abundant food source on predator numbers, and carrying capacity dynamics are governed by multiple mechanisms. Secondly, on many game estates predator densities may be reduced through both legal and illegal culling by gamekeepers. However, the efficacy of this is variable and the effort inputted sometimes low and seasonal, particularly on estates that focus on rearing operations rather than wild breeding birds. Background levels of fox density and estate management in relation to predator control vary regionally, but approximately 190,000 foxes are culled each year in the UK (Heydon & Reynolds 2000b, Heydon & Reynolds 2000a, Heydon *et al.* 2000, Pye-Smith 1997). Webbon *et al.* (2004) estimate the UK adult fox population in spring at 258,000. An additional 425,000 cubs are born each spring, meaning that the same amount must die each year to keep the population stable (Tapper 1999).

6.3 Increased food resources for generalist predators

6.3.1 Increases in generalist predators

The impact of birds of prey and other predators on gamebird populations have been quite extensively studied (Kenward *et al.* 2001, Park *et al.* 2008, Redpath & Thirgood 2009, Watson *et al.* 2007). However, the opposite mechanism has rarely been addressed; what is the impact of abundant game on predator populations (but see Redpath & Thirgood 2009). While predators may decrease the numbers of gamebirds, gamebirds at high densities may increase the numbers of predators, because unlike natural predator-prey systems, there is no density dependence, since released prey numbers are independent of predator abundance (Gibbons *et al.* 2007). Artificially abundant prey sources are therefore expected to enable predator populations to expand. This is likely to be particularly apparent where predators have few direct competitors or predators of their own. Today in the UK, higher food-chain carnivores such as wolf and lynx that would predate upon, and compete with foxes and other predators are extinct. Foxes are therefore thought to be more numerous than natural ecological systems would usually permit, and corvids that have undergone population increases over the last 50 years may be governed by a similar mechanism.

Foxes tend to concentrate on the most available prey source (Delibes-Mateos *et al.* 2008, Ferrari & Weber 1995, Lanszki *et al.* 2007). At the end of the shooting season, as gamebird numbers collapse, it is likely that predators temporarily switch to other prey types. This coincides with the critical period when birds are nesting, and easy prey may be found in eggs and chicks. Elevated predator abundance, caused by high densities of gamebirds, are therefore likely to cause imbalance in the predator-prey relationship, resulting in increased predation of native birds or their nests, and this may subsequently negate the positive effects of predator control on game estates. Ground nesting birds are particularly susceptible to predation (Gibbons *et al.* 2007), so these species are expected to be disproportionately effected by abundant generalist predators (also see section 2.4). Additionally in moorland fringe habitat where a pattern of greater pheasant and red-legged partridge releases has recently been seen, increased predator densities could have substantial impacts on fragile upland wader communities (see Sim *et al.* 2005).

According to Turner & Sage (2004), 36% of released pheasants (N=325) are either predated or scavenged by foxes. If this figure is indicative of the UK pheasant population, this suggests that 12.6 of the 35 million pheasants released annually may fall prey to foxes. 12.6 million pheasants represent the entire yearly food requirement for over 100,000 foxes. However, in the UK foxes appear to favour rabbits (74% of ingested biomass, although this may be less in the uplands eg. 25% - Leckie *et al.* 1998), with pheasants consisting approximately 11% of their diet (Baker *et al.* 2006b). 12.6 million pheasants could therefore support the diet of approximately 950,000 foxes. Assuming pheasants comprise more than 11% of fox diet where pheasant densities are high, the actual figure is likely to be closer to the estimated 258,000 spring adult population size. Therefore, pheasant surplus could in theory support the entire fox population of the UK, while simultaneously reducing predation pressure on other introduced prey such as rabbits. In the uplands the proportion of the diet that is gamebirds can often be greater (e.g. 26% - Leckie *et al.* 1998) and therefore the influence of this abundant source of prey in an upland environment, where prey is comparatively scarce compared to the lowlands, may have an even greater effect in supporting elevated numbers of predators.

6.3.2 Increases in predators of conservation interest

While foxes and corvids are often considered pests in the UK and their numbers can be legally controlled, species of conservation interest may benefit from the increased abundance of gamebirds. Protected birds of prey, such as buzzard, red kite, goshawk, golden eagle, and white-tailed eagle may not only profit from increased food resources, but also reduce their attention towards other prey species that may be of conservation interest, particularly grey partridge. However, evidence does not support this, as Mrlik (1992) found that raptors did not concentrate in areas where pheasants were released, and Kenward (2001) found that only 8% of buzzards that had a release pen within their home-range associated frequently with release pens. On average 5.6-7% of pheasants are predated by birds of prey, although occasionally heavy predation 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, and deciduous canopies with lots of perching sites (Kenward *et al.* 2001), but in general buzzards may be more likely to scavenge dead gamebirds than kill them. In contrast, in Sweden Goshawk predation on pheasants can be particularly heavy (Kenward *et al.* 1981, Valkama *et al.* 2005). BASC provide practical advice to game managers who are concerned about the impacts of birds of prey. These include encouraging plenty of thick cover within pens, and reducing the number of perching opportunities for birds of prey.

Although the relationship between abundant gamebird prey and predator carrying capacity dynamics are not well understood, it is possible that predators of conservation interest may also experience unnatural population expansions, and subsequently have similar detrimental effects to those described for generalist predators (section 6.3.1)

6.4 Illegal persecution of predators by gamekeepers

To protect the shooting industry, and the revenue generated by selling shooting to paying customers, landowners and shooting organisations are pressing government, most recently in Scotland, to allow derogations from the protection afforded to buzzards and goshawks. The basis for such requests is questionable, with the available evidence suggesting that predation by raptors accounts for only small losses of gamebirds, and other causes of mortality are of far greater significance in terms of any impacts on shooting revenue. Despite this, protected raptors are routinely illegally killed by employees of some game estates, though the impacts and incidence of this have not been extensively quantified for lowland release operations due to the difficulty in detecting illegal prosecution. In 2008, there were 210 reports of persecution on birds of prey and their nests, a large proportion of which were buzzards (RSPB 2009). In 2009, over a three-month period on a game estate in Shropshire, a gamekeeper was prosecuted for killing over 100 buzzards, in addition to other protected species including ravens and badgers. In a study of 136 radio-tagged buzzards, about 10% of first year buzzards were killed illegally near pheasant pens (Kenward *et al.* 2001).

Illegal killing of raptors on lowland estates may be less common than in uplands, where population level effects of persecution on some raptor species are known (Smart *et al.* 2010). Subsequently, gamebird releasing has probably aided the re-colonisation of lowlands by buzzards over the last 20 years, because game managers usually release extra birds in order to compensate for losses to raptors.

6.5 Conclusions

The predator-prey mechanism whereby predator numbers are increased by abundant game creates an adverse situation for gamebird managers, because the release of more birds is not always likely to increase the numbers of birds available during the shooting season due to predation pressure. It is therefore likely that an optimum release density will yield the most efficient release density-predation trade-off (also see section 8.2).

The idea of higher predator numbers supported by gamebird releases is purely speculative, but numbers of some predators have increased in recent years. Although fox have suffered an overall decline in recent years, numbers have increased in south-east England, where gamebird releasing is more ubiquitous (Newson & Noble 2005). Abundant gamebird food resources may benefit protected species, particularly raptors such as buzzard and red-kite, but this is again speculative.

6.6 Key knowledge gaps and recommendations

- It is unknown whether 44,000 tonnnes of gamebirds released each year influence predator populations, or the impact of increased predator populations on ground-nesting species of conservation concern.
- There is pressing need for a well designed field study, conducted over a large geographical scale and number of estates, to examine the relationship between gamebird release intensity and the abundance of generalist predators e.g. foxes. This work is currently (March 2010) in proposal stages between the RSPB and GWCT.
- There is also the urgent need to understand whether releases of red-legged partridges on moorlands are elevating predator abundance. The current releasing practices of red-

legged partridge onto the moorland edge results in discrete areas of moorland with and without releases, this provides an opportunity to carry out the research necessary to explore their likely impacts.

7. Socio-economic impacts

7.1 Background

The gamebird industry has both negative and positive socio-economic impacts. The industry is important to rural economies throughout the UK, and supports numerous employment opportunities. Gamebirds may however have negative socio-economic effects, including causing road traffic accidents, increasing disease incidence, and lead in game for human consumption may also represent a health risk. Socio-economic benefits of the gamebird shooting industry have been quite thoroughly investigated (PACEC 2006), though certain adverse effects have been overlooked.

7.2 Economic input and employment

In the UK, 480,000 people shoot game. It is estimated that this industry supports the equivalent of 70,000 full-time jobs, and shooters spend £2 billion each year on goods and services (PACEC 2006). However, the objective of just 18% of shoot providers is to make a profit, as for most, shooting is a hobby (PACEC 2006).

Each year £91 million is spent on eggs/chicks reared either in the UK or France, and of birds shot approximately 46% of pheasants and 37% of partridge are sold to dealers for human consumption, the rest being consumed by the shooting providers and the shooters.

2.7 million man-days per year are spent undertaking estate management, equivalent to 12,000 full-time jobs, and each year £12 million is spent on cover crop seed.

7.3 Road traffic accidents

In studies of radio-tagged pheasants, Hill & Robertson (1988) found that 2.7% are killed on the road, and Turner & Sage (2004) found that about ~5% are killed on the road. This equates to between 0.9 and 1.8 million pheasants each year, and the Post Office® estimate that 3 million pheasants are killed on the road each year (PostOffice 2008). The insurance and damage costs of these collisions are unknown, but likely to be substantial. Additionally, in 2008 a motorcyclist was killed when a pheasant flew into his helmet[1], and it is likely that other such fatal accidents have been caused by gamebirds.

7.4 Gamebirds and disease risk for humans

In 2008, Lyme disease was officially detected in 722 people in England and Wales (mostly in southern England), but the Health Protection Agency estimate the actual total to be 2 to 3 times higher (HPA 2009). The incidence of Lyme disease has shown annual increases since the beginning of the century despite increases in awareness and prevention methods (Smith & Takkinen 2006). Due to modern medicine, Lyme disease is rarely fatal, but does cause severe arthritis, depression, and heart strain. As pheasants are effective hosts of the bacteria that causes Lyme disease, it is likely that some of the cases of human disease are gamebird related although it would be impossible to quantify how many.

7.5 Lead and humans

Until recently it has not been recognised that when lead shot passes through the tissue of shot game, the shot undergoes considerable fragmentation, leaving small pieces of lead far from the bullet entry canal (Hunt *et al.* 2009). These tiny fragments therefore go undetected while preparing game for human consumption, and are subsequently ingested by humans. Nearly all evidence concludes that people who regularly eat game have higher blood lead levels than

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[1] http://www.telegraph.co.uk/news/uknews/3372872/Motorcyclist-dies-when-pheasant-hits-him-in-million-to-one-accident.html

those who do not (Iqbal *et al.* 2009, Kosnett 2009, Tranel 2009) with the exception of Haldimann *et al.* (2002) who found no significant differences between hunter and non-hunter groups. Tranel *et al.* (2009) however, summarises extensive evidence from multiple studies that all demonstrate correlations between game consumption and elevated blood lead levels.

Lead affects humans in much the same way as it does wildlife. Young children are most vulnerable, and repeated exposure of just low concentrations, can lead to toxicosis (Pokras & Kneeland 2009). According to the European Union, maximum allowable lead levels in meat for human consumption are set at 100 µg/kg (wet-weight) in non-game meats (including poultry), but 100 times greater (10,000 µg/kg) in game meats (EC 2006). There are no obvious reasons for this discrepancy, except maybe that game consumption is limited to certain sub-groups of the population. However, with the growing availability of game in high-street supermarkets, lead exposure risk is likely to be expanding.

7.6 Conclusions

The economic benefits of the game shooting industry may be more considerable than the negative social effects that are likely to be localised, of low incidence or of minor impact. In general, where positive economic impacts outweigh negative social impacts is completely the artefact of personal opinion, as the understanding of the balance of these effects is governed by personal prioritisation. This is unlike the other chapters of this review that address only the affects on biodiversity conservation. It is important to note also that the most recent and comprehensive report (PACEC 2006), where many of the figures used in this section are derived, has recently come under scrutiny, questioning some of the economic multipliers used to scale-up the results of questionnaires. These multipliers may cause over-estimation in many cases. Therefore the data presented in this section, and the land-use data in section 1.4.3, should be interpreted with caution.

7.7 Key knowledge gaps and recommendations

- A compilation of insurance claims relating to gamebird collisions may be of use.
- Road transects could be used to calculate gamebird deaths on the road.

8. Interpreting the primary findings of this review

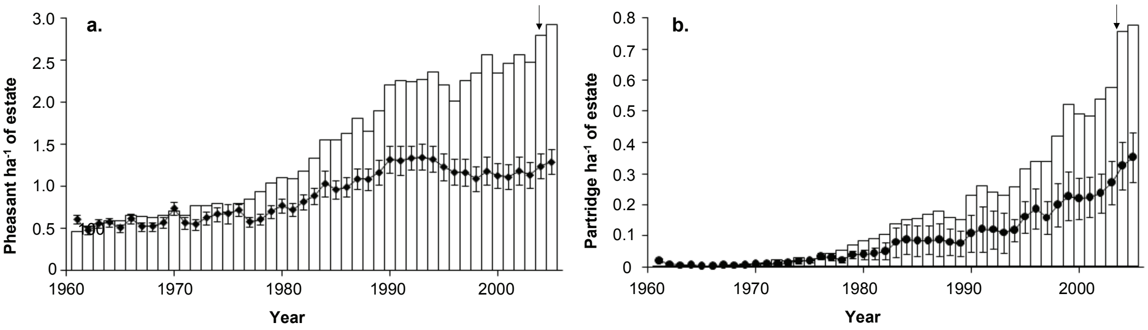

Fig. 1.1.1. (Repeated from section 1 to aid understanding of section 8). Annual density of a. pheasants and b. red-legged partridges released per hectare of game estate (bars), and numbers shot (dots ± 95% CI) from 1961 to 2005 in the UK. In 2004 (depicted by arrow), 35 million pheasants were released and 15 million shot, and 6.5 million red-legged partridges were released, and 2.6 million shot (PACEC 2006) (graphs from www.gct.org.uk).

8.1 Understanding actual bird densities on estates

There is a relatively common pattern whereby negative direct impacts are experienced at releases greater than 1000 birds ha^{-1} of release pen. 35 million pheasants divided by the estimated 30,000 ha of release pens equals an average stocking density of 1166 birds ha^{-1}. However, stocking densities are highly variable, and it is unknown what proportion of pens may be above this average figure, as most literature suggests that densities are more normally around 2000ha^{-1} (Davey 2008, Sage *et al.* 2005). Birds per ha of release pen is the measure used in several studies, but is not particularly appropriate when considering habitat level impacts outside of pens. An ideal figure would be density within the area regularly used by gamebirds, however this is inevitably difficult to calculate. Fig. 1.1.1. measures pheasant abundance in terms of gamebirds per ha of estate, which is likely to be the best alternative measure in most circumstances. However, in the case of impacts on linear boundaries such as hedgerows, the actual numbers of birds passing hedges may further provide a more intuitive understanding of impacts. Release numbers and methods for calculating density are therefore often confounded, and are subsequently difficult to interpret and compare between studies.

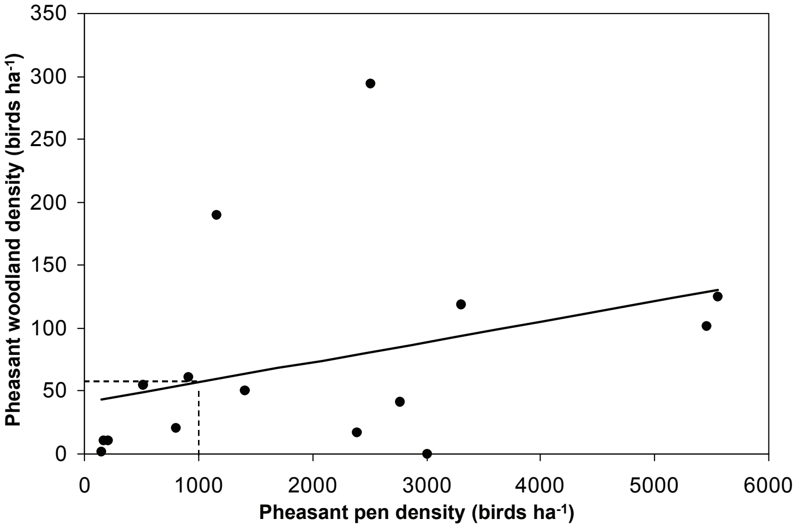

Fig. 8.1.1. Relationship between density of pheasants within pens, and density of pheasants within the entire woodland that contains the pens (N=15). Dashed line shows approximate woodland density of 55 ha^{-1} were pens are stocked at 1000 birds ha^{-1}. This subsequently enables the comparison of different measures of pheasant abundance. Data after Davey 2008.

Fig. 8.1.1 uses the only available data to consolidate the different density measures and enable comparisons between studies. This shows that the average expected woodland density at a release pen stocking density of 1000 birds ha^{-1} is in the region of 55 birds ha^{-1} of woodland. Data are not available for entire estate, but woodland density is in many ways more applicable, and should suffice as at least a comparable measure across studies and a potential proxy for estate density. The GWCT recommend that on a 450 ha estate with 30 ha (6.7%) of woodland (typical for lowland England), the total area of release pens should not exceed one third of the woodland (10 ha). They also suggest it would be appropriate to release between 7,000 and 10,000 pheasants, equal to 15-22 birds ha^{-1} of the entire estate, or 233-333 birds ha^{-1} of estate woodland. If the average estate in Britain contains 6.7% woodland, using the number derived from Fig. 8.1.1, 55 birds ha^{-1} of woodland would equate to 3.7 birds ha^{-1} of estate, or 1650 birds per 30 ha of woodland, or 450 ha estate. This is therefore approximately five times lower than the GWCT recommendation. Although these calculations make broad assumptions about release estate characteristics, Fig. 1.1.1 does show that recent pheasant estate densities are approximately 2.8 birds ha^{-1}, and are thus within both types of recommendations. However, this simplistic inference does not account for the aggregation and clumped distribution of birds within the estate. Furthermore, while the figure of 700-1000 birds ha^{-1} 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. Impacts are of course highly site specific, and convoluted by numerous factors, such as the number of pens as opposed to area of pens. Fig. 8.1.2 attempts to theorise the differing levels of gamebird release and the overall impact at different densities and management practices. When habitat management is best, it is likely that impacts are mitigated by management to a certain degree. In normal circumstances, when birds are stocked at more than 1000 birds ha^{-1} of release pen (equal to approximately 55 birds ha^{-1} of woodland, or 3.7 birds ha^{-1} of estate), negative impacts are likely. When no habitat management is implemented, negative effects may occur at lower densities. At low release numbers habitat management may offset negative effects, and the net impact may be positive.

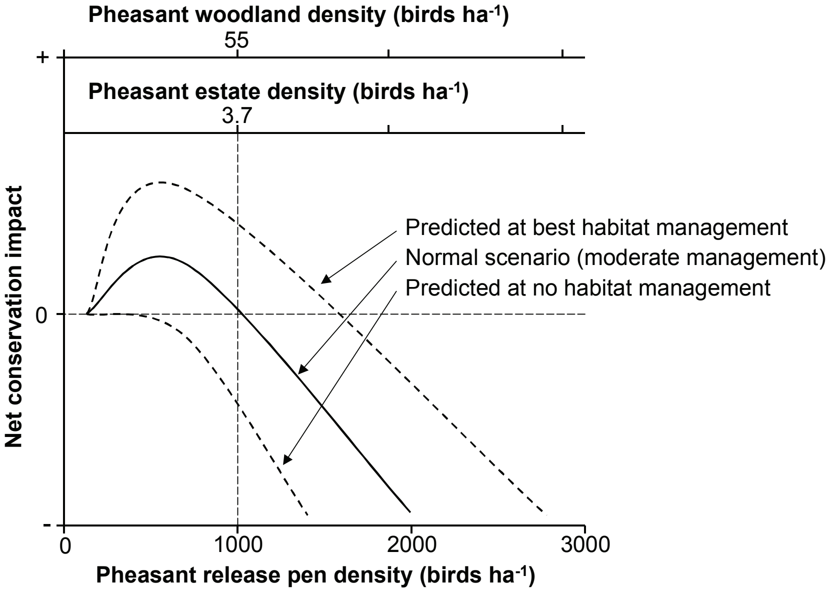

Fig. 8.1.2. Theoretical conservation impacts of three gamebird management scenarios at differing pheasant release densities.

8.2 The release, dispersal, shoot, surplus, financial-return trade-off

At high release densities, gamebirds disperse further from release pens (Sage & Swan 2003), often spilling-over to neighbouring farmland and other rural land types. Game managers will not receive revenue from birds that have dispersed beyond estate boundaries, and so there is a trade-off between release density and financial returns, as well as environmental impacts. Predator densities may be higher outside of release estates where there is less likely to be predator control, and birds that disperse further are more likely to succumb to death on the road also. The conservation implications of this relate closely to the direct impacts experienced at high release densities (see section 3), as well as the effect of abundant prey on predator communities (see section 6). This trade-off is illustrated by Fig. 8.2, which combines numbers from Fig. 1.1.1 and the understanding that pheasants at higher densities disperse further from release areas. Fig. 1.1.1 indicates that between 1990 and 2005, despite increasing densities of pheasant release, the number of pheasants shot remained stable at around 1.25 birds shot per hectare of estate. Therefore, over this period, the number of surplus birds has increased, and subsequently the net revenue to estates has decreased. The relationship between dispersal distance, and whether birds spread beyond estate boundaries is obviously highly dependent on multiple variables including estate size and shape, as well as release density. Consequently, Fig. 8.2 is a generalisation, and is only likely to be comparable among average circumstances across multiple estates. However, the equations that can be derived from this graph (once dispersal data is added) could be used to calculate the optimum release density (in financial terms) for any given estate. Adding optimum densities in terms of environmental impacts may complicate this calculation. However, given that the mean financial optimum release density appears to be in the region of 2.2 pheasants per hectare, financial and environmental optimums do not appear to be independent of one-another, and would suggest that on a nationwide scale, current release densities are well within recommendations regarding direct impacts.

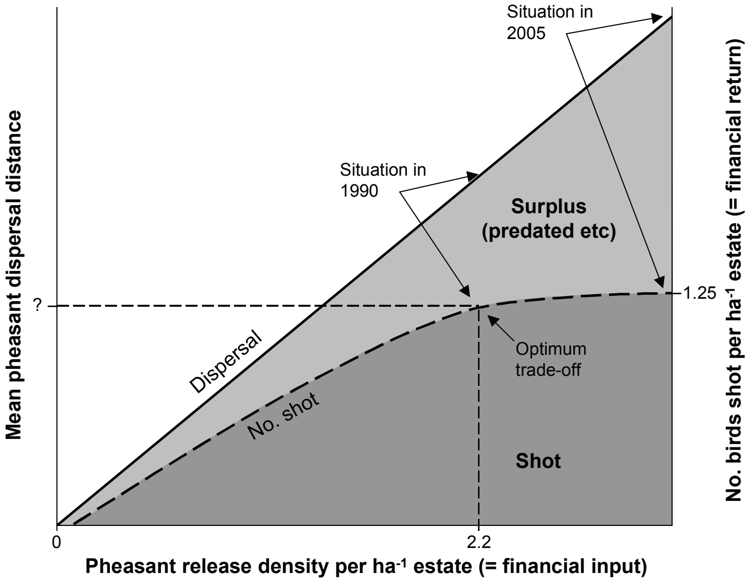

Fig. 8.2. Relationship between pheasant release density, dispersal away from release area, and the subsequent number of birds shot on the estate. At approximately 2.2 birds released per hectare of estate, the number of birds shot (~1.25 ha^{-1}), in relation to the number released is optimal, and subsequently financial returns are also greatest.

8.3 Multi-level interactions between gamebirds, management, and habitats

As expected in all functioning ecosystems, gamebirds interact on multiple levels, with multiple biotic and abiotic components within their habitats. Additionally, because gamebird shooting involves active habitat management, there is a further anthropogenic aspect within the functional ingredients of the interaction system. The shooting industry necessitates this for two reasons. Firstly, pheasant and red-legged partridge are not native to the UK, and therefore are not well adapted to these surroundings. Secondly, the gamebird shooting industry has grown to levels whereby millions of birds must be released, and sustained each year. Fig. 8.2 visually displays some of the primary interactions expected within this system. The strength of many of these interactions is not well understood, and as such cannot be illustrated.

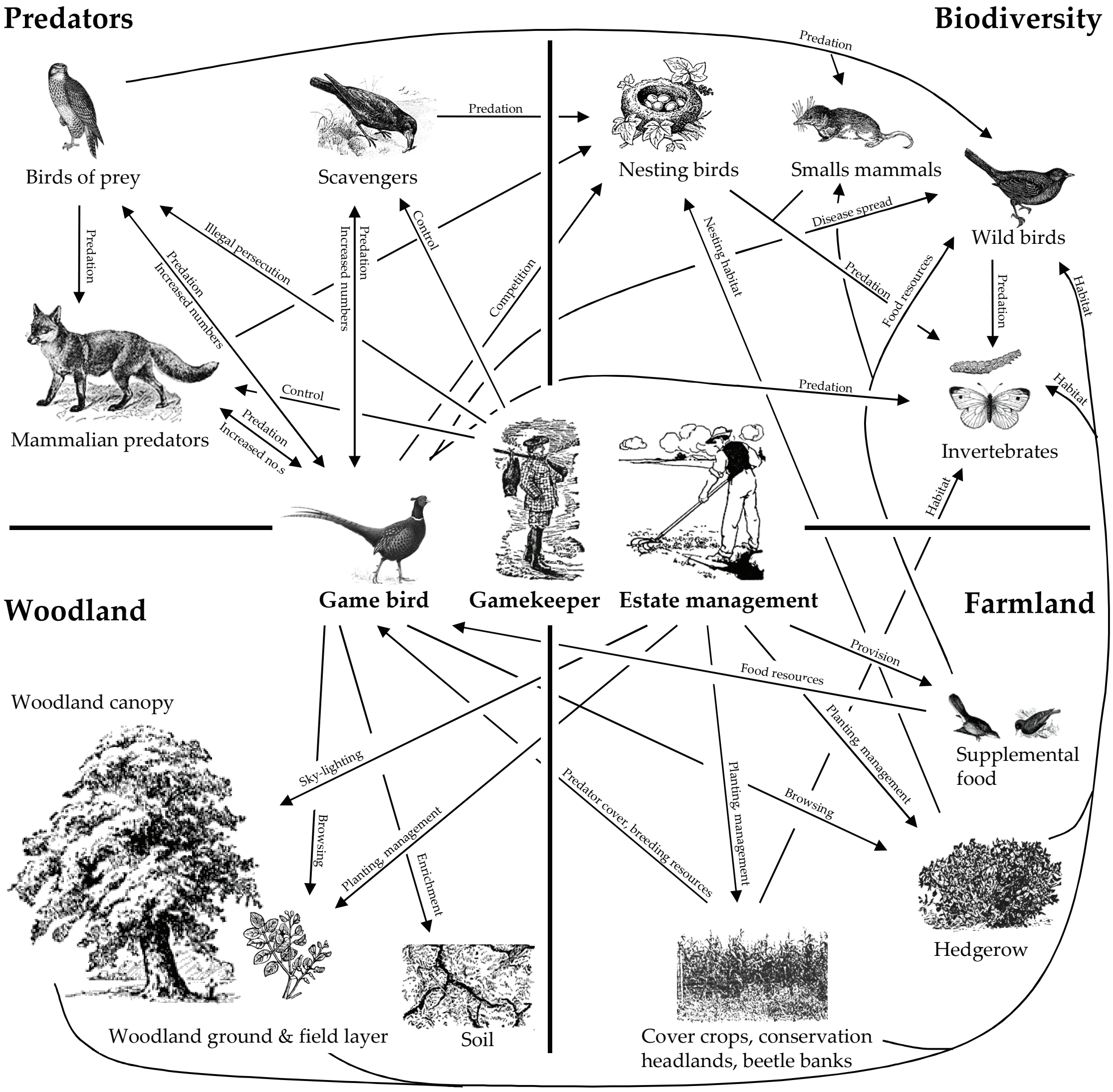

Fig. 8.3. Interactions between gamebird associated effects in the middle (gamebirds, gamekeeper, game estate management), and the environmental elements they may affect (divided into predators, biodiversity, woodland and farmland). Arrows point in the direction of the effect with short description of the interaction.

8.4 General conclusions

Evidence suggests that the management of land and wildlife for pheasant and partridge shooting delivers genuine benefits to certain habitats and species. However, there is also a significant and growing body of evidence indicating that the negative impacts of non-native gamebird release and related activities are considerable.

Although a very general analysis suggests that current nationwide average release densities appear to be below the levels where known negative impacts occur, accurate data on this are severely lacking making it impossible to ascertain what the real average release density is across all landholdings. However, there is a great deal of variation in release densities, from estates where just a few birds are released for private shooting, to estates where over 50,000 birds may be released each year. Therefore, despite the nationwide average, areas where negative impacts from gamebirds occur are likely to be sporadically distributed, and spatially correlated with estates that release large numbers of birds. These impacts do not only relate to direct impacts of gamebirds; in areas where more birds are released, there is likely to be more lead in the environment, greater risk of disease transfer, greater impacts on predator numbers, and more collisions between gamebirds and vehicles. If these negative effects are not successfully mitigated, then positive impacts of game estate management are likely to be negated. Such mitigation will require the compilation and review of best practice advice relating to non-native gamebird management. Conversely, on estates that operate good habitat management, and release birds at sustainable densities, the positive effects of habitat management are likely to result in a positive net conservation impact.

9. References

Aebischer, N. & Sotherton, N. 2002. Grey partridge recovery. *The Game Conservancy Trust Review of 2001* **33**: 53-57.

Amar, A., Hewson, C. M., Thewlis, R. M., Smith, K. W., Fuller, R. J., Lindsell, J. A., Conway, G., Butler, S. & MacDonald, M. A. 2006. What's happening to our woodland birds? Long term changes in the populations of woodland birds. *RSPB research report number 19, BTO research report number 169.*

Arroyo, B. & Beja, P. 2002. Impact of hunting management practices on biodiversity. *In* Viñuela, J, (Eds) *Reconciling gamebird hunting and biodiversity (REGHAB).* Instituto de Investigación en Recursos Cinegéticos, Ciudad Real, Spain.

Baillie, S. R., Marchant, J. H., Leech, D. I., Joys, A. C., Noble, D. G., Barimore, C., Grantham, M. J., Risely, K. & Robinson, R. A. 2009. Breeding birds in the wider countryside: their conservation status 2008. *BTO Research Report No. 516*.

Baker, H., Stroud, D. A., Aebischer, N. J., Cranswick, P. A., Gregory, R. D., McSorley, C. A., Noble, D. G. & Rehfisch, M. M. 2006a. Population estimates of birds in Great Britain and the United Kingdom. *British Birds* **99**: 25-44.

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

Beer, J. V. 1988. *Diseases of gamebirds and wildfowl. Advisory Green Guide 6*. The Game Conservancy Trust, Fordingbridge.

Bradbury, R. B., Kyrkos, A., Morris, A. J., Clark, S. C., Perkins, A. J. & Wilson, J. D. 2000. Habitat associations and breeding success of yellowhammers on lowland farmland. *Journal of Applied Ecology* **37**: 789-805.

Brickle, N. W. 1997. The use of game cover and game feeders by songbirds in winter. *In* (Eds) *1997 Brighton Crop Protection Conference - Weeds, Conference Proceedings Vols 1-3.*

Brickle, N. W. & Harper, D. G. C. 1999. Diet of nestling Corn Buntings *Miliaria calandra* in southern England examined by compositional analysis of faeces. *Bird Study* **46**: 319-329.

Brickle, N. W., Harper, D. G. C., Aebischer, N. J. & Cockayne, S. H. 2000. Effects of agricultural intensification on the breeding success of corn buntings *Miliaria calandra*. *Journal of Applied Ecology* **37**: 742-755.

Bunbury, N., Jones, C. G., Greenwood, A. G. & Bell, D. J. 2008. Epidemiology and conservation implications of *Trichomonas gallinae* infection in the endangered Mauritian pink pigeon. *Biological Conservation* **141**: 153-161.

Butler, D. A., Sage, R. B., Draycott, R. A. H., Carroll, J. P. & Potts, D. 2005. Lead exposure in ring-necked pheasants on shooting estates in Great Britain. *Wildlife Society Bulletin* **33**: 583-589.

Cade, T. J. 2007. Exposure of California condors to lead from spent ammunition. *Journal of Wildlife Management* **71**: 2125-2133.

Callegari, S. 2006. Impact of released gamebirds on chalk grassland. *The Game Conservancy Trust Review of 2005* **37**: 42-43.

Canning, P. 2005. The UK game bird industry - A short study. *Report to DEFRA*.

Carpenter, J. W., Charman, E., Smart, J., Amar, A., Gruar, D. & Grice, P. 2009. Habitat assosiations of woodland birds 2. *RSPB research report number 36*.

Carpenter, J., Smart, J., Amar, A., Gosler, A., Hinsley, S. & Charman, E. 2010 National-scale analyses of habitat associations of Marsh Tits *Poecile palustris* and Blue Tits *Cyanistes caeruleus*: two species with opposing population trends in Britain. Bird Study **57:** 31-43.

Cerradelo, S., Munoz, E., To-figueras, J., Mateo, R. & Guitart, R. 1992. Intoxicacion por ingestion de perdigones de plomo en dos aguilas reales, Donana. *Acta Vertebrat.* **19**: 122-127.

Chamberlain, D. E., Vickery, J. A., Marshall, E. J. P. & Tucker, G. M. (2001) The effect of hedgerow characteristics on the winter hedgerow bird community. In: *10th Annual*

Conference of the International-Association-for-Landscape-Ecology, (Barr, C. & Petit, S., eds.). pp. 197-206. Birmingham, England.

Clapham, P. A. 1935. Some helminth parasites from partridges and other English birds. *Journal of Helminthology* **13**: 139-148.

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

Coles, C. 1984. *The complete book of game conservation*. Stanley Paul, London.

Collinge, W. E. 1927. *The food of some Britsh wild birds: a study in economic ornithology*. Published by the author, York, UK.

Dalke, P. D. 1937. Food habits of adult pheasants based on crop analyses. *Ecology* **18**: 199-213.

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

Davey, P. & Aebischer, N. 2008. National gamebag census: game species trends. *The Game and Wildlife Consevation Trust Review of 2007* **39**: 30-33.

Davey, P. & Aebischer, N. 2009. National gamebag census: predators. *The Game and Wildlife Consevation Trust Review of 2008* **40**: 34-37.

Dawes, A. & Szekely, T. 2008. The UK Great bustard reintroduction. *Avian Biology Research* **1**: 42-42.

De Francisco, N., Troya, J. D. R. & Aguera, E. I. 2003. Lead and lead toxicity in domestic and free living birds. *Avian Pathology* **32**: 3-13.

DEFRA. 2010. *Avian influenza advice from the British Veterinary Poultry Association (BVPA) and Defra to gamekeepers and their veterinarians.* Online at: http://www.defra.gov.uk/foodfarm/farmanimal/diseases/atoz/ai/keptbirds/game-advice.htm. Accessed on: 01/02/10.

Delibes-Mateos, M., de Simon, J. F., Villafuerte, R. & Ferreras, P. 2008. Feeding responses of the red fox (*Vulpes vulpes*) to different wild rabbit (*Oryctolagus cuniculus*) densities: a regional approach. *European Journal of Wildlife Research* **54**: 71-78.

Draycott, K. 2005. The use of feed hoppers. *The Game Conservancy Trust Review of 2004* **36**: 18-19.

Draycott, R. A. H. & Hoodless, A. N. 2005. Effect of pheasant management on wildlife in woods. *The Game Conservancy Trust Review of 2004* **36**: 38-39.

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

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

Duff, J. P., Holmes, P., Brown, I. & Gayford, P. 2003. Surveillance scheme for wildlife disease in England and Wales. *Veterinary Record* **153**: 538-538.

Eaton, M. A., Brown, A. F., Noble, D. G., Musgrove, A. J., Hearn, R., Aebischer, N. J., Gibbons, D. W., Evans, A. & Gregory, R. D. 2009. Birds of conservation concern 3: the population status of birds in the United Kingdom, Channel Islands and the Isle of Man. *British Birds* **102**: 296-341.

EC. 2006. Commission regulation (EC) No. 1881: setting maximum levels for certain contaminants in foodstuffs. *Official Journal of the European Union* **1881**: L364/5-L364/24.

Elliott, J. E., Langelier, K. M., Scheuhammer, A. M., Sinclair, P. H. & Whitehead, P. E. 1992. Incidence of lead poisoning in bald eagles and lead shot in waterfowl gizzards from British Columbia, 1988–91. *CWS Prog. Note 200. Canadian Wildlife Service, Quebec.*

Evans, A. D., Smith, K. W., Buckingham, D. L. & Evans, J. 1997. Seasonal variation in breeding performance and nestling diet of Cirl Buntings *Emberiza cirlus* in England. *Bird Study* **44**: 66-79.

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

Ferrari, N. & Weber, J. M. 1995. Influence of the abundance of food resources on the feeding-habits of the red fox, *Vulpes-vulpes*, in western Switzerland. *Journal of Zoology* **236**: 117-129.

Firbank, L. G. 1999. *Lowland game shooting study*. Institute of terrestrial ecology, Grange over Sands, Cumbria.

Fisher, I. J., Pain, D. J. & Thomas, V. G. 2006. A review of lead poisoning from ammunition sources in terrestrial birds. *Biological Conservation* **131**: 421-432.

Flowerdew, J. R. 1972. The effect of supplementary food on a population of wood mice (*Apodemus sylvaticus*). *Journal of Animal Ecology* **41**: 553-566.

Fuller, R. J. & Green, G. H. 1998. Effects of woodland structure on breeding bird populations in stands of coppiced lime (Tilia cordata) in western England over a 10-year period. *Forestry* **71**: 199-218.

Fuller, R. J., Gregory, R. D., Gibbons, D. W., Marchant, J. H., Wilson, J. D., Baillie, S. R. & Carter, N. 1995. Population declines and range contractions among lowland farmland birds in Britain. *Conservation Biology* **9**: 1425-1441.

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

Fuller, R. J., Stuttard, P. & Ray, C. M. 1989. The distribution of breeding songbirds within mixed coppiced woodland in Kent , England, in relation to vegetation age and structure. *Annales Zoologici Fennici* **26**: 265-275.

Fuller, R. J. & Warren, M. S. 1993. *Coppiced woodlands: their management for wildlife. 2nd ed.* JNCC, Peterborough.

GCT. 2001. Farmland research. *The Game Conservancy Trust Review of 2000* **32**: 75-84.

GCT. 2003. Uplands research. *The Game Conservancy Trust Review of 2002* **34**: 18-30.

Gibbons, D. W., Amar, A., Anderson, G. Q. A., Bolton, M., Bradbury, R. B., Eaton, M. A., Evans, A. D., Grant, M. C., Gregory, R. D., Hilton, G. M., Hirons, G. J. M., Hughes, J., Johnstone, I., Newbury, P., Peach, W. J., Ratcliffe, N., Smith, K. W., Walton, P. & Wilson, J. D. 2007. The predation of wild birds in the UK: a review of its conservation impact and management. *RSPB Research Report Number 23*.

Gill, C. E. & Langelier, K. M. 1994. Acute lead-poisoning in a bald eagle secondary to bullet ingestion. *Canadian Veterinary Journal-Revue Veterinaire Canadienne* **35**: 303-304.

Gillings, S., Newson, S. E., Noble, D. G. & Vickery, J. A. 2005. Winter availability of cereal stubbles attracts declining farmland birds and positively influences breeding population trends. *Proceedings of the Royal Society B, Biological Sciences* **272**: 733–739.

Ginsberg, H. S. 1994. Lyme-disease and conservation. *Conservation Biology* **8**: 343-353.

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

Grandjean, P. 1976. Possible effect of lead on eggshell thickness in kestrels *Bulletin of Environmental Contamination and Toxicology* **16**: 101-106.

Green, R. E. 1984. The feeding ecology and survival of partridge chicks (*Alectoris rufa* and *Perdix perdix*) on arable farmland in east Anglia. *Journal of Applied Ecology* **21**: 817-830.

Green, R. E., Osborne, P. E. & Sears, E. J. 1994. The distribution of passerine birds in hedgerows during the breeding-season in relation to characteristics of the hedgerow and adjacent farmland. *Journal of Applied Ecology* **31**: 677-692.

Gryczynska, A., Zgodka, A., Ploski, R. & Siemiathkowski, M. 2004. *Borrelia burgdorferi* sensu lato infection in passerine birds from the Mazurian Lake region (Northeastern Poland). *Avian Pathology* **33**: 69-75.

Haldimann, M., Baumgartner, A. & Zimmerli, B. 2002. Intake of lead from game meat - a risk to consumers' health? *European Food Research and Technology* **215**: 375-379.

Heydon, M. J. & Reynolds, J. C. 2000a. Demography of rural foxes (*Vulpes vulpes*) in relation to cull intensity in three contrasting regions of Britain. *Journal of Zoology* **251**: 265-276.

Heydon, M. J. & Reynolds, J. C. 2000b. Fox (*Vulpes vulpes*) management in three contrasting regions of Britain, in relation to agricultural and sporting interests. *Journal of Zoology* **251**: 237-252.

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

Hill, D. A. & Robertson, P. A. 1988. *The Pheasant: Ecology, management and conservation*. Blackwell Scientific Publications, London.

Hinsley, S. A. & Bellamy, P. E. 2000. The influence of hedge structure, management and landscape context on the value of hedgerows to birds: A review. *Journal of Environmental Management* **60**: 33-49.

Hinsley, S. A., Bellamy, P. E., Sparks, T. H. & Rothery, P. 1999. A field comparison of habitat characteristics and diversity of birds, butterflies and plants between game and nongame areas. *In* Firbank, L. G. (Eds) *Lowland Game Shooting Study: Final Report.* Institute of Terrestrial Ecology, Grange over Sands, UK.

Hoodless, A. N. & Draycott, K. 2006. Effects of pheasant management at woodland edges. *The Game Conservancy Trust Review of 2005* **37**: 30-31.

Hoodless, A. N. & Draycott, R. A. H. 2007. Effects of pheasant management on hedges. *The Game and Wildlife Conservation Trust Review of 2006* **38**: 68-69.

Hoodless, A. N., Lewis, R. & Palmer, J. 2006. Songbird use of pheasant woods in winter. *The Game Conservancy Trust Review of 2005* **37**: 28-29.

Hoffman, D. J., Franson, J. C., Pattee, O. H., Bunck, C. M. & Anderson, A. 1985a. Survival, growth, and accumulation of ingested lead in nestling American kestrels (*Falco sparverius*). *Archives of Environmental Contamination and Toxicology* **14**: 89-94.

Hoffman, D. J., Franson, J. C., Pattee, O. H., Bunck, C. M. & Murray, H. C. 1985b. Biochemical and hematological effects of lead ingestion in nestling American kestrels (*Falco sparverius*). *Comparative Biochemistry and Physiology C-Pharmacology Toxicology & Endocrinology* **80**: 431-439.

Hofle, U., Gortazar, C., Ortiz, J. A., Knispel, B. & Kaleta, E. F. 2004. Outbreak of trichomoniasis in a woodpigeon (*Columba palumbus*) wintering roost. *European Journal of Wildlife Research* **50**: 73-77.

HPA. 2009. *Lyme borreliosis in England and Wales: 2008.* Online at: www.hpa.org.uk. Accessed on: 14/12/09.

Hudson, P. J. 1997. *Wildlife diseases: The epidemiology of infectious diseases in wild animals and how they relate to mankind*. British Ecological Society, London.

Humair, P. F., Turrian, N., Aeschilimann, A. & Gern, L. 1993. *Borrelia burgdorferi* in a focus of Lyme borreliosis: epizootiologic contribution of small mammals. *Folia Parasitologica* **40**: 65-70.

Hunt, W. G., Watson, R. T., Oaks, J. L., Parish, C. N., Burnham, K. K., Tucker, R. L., Belthoff, J. R. & Hart, G. 2009. Lead bullet fragments in venison from rifle-killed deer: potential for human dietary exposure. *In* Watson, M., Fuller, M., Pokras, M. & Hunt, G. (Eds) *Ingestion of Lead from Spent Ammunition: Implications for Wildlife and Humans.* The Peregrine Fund, Boise, Idaho, USA.

Iqbal, S., Blumenthal, W., Kennedy, C., Yip, F. Y., Pickard, S., Flanders, W. D., Loringer, K., Kruger, K., Caldwell, K. L. & Brown, M. J. 2009. Hunting with lead: Association between blood lead levels and wild game consumption. *Environmental Research* **109**: 952-959.

Jansson, D. S., Brojer, C., Gavier-Widen, D., Gunnarsson, A. & Fellstrom, C. 2001. *Brachyspira* spp. (*Serpulina* spp.) in birds: A review and results from a study of Swedish game birds. *Animal Health Research Reviews* **2**: 93-100.

Keesing, F., Holt, R. D. & Ostfeld, R. S. 2006. Effects of species diversity on disease risk. *Ecology Letters* **9**: 485-498.

Kendall, R. J., Lacher, T. E., Bunck, C., Daniel, B., Driver, C., Grue, C. E., Leighton, F., Stansley, W., Watanabe, P. G. & Whitworth, M. 1996. An ecological risk assessment of lead shot exposure in non-waterfowl avian species: Upland game birds and raptors. *Environmental Toxicology and Chemistry* **15**: 4-20.

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

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

Keymer, I. F., Rose, J. H., Beesley, W. N. & Davies, S. F. M. 1962. A survey and review of parasitic diseases of wild and game birds in Great Britain. *Veterinary Record* **74**: 887-894.

Knott, J., Gilbert, J., Green, R. E. & Hoccom, D. G. 2009. Comparison of the lethality of lead and copper bullets in deer control operations to reduce incidental lead poisoning; field trials in England and Scotland. *Conservation evidence* **6**: 71-78.

Komosa, A. & Kitowski, I. 2008. Elevated lead concentration in skeletons of diurnal birds of prey Falconiformes and owls Strigiformes from Eastern Poland. *Ecological Chemistry and Engineering S-Chemia I Inzynieria Ekologiczna S* **15**: 349-358.

Kosnett, M. J. 2009. Health effects of low dose lead exposure in adults and children, and preventable risk posed by the consumption of game meat harvested with lead ammunition. *In* Watson, M., Fuller, M., Pokras, M. & Hunt, G. (Eds) *Ingestion of Lead from Spent Ammunition: Implications for Wildlife and Humans.* The Peregrine Fund, Boise, Idaho, USA.

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

Kurtenbach, K., Peacey, M., Rijpkema, S. G. T., Hoodless, A. N., Nuttall, P. A. & Randolph, S. E. 1998b. Differential transmission of the genospecies of *Borrelia burgdorferi* sensu lato by game birds and small rodents in England. *Applied and Environmental Microbiology* **64**: 1169-1174.

Lanszki, J., Zalewski, A. & Horvath, G. 2007. Comparison of red fox *Vulpes vulpes* and pine marten *Martes martes* food habits in a deciduous forest in Hungary. *Wildlife Biology* **13**: 258-271.

Leech, D. & Barimore, C. 2008. Is avian breeding success weathering the storms? . *BTO News* **279**: 19–20.

Leckie, F.M., Thirgood, S.J., May, R. & Redpath, S.M. 1998. Variation in the diet of red foxes on Scottish moorland in relation to prey abundance. *Ecography* **21**: 599-604.

Lewis, L. A., Poppenga, R. J., Davidson, W. R., Fischer, J. R. & Morgan, K. A. 2001. Lead toxicosis and trace element levels in wild birds and mammals at a firearms training facility. *Archives of Environmental Contamination and Toxicology* **41**: 208-214.

Low, P., Low, N. & Owens, N. 2003. *Study of vegetation in the pheasant pen, Old Sulehay Forest*. Sulehay Nature Reserve, Northhamptonshire.

Ludulf, I. C., Payne, S. & Robertson, P. A. 1989. The effects of pheasant release pens and strawed rides on ancient woodland ground floras. *The Game Conservancy Trust Review of 1988* **20**: 42-46.

Ma, W. C. 1987. Heavy-metal accumulation in the mole, *Talpa europea*, and earthworms as an indicator of metal bioavailability in terrestrial environments. *Bulletin of Environmental Contamination and Toxicology* **39**: 933-938.

Ma, W. C., Denneman, W. & Faber, J. 1991. Hazardous exposure of ground-living small mammals to cadmium and lead in contaminated terrestrial ecosystems. *Archives of Environmental Contamination and Toxicology* **20**: 266-270.

Manchester, S. J. & Bullock, J. M. 2000. The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology* **37**: 845-864.

Mateo, R. 2009. Lead poisoning in wild birds in Europe and the regulations adopted by different countries. *In* Watson, M., Fuller, M., Pokras, M. & Hunt, G. (Eds) *Ingestion of Lead from Spent Ammunition: Implications for Wildlife and Humans.* The Peregrine Fund, Boise, Idaho, USA.

Mateo, R., Cadenas, R., Manez, M. & Guitart, R. 2001. Lead shot ingestion in two raptor species from Donana, Spain. *Ecotoxicology and Environmental Safety* **48**: 6-10.

Mateo, R., Estrada, J., Paquet, J. Y., Riera, X., Dominguez, L., Guitart, R. & Martinez-Vilalta, A. 1999. Lead shot ingestion by marsh harriers *Circus aeruginosus* from the Ebro Delta, Spain. *Environmental Pollution* **104**: 435-440.

Millan, J. 2009. Diseases of the Red-Legged Partridge (*Alectoris rufa* L.): a Review. *Wildlife Biology Practice* **5**: 70-88.

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

Mrlik, V. & Koubek, P. 1992. Relation of birds of prey to the place of release of artificially-bred pheasant chicks. *Folia Zoologica* **41**: 233-252.

Musil, D. D. & Connelly, J. W. 2009. Survival and reproduction of pen-reared vs translocated wild pheasants *Phasianus colchicus*. *Wildlife Biology* **15**: 80-88.

Natural England. 2010. *Reasons for adverse condition summary.* Online at: http://www.sssi.naturalengland.org.uk/Special/sssi/reportAction.cfm?Report=sdrt17&Category=N&Reference=0. Accessed on: 28/01/10.

Newson, S. E. & Noble, D. G. 2003. Producing statistically valid maps of species abundance from UK Breeding Bird survey counts using Geostatistical Analyst in ArcGIS. *BTO Research Report No. 318*.

Newson, S. E. & Noble, D. G. 2005. The production of population trends for UK mammals using BBS mammal data: 1995-2003 update. *BTO Research Report No. 404.*

Newson, S.E., Rexstead, E.A., Baillie, S.R., Buckland, S.T. & Aebischer, N.J. 2010. Population change of avian predators and grey squirrels in England: is there still evidence for an impact on avian prey populations? *Journal of Applied Ecology*. In press.

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

Oltrogge, V. 2009. Success in developing lead free, expanding nose centerfire bullets. *In* Watson, M., Fuller, M., Pokras, M. & Hunt, G. (Eds) *Ingestion of Lead from Spent Ammunition: Implications for Wildlife and Humans.* The Peregrine Fund, Boise, Idaho, USA.

Ostfeld, R. S. & Keesing, F. 2000. Biodiversity and disease risk: The case of lyme disease. *Conservation Biology* **14**: 722-728.

PACEC. 2006. *Economic and environmental impact of sporting shooting in the UK*. PACEC, Cambridge, UK.

Pain, D. J. 1991. Why are lead-poisoned waterfowl rarely seen? The disappearance of waterfowl carcasses in the Camargue, France. *Wildfowl* **42**: 118-122.

Pain, D. J., Amiard-Triquet, C., Bavoux, C., Burneleau, G., Eon, L. & Nicolau-Guillaumet, P. 1993. Lead poisoning in wild populations of marsh harriers *Circus aeruginosus* in the Camargue and Charente-Maritime, France. *Ibis* **135**: 379O-386.

Pain, D. J., Amiardtriquet, C. & Sylvestre, C. 1992. Tissue lead concentrations and shot ingestion in 9 species of waterbirds from the camargue (france). *Ecotoxicology and Environmental Safety* **24**: 217-233.

Pain, D. J., Carter, I., Sainsbury, A. W., Shore, R. F., Eden, P., Taggart, M. A., Konstantinos, S., Walker, L. A., Meharg, A. A. & Raab, A. 2007. Lead contamination and associated disease in captive and reintroduced red kites *Milvus milvus* in England. *Science of the Total Environment* **376**: 116-127.

Pain, D. J., Fisher, I. J. & Thomas, V. G. 2009. A global update of lead poisoning in terrestrial birds from ammunition sources. *In* Watson, M., Fuller, M., Pokras, M. & Hunt, G. (Eds) *Ingestion of lead from spent ammunition: Implications for wildlife and humans.* The Peregrine Fund, Boise, Idaho, USA.

Pain, D. J., Sears, J. & Newton, I. 1995. Lead concentrations in birds of prey in Britain. *Environmental Pollution* **87**: 173-180.

Parish, D. 2004. Songbirds in Scotland prefer game cover crops. *The Game Conservancy Trust Review of 2003* **35**: 22-23.

Parish, T., Lakhani, K. H. & Sparks, T. H. 1994. Modeling the relationship between bird population variables and hedgerow and other field margin attributes 1. Species richness of winter, summer and breeding birds. *Journal of Applied Ecology* **31**: 764-775.

Parish, T., Lakhani, K. H. & Sparks, T. H. 1995. Modeling the relationship between bird population variables and hedgerow, and other field margin attributes 2. Abundance of individual-species and of groups of similar species. *Journal of Applied Ecology* **32**: 362-371.

Park, K. J., Graham, K. E., Calladine, J. & Wernham, C. W. 2008 Impacts of birds of prey on gamebirds in the UK: a review. In: *BOU Annual Meeting on Birds as Predators and as Prey.* pp. 9-26. Leicester, UK.

Pattee, O. H., Wiemeyer, S. N., Mulhern, B. M., Sileo, L. & Carpenter, J. W. 1981. Experimental lead-shot poisoning in bald eagles. *Journal of Wildlife Management* **45**: 806-810.

Pennycott, T. W. 1998. Population density and infectious disease at bird tables. *Veterinary Record* **142**: 523-523.

Pennycott, T. W. & Duncan, G. 1999. *Salmonella pullorum* in the common pheasant (*Phasianus colchicus*). *Veterinary Record* **144**: 283-287.

Pennycott, T. W., Ross, H. M., McLaren, I. M., Park, A., Hopkins, G. F. & Foster, G. 1998. Causes of death of wild birds of the family Fringillidae in Britain. *Veterinary Record* **143**: 155-158.

Pokras, M. A. & Kneeland, M. R. 2009. Understanding lead uptake and effects across species lines: A conservation medicine approach. *In* Watson, M., Fuller, M., Pokras, M. & Hunt, G. (Eds) *Ingestion of Lead from Spent Ammunition: Implications for Wildlife and Humans.* The Peregrine Fund, Boise, Idaho, USA.

PostOffice. 2008. Post Office counts the cost of wild animal accidents. *Press release*.

Potts, D. 1997. Cereal farming, pesticides and grey partridges. *In* Pain, D. J. & Pienkowski, M. W. (Eds) *Farming and birds in Europe: A common agricultural policy and its implications for bird conservation.* Academic Press, London.

Potts, G. R. 2005. Incidence of ingested lead gunshot in wild grey partridges (*Perdix perdix*) from the UK. *European Journal of Wildlife Research* **51**: 31-34.

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

Pye-Smith, C. 1997. *Fox-hunting. Boyond the propoganda.* Wildilfe Network, Oakham, Rutland.

Rands, M. R. W. 1985. Pesticide use on cereals and the survival of grey partridge chicks - a field experiment. *Journal of Applied Ecology* **22**: 49-54.

Rands, M. R. W. 1986. Effect of hedgerow characteristics on partridge breeding densities. *Journal of Applied Ecology* **23**: 479-487.

Real, J., Manosa, S. & Munoz, E. 2000. Trichomoniasis in a Bonelli's eagle population in Spain. *Journal of Wildlife Diseases* **36**: 64-70.

Redpath, S. & Thirgood, S. 2009. Hen harriers and red grouse: moving towards consensus? *Journal of Applied Ecology* **46**: 961-963.

Robertson, P. A. 1997. *A natural history of the pheasant*. Swan Hill Press, London.

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

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

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

RSPB. 2009. Birdcrime 2008: Offences against wild bird legislation in 2008.

Sage, R. B. 2003. Releasing gamebirds and wildlife conservation. *The Game Conservancy Trust Review of 2002* **34**: 57-60.

Sage, R. B. 2007. *Guidlines for sustainable gamebird releasing. GWCT conservation guides*. The Game and Wildlife Conservation Trust, Fordingbridge, Hampshire.

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

Sage, R. B., Putaala, A. & Woodburn, M. I. A. 2002a. Comparing growth and condition in post release juvenile common pheasants on different diets. *Poultry Science* **81**: 1199-1202.

Sage, R. B. & Swan, M. 2003. *Woodland conservation and pheasants. GWCT conservation guides*. The Game Conservancy Trust, Fordingbridge, Hampshire.

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

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

Sansano-Maestre, J., Garijo-Toledo, M. M. & Gomez-Munoz, M. T. 2009. Prevalence and genotyping of *Trichomonas gallinae* in pigeons and birds of prey. *Avian Pathology* **38**: 201-207.

Scheuhammer, A. M. & Norris, S. L. 1996. The ecotoxicology of lead shot and lead fishing weights. *Ecotoxicology* **5**: 279-295.

Schmidt, K. A. & Ostfeld, R. S. 2001. Biodiversity and the dilution effect in disease ecology. *Ecology* **82**: 609-619.

Sim, I.M.W., Gregory, R.D. & Brown, A.F. 2005. Recent changes in the abundance of British upland breeding birds. *Bird Study* **52**: 261-275.

Sinski, E. & Karbowiak, G. 1994. Zarazenie Borrelia burgdorferi w wybranej populacji drobnych ssako´w w po´lnocnej Polsce. *Biuletyn Instytutu Medycyny Morskiej i Tropikalnej* **27**: 70.

Sinski, E., Pawelczyk, A., Bajer, A. & Behnke, J. M. 2006. Abundance of wild rodents, ticks and environmental risk of Lyme borreliosis: A longitudinal study in an area of Mazury Lakes district of Poland. *Annals of Agricultural and Environmental Medicine* **13**: 295-300.

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

Smart, J., Taylor, E., Amar, A., Smith, K., Bierman, S., Carpenter, J., Grice, P., Currie, F., Smithers, R., Fuller, R. & Hewson, C. 2007. Habitat associations of woodland birds: Implications for woodland management for declining species. *RSPB Research Report Number 26*.

Smart, J., Amar, A., Sim, I.M.W., Etheridge, B., Cameron, D., Christie, G. & Wilson, J.D. 2010. Illegal killing slows population recovery of a re-introduced raptor of high conservation concern – The red kite *Milvus milvus*. *Biological conservation* (In press).

Smith, R. & Takkinen, J. 2006. Lyme borreliosis: Europe-wide coordinated surveillance and action needed? *Eurosurveillance* **11**.

Sotherton, N. 1991. Conservation headlands: A practical combination of intensive cereal farming and conservation. *In* Firbank, L. G., Carter, N., Darbyshire, J. F. & Potts, G. R. (Eds) *Ecology of temperate cereal fields.* Blackwell Scientific Publications, Oxford.

Stark, G., Connolly, H., Duckworth, J. C. & Bunce, R. G. H. 1999. Relationships between lowland game management, landscape structure and botanical diversity. *In* Firbank, L. G. (Eds) *Lowland Game Shooting Study.* Institute of Terrestrial Ecology, Grange over Sands, Cumbria.

Stoate, C. 2004. Loddington game and songbird counts. *The Game Conservancy Trust Review of 2003* **35**: 28-29.

Stoate, C. 2005. Loddington game and songbirds. *The Game Conservancy Trust Review of 2004* **36**: 16-17.

Stoate, C. 2006. Loddington game and songbirds. *The Game Conservancy Trust Review of 2005* **37**: 16-17.

Stoate, C. 2007. Songbirds at Loddington. *The Game Conservancy Trust Review of 2006* **38**: 54-55.

Stoate, C., Moreby, S. J. & Szczur, J. 1998. Breeding ecology of farmland Yellowhammers Emberiza citrinella. *Bird Study* **45**: 109-121.

Stoate, C. & Parish, D. 2002. Songbirds gain from game crops. *The Game Conservancy Trust Review of 2001* **33**: 90-93.

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

Stoate, C. & Szczur, J. 2001b. Whitethroat *Sylvia communis* and Yellowhammer *Emberiza citrinella* nesting success and breeding distribution in relation to field boundary vegetation. *Bird Study* **48**: 229-235.

Stoate, C. & Szczur, J. 2006. Spotted flycatchers at Loddington. *The Game Conservancy Trust Review of 2005* **37**: 18-19.

Stoate, C., White, P. & Szczur, J. 2008. Songbirds at Loddington. *The Game and Wildlife Conservation Trust Review of 2007* **39**: 72-73.

Stoate, C., White, P. & Szczur, J. 2009. Blackbird habitat and predation. *The Game and Wildlife Conservation Trust Review of 2008* **40**: 72-73.

Sutherland, W. J., Armstrong-Brown, S., Armsworth, P. R., Brereton, T., Brickland, J., Campbell, C. D., Chamberlain, D. E., Cooke, A. I., Dulvy, N. K., Dusic, N. R., Fitton, M., Freckleton, R. P., Godfray, H. C. J., Grout, N., Harvey, H. J., Hedley, C., Hopkins, J. J., Kift, N. B., Kirby, J., Kunin, W. E., Macdonald, D. W., Marker, B., Naura, M., Neale, A. R., Oliver, T., Osborn, D., Pullin, A. S., Shardlow, M. E. A., Showler, D. A., Smith, P. L., Smithers, R. J., Solandt, J. L., Spencer, J., Spray, C. J., Thomas, C. D., Thompson, J., Webb, S. E., Yalden, D. W. & Watkinson, A. R. 2006. The identification of 100 ecological questions of high policy relevance in the UK. *Journal of Applied Ecology* **43**: 617-627.

Tapper, S. 1992. *Game heritage. An ecological review from shooting and gamekeeping records*. The Game Conservancy Trust, Fordingbridge, Hampshire.

Tapper, S. 1999. *A question of balance: Game animals and their role in the British countryside*. The Game Conservancy Trust, Fordingbridge.

Thomas, M. B., Wratten, S. D. & Sotherton, N. W. 1992. Creation of island habitats in farmland to manipulate populations of beneficial arthropods - predator densities and species composition. *Journal of Applied Ecology* **29**: 524-531.

Thompson, P. & Sage, R. B. 2000. Songbirds in cover crops. *The Game Conservancy Trust Review of 1999* **31**: 28-29.

Tompkins, D. M., Dickson, G. & Hudson, P. J. 1999. Parasite-mediated competition between pheasant and grey partridge: a preliminary investigation. *Oecologia* **119**: 378-382.

Tompkins, D. M., Draycott, R. A. H. & Hudson, P. J. 2000a. Field evidence for apparent competition mediated via the shared parasites of two gamebird species. *Ecology Letters* **3**: 10-14.

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

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

Tranel 2009. Impacts of lead ammunition on wildlife, the environment, and human health - A literature review and implications for Minnesota. *In* Watson, M., Fuller, M., Pokras, M. & Hunt, G. (Eds) *Ingestion of lead from spent ammunition: Implications for wildlife and humans.* The Peregrine Fund, Boise, Idaho, USA.

Turner, C. & Sage, R. B. 2004. Fate of released pheasants. *The Game Conservancy Trust Review of 2003* **35**: 74-75.

Vickery, J. A., Bradbury, R. B., Henderson, I. G., Eaton, M. A. & Grice, P. V. 2004. The role of agri-environment schemes and farm management practices in reversing the decline of farmland birds in England. *Biological Conservation* **119**: 19-39.

Vickery, J. A., Feber, R. E. & Fuller, R. J. 2009. Arable field margins managed for biodiversity conservation: A review of food resource provision for farmland birds. *Agriculture Ecosystems & Environment* **133**: 1-13.

Valkama, J., Korpim, auml, ki, E., Arroyo, B., Beja, P., Bretagnolle, V., Bro, E., Kenward, R., Mañosa, S., Redpath, S. M., Thirgood, S., & Viñuela, J. 2005. Birds of prey as limiting factors of gamebird populations in Europe: a review. *Biological Reviews* **80:** 171-203.

Villanúa, D., Acevedo, P., Hofle, U., Rodriguez, O. & Gortazar, C. 2006a. Changes in parasite transmission stage excretion after pheasant release. *Journal of Helminthology* **80**: 313-318.

Villanúa, D., Casas, F., Vinuela, J., Gortazar, C., De La Morena, E. G. & Morales, M. 2007. First occurrence of *Eucoleus contortus* in a Little Bustard *Tetrax tetrax*: negative effect of Red-legged Partridge *Alectoris rufa* releases on steppe bird conservation? *Ibis* **149**: 405-406.

Villanúa, D., Hofle, U., Perez-Rodriguez, L. & Gortazar, C. 2006b. *Trichomonas gallinae* in wintering Common Wood Pigeons *Columba palumbus* in Spain. *Ibis* **148**: 641-648.

Vrezec, A. 2006. Is the grey partridge *Perdix perdix* population decline in Slovenia the result of apparent competition with the pheasant *Phasianus colchicus*? *Acrocephalus* **27**: 73-81.

Warren, M. S. 1989. Pheasants and fritillaries: Is there really any evidence that pheasant rearing may have caused butterfly declines. *British Journal of Entomology and Natural History* **2**: 169-175.

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

Webbon, C. C., Baker, P. J. & Harris, S. 2004. Faecal density counts for monitoring changes in red fox numbers in rural Britain. *Journal of Applied Ecology* **41**: 768-779.

Wong, J. & Dickinson, B. 2003. *Current status and development potential of woodland and hedgerow products in Wales*. Wild Resources Limited, Bangor, Wales.

Woodburn, M. I. A. 1995. Do parasites alter pheasant breeeding success? *The Game Conservancy Trust Review of 1994* **26**: 96-97.

Woodburn, M. I. A. & Sage, R. B. 2005. Effect of pheasant releasing on edge habitats. *The Game and Wildlife Conservation Trust Review of 2004* **36**: 36-37.

10. Appendix

10.1 Scientific names of species mentioned in the text

Birds

Bald eagle	*Haliaeetus leucocephalus*
Blackbird	*Turdus merula*
Blackcap	*Sylvia atricapilla*
Blue tit	*Cyanistes caeruleus*
Bonelli's eagle	*Hieraaetus fasciatus*
Bullfinch	*Pyrrhula pyrrhula*
Buzzard	*Buteo buteo*
Californian condor	*Gymnogyps californianus*
Carrion crow	*Corvus corone*
Chaffinch	*Fringilla coelebs*
Chiffchaff	*Phylloscopus collybita*
Chukar partridge	*Alectoris chukar*
Corn bunting	*Emberiza calandra*
Dunnock	*Prunella modularis*
Garden warbler	*Sylvia borin*
Goldcrest	*Regulus regulus*
Golden eagle	*Aquila chrysaetos*
Goshawk	*Accipter gentilis*
Great bustard	*Otis tarda*
Great tit	*Parus major*
Great spotted woodpecker	*Dendrocopos major*
Green woodpecker	*Certhia familiaris*
Grey partridge	*Perdix perdix*
Hawfinch	*Coccothraustes coccothraustes*
Hen harrier	*Circus cyaneus*
Herring gull	*Larus argentatus*
Honey buzzard	*Pernis apivorus*
House sparrow	*Passa domesticus*
Jackdaw	*Corvus monedula*
Jay	*Garrulus glandarius*
Northern bobwhite quail	*Colinus virginianus*
Nuthatch	*Sitta europea*
Linnet	*Sylvia communis*
Little bustard	*Tetrax tetrax*
Long-tailed tit	*Aegithalos caudatus*
Magpie	*Pica pica*
Marsh tit	*Poecile palustris*
Nuthatch	*Sitta europaea*
Peregrine falcon	*Falco peregrinus*

Pheasant	*Phasianus colchicus*
Pied flycatcher	*Ficedula hypoleuca*
Red grouse	*Lagopus lagopus*
Red-kite	*Milvus milvus*
Red-legged partridge	*Alectoris rufa*
Redstart	*Phoenicurus phoenicurus*
Reed bunting	*Emberiza schoeniclus*
Robin	*Erithacus rebecula*
Rook	*Corvus frugilegus*
Ruffed grouse	*Bonasa umbellus*
Scaled quail	*Callipepla squamata*
Siskin	*Carduelis spinus*
Skylark	*Alauda arvensis*
Solitary vireo	*Vireo solitarius*
Song thrush	*Turdus philomelos*
Spanish imperial eagle	*Aquila adalberti*
Sparowhawk	*Accipiter nisus*
Spotted flycatcher	*Muscicapa striata*
Starling	*Sturnus vulgaris*
Tawney owl	*Strix aluco*
Treecreeper	*Certhia familiaris*
Tree pipit	*Anthus trivialis*
Turtle dove	*Streptopelia turtur*
Western marsh-harrier	*Circus aeruginosus*
White-tailed eagle	*Haliaeetus albicilla*
Whitethroat	*Sylvia communis*
Wild turkey	*Meleagris gallopavo*
Willow warbler	*Phylloscopus trochilus*
Woodpigeon	*Columba palumbus*
Wood warbler	*Phylloscopus sibilatrix*
Wren	*Troglodytes troglodytes*
Yellowhammer	*Emberiza citrinella*
Yellow wagtail	*Motacilla flava*
Yellow-rumped warbler	*Dendroica coronata*

Mammals

Badger	*Meles meles*
Bank vole	*Myodes glareolus*
Brown rat	*Rattus norvegicus*
Fox	*Vulpes vulpes*
Grey-squirrel	*Sciurus carolinensis*
Lynx	*Lynx lynx*
Rabbit	*Oryctolagus cuniculus*
Shrew	*Sorex araneus*

Stoat	*Mustela erminea*
Vole	*Microtus agrestis*
Woodmouse	*Apodemus sylvaticus*
Wolf	*Canis lupus*

Invertebrates

Adonis blue butterfly	*Lysandra bellargus*

Plants

Annual meadow grass	*Poa annua*
Bluebell	*Hyacinthoides non-scripta*
Broadleaf dock	*Rumex obtusifolius*
Common chickweed	*Stellaria media*
Stinging nettle	*Urtica dioica*