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Consequences of game bird management for non-game species in Europe

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Abstract

1. Game bird management has the potential to benefit conservation, as management practices specifically targeted at reducing the factors limiting game populations may have positive effects on non-game species. However, such management may also have costs to species.
2. We review the literature that examines the effect of different forms of game bird management on non-target species in Europe, including habitat management, predator control, parasite control, provision of water and food, and rear and release. We focus on Europe, where these forms of management are common and sometimes intensive.
3. We identified 35 studies, which recorded 122 individual significant effects. Most studies (80%) focussed on the effects of habitat management and predator control, and >90% were carried out in the UK.
4. 63% of the 122 significant effects on non-game species were positive. Overall, 85% of the effects of habitat management in agricultural habitats were positive, while in non-agricultural habitats 65% of effects were negative. Effects of rear and release were mixed

(8 positive and 7 negative). Legal predator control was almost always positive (96% of effects), or benign, whereas illegal predator control was always negative (8 effects). This continues to be a major cost to conservation. No studies examined the effects of parasite control on non-target wildlife. Three of four significant effects of supplementary feeding were negative.

5. More studies are needed on the impacts of game bird management on non-game species, and particularly of rear and release, the provision of supplementary food and water, and parasite control. We also found few experimental studies examining the specific effects of management for shooting of game birds, and very few studies overall outside the UK. Future studies should aim to fill these gaps.
6. *Synthesis and applications.* The management of game bird populations for shooting is widespread across Europe. Our study shows that effects of such management practices vary between different non-target species. There is a need to understand these trade-offs, find effective strategies to limit the damaging aspects of game bird management, and work to enhance the benefits for the conservation of biodiversity.

Keywords: hunting, uplands, biodiversity conservation, predator control, rear and release, habitat management, parasite control, supplementary feed, game bird management

Introduction

Protected areas lie at the core of strategies for conserving biodiversity (Geldmann et al. 2013). Yet the vast majority of terrestrial and marine systems lie outside protected areas, where conservation means engaging with the legitimate land-uses associated with social and economic activities to promote practices favourable for conserving this biodiversity (Kenward et al. 2011). One form of global land use that cuts across habitat types and landscapes is recreational hunting (Loveridge et al. 2009). Conservation and hunting organisations have shared interests in biodiversity and the value of integrating hunting and game management as a conservation tool has a long history (Leopold 1933; Oldfield et al. 2003; Sotherton et al. 2017). Hunting has the potential to support biodiversity, as management practices specifically targeted at game may have positive effects on non-game species (Oldfield et al. 2003; Arnett & Southwick 2015; Sotherton et al. 2017). Financial investment in management for hunting activities also contributes to the maintenance of some habitat types at the landscape scale, including upland heather moorland and farm woodlands in the UK, and Mediterranean scrub, *dehesas* and *montados* in Spain and Portugal (Robertson et al. 2001; Arroyo et al. 2012). Furthermore, in some parts of the world, hunting can raise considerable funds through taxes and other revenues which feed back into wildlife management (Arnett & Southwick 2015).

Despite the potential mutual benefits, conservation and hunting are often uneasy allies. There are some who question the moral legitimacy of hunting practices (Fischer et al. 2013) or ethical aspects of hunting (McLeod 2007). Others have expressed concern over the wider ecosystem-level effects of management for hunting (Thompson et al. 2016), or the impact of hunting on game species (Benítez-López et al. 2017). The release of game birds may also be detrimental to the wild stock of that species (Diaz-Sanchez et al. 2012), and disturbance due to human hunting activity can negatively influence other species (Tarjuelo et al. 2015). This paper focusses on the population and community level impacts of game bird management practices on non-target species.

There have been previous attempts to synthesise the evidence regarding biodiversity consequences of game bird management (Arroyo & Beja 2002; Gallo & Pejchar 2016; Thompson et al. 2016; Sotherton et al. 2017). However, these studies have either focussed on management for a particular species of game bird (Thompson et al. 2016; Sotherton et al. 2017), been mostly focussed on the impacts on non-game vertebrates (Arroyo & Beja 2002), or have not included the full range of common management practices (Gallo & Pejchar 2016). As such, and given that the potential costs and benefits of game management for biodiversity are contested, our aim here is to provide an assessment of the impacts of the broad suite of common management practices carried out for a range of game bird species, on plants and on vertebrate and invertebrate animals.

The shooting of game birds for recreation, food, or to reduce damage to crops is geographically widespread, and is focussed on members of the order *Galliformes*, waterfowl (*Anseriformes*) and pigeons and doves (*Columbidae*). Management occurs across Europe and North America, although it is at its most intensive and includes a wider range of practices in Europe (Mustin et al. 2012). Furthermore, management practices are primarily targeted at G*alliformes* (Arroyo & Beja 2002; PACEC 2014). Our assessment therefore focuses on the impact of management targeted on *Galliformes* (hereafter game birds) on non-target species across Europe. In Europe, shooting practices vary from "driven shooting", where birds at high density are driven towards the hunters, to "rough shooting", in which the hunter flushes and shoots birds, with or without the aid of a dog. Driven shooting of *Galliformes* is particularly common in parts of central and southern Europe and the UK, and involves more intensive management and attracts high revenues, in contrast to rough shooting where less investment occurs and revenues are lower (Sotherton et al. 2009; PACEC 2014).

The purpose of management is primarily to minimise the effects of the factors limiting the game bird population and to increase the numbers available for shooting. The main types of management for game bird shooting in Europe are: habitat management, predator control, parasite control, rear and release, and the provision of supplementary food and/or water. In agricultural habitats, habitat management for game birds can take the form of reduced pesticide usage (e.g. Sotherton 1991), planting of cover and food crops (e.g. GWCT 1986), and set-aside (e.g. Sotherton et al. 1992), amongst other actions. In non-agricultural habitats, habitat management for game birds mostly involves prescribed burning, mechanical vegetation control and /or grazing control (Arroyo & Beja 2002; Newey et al. 2016). However, in both agricultural and non-agricultural habitats, these types of management practices are not necessarily associated with game bird management, as they are often adopted specifically for conservation or other management purposes.

Legal predator control is common across Europe, in relation to the management of important socio-economic game birds such as red-legged partridge (*Alectoris rufa*), pheasant (*Phasianus colchicus*) and red grouse (*Lagopus lagopus scoticus*). However, the predator species involved vary (Arroyo & Beja 2002). In the UK, Portugal and Spain predators that can be legally controlled include red foxes, feral cats, magpies and carrion crows, amongst others (Arroyo & Beja 2002). However, the illegal killing of protected predators, particularly birds of prey, by those interested in the production of game is of concern to conservation (Whitfield et al. 2004; Redpath et al. 2010; Amar et al. 2011; Marquez et al. 2013), although its extent and impact is hard to evaluate.

Parasite and disease control is carried out either directly through provision of medication, or indirectly via vaccinating or culling individuals of species which are intermediary hosts (Harrison et al. 2010; Gilbert 2016). The rearing and releasing of game birds for shooting is most common in farmland habitats, where it involves tens of millions of farm-reared pheasants and partridges in Europe each year (Arroyo & Beja 2002; PACEC 2014). Finally, the provision of grain for game birds, either over winter or in the breeding period, is a common management practice throughout Europe, and supplementary water may also be provided (Arroyo & Beja 2002).

These interventions in ecosystems can potentially have profound effects on a wide range of other non-target species. Some effects will be direct. For example, we would expect predators and parasites of game birds to be adversely affected by management practices that aim to reduce or eradicate these species. Similarly, management targeted at improving food availability should benefit those plants and invertebrates that act as food. Other effects will be indirect. Thus, ground-nesting birds may benefit from the removal of generalist predators (Côté & Sutherland 1997), a range of species may be expected to benefit from habitat improvements, granivorous species may benefit from the provision of grain (Siriwardena et al. 2007), and a range of species in arid areas may benefit from the provision of water (Borralho et al. 1998). On the other hand, it has been suggested that concentrations of animals around feeders and watering points might have negative impacts by increasing the risk of disease transmission and predation (Arroyo & Beja 2002). Similarly, the release of large numbers of game birds may have impacts on food species, competitors and via the attraction of generalist predators (Draycott et al. 2008; Sage et al. 2009; Draycott et al. 2012; Lees et al. 2013; Delibes-Mateos et al. 2015). Finally, depending on the structure of the predator guild, the removal of predators may have unexpected consequences owing to meso-predator release and trophic cascades (Schmitz et al. 2000; Salo et al. 2010).

Here we review the existing evidence of the consequences of these game bird management practices for non-game species, as published in peer-reviewed literature, and focus on the impacts on diversity, abundance, survival and productivity. We discuss the potential ecological consequences of these impacts, and highlight key knowledge gaps and future research needs.

Materials and Methods

Literature search

We conducted a systematic literature review, using Web of Science, following the guidelines laid out in Pullin and Stewart (2006). After an initial scoping search, testing different combinations of the search terms, a final search string was constructed (see Appendix S1 in Supporting Information for a full description). The database search was conducted between mid-May 2016 and mid-June 2016 (last search on 17th of June 2016). To check for references that might have been missed by our systematic search, we then also conducted a second more general, independent review. The second review used Web of Knowledge and Google Scholar, and the search terms "abundance", "diversity", "breed*" or "surviv*" with keywords relating to each management practice, with and without the terms "game bird" and "manage*". In both reviews, only primary research, published in peer-reviewed journals was considered, on the basis that grey literature lacks the scrutiny of peer review, and therefore cannot be given equal weight. This is especially important when considering a subject that is often value-laden and controversial.

Study inclusion criteria

The systematic search returned 1,735 results which were then filtered by JK, based first on the title, and then on the abstract (Figure S1). A random subsample of 10% of the initial 1,735 search results were assessed by a second reviewer (SR) and results were consistent (Kappa K= 0.89) (Pullin & Stewart 2006). Each assessment was made in relation to six *a priori* criteria, such that studies were included that: investigated the effects on the abundance, diversity, breeding success or survival of any non-target species or group of at least one of the afore-mentioned management practices, being applied specifically to manage G*alliforme* populations for hunting in Europe (see Appendix S1).

After this initial filtering, 44 studies had been identified for potential inclusion in the review (see Figure S1). However, 21 of the 44 studies were subsequently excluded as they contained no primary data (6 studies); or after detailed reading, were found not to meet the inclusion criteria (15 studies). The reference lists of all 44 papers were checked for additional studies, which added seven studies that met the inclusion criteria. We also used Google Scholar to check for more recent papers that cited any of these studies, and this led to three further papers that met the inclusion criteria being added to the review. As such, a total of 33 papers that met the inclusion criteria were identified through this search (see Figure S1). The non-systematic literature search identified two additional papers that met the selection criteria but were not captured in the systematic review. Therefore, a total of 35 studies were passed to the data extraction and analysis stage (see Figure S1).

Data extraction and analysis

The data extraction process was carried out twice, independently, once by JK and once by KM to ensure consistency, and is fully described in Appendix S1. Data on the type of study, management practice/s applied and non-game taxa studied were extracted.

For each management practice, we counted the number of significant positive, significant negative, and non-significant (at the $p<0.05$ level) effects reported. These data were then summed for the main taxonomic groups and based on national conservation status (see Appendix S1; Figure 1). Only results where a significance test had been performed were included. As some studies reported effects of multiple management actions and/or effects on more than one species or taxon, multiple effects were sometimes reported for the same study. A wide range of management practices, study species, and types of outcome were assessed using a variety of methodological approaches in different ecological settings. Furthermore, only half of the studies reported effect sizes. Therefore, a formal, quantitative meta-analysis could not be carried out, and instead a more qualitative synthesis is presented.

Results

Of the 35 studies, four (11%) reported results of experimental studies (Figure 1). Most studies (N=21) reported on species abundance, with 14 reporting on breeding success, 10 on diversity and two on survival. Seventy-six positive effects were reported from 23 studies, and 46 negative effects from 18 studies. Most studies were from the UK, with just two from Spain (Virgós & Travaini 2005; Estrada et al. 2015) and one from Portugal (Beja et al. 2009).

HABITAT MANAGEMENT

In total, we found 13 studies that examined the impact of game bird habitat management on non-target species (Figure 1). These were all from the U.K. Eight of the studies were carried out in agricultural landscapes and five in the uplands. Of the former, six examined the effects of game bird crops, together reporting 28 significant positive effects on the abundance of non-game birds (Stoate et al. 2003; Parish & Sotherton 2004b, a; Sage et al. 2005b; Parish & Sotherton 2008; Aebischer et al. 2016). The remaining two studies focussed on un-sprayed field margins known as “conservation headlands” (White et al. 2008), and on woodland management for pheasants *Phasianus colchicus* – including for example the clearing of open “rides” (Robertson et al. 1988). In summary, the evidence reviewed here suggests that habitat management practices in agricultural landscapes are often beneficial for non-target species (Figure 1, see Table S1).

In the uplands, five studies examined the impact of rotational burning of heather (muirburn) in the UK on birds and invertebrates (Figure1). Muirburn was significantly correlated with reductions in the abundance of some bird and invertebrate species and increases in others (Smith et al. 2001; Tharme et al. 2001; Brown et al. 2013; Ramchunder et al. 2013). In summary, the evidence reviewed here suggests that the effects of muirburn are mixed, being positive for some non-target species and negative for others (Figure 1, see Table S1).

PREDATOR CONTROL

Thirteen studies showed legal predator control to be almost exclusively positive or non-significant for non-game populations (Figure 1). These studies were focussed largely on the abundance and breeding success of birds. Two experimental studies, both from the UK, suggest that the impacts of legal predator control are variable between bird species, though stronger positive effects were usually found on breeding success than on population density (Parr 1993; Fletcher et al. 2010). Correlational evidence provided further support for the positive impacts of legal predator control on abundance and breeding success of non-target birds (Stoate & Szczur 2006; White et al. 2008; Douglas et al. 2014; White et al. 2014; Aebischer et al. 2016). The only negative effect was reported for yellowhammer nest success with sporadic removal of corvids (Eurasian magpie and carrion crow) (White et al. 2014). However, the same authors reported increased nest success for yellowhammer with the systematic removal of legally controllable predators (Eurasian magpie *Pica pica*, carrion crow, brown rat *Rattus norvegicus*, stoat, weasel, red fox, American mink *Neovison vison* and eastern grey squirrel *Sciurus carolinensis*), and they conclude that changes in the behaviour and structure of the remaining predator assemblage as a result of corvid removal may explain the negative impact.

The illegal killing of some species of predators, particularly birds of prey such as hen harriers and peregrines, was also associated with game bird management in some areas. Given the difficulties in directly detecting and quantifying illegal activities, the evidence is based on seven correlative studies, which together reported nine significant negative effects (Figure 1, see Table S1) (Etheridge et al. 1997; Green & Etheridge 1999; Virgós & Travaini 2005; Whitfield et al. 2008; Anderson et al. 2009; Beja et al. 2009; Amar et al. 2011). There were no positive effects reported.

PARASITE CONTROL

We found no studies that examined the effects of parasite control on non-target wildlife.

SUPPLEMENTARY FEEDING

Only two studies examined the effects of the provision of supplementary feed and water (Figure 1, see Table S1), reporting one positive effect on abundance of granivorous steppe-birds in central Spain, and three negative effects on several groups of species of conservation concern in UK farmland where winter grain provision took place (Estrada et al. 2015; Aebischer et al. 2016).

REAR AND RELEASE

Seven studies reported on effects of rear and release (Figure 1, Table S1), including two experimental studies that both reported no significant effects (Clarke & Robertson 1993; Callegari et al. 2014), and five correlational studies reporting a mix of significant positive and significant negative effects on non-game abundance and diversity (Sage et al. 2005a; Draycott et al. 2008; Sage et al. 2009; Draycott et al. 2012; Neumann et al. 2015).

Discussion

Our review highlights that the management of game birds leads to trade-offs for biodiversity conservation. As expected, there are some clear benefits of game bird management for a range of non-target species and negative impacts on other species. Of the 122 significant effects found, 63% were positive, and 37% were negative. For species of conservation importance, there were 16 reported positive effects and 14 negative, of which eight were related to illegal killing of predators. The illegal killing of birds of prey in relation to intensive management continues to be a major cost to conservation.

Habitat management was beneficial for a range of species, especially in agricultural systems, whilst the effects were more mixed in the uplands. In a specific review of habitat management for game species generally, Gallo & Pejchar (2016) similarly found mixed effects and argued for more consistent monitoring of non-target effects to mitigate against the negative effects of management. One of the challenges in identifying the impact of habitat management is that most practices are not exclusive to management for hunting (Stoate et al. 2009). In the UK there has been some quantification of the relative extent of these practices in game and non-game settings, though more clarification is required. In the context of the Agri-Environment Scheme (AES) in the UK, participants who are members of the Game & Wildlife Conservation Trust Partridge Count Scheme have been found to be more likely to employ the best AES options for game and other wildlife than other farms (Ewald et al. 2010). There is also evidence that farms where shooting takes place have a greater willingness to undertake practices specifically for conservation purposes, and to modify farming from optimum levels for the benefit of conservation objectives (Howard & Carroll 2001). In Spain and Portugal, habitats such as dehesas and montados, which are of conservation value, may also be supported by hunting (Robertson et al. 2001; Arroyo et al. 2012). However, in many parts of southern Europe, land owners are not necessarily the owners of the hunting rights, so management decisions in hunting estates may be decoupled from land owner decisions related to habitat (Arroyo et al. 2012), and we know of no specific studies that investigate this phenomenon outside the UK. This warrants further investigation.

Legal predator control is mostly positive for a range of species in the UK. Of the 30 effects reported, only one effect was negative, possibly as a result of compensatory increase in predation by other species (White et al. 2014). Similarly, Madden et al. (2015) showed that corvid removal was much more likely to show a positive impact on prey when it was associated with the removal of other generalist predators. However, all reported studies were carried out in the UK, which has a depauperate predator fauna, with no apex predators. Therefore, there are reasons to expect patterns to vary in other parts of Europe where predator communities are more diverse (see e.g. Bodey et al. 2011). The removal of apex predators may produce an increase in numbers of meso-predators or compensatory responses by other predators (Ritchie & Johnson 2009; Salo et al. 2010), so that overall predation rates are not reduced as expected with possible detrimental effects on other species (Norrdahl & Korpimaki 1995; Palomares et al. 1995; Palomares et al. 1996; Henke & Bryant 1999). More work is therefore needed to establish the consequences of predator control on the meso-predator guild, and potential compensatory effects, to elucidate the subsequent impacts on game and non-game populations.

We found no published studies specifically designed to test the effects of parasite control on non-target wildlife, although concerns have been raised about the potential risks (Thompson et al. 2016; but see Sotherton et al. 2017). However, species that share parasites with game birds may be targeted by management. For example, in Scotland, mountain hares *Lepus timidus* and red deer *Cervus elaphus* are killed in an attempt to reduce the risk of red grouse being bitten by ticks and infected with Louping-ill (Harrison et al. 2010; Gilbert 2016). A survey of Scottish landowners and managers showed that 50% of hares reported killed over 2006/07 were killed as part of tick and tick-borne disease control (Patton et al. 2010). Furthermore, there is evidence that significant declines in hare numbers have been documented on some grouse moors in north-east Scotland where culls have taken place to control ticks and tick-borne diseases (Noble et al. 2012; Watson 2013; Wright et al. 2014).

Millions of game birds are reared and released for shooting across Europe (Caro et al. 2014; PACEC 2014). Concerns have been expressed that this activity may affect non-target species through herbivory by high densities of birds, the spread of diseases and parasites, increased competition for resources, indirect effects via generalist predators, and through potentially lowering investment in habitat management or protection (Draycott et al. 2000; Villanua et al. 2007; Diaz-Fernandez et al. 2012; Diaz-Sanchez et al. 2012; Lees et al. 2013). We reviewed seven studies which found both positive (N=8) and negative (N=7) effects, but the majority (N=46) were non-significant. Overall the evidence for a negative impact of game bird releases on non-game species is not compelling, though appropriate large-scale experiments are absent.

However, game bird releases may affect wild populations of game birds via hybridisation and genetic introgression (hybrids repeatedly cross-breed with one parent species). For example, Barbanera et al (2010) compared genetic diversity across the entire range of red-legged partridge between ancient (museum specimens collected between 1856 and 1934) and modern times using mitochondrial DNA and found significant changes in the haplotype profile. They also found evidence of introgression in the modern samples with chukar partridge across almost the entire species range. They recommend that the import of exotic species and non-local populations of native species for release be banned in the case of intensively managed game species. Hybrids of red-legged and chukar partridge are bred to increase the productivity of farmed birds and it is thought that game releases have spread these hybrids to the wild (Casas et al. 2011). Subsequent fitness consequences have arisen in central Spain, where survival of hybrids is lower, mainly due to higher predation rates (Casas et al. 2011). However, despite lower survival the authors find that the hybrids do breed in natural populations and they conclude that the release of farmed hybrids has consequences for the long-term conservation of wild red-legged partridge populations.

Few studies have examined the effects of supplementary food and water provision for game birds on non-target species. While these practices might be expected to have positive impacts on a range of species (Borralho et al. 1998; Siriwardena et al. 2007; but see Aebischer et al. 2016), they could also have negative impacts by concentrating birds around feeders and watering points and thus increasing the risk of disease transfer and predation (Pennycott et al. 1998; Arroyo & Beja 2002). There is evidence that a wide range of non-game species use watering points, though use by non-game species may be lower than use by target game (Gaudioso Lacasa et al. 2010), and evidence from both within and outside Europe suggests that associating with water developments does not necessarily present a high risk of predation (Destefano et al. 2000; Krausman et al. 2006; Gaudioso Lacasa et al. 2010). Nevertheless, this review highlights clearly that more evidence is required to properly evaluate the population and community level impacts of both food and water provisioning, and the control of parasites and disease.

Identifying the trade-offs.

Whilst many species do clearly benefit from game bird management, others are negatively impacted. The most obvious cost of game bird management to conservation is the impact on protected species of predators through illegal killing. This activity appears to be widespread in certain parts of Europe (Virgós & Travaini 2005; Anderson et al. 2009; Amar et al. 2011; Marquez et al. 2013; Fairbrass et al. 2016). By its very nature, the extent of this activity is hard to quantify, but there is correlational evidence that levels of direct or indirect illegal killing have an impact on the breeding success, survival and abundance of a range of predatory species, many of which are scarce and threatened. This issue has a strong influence on the position of conservationists and conservation organisations towards game bird hunting (Thirgood & Redpath 2008), with some calling for driven red grouse shooting to be banned (Avery 2015) or licenced (Thompson et al. 2016).

In addition to the direct negative effects of management, other species will be indirectly negatively affected. Burning provides a good illustration of the trade-offs. In the UK, fire is used for the management of red grouse habitat, and the management practice is increasing (Douglas et al. 2015). This activity certainly benefits its target species, but it has mixed effects on a range of other birds (Smith et al. 2001; Tharme et al. 2001), vegetation and invertebrates (Hobbs & Gimingham 1984; Shaw et al. 1996; Stevenson et al. 1996; Tucker 2003; Marrs et al. 2004), along with impacts on hydraulic conductivity and macropore flow (Holden et al. 2014) and the degradation of peat-forming bog communities (Ratcliffe & Thompson 1988; Stewart et al. 2004). Too-infrequent burning may also lead to increased risk of more serious wildfires (Davies et al. 2008) and lead to scrub and woodland encroachment, ultimately leading to the loss of heather habitats (Hester & Sydes 1992). Management decisions must therefore be carefully made, and should depend on both the management goal and on the species that we want to benefit.

Limitations & future challenges

We see four main limitations of the available evidence summarised by this review. First, there were relatively few studies and only half of these reported effect sizes. Second, it was not always possible to identify the relative contribution of each specific management practice, although general game bird management may be positively associated with species richness and abundance (Hinsley et al. 1999; Stoate & Szczur 2001; Stoate 2002; Stoate et al. 2002; Caro et al. 2015). Third, the majority of the available evidence is correlational. We urgently need more experimental studies where effects can be separated and attributed to the individual management action itself (Clarke & Robertson 1993; Parr 1993; Fletcher et al. 2010; Reynolds et al. 2010). Fourth, there are important geographic biases in the literature, with most of the studies coming from the UK. As such, there is a need for further research from elsewhere to establish the generalisability of findings across contexts.

Management for game birds varies substantially in its intensity and it is the most intensive forms of management that are the focus of much debate of the wider ecosystem-level impacts, alongside economic and social arguments (Thompson et al. 2016; Mustin et al. 2017; Sotherton et al. 2017). From a conservation perspective, arguments revolve around minimising the costs of game bird management for ecosystems and species of most concern (Thompson et al. 2016). From a management perspective, arguments revolve around the benefits of high intensity shooting for livelihoods and communities as well as highlighting the species that benefit from management (Sotherton et al. 2017).

The recreational shooting of game birds is a controversial activity. Even if associated biodiversity benefits were overwhelmingly positive, it is still likely that some people would not support it for reasons of divergent values or morals, or concerns over the wider ecosystem-level effects (McLeod 2007; Fischer et al. 2013; Thompson et al. 2016). In this review, we have focused on the biodiversity consequences of game bird management. However, aspects such as morality and legitimacy of hunting play a critical role in how individuals and organisations engage and collaborate to benefit biodiversity, and are at the centre of current debates about the future direction of conservation. From a conservation perspective, the challenge is to promote the beneficial practices associated with game bird management as a mechanism to achieve the goals of biodiversity conservation whilst reducing the negative effects on protected species, habitats and the wider landscape.

Authors' contributions

KM carried out the second literature review, led synthesis of results and writing, and contributed to conception of ideas and discussion of results; BA, PB, SN, and RJI contributed to conception of ideas, discussion of results and to writing; JK carried out the systematic literature review and contributed to synthesis of results; SR led conception of ideas and discussion of results, and contributed to the systematic literature review, synthesis of results and writing. All authors contributed critically to the drafts and gave final approval for publication.

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Supporting Information

Additional supporting information may be found in the online version of this article:

Appendix S1. Full details of the review methodology

Table S1. Details of the data extracted from each of the 35 references

Figure S1. Schematic to show the process of study exclusion/inclusion

Data accessibility

Data have not been archived because this article does not use data.

Figure legends

Figure 1: Reported significant positive and negative effects of game bird management practices on non-target wildlife abundance, diversity, breeding success and survival. "Listed species" are those listed as of conservation concern in the country where the study took place (UK – Biodiversity Action Plan species, Red-Listed Birds of Conservation Concern or Schedule 1 of the Wildlife and Countryside Act; Spain and Portugal - national red lists of vertebrates). Symbols represent the management practices (agricultural habitat management, non-agricultural habitat management, legal and illegal predator control, supplementary feeding and rear and release), and taxa affected (birds, invertebrates, plants, and mammals).

The number of non-significant effects and the number of correlational and experimental studies are also given. Some studies report on the effects of more than one management practice or present more than one result, i.e. for multiple species or groups of species, and these results have been counted separately. References cited in the figure are as follows: [1]White *et al.*, 2008, [2]Robertson *et al.*, 1988, [3]Stoate *et al.*, 2003, [4]Parish & Sotherton, 2004a, [5]Parish & Sotherton, 2004b, [6]Sage *et al.*, 2005b, [7]Parish & Sotherton, 2008, [8]Aebischer et al, 2016, [9]Smith *et al.*, 2001, [10]Tharme *et al.*, 2001, [11]Ramchunder *et al.*, 2013, [12]Brown *et al.*, 2013, [13] Newey et al. 2016, [14]White *et al.*, 2014, [15]Green & Etheridge, 1999 ,[16]Stoate & Szczur, 2006, [17]Whitfield *et al.*, 2008, [18]Estrada et al. 2015, [19]Reynolds, Stoate et al. 2010, [20]Baines and Richardson 2013, [21]Douglas *et al.*, 2014, [22]Parr, 1993, [23]Fletcher et al. 2010, [24]Etheridge *et al.*, 1997, [25]Virgós & Travaini, 2005, [26]Anderson *et al.*, 2009, [27]Beja *et al.*, 2009, [28]Amar *et al.*, 2011, [29]Neumann et al, 2015, [30]Sage *et al.*, 2005a, [31]Draycott *et al.*, 2008, [32]Sage et al, 2009, [33]Draycott *et al.*, 2012, [34]Callegari *et al* 2014, [35]Clarke & Robertson, 1993.

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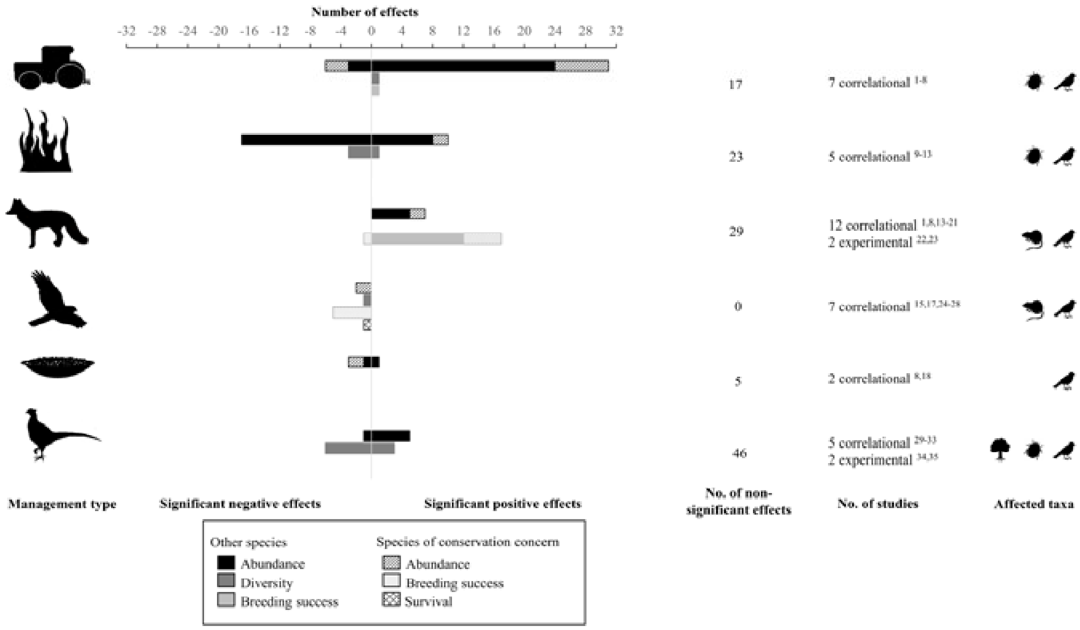
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No. of studies
Affected taxa
Other species
Abundance
Diversity
Breeding success
Species of conservation concern
Abundance
Breeding success
Survival