Predator management for breeding waders: a review of current evidence and priority knowledge gaps

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ABSTRACT

Rapid declines in breeding wader populations across the world have prompted the development of a series of conservation tools, many of which are designed to influence productivity. Across western Europe, efforts to reverse population declines are typically limited by high levels of nest and chick predation and, managing this predator impact has been a major research focus in the last two decades. A workshop held at the 2019 International Wader Study Group conference aimed to synthesise current understanding of predator management tools and to use expert knowledge to identify and prioritise important knowledge gaps in this area. Here we review the four predator management tools that were described (predator diversion, exclusion, lethal control and headstarting), together with insights into the potential responses of mammalian predators to these management tools. The expert assessment of important areas for future work highlighted the need to increase our knowledge of predators and their responses to management interventions; to ensure our science connects to policy, practitioners and members of the public; and the need for clear and consistent goals for the future of breeding wader populations to inform the development and deployment of these management tools.

INTRODUCTION

Across western Europe, widespread drainage and agricultural intensification have driven declines in wetland biodiversity, and breeding wader populations have been a particularly prominent casualty of these processes (Wilson *et al.* 2004, Smart *et al.* 2008). Once common and widespread, breeding wader populations are increasingly confined to nature reserves (Smart *et al.* 2006), and have continued to decline despite the creation and management of conditions suitable for breeding in nature reserves and, through agri-environment schemes, in the wider countryside (O'Brien & Wilson 2011, Smart *et al.* 2014). The life history of waders is generally characterised by low fecundity and high adult survival but, while variation in survival rates contributes greatly to population dynamics, manipulating survival is rarely feasible. By contrast, management to enhance productivity is common, with the ultimate goal of increasing numbers of breeding individuals. One of the primary reasons associated with the failure of declining wader populations to recover is unsustainably high levels of nest and chick predation, and consequent low levels of recruitment into breeding populations

(MacDonald & Bolton 2008a, Laidlaw *et al.* 2017, Kentie *et al.* 2018). There is evidence that nest predation rates have increased in recent decades (Roodbergen *et al.* 2012), and a recent review of predator impacts on bird populations found that waders were commonly limited by predation (Roos *et al.* 2018). The predators of wader eggs and chicks are typically generalist mammalian and avian predators and, consequently, managing their impacts on specific populations (which may comprise only a small part of their diet) is challenging. In addition, several of the avian predators that can be important predators of wader chicks (Mason *et al.* 2018) are themselves of protected conservation status (especially raptors). A series of different conservation tools have been used to try to reduce predator impacts on breeding waders (e.g. see Colwell (2019) for *Charadrius* Plover examples). The aim of a recent predator management workshop held during the 2019 International Wader Study Group (IWSG) conference was to synthesise current understanding of the deployment and effectiveness of a selection of these tools, and to identify and prioritise knowledge gaps that need to be addressed.

The predator management tools considered at the workshop included (1) diversionary techniques, which aim to reduce levels of nest and chick predation by altering the relative attractiveness of the landscape or resource base; (2) exclusion techniques, which aim to create barriers between predators and nesting waders; and (3) lethal control techniques which aim to reduce local predator abundance. Our understanding of these tools are summarised below, together with details of their design and deployment. Studies of these tools have focussed almost entirely on their effectiveness at reducing predation levels, and very little attention has been paid to how predators might respond to the use of these tools. Consequently, this issue is also considered below. Finally, a more recently developed emergency intervention tool for increasing hatching, fledging and recruitment rates is headstarting, which involves removing eggs and rearing chicks in captivity through the period of greatest vulnerability to predation. This technique is also described.

Attendees at the workshop spanned a broad range of stakeholders in breeding wader conservation, and included researchers, landowners, conservationists and representatives of organisations involved in the development of conservation policy. Following presentations on each of four predator management techniques, attendee discussion was used to identify knowledge gaps and the long-list of questions resulting from this process was subsequently reduced to 12 through round-table discussion by the plenaries. Attendees were then asked to rank each of the 12 short-listed questions on three criteria; urgency, importance and feasibility (Table 1). The resulting scores (numbers of attendees ranking high, medium or low for each criterion applied to each question) were then synthesised and discussed. Here we present (i) reviews of the evidence for the effectiveness of each of the predator management techniques, including potential implications for the responses of mammalian predators to these activities, and (ii) for each identified question, the outcome of the scoring of criteria and the main points arising from the discussion of these issues.

1. PREDATOR DIVERSION

Managing breeding wader habitat

Strategic habitat management in landscapes that support breeding waders is likely to influence how predators interact with waders and other prey. Relatively simple forms of strategic habitat management aim to reduce accessibility of sites to predators, availability of predator breeding locations (e.g. trees, dry banks or reedbeds) and/or opportunities for predators to hunt effectively (e.g. through removal of perches for avian predators).

Landscape-scale habitat management can potentially be used to influence the impact of predators on breeding waders. In Dutch grasslands, numbers of breeding Black-tailed Godwits *Limosa limosa limosa* are declining rapidly (Kentie *et al.* 2016, Roodbergen & Teunissen 2019), and densities increase along a gradient of land-use intensity from herb-poor meadows and grassland monocultures to herb-rich meadows (Groen *et al.* 2012), with important habitat-specific differences in demographic rates. Black-tailed Godwits breeding in monocultures tend to experience lower nest survival (Kentie *et al.* 2015) and lower survival of chicks, possibly due to a combination of low food availability and higher predation rate (Kentie *et al.* 2013), compared to herb-rich meadows where population growth rates can be positive (Kentie *et al.* 2018). In this example, landscape-scale variation in land-use intensity is having population-level effects through complex interactions between management, predation and breeding success, and strategic management of landscape structure could potentially be used to alter these relationships. Similar processes also operate in other species and study systems. For example, the abundance of wet features positively influences the breeding density of some wader species on wet grasslands (e.g. Smart et al. 2006, Eglington et al. 2008) with important density-dependent reductions in predation rates of nests and chicks (MacDonald & Bolton 2008b, Eglington *et al.* 2009, Laidlaw *et al.* 2017). Reducing the accessibility of wader breeding areas, for example by surrounding them with water, may deter some mammalian predators, although both European Badgers *Meles meles* (hereafter, Badgers) and Foxes can and do swim, if necessary.

Managing non-wader prey

The availability of small mammal prey in wader landscapes could also have important implications for the generalist predators that prey on small mammals and waders (e.g. Foxes, Stoats *Mustela erminea*, Weasels *M. nivalis* and raptors), so understanding how management influences small mammal distribution is important. For example, the presence and activity of Common voles *Microtus arvalis* can vary across grazing regimes, and grazing management can be used to manipulate vole presence (Lagendijk *et al.* 2019). There is also a need to understand the influence of agricultural activities on the availability of key resources for predators (Pringle *et al.* 2019).

Wet grasslands managed for waders are generally unsuitable for small mammals (too short and wet), which mostly occur in the taller and denser vegetation of verges outside of grazed fields (Laidlaw *et al.* 2013). Northern Lapwing *Vanellus vanellus* (hereafter Lapwing) nest predation rates have been shown to be lower on wet grassland fields with more surrounding verge habitat (Laidlaw *et al.* 2015), and the magnitude of this effect is such that increasing the amount of verge in wet grassland landscapes could, in theory, reduce nest predation rates by up to ~20%, but only in areas with high lapwing nesting densities (Laidlaw *et al.* 2017). Managing habitat to benefit the non-wader prey of key predators could therefore have implications for wader demography.

Potential predator responses to diversion techniques

In the case of raptors, which are species of conservation importance protected by law but important predators of wader chicks (Mason et al. 2018), diversionary techniques to reduce their impact may be most appropriate, particularly when raptor predation pressure is localised and substantial. In these situations, providing diversionary food directly to focal raptors during the breeding season, with the aim of reducing their need to hunt, has been shown to significantly reduce predation rates on chicks (e.g. Red Kites *Milvus milvus* predating Lapwings*:* RSPB unpublished data; Kestrels *Falco tinnunculus* predating Little Terns *Sternula albifrons*: Smart & Amar 2018). There are other potential methods for diverting avian predators away from important breeding areas. For example, laser-hazing involves directing a laser beam at the body of the predator to dissuade them from hunting, but trials of the efficacy of this method (at Tern colonies) have thus far been inconclusive because it has proven difficult to haze a sufficiently large proportion of predators, and there appear to be inconsistent effects of hazing on predation attempts and success (RSPB unpublished data).

In the case of mammalian predators, the cover provided by shrubs and trees, and the availability of suitable areas for breeding (e.g. subterranean earths for foxes) can be very important, and removal of these features could potentially divert them away from wader breeding areas. However, the area over which such features may have to be removed could be extensive and, may therefore not be financially or practically feasible. Reducing the attractiveness to predators of wader breeding areas through, for example, provision of alternative high quality and accessible foraging habitats could, in theory, encourage predators to focus their activity away from wader breeding areas (Mukherjee *et al.* 2009), but predator dissuasion is likely to depend on predator abundance and the spatial and temporal distribution of resources.

Manipulating habitats to enhance small mammal populations could have the unintended effect of allowing the area to support higher densities of predators due to an increase in prey abundance, and changes in the availability of key prey species could influence mammalian predator responses to diversion techniques. For example, Rabbits *Oryctolagus cuniculus* and small mammals are a key component of the diet of rural Foxes (Soe *et al.* 2017), and rabbit populations have declined across Europe (Smith & Boyer 2007); in the UK, a 62% decline has been reported between 1996 and 2017 (Harris *et al.* 2019), in part linked to the recent occurrence of Rabbit Haemorrhagic Disease (RHD). Blanco-Aguiar et al. (2012) documented an avian predatory switch from Rabbits to gamebirds as a consequence of Rabbit declines from RHD in Spain. Additionally, Water Voles *Arvicola amphibius*, a wetland vole species which are likely to have been alternative prey for Foxes foraging in wetland habitats (Short & Porteus 2018), have seen dramatic declines in distribution and numbers in the UK (90% decline since 1970's; Jefferies *et al.* 2003). It is unknown if current mammalian prey declines are causing shifts in the diet of predators towards breeding waders.

While there has been considerable research into some aspects of predator diversion tools there are still several important questions that need to be addressed. Key knowledge gaps include the behavioural and demographic responses of predators to the deployment of these tools, especially increased provision of non-wader prey, the potential for predator dietary shifts in relation to changes in prey availability, and the scale of deployment of habitat management, diversionary feeding, or predator dissuasion that would be required to achieve local population growth of waders.

2. PREDATOR EXCLUSION

Over the last two decades, the potential for predator fencing to improve wader breeding success by excluding mammalian predators (particularly Foxes and Badgers) from nesting areas has been widely explored. A variety of fence types and designs have been employed, to address a wide range of contexts. In particular, fences can be designed to operate at different spatial and temporal scales. Spatially, fences to exclude large mammalian predators can be deployed from individual nests up to whole sites, and temporally, deployment can range from temporary (e.g. covering only the period when nests are active) to seasonal (e.g. covering some or all of the breeding season) to permanent (Figure 1). In addition, fences can operate through electrification or by creating a physical barrier that is impenetrable to the larger mammalian predators of nests and chicks. Nest enclosures (i.e. physical barriers to predators placed directly over nests) can enhance hatching success, but nest abandonment and predation of incubating adults have also been recorded (Isaksson *et al.* 2007, Barber *et al.* 2010), and so the overall benefit of this management approach remains unclear.

Temporary fencing at smaller spatial scales (individual nests to fields) tends to involve electrified fences that are easy to construct and move around (e.g. stranded wire livestock fences) while permanent, site-scale fencing tends to involve barrier fencing, which can be of sufficient height and/or buried depth to exclude mammals capable of jumping and/or digging, or existing permanent livestock fences can be electrified (the latter are often termed 'combination' fences). Combination fences provide both a physical and an electric barrier and are commonly used in conservation settings. Further details on fencing design, installation and maintenance together with the advantages and disadvantages of different fence types can be found in the detailed guidance produced by the RSPB (White & Hirons 2019). In general, temporary electrified fences are relatively cheap and easy to deploy but require reliable electricity supplies (mains or battery, potentially with solar panel charging) and regular monitoring, and batteries can drain rapidly if vegetation is not kept sufficiently short to avoid contact with the fences. By contrast, permanent barrier or combination fences are generally more durable and easier to maintain but are also more expensive to construct and can restrict movements of non-target species. Fence designs have developed greatly in recent years and following the most recent guidelines closely is likely to be extremely important. In addition, ongoing maintenance and management of all fence types is essential to ensure that an effective barrier is maintained.

Several studies have explored the effectiveness of fences at excluding mammalian nest predators, typically by comparing either fenced and unfenced areas, or comparing areas before and after fence deployment. These studies typically report substantial improvements in hatching success inside fences for all scales and types of fences, with hatching success rates of around 80% being regularly reported in fenced areas (Maslo & Lockwood 2009, Rickenbach *et al.* 2011, Malpas *et al.* 2013). Consequently, fencing has rapidly become a key component of breeding wader conservation actions across western Europe. Fences do not exclude avian predators and smaller mammalian predators (e.g. mustelids) and so the consistently high hatching success achieved within fences supports the previous evidence that larger mammals are responsible for the majority of wader nest predation in these areas. A much larger range of predators (including avian predators) can be responsible for chick predation. Fences do not exclude many of these chick predators and the precocial chicks of waders can leave fenced areas, but the evidence to-date suggests that the increase in hatching success achieved with fencing can translate into high levels of fledging (Rickenbach *et al.* 2011, Malpas *et al.* 2013), although this is not always the case (e.g. Hoodless & MacDonald 2016).

While a great deal of trialling and testing of predator-exclusion fencing has been conducted, and while there is strong evidence of the effectiveness of fences as a nest protection tool, several important questions have yet to be addressed. These include the capacity of fences to facilitate breeding wader population recovery, the deployment strategies that could deliver such a goal and the extent to which fences need to be deployed in combination with other predator management techniques (e.g. lethal predator control to reduce predator pressure and/or predator diversion techniques to avoid high levels of chick predation).

Potential predator responses to exclusion

Fencing is one of the most effective exclusion interventions for mammalian predators (Khorozyan & Waltert 2019), but it’s effectiveness could potentially be improved by being used in combination with predator dissuasive tools, e.g. acoustic (high pitched sounds), visual (e.g. flashing lights) or chemical (scent based), that aim to deter predators by overwhelming their senses. The success of these deterrents is typically context-dependent, and over-exposure can sometimes lead to habituation (Khorozyan & Waltert 2019), and the effects of such deterrents on breeding waders is unknown. Temporary fencing could potentially exclude mammalian predators from areas which were previously part of a home range, which could result in range shifts and increased between-group aggression, reductions in body condition and survival and increases in stress and disease occurrence (Williamson & Williamson 1984). If the patch excluded is large and/or high quality this could result in tenacity to penetrate the barrier. For some terrestrial predators, persistence can result in individuals assessing fences for weak spots where fences can be breached. Fencing without consideration of the quality and extent of the remaining landscape for predator use may therefore increase risks of fence breaches.

3. PREDATOR CONTROL

The concept of increasing wader productivity and population size through lethal control of predators stems from wild gamebird management, where culling of predators is regarded as fundamental, alongside the provision of nesting and brood-rearing cover (Potts 1980). Control typically involves the removal of Foxes and corvids from the area where waders breed, and often from a buffer strip of 500-1,000 m surrounding this core area. It may also involve control of small native mustelids (Stoat and Weasel) or the invasive American Mink *Neovison vison* which, as an exotic predator, potentially renders evolved defence mechanisms of waders less effective. Methods and seasonal timing of control vary between countries owing to differences in national and regional legislation. Methods used include day/night shooting and various live-capture traps and neck-snares for Foxes, shooting and cage-trapping for corvids, and killing or live-capture traps for mustelids. During the last five years, night vision and thermal-imaging rifle-scopes have become more widely used and have started to replace traditional spotlighting for Fox control (GWCT, unpublished data). These new technologies, in combination with the use of trail cameras to detect predator presence and trap alarm systems, have generally led to improved efficiency of predator control.

When implemented at the landscape level, lethal control can result in local and regional predator suppression (Heydon & Reynolds 2000a, b, Heydon *et al.* 2000, Porteus *et al.* 2019). Lethal control has been shown to be effective at increasing breeding productivity of several wader species above the level required for stable populations in different countries and situations (e.g. Niemczynowicz et al. 2017). In the UK uplands, for example, experimental control of Foxes, corvids and small mustelids resulted in an average threefold increase in the breeding success of Lapwing, Golden Plover *Pluvialis apricaria* and Curlew *Numenius arquata*. Importantly, greater breeding success translated into increases in breeding numbers (≥14% per annum) for these three species, compared to ongoing declines in numbers (≥17% per annum) in the absence of predator control, although no effect was recorded for Snipe *Gallinago gallinago* (Fletcher *et al.* 2010). Large-scale surveys indicate that predator control on grouse moors in the UK uplands leads to higher breeding wader densities than on moorland with no predator control, and increases in wader populations have been documented following the reinstatement of predator control (Tharme *et al.* 2001, Littlewood *et al.* 2019, Ludwig *et al.* 2019).

On lowland wet grassland at the Dümmer reserve, NW Germany, Black-tailed Godwit fledging success during six years of Fox control averaged 0.83 chick/pair (n = 136 pairs), compared to 0.27 chick/pair (n = 62 pairs) over seven years without Fox control (Belting pers. comm.). Across Lower Saxony, monitoring of 2,537 pairs of Black-tailed Godwit over 14 sites during 2012-2017 revealed fledging success greater than 0.7 chick/pair only at the four sites, supporting 853 pairs, where efficient Fox control was undertaken (Belting pers. comm.). However, an effect of predator control is not always apparent (e.g. Bodey et al. 2011). In an eight-year experiment across 11 nature reserves, Bolton et al. (2007) found that reducing Fox and Carrion Crow *Corvus corone* numbers had no overall effect on Lapwing nest survival rates or population trends, although twice as many pairs fledged young at six sites during periods of predator control. In addition, reductions in nest survival in the presence of predator control were apparent when controlling for the background density of Foxes and Carrion Crows, indicating that the impact of predator control on nest survival rates may vary depending on the density of predators present at that time (Bolton *et al.* 2007) .

Several meta-analyses of the effect of lethal control on bird populations, all including studies on breeding waders and other ground-nesting birds, have concluded that the average overall effect is positive but that there is great variation in effect sizes among species and locations (Côté & Sutherland 1997, Holt *et al.* 2008, Smith *et al.* 2010). There are many possible causes for these variable responses to predator removal, including annual variation in the abundance of predators or alternative prey, abiotic factors, such as poor weather at hatching or catastrophic losses due to flooding, an impact from other predators which have not been targeted, density-dependent effects, individual variation in predator behaviour, or inefficient predator control.

Lethal control is the most emotive and controversial of the conservation tools for increasing wader productivity but may be the only feasible option in certain landscapes and for species which breed at low density and whose broods wander over large areas. For instance, exclusion fencing is largely impractical for Lapwings nesting in arable fields and for Curlews in upland areas, whereas lethal control has the advantage that it affords protection to both nests and chicks. In situations where a wader population is critically low, lethal control can buy time to address habitat issues and, if conducted efficiently at a large enough scale, it might reduce the predation problem at the landscape scale (Heydon *et al.* 2000). The need for lethal control also needs to be clearly explained, to maintain support for a recovery project. Disadvantages are that it requires competent practitioners following best practice and, even then, some methods risk the capture of non-target species. The outcome of lethal control in a given location is difficult to predict, and there is a risk that by removing Foxes and corvids, predation by species that are protected (e.g. Badger, Buzzard *Buteo buteo*) or more difficult to control (e.g. Stoat) increases. It is therefore essential to undertake adequate monitoring of predation rates, to avoid unintended consequences such as compensatory predation (Dion *et al.* 1999).

Monitoring before, during and after deployment of lethal control is important to check that predation is the main cause of low wader productivity, to identify the predator species responsible, and then to ensure that lethal control results in the desired outcome. In some cases, the main predator may be a legally protected species and alternative management tools will have to be considered. If lethal control is identified as a necessary tool to boost a wader population, clear aims should be defined at the outset, encompassing the methods to be used, scale, timeframe, cost and method of measuring the outcome. Where legislation permits, control leading up to and during the wader breeding period (January-July) is considered most appropriate as the aim should be seasonal predator suppression rather than local eradication. In the study by Fletcher et al. (2010), for example, the increase in wader numbers was achieved with a 43% reduction in spring Fox numbers and a 78% reduction in Carrion Crows. Implementation of lethal control must be legal, proportionate and, because it is controversial, with the potential for detrimental impact on a project or conservation organisation, justifiable. Collection of data on wader productivity, predator density and numbers of predators killed is, therefore, essential so that the approach taken can be evaluated and justified. For example, while the RSPB considers Fox control to be important on some of its key breeding wader reserves, it has a policy of ensuring that practitioners must ensure no orphaning of dependent cubs. Monitoring on its reserves during 2012-2018 showed that annual Lapwing productivity on reserves with Fox control averaged 0.78 ± 0.15 chick/pair compared with 0.47 ± 0.06 chick/pair on reserves with no Fox control, which, in conjunction with the number of Foxes removed, justified this approach.

Ultimately, to reduce the need for lethal control, and possibly other interventions, it is important to investigate why generalist predators occur at such high densities in the landscape and what has driven increases in their numbers, and impacts on ground-nesting birds, over the last 30-40 years. Better understanding of predator populations will inform the development of more sustainable solutions for recovery of declining wader populations in the long-term. In the short-term, the focus should be on filling knowledge gaps that will help make lethal control more efficient and effective. More studies are needed on the behaviour and detectability of predator species, including how predators use landscapes, which may enable practitioners to target their management better (e.g. Reynolds et al. 2004) and measure its impact. Further research is needed on the effects of controlling predators on the wider ecological community. For example, it is currently unclear whether controlling some predators, particularly Foxes, results in functional or numerical responses of other meso-predators, leading to compensatory predation on wader eggs and chicks (see Trewby et al. 2008, Ritchie & Johnson 2009). Meso-predator increases would be especially detrimental to wader populations if the new suite of predators were legally protected and/or could not be controlled effectively. Finally, it is important to understand the situations in which lethal control is most effective and when it should be combined with other techniques, such as exclusion fencing.

Potential predator responses to control

In the UK, the National Game Bag Census suggests Fox numbers are relatively stable after a period of increase during 1960-early 1990s (Aebischer *et al.* 2011). Foxes are territorial with a social group that defends the territory against surrounding groups. In addition, there is often a smaller proportion of itinerant individuals that do not hold a home range but move across multiple social groups (Storm *et al.* 1976). Loss of an individual in a territorial social group through culling can affect the social unit, leading to changes in movements and territories (Ham *et al.* 2019) and potentially breeding opportunities. Dependent cubs could also perish, likely through starvation and dehydration, although there is usually a sex bias towards males during culling (Kämmerle *et al.* 2019). Lethal control can reduce social group size and thus group capacity to defend the territory, potentially creating a territory vacuum or 'sink' into which new individuals can move, with consequences for the level of culling likely to be required to maintain suppressed Fox numbers (Porteus *et al.* 2019). In studies of Badgers, culls have been shown to result in greater movement of individuals between social groups (Tuyttens *et al.* 2000). Understanding the economic costs of culling and its relative effectiveness needs to be compared to other non-lethal approaches, alongside the ethical considerations of culling one native species to protect another native species. Finally, culling one predator type can potentially lead to increases in other predators within the community, through competitor release and changes in trophic interactions (e.g. Molsher et al. 2017).

4. HEADSTARTING

Headstarting waders to increase the productivity of a wild population is a relatively new concept but this technique has been used in the amphibian, reptile and fish world for over 50 years (Huff 1989, Heppell *et al.* 1996, Fraser 2008). There are various definitions of 'headstarting' but of most relevance to waders is "a conservation technique in which young animals are raised artificially and subsequently released into the wild. The technique allows a greater proportion of young to reach independence, without predation or loss to other natural causes" (Alberts *et al.* 2004). Species and populations that are most suited to headstarting are those that: (i) experience high mortality during early growth stages, (ii) can be successfully raised in captivity with a high fledging rate, (iii) have relatively high survival in later life stages and are long-lived, (iv) mature quickly, (v) would be expected to recruit to the release population or area (i.e. show a degree of natal philopatry) and (vi) where the number of headstarted individuals contribute a reasonable proportion of the population size to which they are expected to recruit.

Headstarting has been used in various forms for a variety of wader species, including Piping Plover *Charadrius melodus* (Powell *et al.* 1997), American Oystercatcher *Haematopus palliatus* (Collins *et al.* 2016), Spoon-billed Sandpiper *Calidris pygmaea* (Pain *et al.* 2018), Black-tailed Godwit and Curlew. The impact of headstarting will vary depending on the size of the target population, productivity in the wild, and the ability of captive operations to increase the survival of eggs and/or chicks and release healthy birds capable of survival in the wild. Preliminary analysis suggests headstarting, often involving early removal of clutches from just 10 adult pairs per year who then go on to re-lay in the wild, may be slowing the global decline of the Spoon-billed Sandpiper (Clark *et al.* 2018) and is increasing the productivity of UK Black-tailed Godwits from 0.34 to 1.1 fledglings per pair (RSPB/WWT unpublished data). These projects both involve marking and tracking of headstarted individuals and, for both these migratory species, headstarted individuals have migrated successfully and returned to project areas to breed and have produced their own young.

While headstarting can be a powerful conservation tool, it is associated with a number of significant risks and, like other conservation methods, will only result in long-term benefits if conducted as part of a wider conservation effort that addresses the underlying cause(s) of decline. Risks include inadequate care or housing during the captive phase that results in mortality or low fitness in released birds, behavioural modifications, infectious disease, lack of imprinting on natal areas and negative impacts on the source population. Many of these risks can be successfully managed by ensuring headstarting operations (i) are conducted by experienced, multi-disciplinary teams (including animal care specialists, veterinarians, site managers and scientists), (ii) are well-planned and based on a clear conservation case determined using population modelling, and (iii) include comprehensive disease management and post-release monitoring.

The high-degree of uncertainty associated with headstarting raises many questions such as will released birds return, survive as well as their wild counter-parts, breed successfully or will their treatment in captivity affect later behaviour? The uncertainty of headstarting presents two key challenges, the first of which is good decision-making, ensuring that headstarting is undertaken in circumstances where it can be effective ,but also ensuring opportunities to benefit a population through headstarting are not missed. Taking a risk-based approach, using population modelling and completing a comprehensive feasibility assessment can aid decision-making. The second key challenge is increasing our understanding of headstarting when experiments are often not possible due to the target population being threatened, and time and resources being limited. As such, it is vital that headstarting efforts are designed as trials and learning is maximised through close monitoring and detailed reporting of the failures as well as the successes.

There are a number of guidance documents available to help manage the risks and meet the challenges associated with headstarting (e.g. Lee et al. 2012, IUCN/SSC 2013, National Species Reintroduction Forum 2014a, 2014b). Figure 2 presents a set of processes that should be followed from project initiation through to monitoring outcomes (specifically marking and tracking of headstarted individuals to quantify subsequent survival and recruitment), adapted from the Scottish Code for Translocations (National Species Reintroduction Forum 2014a).

DISCUSSION OF KNOWLEDGE GAPS

Workshop attendees were asked to rank each of the 12 short-listed questions on three criteria: urgency, importance and feasibility (Table 1). These questions were derived through plenary discussion from the knowledge gaps raised during the presentations and group discussions. Attendees assigned high, medium or low classifications for each criterion applied to each question (Figure 3). We present the knowledge gaps in order of the proportion of the audience that considered the urgency to address that knowledge gap to be high (Figure 3). The scores assigned to the questions for urgency and importance were broadly similar, indicating that questions tended to be considered as high in both urgency and importance, or medium/low in both urgency and importance (Figure 3a & b). Most questions were considered to have medium or low feasibility, with none receiving a majority score of highly feasible (Figure 3c). The process of determining and prioritising knowledge gaps revealed that there was particular importance assigned to determining an appropriate and achievable vision for breeding wader populations in the future, which we address initially below. We then discuss the remaining knowledge gaps that are focused around three topic areas: (i) increasing our understanding of predator responses to management, (ii) connecting to policy, uptake and transferability of management options and (iii) the wider implications of predator management.

Determining an appropriate and achievable future vision for breeding waders and predator management is integral to determining whether we are carrying out the most urgent and important work required to attain our desired outcome (Figure 3; [Q11], with square brackets hereafter referring to numbered knowledge gap). While targets and goals provide something to aim for, they are often narrow in focus and concentrate on site, landscape or regional levels. While it may seem comfortable to have a realistic target of a certain number of pairs in a local population, the setting of targets will likely be influenced by our preconceptions, and ultimately it may not be the place of practitioners alone to determine these targets. Achieving the goal of increasing local populations on managed areas using the management tools discussed here may be feasible, but if our fundamental objective is to re-establish wader populations in the wider countryside, then we are likely to need to extend beyond the currently available management tools (Lyons *et al.* 2008). Having an appropriate *vision* for the future may also allow us to harness the efforts of people working or living across different countries and habitats towards the same outcomes. Consideration of our collective vision is an important first step as it has the potential to influence how questions concerning the remaining knowledge gaps we present might be framed.

Increasing our understanding of predator responses to management

The knowledge gaps concerning predator responses to management were focussed on understanding the causes of high predator densities [Q1], and the potential role of gamebird releases in parts of Europe in which they occur was highlighted. Determining factors that influence predator behaviour and predator detectability [Q2] and how predator communities respond to predator management interventions [Q4], which includes the possibility of meso-predator release (Crooks & Soulé 1999), were also important areas of future research that could greatly influence the design and deployment of predator management tools. Three of the four knowledge gaps that scored highest on urgency and importance concerned the need for improved understanding of predators ([Q1], [Q2] and [Q4]). Attendees considered there to be particularly low feasibility for determining the impact of apex predators on meso-predator effects on waders ([Q9]; highest number of votes given to "low"; Table 1).

Connecting to policy and transferability of knowledge

Knowledge gaps that were concerned with dissemination of information regarding predator management were also highlighted during discussions. How interventions can be supported by policy [Q3], and how we can influence the uptake, use and understanding of these tools [Q6] both scored highly on the metrics of urgency and importance (Figure 3). Determining how information regarding predator management could be used to influence public perception and behaviour [Q8] and how transferrable our current knowledge is [Q10] to other habitats and species facing the issue of predation were also issues considered important, but slightly less urgent than other issues.

The three knowledge gaps with the highest degree of agreed feasibility (largest proportion of audience considering there to be high feasibility) were those regarding the dissemination of information through policy support, update and understanding of management tools and influencing public perception of management ([Q3], [Q6] and [Q8]; Figure 3). However, attendees considered there to be particularly low feasibility for how transferrable our current knowledge of predator impacts on waders is to other systems [Q10].

Wider impacts of predator management

Consideration of the wider impacts of predator management focussed on how deployment strategies can be designed to achieve specific goals [Q5] and when they should be combined for greater impact [Q12]. The response of waders to both management interventions [Q7] and to meso-predators in the presence of apex predators [Q9] were also key knowledge gaps exploring beyond the direct impacts of management upon predators. Discussion of these issues highlighted the importance of identifying the goals of predator management for breeding waders, as this will influence the design, the spatial and temporal scales of deployment and the geographical targeting of management approaches.

SUMMARY

This workshop provided a very valuable opportunity to identify the most pressing questions in this issue of fundamental importance to recovering breeding wader populations in western Europe. We consider all 12 knowledge gaps to be priorities, especially as their importance, urgency or feasibility may vary geographically. We hope that this work provides a platform for the rapid development of studies to address many of these knowledge gaps and will help to facilitate the collaborations that will undoubtedly be needed to reduce predator impacts on breeding waders before it is too late.

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Table 1 Table showing the criteria for being prioritised as high, medium or low priority for the different classifications of urgency, importance and feasibility. Feasibility included a range of attributes: whether there was existing data availability (need for additional empirical studies), the logistics, cost, scale and time requirements and also the legal constraints (licensing requirements etc.)

	Urgency	**Importance**	**Feasibility**
High	Likely to require swift action	Has potential to greatly influence outcomes of interventions, or alter current practices	Relevant data exist or could be easily gathered (low cost / time / logistic requirements), with few / no legal constraints
Medium	May require swift action in some or all aspects	May influence some or all aspects of outcomes of interventions, or alter some or all current practices	Some relevant data exist and / or could be gathered but some logistic or legal aspects likely to be complex / challenging
Low	Unlikely to require swift action	Unlikely to greatly influence outcomes of interventions or current practices	Relevant data not available and gathering those data would be complex / challenging

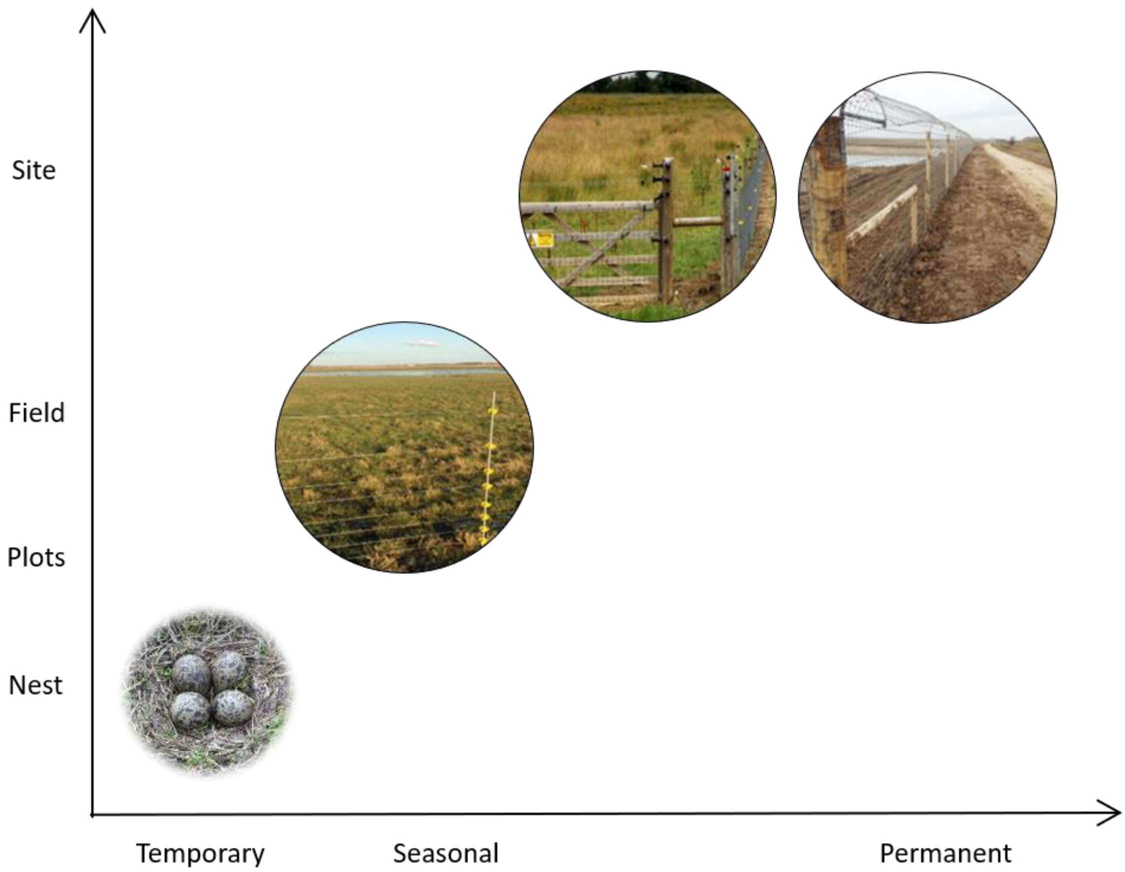

Figure 1 Plot showing relative spatial and temporal scales for the different fence types. Photos of combination and barrier fences from White & Hirons (2019).

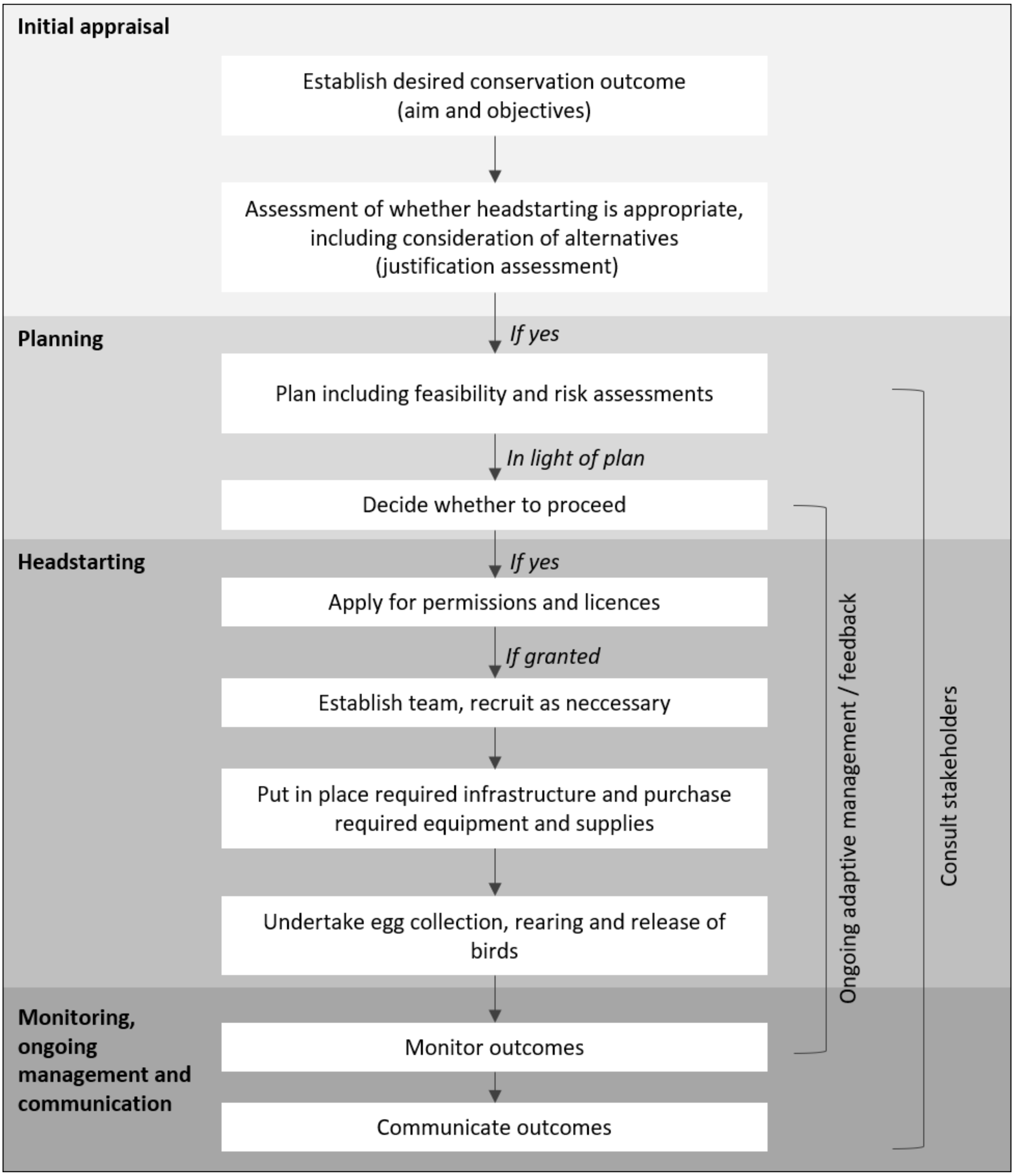

Figure 2 A flow chart of the processes that should be followed for any headstarting project from initial concept through planning, doing and monitoring outcomes. Adapted from the Scottish Code for Translocations (National Species Reintroduction Forum 2014a).

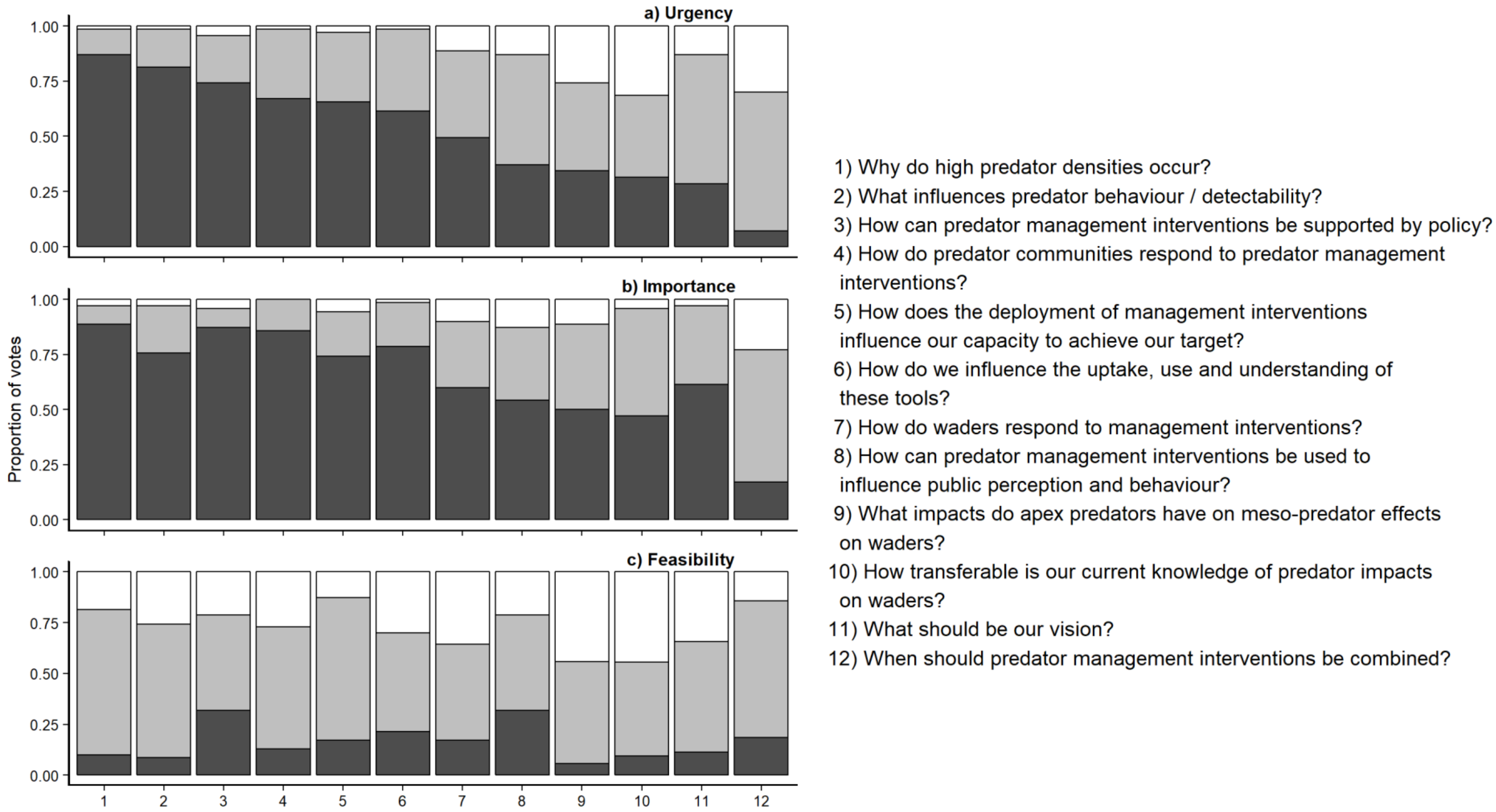

Figure 3 Proportion of the 70 workshop attendees that voted for each of the 12 knowledge gaps over the three classifications of a) urgency, b) importance and c) feasibility on the three priority levels of high (dark grey), medium (light grey) and low (white; see Table 1 for definitions). The knowledge gaps are in descending order of high urgency vote proportion.